Department of Environmental Biology

Fire impacts on restored shrublands following mining for heavy minerals near Eneabba, southwestern Australia

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DECLARATION

To the best of my knowledge and belief this thesis contains no material previously published by any other person except where due acknowledgment has been made.

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university.

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ABSTRACT

Following mineral-sand mining in the northern sandplains near Eneabba, southwestern Australia, rehabilitation managers have the difficult task of restoring shrubland communities of exceptional plant species richness. Management aims to restore a fully functional and self-sustaining shrubland community with similar vegetation and resilience properties to that of the surrounding natural vegetation. This thesis examines the performance of the restoration program by Iluka Resources Ltd. (and their predecessors) by comparing current vegetation properties and their response to fires on previously mined land *versus* the surrounding natural shrubland. As biomass accumulates post-restoration, fires will return as a natural disturbance factor and, as a result, a desirable measure of restoration success might include the ability of the postmined lands to recover from disturbance.

Pre-burnt plant species diversity, composition, structure and key functional attributes in four mined sites rehabilitated 8 (R8) to 24 (R24) years ago were compared with those of surrounding natural areas classified on the basis of substrate type (low and high sand dunes, shallow sand swales, sand over laterite and sand over limestone). The rehabilitated sites (except R8) had more species (about 140) than natural sites (about 100) with 12–37% species in common with natural sites. Floristic composition was most similar to the local swales and dunes (physically closest). Two strong colonizers, the fire-killed *Acacia blakelyi* and the fire-tolerant *Melaleuca leuropoma*, were universally present. Plant densities were about a quarter to half those of natural sites. Fire-resprouters were under-represented. Growth-form distributions were most similar to those of the dunes, with some woody shrubs up to 2.5 m tall present. Greater iron levels and soil hardness (penetrability) were the only soil factors consistently greater in rehabilitated sites.

Following experimental fires at the same study sites, species richness fell by 22–41% in rehabilitated sites but increased by 4–29% in natural sites. Species present before fire were reduced by 40–56% in rehabilitated sites and 4–12% in natural sites. Only 42–66% of resprouting species recovered in rehabilitated sites, whereas 96–100% recovered in natural sites. Nonsprouting species recruitment was also lower in

rehabilitated (18–57%) than natural (67–85%) sites. Seedling mortality over the first summer after fire was higher in rehabilitated sites (59-86% death of individuals) than in natural sites (14-60%). PCoA ordination showed that fire altered the floristic composition of rehabilitated sites much more than it did in natural sites, mostly attributable to the loss of the extant resprouter species.

It was found that the smaller lignotuber size (source of dormant buds) recorded in rehabilitated (*vs.* natural) resprouters was responsible for their higher post-fire mortality. For equivalent crown size in ten common lignotuberous shrub species, lignotuber circumferences were, on average, 50% smaller at rehabilitated sites. As a result, overall persistence in these species was much lower in rehabilitated (mean of 52% alive, range of 11–93%) *versus* natural sites (mean of 96%, range of 79–100%), but improved with time since restoration for five of the ten selected species. Apart from differences in the age of the plants (natural sites having much older plants recruited after previous fires), the lower soil penetrability at rehabilitated sites may have restricted lignotuber development. A tradeoff favoring a higher crown volume to lignotuber size ratio was also apparent in nine of the ten species with greater crown volumes (by 37%) and smaller lignotubers (by 36%) in rehabilitated sites.

Demographic attributes for six selected woody species were compared between rehabilitated and natural sites (~3-30 years since disturbance) to investigate growth patterns and optimum fire-return intervals. At matched years since restoration *or* last fire, nonsprouter species in rehabilitated sites grew larger (1.1 to 4.7 times) and produced/stored more viable seeds per plant (1.1 to 10.9 times). Despite older aged individuals in natural sites at matched years since restoration *vs.* last fire, restored resprouters were larger (1.1 to 3.6 times) and produced/stored more viable seeds (1.1 to 6.9 times). Although greater growth and fecundity rates were recorded in rehabilitated sites, the estimated optimum fire-return interval based on maximum seed production was similar in rehabilitated and natural sites for five out of six species. However, mean fire intervals typical of surrounding natural vegetation near the Eneabba area (13 years over the last 40 years) may not be suitable for rehabilitated minesites at Eneabba, whereby longer initial fire intervals (20–30 years) would better ensure persistence of resprouter individuals via the seedling recruitment strategy and resprouting strategy.

My study indicated that the returned vegetation can at present be classified as “rehabilitated” or “partially restored” but not “completely restored” since the original plant diversity, composition, structure, and resilience properties to fire have not yet been achieved. It may not be possible/realistic to achieve complete restoration since mining is such a destructive disturbance type that some complex ecological attributes may take centuries to develop. I discuss six key factors as important in improving the overall restoration success at Eneabba: 1) restoration of a deeper topsoil and looser subsoil profile; 2) collection of appropriate amounts of only local provenance species, mulch and topsoil; 3) control of highly competitive species; 4) management of fertilizer additions; 5) reseeding and replanting in subsequent years after the initial restoration treatments, including after initial fires; and 6) delaying the introduction of management fires until the restored vegetation develops sufficient fire-resilience properties.

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Theoretical Context

1.1 Overview of study

As human populations have been increasing, there has been a proportional increase in anthropogenic activities resulting in environmental degradation and destruction of the Earth's biota (Novacek and Cleland 2001). For many people, biological diversity has intrinsic value and, because of increased community awareness and legislative obligations, the field of Restoration Ecology has emerged. This discipline is defined by SER (2004) as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed". The ultimate goal is to restore a self-supporting ecosystem that is resilient to environmental pertubations without the need for further assistance (Urbanska et al. 1997; SER 2004)

This thesis presents a study on shrubland restoration after heavy-mineral sand-mining near Eneabba, southwestern Australia. Four themes were highlighted in this thesis. First, restoration efforts were evaluated by comparing compositional, structural, and functional attributes between rehabilitated and surrounding natural analogues. Second, experimental fires were introduced to study sites to determine the vegetation's resilience to natural disturbances. Third, growth and reproductive capacities of common species were compared between rehabilitated and natural sites. Fourth, recommendations are given to facilitate the restoration of natural analogues, and to improve the overall persistence of the restored shrublands after fire.

1.2 Research objectives

The overall objective of this thesis was to evaluate the performance of the Iluka Resources Ltd. restoration program by comparing existing vegetation patterns and the response of the vegetation to fires on shrublands restored on previously mined land *versus* (*vs.*) surrounding natural analogues. In Mediterranean regions such as southwestern Australia, it is inevitable that fire will return as a natural disturbance factor on vegetation restored on previously mined lands. Therefore, a fundamental measure of restoration success concerns the ways in which these restored ecosystems respond to fire (EPA 1995; Grant et al. 2007). The specific objectives of this study were to:

1) Compare species diversity, composition, structure, and some functional attributes of post-mine rehabilitated sites with that of surrounding natural vegetation types (on high dune, low dune, swale, laterite and limestone substrates);
2) Compare the floristic response of vegetation to fires on post-mine rehabilitated sites with that of surrounding natural vegetation types;
3) Compare post-fire persistence of resprouter individuals for ten common lignotuberous shrub species on post-mine rehabilitated sites *vs.* surrounding natural sites;
4) Compare demographic attributes (with time, i.e., plant size, fecundity and longevity) and optimum fire intervals for selected species from post-mine rehabilitated and natural shrubland sites;
5) Provide management recommendations to Iluka Resources Ltd to facilitate the restoration of natural analogues, and improve the overall persistence of the restored shrublands after fire.

1.3 Thesis structure

This thesis consists of 6 chapters with this first one introducing the subject matter of the thesis, the basic theoretical background to restoration ecology, a brief description of the mining history and restoration of vegetation methods used at Eneabba, and an environmental description of the Eneabba study area. In Chapter 2, comparisons are made on the plant diversity, composition, structure, and some functional attributes of the rehabilitated sites with that of surrounding natural sites (classified on the basis of substrate types: high dune, low dune, swale, laterite and limestone). Following an assessment of the vegetation patterns present, fires were introduced to the same study sites to examine post-fire responses of species recruitment and vegetation recovery patterns (Chapter 3). Post-fire persistence of ten common resprouting species were studied (rehabilitated *vs.* natural sites) in more detail in Chapter 4. Chapter 5 examines how six common woody species with different regeneration and seed storage modes perform (i.e., how they develop: size, fecundity and longevity) over time in rehabilitated sites, and how their reproductive capacities affect their optimum fire-return intervals for population recruitment after fires. Chapter 6 provides a synthesis of the preceding chapters linking the ecological processes identified in the study to the management of the post-mined restored shrubland sites. Areas requiring further research are also discussed.

1.4 Restoration ecology

1.4.1 Human disturbance and the development of restoration ecology

Human-caused disturbance is a world-wide phenomenon which occurs when our actions result in physical (and biological) changes to the natural environment. It varies in extent and severity from mild impacts on native vegetation and habitats such as walking tracks to more serious impacts such as mining of mineral sands. According to Vitousek et al. (1997), approximately one-third to one-half of the earth's land surface has been somehow transformed by human action. A study conducted by the United Nations Environment Programme, designed specifically to quantify the extent of disturbance on a world-wide basis, found that approximately 2000 million ha of land had been disturbed by 1983, and the rate of disturbance was 5-7 million ha per year (Oldeman 1994).

Australia has been severely affected by human disturbance. Since European settlement only a little more than 200 years ago, the natural vegetation estate has been reduced as alternative land uses have taken its place. According to Hobbs and Hopkins (1990), only 26% of the Australian land surface has not been utilized by humans, and much of this land occurs in the arid zone where the environment is inhospitable and of low productivity. Hobbs and Hopkins (1990) identified four general land use categories in Australia based on the degree of environmental impact:

1. REMOVAL: involves complete removal of native vegetation cover and disruption of all ecological processes. This is the most radical form of human disturbance (e.g., urban development, transport construction, and mining)
2. REPLACEMENT: involves removal of vegetation and replacement with intensively managed vegetation systems (e.g., agriculture, horticulture and plantation forestry;
3. UTILIZATION: involves utilization of existing vegetation with some consequent modification (e.g., pastoralism, recreation, timber harvesting); and
4. CONSERVATION: involves conservation with minimum deliberate modification to habitats (e.g., nature reserves and national parks).

There is a consensus in the scientific community, that the current rate of environmental degradation of the Earth's biota is taking place on a catastrophically short timescale (Novacek and Cleland 2001). It is now also widely accepted that biological diversity has a variety of significant values, ranging from the economic (e.g industrial, medical, food) to ethical, moral and intrinsic (Nash 1990), including an obligation to preserve biodiversity for future generations. This relatively recent increase in awareness and community concern for the environment has resulted in the emerging discipline of Restoration Ecology, which aims primarily to use our knowledge of the ecology of species and ecosystems to restore former patterns of biodiversity and ecological processes to disturbed sites. Over the past few decades in particular, there has been a tremendous upsurge in restoration activity, as for example in the Australian mining industry where there has been growing emphasis on the planning, funding and research on restoration of post-mined lands (Brooks et al. 1996). Public scrutiny has also resulted in legal requirements concerning the restoration of post-mined areas (Bell 1994).

1.4.2 Definitions of restoration, rehabilitation, and reclamation

Communication among scientists often relies on careful definition of terms to ensure consistency of usage. However, defining restoration has proven problematic for many ecologists (Hobbs and Hopkins 1990; Hobbs and Norton 1996). In the literature, revegetation, reclamation, rehabilitation, and restoration appear to have overlapping meanings and are used interchangeably and differently by different authors (Hobbs and Hopkins 1990). To reduce some of the ambiguity in these definitions, I have constructed a model which attempts to summarize how most international practicioners view these definitions (Figure 1.1). In this model, the definitions are separated depending on the final condition to which the system is repaired (e.g., Hobbs and Hopkins 1990). The overall process or procedures involved in repairing damaged ecosystems can be described as "restoration" (Hobbs and Hopkins 1996), but the degree to which the system is repaired to that of a pre-disturbed ecosystem dictates whether the system has been 'revegetated', 'reclaimed/rehabilitated', or 'completely restored'. Most international practicioners view a 'restored (completely) site' to be distinct from a 'revegetated', 'rehabilitated', or 'reclaimed' site as the latter systems do not entirely represent a pre-existing indigenous ecosystem (EPA 1995; Lubke and Avis 1999; McDonald 2000). On the basis of Hobbs and Norton's (1996) proposal, and of the other literature cited here, I use the term "restoration" as reflecting the primary objective of post-mining land management in this thesis, and a completely restored site, that is, one which closely models that of the pre-disturbed system, is the ultimate goal. However, it is important to understand that such a destructive disturbance severely constrains complete restoration of a pre-disturbed functional ecosystem. It is more realistic that a closely matched species composition may be achieved, and that a functional system not exactly the same as that previously present, will eventuate (i.e., indvidualistic hypothesis: Gleason 1926)

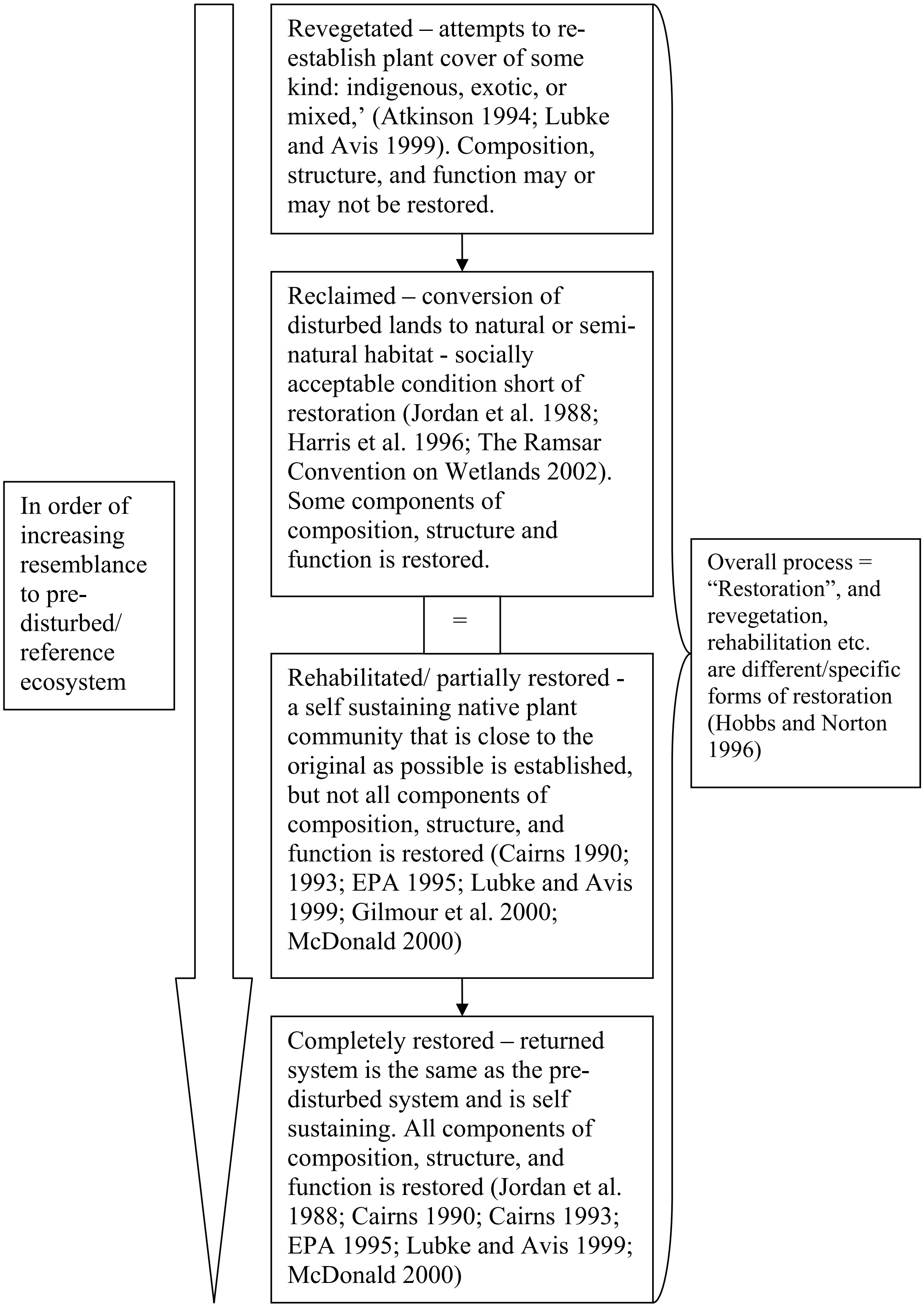


Figure 1.1: Common definitions (describing the repair of damaged systems) interpreted by many international (restoration) practitioners

1.4.3 Restoration of ecosystem attributes

Successful restoration requires the return of the following three ecosystem components to disturbed sites: species diversity, structure (including composition), and function (Ewel 1987; Anderson 1993; Hobbs and Norton 1996; Lockwood and Pimm 1999; Reay and Norton 1999; Ruiz-Jaen and Aide 2005). These attributes depend on, and interact with, each other. For instance, recreating structure and diversity without restoring function, or recreating function in the absence of structure and diversity fails to constitute complete or successful restoration (Lamont 1995b; Hobbs and Norton 1996; Reay and Norton 1999; Lockwood and Pimm 1999). E.g., without fire (function), prairie vegetation can succeed to woodland and forest under some climate and soil conditions (Lockwood and Pimm 1999).

Species diversity is the most frequently used ecosystem component utilized in restoration programs (e.g., Bowles and Jones 1999; Reay and Norton 1999; Walli 1999; Woodward et al. 1999). It can be divided into two components: richness (number of species in the reference area), and their relative abundances (Odum 1996; Walli 1999; Nichols and Nichols 2003). Indices commonly used to quantify diversity include the Shannon-Weaver index (Shannon and Weaver 1949), Simpson’s index (Simpson 1949), and Pielou’s Evenness index (Pielou 1966). Although species diversity is used as an indicator of perceived community or system quality, vegetation composition and structure may be equally important in assessing restoration success (Henderson 1999). For instance, species richness can be restored via broadcasting seeds selected from a variety of nearby natural substrate types containing different species, but this will be meaningless (or misleading) if the original composition typical of the predisturbed substrate type is not restored. It is therefore essential to target the species of the pre-disturbed community. Other structural attributes are usually determined by measuring species cover, species density, growth-forms (such as trees, shrubs, herbs), biomass or vegetation profiles (Salinas and Guirado 2002; Kruse and Groninger 2003; Wilkins et al. 2003). These measures are also useful in predicting the on-going direction of plant succession (Ruiz-Jaen and Aide 2005).

Basic ecological processes such as biological interactions (e.g., mycorrhizeae activity, herbivory), energy and water flows, and nutrient cycles are vital for the success of restoration projects because they provide information on the developing properties and resilience of the restored ecosystem (Ruiz-Jaen and Aide 2005 Individual species may perform a range of ecological functions (Peterson et al. 1998), e.g.; regulate biogeochemical cycles (Vitousek 1990; Zimov et al. 1995), alter disturbance regimes (Dublin et al. 1990; D'Antonio and Vitousek 1992), or modify the physical environment (Jones et al. 1994; Naiman et al. 1994). An increase in species richness is therefore often associated with an increase in functional diversity, and perhaps producing an increase in ecological stability or resilience which is often cited as being positively correlated with diversity (Darwin 1859; Schindler 1990; Frost et al. 1995; Ewel and Bigelow 1996; Tilman 1996; Peterson et al. 1998). However, few studies in Australia (or elsewhere) have tested the resilience properties of restored vegetation by re-introducing major disturbance processes, such as fire, which are important in surrounding natural areas (Grant and Loneragan 1999; Smith 2001; Ross et al. 2004).

In an attempt to evaluate how restoration success has been measured around the world (based on ecosystem attributes), Ruiz-Jaen and Aide (2005) examined 468 restoration studies published in Restoration Ecology from 1993 to 2003. Only 68 measured restoration success after seeding or planting (see Table 1.1). Most studies were conducted in North America (53%), Australia (19%) and Europe (16%), most likely because these more developed countries enforce environmental laws and have financial resources to fund restoration work. Only 38% of these studies measured three ecosystem attributes (diversity, structure, function), 59% measured two attributes (28 studies: diversity and vegetation structure, 6 studies: diversity and ecological processes/function), while 3% of studies measured one ecosystem attribute (2 studies: diversity). Plant (79%) and arthropod (35%) species richness were the most common measures of diversity. Only a few studies recorded recovery in a wider range of taxa (Table 1.1). All studies evaluated the recovery of either vegetation structure or diversity since most laws require that restoration criteria include some specified vegetation component and since it is assumed that native fauna and ecological processes will eventually return following vegetation establishment (Allen 1992; Toth et al. 1995). Furthermore, vegetation is much easier to measure than fauna which can be highly mobile. Plant cover (62%), density (58%), biomass (39%), and height (39%) were the most common measures of vegetation structure. Ecological processes were the least studied attribute because they are generally slower to recover compared with diversity or vegetation structure (Chambers et al. 1994; Kindscher and Tieszen 1998), and may require multiple measurements requiring more time and money (Chambers et al. 1992). Biological interactions (60%), nutrient pools (47%) and soil organic matter (39%) were the most common measures of ecological function.

Table 1.1 (modified from Ruiz-Jaen and Aide (2005)): Key ecosystem attributes evaluated across 68 restoration studies worldwide: 1993–2003. Note: references for authors cited in this table are available in Ruiz-Jaen and Aide (2005).

				Diversity					*Vegetation Structure*					*Ecological processes*			*Replication*		
GeoRe	*Habitat*	*PreUse*	*Techn*	*Plant*	*Arth*	*Bird*	*Othr*	*Guild*	*Cover*	*Density*	*BioM*	*Height*	*Litter*	*BioInt*	*NutPo*	*SOM*	*Rest*	*Ref*	*Authors*
One ecosystem attribute																			
NA	Grass	Agr	Sdl	X													13	2	Sluis (2002)
NA	Ripar	Culve	Sdl	X	X												1	1(1)	Purcell et al. (2002)
Two ecosystem attributes																			
AUS	Dunes	Transp	Sdl	X	X				X	X							5	5	Webb et al. (2000)
EUR	MonF	Mine	Sdl	X	X							X					6	1	Tajovsky (2001)
AFR	Dunes	Mine	Sd	X	X	X	X	X	X			X					3	1	Weiermans and van Aarde et al. (2003)
AFR	Dunes	Mine	Sd	X	X	X	X	X		X		X					6	2	van Aarde et al. (1996)
NA	Conif	Agr	Sdl	X	X			X	X	X	X		X				1	2	Willet (2001)
NA	DryF	Transp	Sd/Sdl	X	X			X	X	X		X					4	7(4)	Longcore (2003)
EUR	Grass	Mine	Sd	X	X			X		X							5		Cullen and Wheater (1993)
EUR	Grass	Transp	Sd	X					X								16		Andres et al. (1996)
NA	Wetld	Mil	Sd	X					X								2	6	Parikh and Gale (1998)
NA	Ripar	Defo	Sd	X		X			X			X					5		Kus (1998)
NA	DryF	Dam	Sd/Sdl	X			X		X	X							1	1	Patten (1997)
NA	Decid	Defo	Sdl	X					X		X						1		Larson (1996)
NA	Wetld	Transp	Sdl	X					X		X						2	3	Forbes (1993)
NA	MonF	Agr	Sdl	X					X	X		X					8		Kruse and Groninger (2003)
NA	Wetld	Mine	Sd/Sdl	X					X	X							6	6	Cooper and Macdonald (2000)
AUS	Grass	Agr	Sd	X					X	X		X	X				5	2(2)	Wilkins et al. (2003)
EUR	Ripar	Defo	Sdl	X					X	X							2	2	Salinas and Guirado (2002)
AUS	WetTep	Dam	Sdl	X						X	X						2	1	Smale et al. (2001)
EUR	Grass	Agr	Sd	X							X						1		Kleijn (2003)
ASIA	Wetld	Defo	Sdl	X						X	X	X					29	5	Walters (2000)
SA	WetTro	Mine	Sd/Sdl	X						X	X	X	X				3	1(1)	Parrota and Knowles (1999)
AUS	DryF	Mine	Sd	X						X							1		Ward et al. (1996)
AUS	DryF	Mine	Sd	X				X	X	X							4	1(2)	Grant and Loneragan (2003)

AUS	Grass	Pastu	Sd/Sdl		X				X	X							1		O'Dwyer and Attiwil (2000)
AUS	DryF	Mine	Sd/Sdl		X			X	X			X	X				6	2	Andersen (1993)
AUS	Grass	Mine	Sd		X			X	X			X	X				7	3	Bisevac and Majer (1999)
ASIA	WetTro	Mine	Sdl			X						X					8	7(4)	Passell (2000)
AUS	Priairie	Agr	Sdl		X			X	X	X		X	X				3	1	Watts and Gibbs (2003)
NA	Sonif	Agr	Sdl	X	X									X			1	1	Meyer and Sisk (2001)
NA	Grass	Defo	Sd	X	X									X			10	11	Leong and Bailey (2000)
NA	MonF	Mine	Sdl	X	X									X			18	5	Holl (1995)
AUS	WetTro	Mine	Sd	X	X		X									X	17	18	Andersen and Sparling (1997)
NA	Decid	Pastu	Sdl	X		X								X			1	1	Germaine and Germaine (2002)
EUR	Alpine	Defo	Sdl	X										X			1		Urbanska and Fattorini (2000)
SA	MonF	Pastu	Sdl						X	X	X	X	X		X	X	1		Rhoades et al. (1998)
NA	Wetld	Dre/Dis	Sdl						X	X	X				X	X	2	1	Sheridan et al. (1998)
NA	Decid	Mine	Sdl						X		X	X		X	X	X	4		Helm (1994)
EUR	Grass	PowS	Sd						X						X		2		Shaw (1996)
NA	Wetld	Mine	Sdl								X				X	X	1	1	Craft et al. (2002)
NA	Wetld	Transp	Sdl								X				X	X	1	1	Zedler and Callaway (1999)
Three ecosystem attributes																			
NA	Wetld	Pastu	Sdl	X	X		X		X					X			1	1	Tanner et al. (2002)
AUS	DryF	Mine	Sdl	X	X	X	X	X		X		X		X			2	2	Nichols and Nichols (2003)
AUS	WetTro	Transp	Sdl	X	X			X		X			X	X			3	2	Jansen (1997)
NA	DryF	Pastu	Sdl	X					X	X	X	X		X	X	X	1		Whisenant et al. (1995)
NA	Wetld	Defo	Sdl	X						X	X	X	X	X	X		2	2	McKee and Faulkner (2000)
NA	Wetld	Mine	Sd/Sdl	X					X	X	X	X			X	X	1	27	Clewell (1999)
AFR	Dunes	Mine	Sdl	X						X					X	X	4	2,1	van Aarde et al. (1998)
EUR	Grass	Mine	Sd	X					X		X				X		1		Chapman and Younger (1995)
NA	Decid	Agr	Sdl	X		X			X	X		X		X			13,7		Twedt et al. (2002)
NA	Conif	Agr	Sdl	X		X			X	X	X		X	X			28	5	Houseman and Anderson (2002)
EUR	Conif	Defo	Sdl	X		X	X			X		X		X			85		Vallauri et al. (2002)
SA	MonF	Transp	Sd	X		X	X			X				X			15		Rosales et al. (1997)

NA	Wetld	Agr	Sdl	X						X	X			X			3	6	Shear et al. (1996)
NA	Wetld	Transp	Sdl	X					X	X						X	6	11	Morgan and Short (2002)
NA	Conif	Mil	Sd/Sdl	X					X		X	X		X			1	1	Fimbel and Kuser (1993)
NA	Priairie	Agr	Sd	X				X	X	X					X	X	2	2	Kindscher and Tieszen (1998)
NA	Priarie	Agr	Sd	X				X							X	X	4	1	Fuhlendorf et al. (2002)
NA	MonF	Mine	Sd	X				X	X		X				X	X	2	1	Chambers et al. (1994)
AUS	MonF	Pastu	Sdl		X				X				X	X			3	2(1)	Reay and Norton (1999)
NA	Wetld	Mine	Sdl		X	X		X	X					X			18		Mulyani and DuBowy (1993)
NA	Ripar	Defo	Sdl		X			X	X			X		X			1	1	Williams (1993)
EUR	MonF	Mine	Sdl				X			X	X				X	X	6	1(1)	Pizl (2001)
NA	MonF	Mine	Sd/Sdl				X						X		X	X	1		Allen (1993)
NA	Priarie	Mine	Sd				X				X			X	X		1		Corbett et al. (1996)
NA	Conif	Mine	Sd				X		X					X	X		1	1	Moynahan et al. (2002)
EUR	Grass	Pastu	Sdl				X			X	X			X			3	4(1)	Greipsson and El-Mayas (2000)
		Total		49	22	9	13	16	37	34	23	23	12	23	18	15			

Geographic region (GeoRe): AFR = Africa; AUS = Australia; EUR = Europe; NA = North America; SA = South America.

Habitat: Decid = decidous forest; Conif = coniferous forest; DryF = dry or coastal forest; Grass = grassland or savanna; MonF = Montane forest; Ripar = riparian forest; Wetld = wetlands or mangroves; WetTep = wet temperate; WetTro = wet tropics.

Previous use (PreUse): Agr = Agriculture; Culve = culverts construction; Defo = deforestation; Dre/Dis = dredging and discharges; Mil = military practices; Nat = natural disturbance; Pastu = pastures; PowS = power stations; Transp = transportation development.

Restoration technique (Techn): Sd = seeds, sdl = seedlings, Sd/Sdl = seeds and seedlings

Diversity: Arth = arthropods; Othr = microbes, fungi, earthworms, reptiles, small mammals, or fish; Guild = guilds or trophic levels.

Vegetation structure: cover = plant cover; Density = plant density; BioM = plant biomass or basal area; Litter = litter layers, cover, biomass, or production.

Ecological processes: BioInt (biological interactions) = herbivory, mycorrhizzae, pollination, predation, seed dispersal, parasitism, or competition; NutPo = nutrient pools; SOM = soil organic matter or carbon isotopic composition.

Replication: Rest = number of restored sites; Ref = number of reference sites. The number in parenthesis represent control sites (e.g. prerestored conditions of areas of natural regeneration).

1.4.4 Reference sites

Evaluation of restoration success ideally requires determining if the key ecosystem components (diversity, composition, structure and function) are similar to, or are on a similar successional trajectory, to those of natural reference sites (Chambers et al. 1994). Restoring a perfect replica of the predisturbed ecosystem may not be realistic because: (i) it is rarely possible to determine what historic or prehistoric ecosystems looked like, or how they functioned, due to the lack of historical data; (ii) ecosystems are dynamic in nature; (iii) environmental change, especially of the substrate, is irreversible for certain sites; (iv) economic and social costs may be prohibitive; and (v) understanding of natural processes is limited (Aronson et al. 1993; Cairns 1993; Pickett and Parker 1994; Jackson et al. 1995; Hobbs and Norton 1996; Wali 1999; Zedler and Callaway 1999; Choi 2004). A more common and realistic means of evaluation is through comparison with neighbouring native reference areas that exhibit the desired ecosystem properties (Chambers et al. 1992; Aronson et al. 1995). The selection of reference sites should occur in similar geographical areas with similar substrate conditions and vegetation assemblages, preferably adjacent to the restored sites since the diversity and structure in the surrounding natural areas are most likely similar to what was in the pre-disturbed area and will decrease with increasing distance away. Replication of reference sites is also desirable in order to account for the natural variation which exists in surrounding natural areas (Ruiz-Jaen and Aide 2005).

Pickett and Parker (1994) argue that choosing a reference state or system is a “pitfall” or “trap” and should be avoided by up-to-date restoration ecologists because ‘there is never one ecologically legitimate or ideal system, or state to be used as blueprint’. However, in an ‘imperfect’ reply to Pickett and Parker, Aronson et al. (1995) raised concerns that if no reference or control is selected, the experiment cannot be evaluated (i.e., if no baseline data are available, how can any restored ecosystem be deemed successful?). They were astounded by the fact that many restoration projects worldwide lacked baseline data with which to be compared, and argued that although Pickett and Parker (1994) were conveying the idea that a new paradigm prevails in ecology today – the “flux” of nature – an ecosystem of reference is still needed to allow some standard of comparison and evaluation, even if the choice made is somewhat arbitrary (Aronson et al. 1993). Therefore, for the purposes of project design and evaluation, it seems desirable to have a reference state against which to evaluate restoration efforts. Restoration ecologists should also take note of Pickett and Parker's (1994) reminder that natural ecosystems are dynamic, and therefore if baseline data are used – one must be careful to define the 'state' of the analogue ecosystem against which the restored system is to be compared.

1.4.5 Completion criteria

Once the properties of the baseline ecosystem have been determined, the next stage involves setting appropriate completion criteria. Completion criteria are specific targets or measures (qualitative or quantitative) generated by land managers/ conservation authorities for monitoring and evaluating the success of restoration projects. They represent milestones aimed at ensuring that a self-sustaining ecosystem with specified properties has been achieved. Increasing resemblance of rehabilitated areas to that of surrounding natural communities is one clear indicator of the success of restoration efforts (van Aarde et al. 1996). The Environmental Protection Agency of the Commonwealth of Australia recognises three criteria in its guidelines for best practice environmental management in the mining industry (EPA 1995):

- The site can be managed for its designated land use without any greater management inputs than other land in the area being used for a similar purpose;
- Restored native ecosystems may be different in structure to surrounding native ecosystems in early phases of restoration, but there should be confidence they will change with time towards the surrounding area; and
- The restored land should be capable of withstanding normal disturbances such as fire or flood.

Completion criteria are not universally applicable, and they vary between different projects depending on the degree of modification of the landforms, the scale of projects, and the local context and the types of communities to be restored (EPA 2006). Ruiz-Jaen and Aide (2005) found considerable variation in the use of completion criteria, with the majority of studies only measuring one or two ecosystem attributes (mostly plant species diversity – 70% - see *1.4.3*).

1.4.6 Post-mine vegetation restoration studies

Strip mining occurs in coastal southern Africa, India, North America, and Australia (Lubke and Avis 1999) across a diverse range of vegetation types including eucalyptus forests (of Australia: Fox et al. 1996; Ross et al. 2004; Bell and Hobbs 2007), dune forests (of South Africa: Mentis and Ellery 1994; van Aarde et al. 1996), shrublands (of Australia: Bellairs and Bell 1993, South Africa: Holmes 2001, USA: Booth et al. 1999), grasslands (of Australia: Hooper 1985; USA: Chambers et al. 1994; Corbett et al. 1996) and riverine forests (USA: Clewell 1999). Although mining accounts for a small portion of the earths land area (<1% of South Africa/USA/Australia: Australian Bureau of Statistics 2002; Bell 1998;) compared to other human land uses such as agriculture (38% of earths land surface: Shiklomanov 2000), mining activities are only run temporarily or usually for short periods (usually only years to decades for a mined block). Therefore, restoration of the mined lands is desired to ensure an aesthethically pleasing landscape which matches well to that of the surrounding landscape, or otherwise returns the land to some viable new land-use.

While mine revegetion studies attempt to restore one or more ecosystem properties, evaluations of completion criteria are rarely published (Nichols 2006), with Alcoa's studies on bauxite-mined jarrah forests of southwestern Australia one of the few exceptions (Elliott et al. 1996). The lack of published completion criteria may be a result of the difficulties associated with restoring such complex ecological relationships. Restoring native species richness is probably the most common target in restoration programs (Jefferies et al. 2001; Koch 2007a) but has proved to be a difficult task. For instance, after restoring vegetation on mined peatlands in the southern rocky mountains of Colorado, mined sites recorded only 30 plant species compared with 122 (four times more) in unmined sites, and 43% of the returned species were not respresented in the undisturbed vegetation. In restored bauxite-mined forests of Brazil, species richness does not approach that of unmined communities after 13 years (Parrotta and Knowles 1999). Similarly, Holl (2002) found that in 15 coal mines of eastern USA, woody species richness was much lower in restored sites even after 35 years.

Alcoa World Alumina Australia (Alcoa) is one of the few companies that have been successful in re-establishing species richness equivalent to that of surrounding natural vegetation (Tacey and Glossop 1980; Nichols and Michaelson 1986; Koch 2007a; Norman et al. 2006). Much of this success is attributed to the substantial research conducted in developing effective techniques for restoration, including deep-ripping procedures to provide sufficient water infiltration and root penetration, propagule-containing topsoil replacement (double stripped) in the same order as was pre-mined, mulching from natural areas to release the seeds of serotinous species onto restored topsoil, and handbroadcasting of seeds of species which are poorly represented in restored sites (Koch 2007b). Koch (2007a) found that approximately 70% of species richness is returned through direct transfer (in pre-mined order) of fresh topsoil (with its store of viable seeds and other plant parts). Addition of local provenance seeds comprising between 78 and 113 native species also contributed significantly to the species richness of post-mined sites. Another 20 species were added using tissue culture and nursery-raised stock planted in to restored sites. As a result of these considerable research efforts, Alcoa has received numerous awards from industry and government, including the 2003 Society for Ecological Restoration International Model Project Award, World Environmental Centre Gold Medal (1990), and the United Nations Environment Program listed Alcoa on the Global 500 Roll of Honour for environmental achievement in 1990 (Bell and Hobbs 2007).

Restoring plant density has also been an important objective in mining restoration programs (Koch and Ward 1994). In reclaimed coal mine lands of Wyoming, USA, Booth et al. (1999) found that the lower densities recorded in mined versus unmined shrubland could be improved by hand-broadcasting of seeds to reach the completion criteria target of 1 shrub/m^2. In southwestern Australian shrublands, Bellairs and Bell (1993) report that the spreading of mulched strips of native vegetation into mined sites contributed most (96%) of the germinable seeds in the soil seedbank because of the dominance of serotinous species in the area.

Species compositional similarity has been one of the most difficult components to restore after mining, especially for vegetation types of high species turnover. Cooper and MacDonald (2000) in restored vegetation of mined peatlands of Colorado, U.S.A, recorded that the restored sites contained too many exotic species uncharacteristic of undisturbed sites. In Alcoa's case, mean compositional similarity between restored and natural forests were lower (about half) than those of forest-to-forest similarity (Norman et al. 2006; Koch 2007a,b). Much of the difference in plant community composition was attributed to the lack of resprouter species in restored sites. Resprouters have proved difficult to restore (Bellairs and Bell 1993; Koch and Ward 1994; Smith 2001) since they are completely removed from the mining process (so that no vegetative resprouting is possible), and generally produce fewer, and often less viable, seeds than do nonsprouter species (Lamont and Wiens 2003). As, a result, restored communities favoured nonsprouters over resprouters, and few restoration programs have managed to restore both species richness and composition after mining. One of these few exceptions was in the Riverine forests of Florida, USA, where after mining for phosphate, Clewell (1999) found that the restored forests comprised 73 species representing all life forms and strata as in the undisturbed reference ecosystem (78 species). It was concluded that an intact ecosystem had been produced and that the sites no longer required restoration assistance.

Even fewer studies have investigated the potential impacts of the return of landscape scale natural disturbances, such as fires, on post-mining restored vegetation. Most of these important studies have occurred in Australia, and have shown that controlled fires at the right age intervals can accelerate the restoration process by boosting species richness, densities, and driving the overall species composition towards that of surrounding natural areas (Grant and Loneragan 1999, 2001; 2003; Smith 2001; Ross et al. 2004). For instance, Grant and Loneragan (2003) found that after burning restored eucalyptus forests (11-13 years since restoration) in Alcoa's minesites, dominance- diversity curves for restored sites exhibited a more similar curve to natural sites than before burning. In eastern Australian eucalypt forests, the time interval between the completion of mineral sand mining and the occurrence of fire was critical for the regeneration of sand dune vegetation, with a minimum of 10 years required by species to reach reproductive maturity and sufficient seed bank size to ensure recruitment after fire. Similarly, Hooper (1985) suggested that burning restored northern savannah vegetation (after mineral sand mining) should be delayed for at least 7-10 years, after which fires should begin to have beneficial effects on the restored native species diversity.

1.5 Mining at Eneabba

1.5.1 Mining history at Eneabba

The presence of heavy minerals in the sandplains near Eneabba was first documented in 1968 by Peers and by Rowston (Baxter 1977). Mapping and feasibility studies by Baxter (1972) showed that the concentration of heavy minerals varied between 0.2 and 40% and the region contained the largest amount of heavy mineral deposit in Australia (30,000,000 tonnes primarily comprising ilmenite, zircon, and up to 15% rutile (Playford et al. 1976). Subsequently, mining companies Ilmenite Pty Ltd, Jennings Mining Ltd, Western Titanium Ltd and Allied Eneabba Pty Ltd took over portions of the land in 1973 (Bellairs 1991).

Allied Eneabba Pty Ltd began production in a pilot plant in 1973 and Jennings Mining and Western Titanium Ltd commenced production in 1974 (Bellairs 1991). Associated Minerals Consolidated Ltd (AMC) took over Western Titanium Ltd in late 1976 (Morley 1981) and began mining in 1977 (Majer et al. 1982). In 1979, AMC took over production of 60% of the mining lease, yielding 45 000 tonnes of rutile per annum. The takeover resulted in two mining companies AMC and Allied Eneabba Pty Ltd (AEL) operating the mines at Eneabba. In Early 1986, Renison Goldfields Consolidated Ltd (RGC), AMC's parent company, took over AEL. They were separate legal entities but management and mining strategies were operated by the same staff. In late 1998, Iluka resources Ltd took over all mining leases and operations at Eneabba. Today, Iluka resources Ltd supplies 38% of the world's zircon and the Eneabba operations supplies about 55% of the company's production of 428,000 tonnes per annum. In 2007, the Eneabba region supplied 220,000 tonnes of zircon, 580,000 tonnes of ilmenite, and 77,000 tonnes of rutile.

1.5.2 Completion criteria and restoration methods at Eneabba

Restoration methods at Eneabba aim to satisfy completion criteria (developed in 1985 by EMRC (1996)) focussing on restoring species richness, plant density, and foliage cover values:

- Species richness – a mean of 6 species per m² and a minimum of 70 species in blocks larger than 10 hectares;
- Plant density – a mean of 12 plants per m² with not more than 10% bare quadrats in each block; and
- Foliage cover: total community projective cover of 32% or more, excluding *Acacia blakelyi*.

These completion criteria are currently under review.

To Dec 2008, 3839 ha has been mined of which 2478 ha has been rehabilitated and 1361 ha remains open. The restoration procedures were generally completed in the following sequence (Bellairs 1991; Jefferies et al. 1991; Bellairs and Bell 1993; EMRC 1996; Bob Wynne and Cameron Payne 2008, Iluka Resources Ltd, personal communication):

1. **Landform re-construction and contouring.** After mining for heavy minerals, the residue sand/tailings are pumped behind the concentrator and deposited back onto mined pits. The residue is used to form an approximation of the original topographic patterns after plans are made to achieve the desired sand placement, final heights and contoured shapes of the landform. Land contouring takes place using elevation scrapers. The subsoil is ripped to break up the agglomerated subsoil structure, important in providing sufficient drainage and adequate root penetration for vegetation establishment (Enright and Lamont 1992a; Holmes 2001).
2. **Topsoil spreading.** Topsoil is stripped using scrapers in natural areas ahead of the mine path. The double-stripping method is employed where topsoil is cut in two layers – the first cut being the top 5 cm of soil, the second cut being approximately 5 – 20 cm of soil (stripped to the soil colour change). Where possible,

topsoil is spread directly over post-mined tailings, otherwise, it is placed in a stockpile for an undefined period, usually between 6 months to several years. Since Iluka Resources acquired the Eneabba operations in 1999, practices have reduced the duration of stockpiling to as short as possible, usually within two years. When topsoil is required, the second cut topsoil is applied first (top 5–20 cm), followed by the first cut topsoil (top 5 cm). The topsoil is biologically important since it contains a rich source of viable seeds, organic matter, nutrients, and microorganisms (Koch et al. 1996; Shaw 1996; Grant and Koch 1997). Therefore conservation and 'fresh' replacement of the topsoil in the correct order is essential for the maintenance of plant diversity on many post-mined lands (Tacey and Glossop 1980; Bellairs and Bell 1993; Ward et al. 1996; Holmes 2001; Smith 2001).

3. **Mulching.** Mulch sourced from natural areas ahead of the mining front is applied at a rate of one hectare of cut mulch (cutting height approximately 30 cm above ground) for one hectare of restoration ground. The high fraction of serotinous (canopy-storing of seeds in woody fruits) species on the Eneabba sandplain renders the mulching process a significant contributor of viable seeds (Bellairs and Bell 1993). Mulching also assists in seed and soil stabilization – important as newly rehabilitated sites are susceptible to erosion since Eneabba regularly experiences strong winds (Bell et al. 1986).

4. **Cover crops and fertilizing.** A cover crop of oats, Sudax (sterile hybrid between Sudan grass and sorghum) and/or cereal ryegrass (*Secale cereale*) together with native *Acacia blakelyi* and *Acacia pulchella*, are randomly broadcast (about 2–3 kg per ha) over restored topsoil via cultidrills. The aim is to provide a quick cover of vegetation to assist in soil stabilization and nitrogen fixation until desired native species dominate. Since Iluka Resources Ltd acquired the Eneabba operations in 1999, the practice of seeding *Acacia* spp as cover crops was halted due to their dominating effects and weedy properties in older restored stands. Synthetic fertilizer is added via cultidrills at a rate of 70–100 kg per ha. It contains N, P (superphosphate), S, K, Cu, Zn and other micronutrients (Croprich, Perth, Western Australia). Fertilizers are added to fastrack the initial growth processes.

5. **Seeding of native species.** A native seed mix comprising 20–30 species were sown onto restored topsoil in the 1970's and 1980's. Gradually, the seed mix increased to about 50 species in the late 1990's. The composition of the species mix and from where it was sourced is unknown but recent practices have (since Iluka

aquired Eneabba operations in 1999) targeted seed mixes from specific community types (swale, dune, laterite or wetland species mix) for particular rehabilitated blocks, depending on what pre-mining vegetation existed there.

6. **Infill planting nursery stock.** Seedlings of species deficient in the topsoil and mulch (as seeds, e.g., resprouter spp.) are planted into the rehabilitated sites. This method increases the chance of species establishment and survival to maturity if the species fails or is absent from the seed mix, mulch or topsoil. Since the late 1980's, 60,000–80,000 seedlings have been cultivated in the nursery each year.

Topsoiling is usually completed in mid to late April, and mulching, fertilizing, seeding and infill planting usually occur in early May before the onset of winter rains. Maintenance of rehabilitated blocks includes constructing windbreak fences on erosion susceptible areas and applying more mulch where restored vegetation densities are low.

1.6 Description of the Eneabba study area

1.6.1 Location

This study was conducted in the mid-west region of south-western Australia, near Eneabba (29° 49' S Latitude, 115° 16' E Longitude), 280 km north of Perth (Figure 1.2). The minesites rehabilitated by Iluka Resources Ltd (and their predecessors) are located approximately 10 km south of the Eneabba town-site, and the natural sites are either adjacent (i.e., high dune, low dune, swale) to the rehabilitated minesites or up to 20 km away (i.e., laterite, limestone) (see Table 2.1 in Chapter 2 for exact co-ordinates of sites). All study sites occurred within the shrubland vegetation in the location of what is commonly known as the Northern Sandplains. Competing land uses for the area include nature conservation, mineral sand-mining, coal mining, agriculture and horticulture (Hnatiuk and Hopkins 1981).

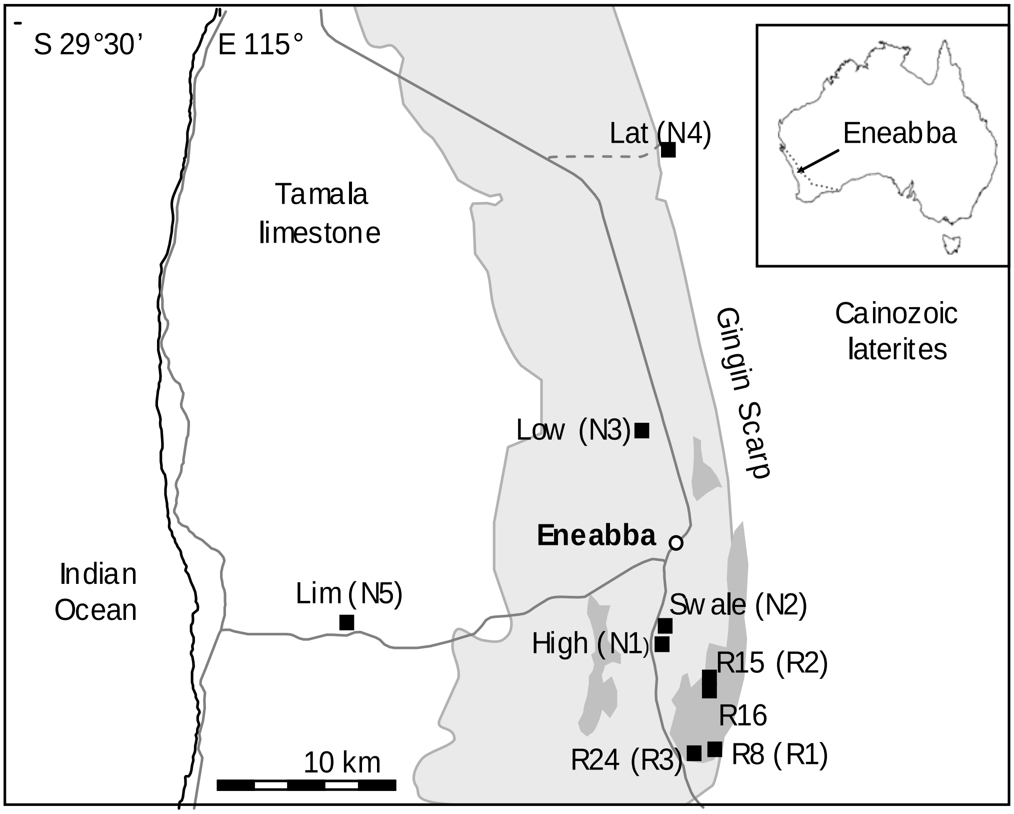


Figure 1.2: Location of study area (near Eneabba) and study sites (see table 2.1 for site descriptions). N = Natural site, R = Rehabilitated site, Eneabba sandplain = pale grey shade, Mined ares = darker grey shade.

1.6.2 Climate

Eneabba experiences a Mediterranean-type climate, with cool-wet winters and hot-dry summers. The mean annual rainfall is 502 mm, and the majority of the rain (75%) occurs in the late-autumn to early spring months to September (BoM 2008, Figure 1.3). June is on average the wettest month of the year (105 mm pa) and January the driest (7 mm pa). Evaporation (over 2000 mm pa) exceeds rainfall at all times of the year (except a few winter months) especially in summer (December, January and February) with daily maximum temperatures averaging 33–36 ºC, and regularly exceeding 40 ºC. January and February are the hottest months of the year (both 36 ºC on average). At the ground surface, temperatures may regularly exceed

60–70 ºC (Hnatiuk and Hopkins 1981) creating difficult conditions for the survival of vegetation (and any seeds) which exist there. Eneabba also experiences strong winds with records of over 16% of twice daily wind-speed readings exceeding 30 km.h-¹ and about 1% exceeding 50 km.h-¹ (Bell et al. 1986). In the afternoon, winds tend to be south-westerly in summer but range form north-westerly to south-westerly in winter (Bellairs 1991). Rain-bearing winds tend to be westerly or north-westerly whereas the easterlies tend to be dry. Compared with much of the rest of Australia, Eneabba experiences a climate with above average daytime temperatures, higher wind speeds and greater numbers of clear days (120 days per annum). Rainfall and humidity levels are also lower than average. Mean minimum temperatures are also comparatively high during the winter months.

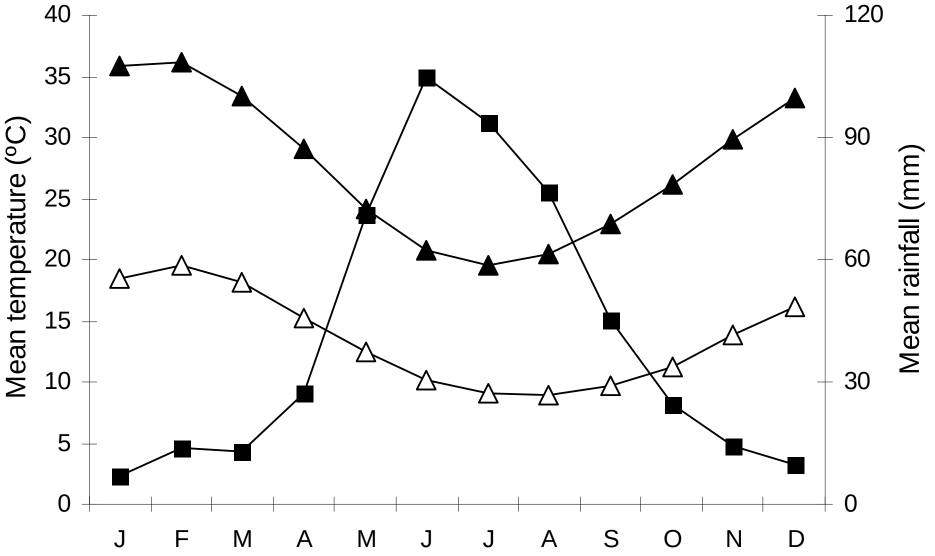


Figure 1.3 (values extracted from BoM 2008): Ombrothermic diagram for Eneabba showing mean rainfall (filled squares), mean minimum (open triangles) and maximum (filled triangles) temperatures over 12 months, from 1964-2008.

1.6.3 Geology and soil types

The study area is part of the eastern edge of the Perth Sedimentary Basin and the northern portion of the Swan Coastal Plain. It is commonly termed the Eneabba Plain (Playford et al. 1976; Commander 1978) and is bounded to the east by the Gingin Scarp (formed by the Darling fault) and to the west by the coastal belt. The Gingin Scarp represents an ancient (Tertiary or Pleistocene) coastline between 91-105 m higher than the present one (Baxter 1972). Therefore, much of Eneabba is built up of a series of early Pleistocene (or late Tertiary) shoreline, lagoon and dune deposits having locally high concentrations of heavy minerals. These deposits are associated with a series of low alluvial fans fronting the Gingin Scarp (Lowry 1974; Playford et al. 1976). The rehabilitated minesites and most natural sites of this study were located within the alluvial fans physiographic unit classified by Lowry 1974, except the laterite site that was located within the dissected region and the limestone site occurring on the coastal belt. Alluvial deposits were formed where westward flowing rivers decreased in gradient as they approached the coastal belt (Bellairs 1991). The Dissected Region is occupied by laterized Mesozoic strata reaching elevations of 250 to 300 m.

Four major soil types can be recognised near the Eneabba area differentiated by the the depth of the sand layer, and the nature of the underlying material, whether clay, laterite or limestone. The soil types can be broadly classified as dunes (high and low sandy dunes), swale, laterite or limestone. Dunes (or deep sands) typically develop on colluvial sand sheets (Bettenay 1984). Sandy (quartz) soils may extend to 10 m deep (i.e., high dune) or about 2–3 m deep (i.e., low dune) over silt and clay. The upper layer (<20 cm) is usually darkened by organic matter and there may be a gradual increase in clay content with depth. The swale soil type can be described as shallow layers of soil up to 50 cm deep followed by high concentrations of silt and clay. Sand may be blown off in windy areas leaving behind agglomerated silt and clay. The top 10–15 cm is darkened by organic matter. The lateric sandplain is generally a sandy surface (usually grey quartz sand) up to 3 m deep, has low to moderate clay content, and a high content of secondary gravels (ferruginous and aluminum nodules) derived from weathering. The underlying material is composed of layers of pisolitic (pea-size) to massive ferruginous laterite up to several metres thick (Lowry 1974). The top 10–15 cm contain darker soils with organic matter. Limestone soil types occur towards the west of Eneabba closer toward the coastline. They are generally calcareous soils from 0–50 cm in depth over Tamala limestone rock. Outcropping limestone is also common in these areas.

All the above soil types are generally infertile due to the lack of tectonic activity since the Cretaceous period allowing the development of deeply weathered soils. Low and Lamont (1985) found that nitrogen and phosphorus levels in the Eneabba soils were lower than that of South African Cape soils (with similar rainfall) and lower than that of the heathlands (or shrublands) in South Australia. Wild oats (*Avena* spp.) fails to grow in the sandy soils without the simultaneous addition of nitrogen, phosphorus and potassium (Lamont 1995a).

1.6.4 Vegetation

The sclerophyll shrublands (heathlands) of southwestern Australia are considered botanically important because of its outstanding species diversity and its high degree of endemism (Diels 1906; Gardner 1944; Marchant 1973; Lamont et al. 1984). They generally have more vascular plants per unit area than any other sclerophyll vegetation type in southern Australia (Parsons and Cameron 1974; George et al. 1979; Whittaker et al. 1979; Rice and Westoby 1983; Lamont et al. 1984). The shrublands of the Eneabba region, in particular, is of paramount botanical importance and has become a focal point for botanical research and ecotourists because of its high biodiversity (George et al. 1979; Hnatiuk and Hopkins 1981; Lamont et al. 1984; Bell and Loneragan 1985) and existing land-use conflicts (Froend 1987; Bellairs 1991).

Comprehensive vegetation surveys in the Eneabba area have been conducted by Lamont (1976), Hnatiuk and Hopkins (1981), Griffin et al. (1983), and Froend (1987). In a survey of about 10 km^2, approximately 20 km south of Eneabba, Lamont (1976) recorded 239 species. In a broader survey of about 20 km^2 overlapping most of Lamont’s study area and extending further north, Hnatiuk and Hopkins (1981) detected 429 species. They recorded only 180 species in common in both surveys highlighting the high floristic richness and beta-diversity of the area. However, both studies recorded similar dominant families (Proteaceae, Myrtaceae, Cyperaceae, Epacridaceae, Anthericaceae and Haemodoraceae) and genera (*Hakea*, *Stylidium*, *Drosera*, *Conostylis*, *Thysanotus*, *Banksia*, *Leucopogon*, *Verticordia*, *Daviesia*, *Dryandra* and *Grevillea*). Griffin et al. (1983) and Froend (1987) also recorded similar common families (plus Papilionaceae and Mimosaceae) and genera (plus *Acacia*, *Hibbertia*, *Melaleuca* and *Petrophile*). All these studies indicated a high number of unnamed (and presumably endemic) species in the Eneabba region (Lamont 1976, 9%; Hnatiuk and Hopkins 1981, 15%; Griffith et al. 1983, 9%; Froend 1987, 8%).

The above authors noted that floristic boundaries are unclear in the Eneabba area. It is likely that a combination of factors, such as soils/landforms, climate, and sometimes fire may influence species distribution patterns. Hnatiuk and Hopkins (1981) found that soil type may explain most of the variation in plant species distribution patterns. However, some of the variation remained unexplained – only 15% of studied species was restricted to one or two soil types indicating a high degree of floristic continuity between soil types. However, Hopkins and Griffin (1984) suspected that simplistic classification of soils into different generic types may have led to an underestimate of the importance of the soil-floristic relationship. Later, Griffin and Hopkins (1985) suggested that the principal influence on species distribution in the Eneabba sandplain is related to the depth of sand over clay having a consequential effect on water relations and root penetration, rather than other factors such as soil type. For instance, the deep sands of the dunes provide greater volume of rooting medium and water at depth, supporting mid-high *Banksia* shrubs and large emergents such as *Xylomelum angustifolium* and *Eucalyptus todtiana* (Bellairs 1991) capable of surviving dry summers by sending deep tap and sinker roots into deep water stores (Lamont and Bergl 1991). These larger growing plants are rarely seen on the shallow sands of the swale substrates (personal observation). Griffin et al. (1983) noted that another factor controlling plant distribution may be associated with climatic patterns such as potential evaporation rates/local rainfall patterns.

1.6.5 Fire regime

Bell et al. (1984) estimated that natural fire frequency in the Northern sandplains to be in the order of 25 to 50 years. In contrast, van der Moezel et al. (1987) recorded shorter fire frequencies of 8 to 15 years in the region possibly because their sites were closer to farmland, whereby burning practices may extend to natural bushland. Using aerial photos and SAT images, a more recent paper by Miller et al. (2007) showed a mean fire interval of 13 years over the last 40 years in the Eneabba region. No information is available on the historical fire season but it is assumed to be summer-autumn when lightning is most likely. Fires can cover thousands of hectares if not brought under control (He et al. 2004). Relatively frequent fires in the region indicate that the vegetation is most likely well adapted to fire.

CHAPTER 2.0

Comparison of post-mine rehabilitated and natural shrubland communities in southwestern Australia

(Journal of Restoration Ecology; in press)

2.1 Abstract

Following mineral sand-mining near Eneabba, southwestern Australia, restoration managers have the difficult task of restoring shrubland communities of exceptional plant species richness. Species diversity, composition, structure, and key functional attributes in four mined sites rehabilitated 8 (R8) to 24 (R24) years ago were compared with those of typical nearby natural areas classified on the basis of substrate type (Low and High sand Dunes, shallow sand Swales, sand over Laterite and sand over Limestone). The rehabilitated sites (except R8) had more species (~140) than natural sites (~100) in 40 × 40 m plots with 12–37% species in common with natural sites. Rehabilitated sites were more similar in composition to each other than they were to the natural sites, with two strong colonizers, the fire-killed *Acacia blakelyi* and the fire-tolerant *Melaleuca leuropoma*, universally present. Dendrograms and ordinations based on composition and cover showed that rehabilitated sites grouped with each other before they did with the Dune and Swale sites (physically closest), and last with the Laterite and Limestone sites. Plant densities for R16 and R24 were about half those of the High Dune and Limestone, and about a quarter those of the Swale and Laterite. Fire-resprouters were under-represented in the rehabilitated sites. Growth form distribution in rehabilitated sites was most similar to those of the Dunes, with some woody shrubs up to 2.5 m tall present. Total iron and soil hardness (penetrability) were the only soil factors consistently different (higher) in the rehabilitated sites.

Key words: restoration, shrubland, Eneabba, species richness, species diversity, composition, structure, function.

2.2 Introduction

Disturbance may be defined as an event that can change community structure and composition by altering the physical environment and/or resource availability (Drake et al. 1989). Mining is a form of exogenous disturbance, and mining of mineral sands in particular has produced substantial areas of disturbed vegetation in Australia

(Clark 1975; Brewer and Whelan 2003; Ross et al. 2004). It negatively affects plant communities since it involves complete mechanical removal of the vegetation and disruption of ecological processes (Hobbs and Hopkins 1990).

Restoration on previously mined land involves techniques such as pre-stripping (from areas ahead of a mining front) and returning propagule-containing topsoil, seeding, fertilizing, mulching, and infill planting of seedlings (Fox et al. 1996). Restoration to a standard equivalent to the pre-mining natural diversity, structure, and function is usually the desired criterion. However, the time and effort required to rehabilitate the structure and function of Australian ecosystems can be considerable (Collins et al. 1985), especially in areas of outstanding species diversity such as the Eneabba sandplain in southwestern Australia (Bellairs and Bell 1993).

Restoration success ideally requires determining if the ecosystem components are similar to, or are on a similar successional trajectory towards, the desired endpoint (Chambers et al. 1994). A common and realistic means of evaluation is through comparison with surrounding undisturbed reference areas that exhibit the desired ecosystem properties (Chambers et al. 1992; Aronson et al. 1995). The selected reference sites should occur in close geographical proximity to the rehabilitated sites since the composition, structure, and environmental circumstances, of vegetation in nearby areas are more likely to be similar to those of the rehabilitated sites prior to disturbance, than are more distant sites. However, distant reference sites may be sampled if their vegetation community is targeted for rehabilitation and/or if seed collections extended to these areas.

Iluka Resources Ltd (and their precedessors) has been conducting mineral sand mining on the Eneabba sandplain since the 1970s. Approximately 2500 hectares have been mined to date, and best-practice restoration is required in this biodiverse region (Lamont et al. 1984). Topsoil is double-stripped and returned in two layers with a 20 cm second cut applied first, followed by a 5 cm first cut (Jefferies et al. 1991). A seed mix is sourced from the surrounding natural vegetation usually from within the mine lease area and spread over restored topsoil. Shrubland vegetation is removed 30 cm above ground, mulched and spread over the restored topsoil as a further source of seeds (Bellairs and Bell 1993). Seedlings of selected species are grown in a nursery and infill planted into rehabilitated sites. There has been a lack of documentation (especially in the 1970s – late 1990s) on the origins of the topsoil, mulch, species in the seed mix, seedlings of species produced in the nursery, and the quantities in which they were applied to each rehabilitated block. However, better documentation and techniques that are more efficient have developed since Iluka Resources Ltd acquired the Eneabba operations in 1999.

Restoration efforts at Eneabba aim to satisfy completion criteria that focus on rehabilitating species richness, plant density, and foliage cover (Jefferies et al. 1991; EMRC 1996). However, restoration of composition, structure and function are also desirable goals. The aim of the work reported here was to compare the species diversity, composition, structure, and functional attributes of post-mine rehabilitated sites with that of surrounding vegetation types near Eneabba, southwestern Australia. On the basis of these results, recommendations are given to further refine the mining company's restoration efforts. Specifically, the objectives were to:

(1) Compare species diversity and composition between four selected rehabilitated and five nearby natural sites representing typical plant community-types;

(2) Compare structural characteristics (plant cover and density, and growth-form distribution) between rehabilitated and natural sites;

(3) Compare key functional attributes (soil nutrient levels and species regeneration modes) between rehabilitated and natural sites; and

(4) Provide recommendations for improving the restoration of species diversity and composition, structure and function towards those of natural analogues.

2.3 Methods

Vegetation sampling

Shrubland vegetation at four rehabilitated and five nearby natural sites was examined near the town of Eneabba, approximately 280 km NNW of Perth (29° 49' S Latitude, 115° 16' E Longitude). The four sites rehabilitated by Iluka Resources Ltd (or their predecessors) were chosen to represent a variety of post-mined rehabilitated ages (R8 (8 y), R15 (15), R16 (16) and R24 (24): Table 2.1). Five natural sites were chosen to represent the main substrate types near the mined areas in order to allow a comparison between the restoration sites and the variability expected in the surrounding natural areas (High dune = High, Low dune = Low, Swale, Laterite = Lat, Limestone = Lim: see Table 2.1 for site details). The Lim and Lat sites were located relatively far away from the minesites (~20 km away) but were included in this study as seed collections extended to these areas (Phil Scott 2007, Iluka Resources Ltd., personal communication) and preliminary observations showed that some site specific Lat/Lim species were present on rehabilitated soils. At all sites, a 40 × 40 m plot was established, and divided into 5 × 5 m subplots. Within each subplot, all individuals (except annuals) were identified to species where possible using a field herbarium based on material in the Western Australian Herbarium, South Perth, Florabase (Western Australian Herbarium 1998+), and field knowledge of B. Lamont, B. Miller, N. Enright, D. Herath, A. Tinker and C. van den Bergh. Species abundance was recorded at all sites, except at Low, R8 and R15, where species cover was assessed instead using the Braun-Blanquet (BB) scale (Braun-Blanquet 1932; Braun-Blanquet 1964) since the human resources were not available to undertake the individual plant approach at all sites. This scale is based on visual assessment of plant cover at coded percentage intervals: 1 = <1%; 2 = 1–5%; 3 = >5–25%; 4 = >25–50%; 5 = >50–75%; 6 = >75–100%. The BB scores were assessed for each species in each subplot, and then converted to their mid-point cover values for each subplot (e.g., 0.5 % for BB class 1, 15 % for BB class 3). The average of the 16 mid point-cover values (from 16 subplots) for each species gives an estimate of their % crown cover area within the whole plot. For all other sites, % crown cover area was calculated using equation 1 based on measured height and crown width (1: N to S and 2: E to W) for all plants and assuming their shape is ellipsoid*.

Equation 1:
% crown cover of spp = [{(average of width 1 × average of width 2) × no. of plants} × 0.7854*] / plot area] × 100

At Low, R8 and R15, up to 10 individuals per species were measured (selected randomly) for determining their mean heights, to allocate species into growth forms (described below).

Soil hardness and nutrients

An Australian standard sand penetrometer (AS 1289 6.3.3) with a 16 mm diameter flat-ended rod driven by a 9 kg mass dropping 600 mm was used to assess soil hardness. After one hammer blow, the distance the rod penetrated the ground was measured. 20 random measurements along the edges of each plot were recorded.

Soil samples (14 × 14 cm area, 0–5 cm depth) were collected from 90 random points within each plot to investigate nutrient levels. All litter was removed at each point. Individual soil samples were dried at 40ºC for 48 h, mixed, and then a 20 mL subsample was extracted and analysed by CSBP Wesfarmers, Perth for total nitrate, ammonium, phosphate, potassium, sulphur, organic carbon, reactive iron, conductivity, pH (in $CaCl_2$ solution) and pH (in H_2O). The same procedures were used for examining nutrient levels at R8, R15 and Low, except that samples from 30 random points were analyzed.

Data analyses

Three widely accepted indices were used as estimates of diversity:

(a) Shannon Weiner diversity (Shannon and Weaver 1949): $H' = -\sum[p_i \times \ln(p_i)]$ where p_i = the fraction of cover of a given species to the total cover of species in the community;

(b) Pielou's evenness index (Pielou 1966): $J' = H'/H_{max}$ where H_{max} = species richness; and

(c) Simpson's index of diversity (Simpson 1949): $D' = 1 - \sum[ni\,(ni - 1)/N(N - 1)]$ where ni = the cover of the ith species, and N = the total cover of all species.

Ordination and classification was used to compare species composition. Ordinations were performed on both presence-absence data (floristic composition), and semi-quantitative data (species % cover values) using SYN-TAX 2000 (Podani 2001). Principal Coordinate Analysis (PCoA) was selected as the ordination technique, a metric multidimensional scaling method that is flexible in its choice of dissimilarity index (Gower 1966). Classifications, using the group average linkage method, were run in SYN-TAX 2000 (Podani 2001) to identify the order of clustering of sites. Classification results were superimposed on the ordinations to offer a more complete picture of the relationships between sites. Each classification was truncated once 5 groups were formed, and the members of each grouping were identified on the ordination. Sorensens distance was selected as the dissimilarity measure for the composition dendrogram and ordination, and Bray Curtis dissimilarity for the composition plus cover dendrogram and ordination (Podani 2001).

All species were classified into one of six growth forms: tall shrubs = woody plants 1–2.5 m on average, small shrubs = woody plants <1 m, herbs = non-woody plant <1 m, graminoid = grass or grasslike plants (including reeds and sedges), parasite = plants depending on a host plant for survival. Regeneration mode (resprouter or non-sprouter) was determined by examining if a lignotuber was present, or extent of recovery after fire, or from existing data sources (Enright et al. 2007). Resprouters recover from fire vegetatively and/or from seeds; nonsprouters are killed by fire and regenerate from seeds only.

Soil attributes were analyzed using Principal Components Analysis (PCA, constrained, non-centred) in PcORD 5.0 (McCune and Mefford 1999). Vectors corresponding to specific nutrient variables were drawn on the ordination with their direction relative to their values at the sites and lengths proportionate to their overall correlation with other variables. One-way analysis of variance (ANOVA) was used (SPSS Inc. 2004) to determine if any variables were consistently and significantly different between rehabilitated and natural sites (excluding the Lim site which was an outlier in terms of its chemistry). For PCA analysis, the number of variables needs to be less than the number of sites, so soil attributes were screened to remove redundant variables (i.e., those highly inter-correlated). These included pH (in $CaCl2$ solution), sulphur, and ammonium.

Table 2.1: Location, selected attributes, and sampling methods employed at study sites in natural and restored shrublands near Eneabba, southwestern Australia. Lat = Laterite, High = High dune, Lim = Limestone, Low = Low dune. See Appendix for site photos

Sites	Location	Substrate type	Age (years)	Plot size (m2)	Subplot size (m2)	Data collected	Size measurements
High	S29°52'23.6" E115°15'02.4'	6–10 m of sand over silt and clay	19 since fire	40 × 40	5 × 5	Species*, no. of individuals, LH*	All, H* ×W1* ×W2*
Low	S29°42'41.4" E115°13'28.1"	2–3 m of sand over silt and clay	8 since fire	40 × 40	5 × 5	Species*, BB score*, LH*	H*
Swale	S29°51'54.8" E115°15'12.5"	10–50 cm of sand over silt and clay	24 since fire	40 × 40	5 × 5	Species*, no. of individuals, LH*	All, H* ×W1* ×W2*
Lat	S29°35'45.7" E115°15'22.4"	50 cm of sand over Lat and clay	15 since fire	40 × 40	5 × 5	Species*, no. of individuals, LH*	All, H* ×W1* ×W2*
Lim	S29°52'15.3" E115°05'27.8"	0-50 cm sand over Lim, some outcropping Lim	19 since fire	40 × 40	5 × 5	Species*, no. of individuals, LH*	All, H* ×W1* ×W2*
R8	S29°55'48.6" E115°17'15.6"	0–30 cm sand over silt and clay	8 since mined	40 × 40	5 × 5	Species*, BB score*, LH*	H*
R15	S29°53'44.6" E115°17'02.9"	0–10 cm sand over silt and clay	15 since mined	40 × 40	5 × 5	Species*, BB score*, LH*	H*

R16	S29°54'08.1" E115°16'56.4"	0–10 cm sand over silt and clay	16 since mined	40 × 40	5 × 5	Species*, no. of individuals, LH*	All, H* ×W1* ×W2*
R24	S29º55'58.9" E115º16'27.8"	0–20 cm sand over silt and clay	24 since mined	40 × 40	5 × 5	Species*, no. of individuals, LH*	All, H* ×W1* ×W2*

Species* = All species identified except annuals

BB score* = Braun-Blanquet cover score (see text) recorded

LH* = Life history characteristics (such as regeneration mode and seed storage) noted

H ×W1 ×W2* = Plant dimensions (height (H), crown width from north to south (W1), crown width from east to west (W2) recorded

H* = Average height of all spp. recorded (up to 10 random individuals measured per spp.)

2.4 Results

Species diversity and composition

In all, 348 species were recorded in the study (Appendix A) and all sites generally displayed high species richness, diversity and evenness (Table 2.2). Rehabilitated sites (except R8) had greater species richness (128–146 spp) than adjacent natural sites (78–113 spp). Shannon Weiner (H') and Simpson's diversity indices (D') both showed little difference between rehabilitated and natural sites, except that R8 had lower diversity than all other sites. Pielou's Evenness index (J') showed that the rehabilitated sites, especially R8 and R16, had lower evenness than natural sites.

The natural sites supported 266 species in 107 genera and 39 families. The most common families were Proteaceae (49 spp), Myrtaceae (45), Cyperaceae (19), Papilionaceae (20), and Epacridaceae (16). The rehabilitated sites supported 237 species in 93 genera and 31 families. Common families were Proteaceae (51 spp), Myrtaceae (39), Cyperaceae (17), Papilionaceae (16), and Epacridaceae (14). The most common genera in natural and rehabilitated sites are listed in Table 2.3, and show that *Stylidium* and *Leucopogon* were better represented in the natural areas and *Acacia* in the rehabilitated areas.

The five most dominant (based on % cover) species in each site are given in Table 2.4. In all rehabilitated sites, *Acacia blakelyi* and *Melaleuca leuropoma* were among the top three dominant species, whereas in the natural sites, *A.blakelyi* was infrequent or absent, and *M.leuropoma* was abundant only at Lim and Low. The identity of dominant species varied more in natural sites than that of rehabilitated sites. The most abundant species in the rehabilitated sites had higher cover values than the most abundant species in the natural sites. Most dominant species in rehabilitated sites were nonsprouters, while resprouters were more dominant in natural sites.

Species composition of rehabilitated sites (excluding R8) were most similar to the Low (32–37% spp in common), Swale (25–37%) and High (26–31%), and least similar to the Lat (16–24%) and Lim (12–16%). Within natural sites, the Lat (12–

28% similarity) and Lim (10–15%) were least similar to the other sites and also had the highest percentage of unique species (Table 2.2). Within the rehabilitated sites, R8 (24–36% similarity) was least similar to the other sites (39–49% with each other). Natural sites had a broader range of unique species (8–43%) than rehabilitated sites (8–13%). R8 had the lowest percentage of soil seed bank (SSB) species (52%) compared with 69–81% for all other sites.

Vegetation density and cover

The Swale and Lat sites had nearly four times the plant density of the rehabilitated sites (R16, R24), and the High and Lim sites were nearly twice as dense (Table 3.2). Total cover of all rehabilitated sites was around 105% and most similar to the High and Lim.

Ordination and classification

PCoA based on Sorensen's distance (Figure 2.1a) for species presence-absence data show Lat and Lim as outliers. The Dune and Swale sites group early as do R15, R16 and R24, while R8, Lat and Lim are unlinked at the 5 group level in the classification. At the 4 group level, the Dune and Swale sites group with R15, R16 and R24, with R8 joining next (3 groups), followed by Lat (2 groups). Although Lim appears relatively close to the Dune samples in the PCoA using Bray-Curtis similarity/cover data (Figure 2.1b), the classification keeps it separate at all linkage levels. All rehabilitated sites group early at 5 groups, as do the Dunes, and the rest remain unlinked. The Dunes link with the rehabilitated sites at 3 groups, with the Swale and Lat joining next at 2 groups.

Growth forms

At all sites, small shrubs and graminoids accounted for most species and greatest density (Figure 2.1a, 2b). Tall shrubs were lacking in the Swale and Lat sites. The R8 site lacked a herb layer and had more tall shrubs compared with other sites. Growth form distributions in the rehabilitated sites (apart from R8) were most similar to the Dune. Small shrubs accounted for most cover at all sites (Figure 2.1c) except R8 and R16 which had greater shrub cover.

Regeneration modes

Regeneration-mode composition was more variable in natural sites than rehabilitated sites (Table 2.2). The percentage of resprouting species in rehabilitated sites (71–74%) was within the range of the natural sites (67–88%) sites. All natural sites had more resprouter (67–98%) than nonsprouter individuals, especially in the Swale (84%) and Lat (98%), whereas rehabilitated sites R16 (39%) and R24 (49%) had fewer resprouter individuals. Cover of resprouters and nonsprouters varied greatly among sites but that of resprouters tended to be lower in rehabilitated (<69%) than most natural (>60%) sites.

Soil hardness and nutrients

PCA analysis of soil properties separated the rehabilitated sites from the natural sites (Figure 2.3). Only reactive iron and soil hardness was significantly different ($P < 0.05$, 1-way ANOVA) between treatments (excluding Lim site). Total reactive iron levels were nearly double in the rehabilitated sites and soil hardness was about twice as hard. Lim was an outlier due to its high pH and nutrient levels.

Table 2.2: Species diversity, plant density, total plant cover, and plant functional attributes in natural and restored shrubland near Eneabba, southwestern Australia.

	Natural					Rehabilitated			
Measure	High	Low	Swale	Lat	Lim	R8	R15	R16	R24
Spp. richness	113	101	109	93	78	83	128	146	144
% Unique spp.	18	8	15	26	43	10	12	8	13
H’	1.53	1.6	1.66	1.47	1.41	1.08	1.51	1.4	1.51
D’	0.96	0.96	0.98	0.95	0.95	0.84	0.94	0.93	0.95
J’	0.74	0.81	0.82	0.75	0.75	0.57	0.72	0.65	0.7
Density (m^2)	6.6	-	14.3	15.2	6.7	-	-	4.0	3.9
% Total cover	125	171	79	68	115	143	102	90	120
% Canopy seed bank spp.	26	25	28	24	19	48	31	30	29
% Soil seed bank spp.	74	75	72	76	81	52	69	70	71
% Resprouter spp	69	79	81	88	67	72	71	74	71
% Resprouter individuals	67	-	84	98	71	-	-	39	49
% Resprouter cover	68	72	87	97	61	49	59	37	68

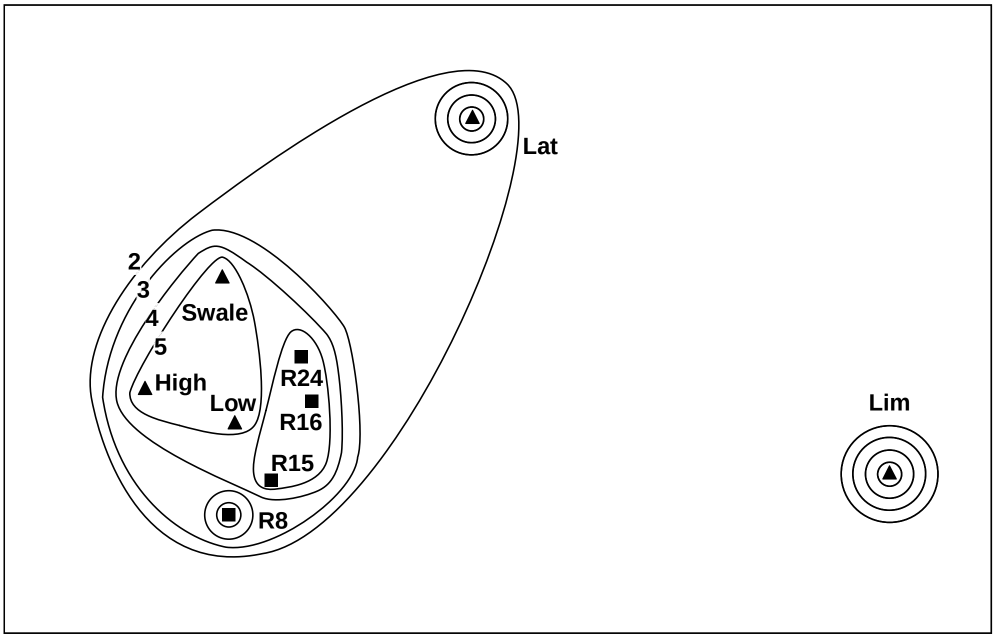
Table 2.3: Five most speciose genera in natural and restored shrubland near Eneabba, southwestern Australia.

Speciose genera (# spp in brackets)	
Natural sites	Rehabilitated sites
1. *Hakea* (11)	1. *Hakea* (14)
2. *Leucopogon* (10)	2. *Dryandra* (9)
3. *Dryandra* (10)	3. *Lepidosperma* (9)
4. *Stylidium* (10)	4. *Acacia* (8)
5. *Lepidosperma* (10)	5. *Conostylis* (7)
Acacia (4)	*Leucopogon* (6)
Conostylis (6)	*Stylidium* (4)

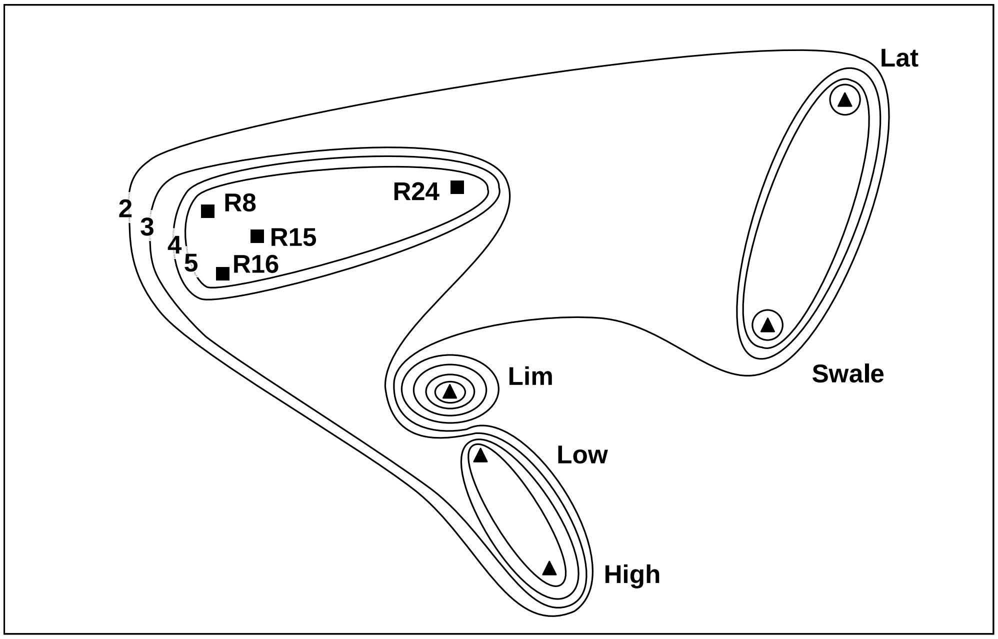
Table 2.4: The five most dominant (by percent crown cover) species (regeneration mode after species) at each natural and restored shrubland site near Eneabba, southwestern Australia. Where dominant at that site, value given in bold, r = resprouter, n = nonsprouter.

	Natural					Rehabilitated			
Species	High	Low	Swale	Lat	Lim	R8	R15	R16	R24
Acacia blakelyi n	0	0.3	0	0	0	**36.4**	**6.7**	**12.2**	**10.2**
Acacia pulchella n	0	0	0	0	0	0.3	1.3	0.2	**5.6**
Acacia spathulifolia n	0	0	0	0	**10.5**	0	0	0	0
Adenanthos cygnorum n	**9**	0	0	0	0	4.81	2.8	**11**	0
Allocasuarina microstachya r	0	0	0	**3.9**	0	0	0.1	0.1	0.4
Banksia attenuata r	**13.2**	**15.7**	0	0	0	1	0	0	1.4
Banksia hookeriana n	**15**	0	0	0	0	0	0	**3.9**	1.4
Banksia lanata n	0.2	0	0	0	0	**6.8**	0	0	0
Banksia leptophylla n	0	0	0	0	**8.8**	6.3	**16.5**	**14**	0
Beaufortia elegans n	4.1	28	**4.5**	0	0	0.44	1.3	1.38	2.77
Cassytha pubescens n	0	**8.2**	0	0	0	3.7	**8.5**	0.4	**7.8**
Desmocladus semiplanus r	1.7	**10.4**	0.7	0	4.9	0.15	0.1	0.12	0.28
Dryandra falcata r	0	0	0	0	0	**9.2**	0	0	0
Dryandra shuttleworthiana r	0.14	0	**3.3**	1.6	0	2	3.3	1	0.3

Ecdeiocolea monostachya r	0	0	**5.9**	**11.3**	0	0	0.3	0.2	**18.1**
Eremaea beaufortioides r	0	4	0.7	0.8	0	0.1	**2.8**	1.4	3.5
Georgeantha hexandra r	0	0	0	**5.7**	0	0	0	0	0
Hakea polyanthema n	0.4	**10.8**	0.6	0	0	0	0	0	0.2
Hibbertia hypericoides r	**10.1**	1.8	2.9	**3.5**	**14.7**	0.44	0.98	0.33	2.25
Labichea cassioides r	0	0	0	0	**8.7**	0	0.06	0	0
Lepidobolus preissianus r	0	0	**5.9**	0	0	0.1	0	0.1	0
Melaleuca leuropoma r	4.7	**15.5**	4.1	2.1	**9.4**	**30**	**24.4**	**8.9**	**12.5**
Melaleuca scabra r	0	0	0	**3.3**	0	0	0.1	0	0.1
Mesomelaena stygia r	0.4	0	**4.9**	0	0	0	0.25	0.01	0.21
Petrophile drummondii n	4.4	3.0	0.1	0	0	**7.8**	3.53	1.62	0.4
Xylomelum angustifolium r	**6**	0	0	0	0	0	0	0	0



a)



b)

Figure 2.1: PCoA on: a) presence–absence data using Sorensen's distance (horizontal axis = 20.8% of total variance (tv), vertical axis = 17.4% tv), and b) composition plus cover data per species using Bray–Curtis dissimilarity (horizontal axis = 26.3% tv, vertical axis = 19.4% tv). Lines represent dendrogram groupings (numbers indicate the # of groups formed).

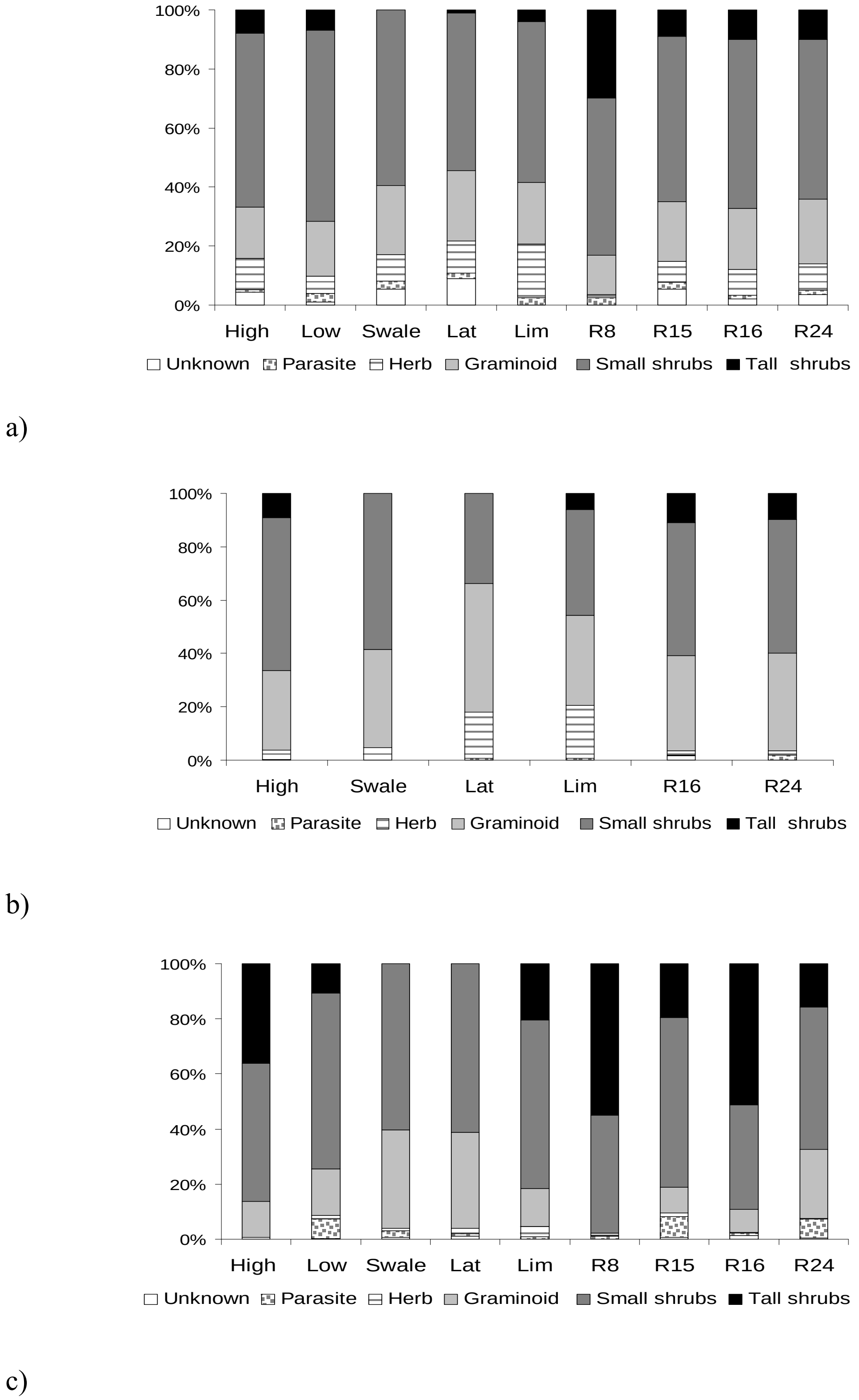


Figure 2.2: Growth form percentage: a) species composition; b) density; and c) crown cover in natural and restored shrubland sites near Eneabba, southwestern Australia.

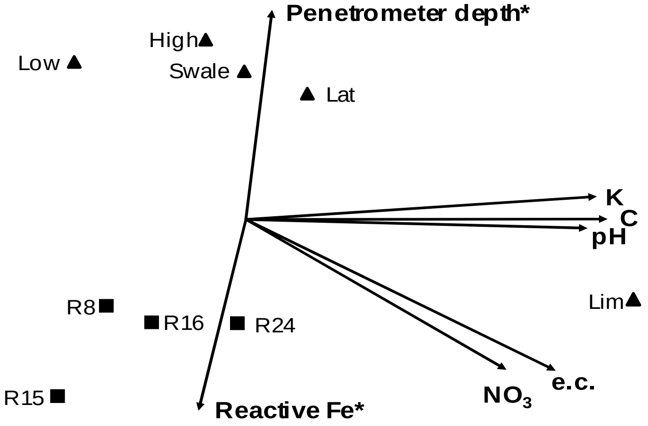


Figure 2.3: Principal components analysis (constrained, non centred) of natural shrubland and sand-mine restoration sites near Eneabba, southwestern Australia based on soil properties. Axis 1 = 57.3 % tv, axis 2 = 25.7 % tv, * $P < 0.05$ one way ANOVA, Fe = iron, NO_3 = nitrate, e.c. = electrical conductivity, K = potassium and C = organic carbon.

2.5 Discussion

The rehabilitated sites (except R8) had more species than natural sites. The high species richness reflects a concerted effort to satisfy the stipulation by the Government conservation authority that the pre-mining species richness should be restored (Jefferies et al. 1991; EMRC 1996). Species arose from seeding, mulching, topsoil replacement, and infill planting. Bellairs and Bell (1993) showed for a rehabilitated block at the same mine that topsoil (3%) and mulch (96%) contributed 99% of the total germinable perennial seeds. However, seeding and interplanting of nursery stock seedlings were also important in supplementing species (~46% of spp.) that were poorly or not represented in topsoil and mulch (recalcitrant species). Other studies have also highlighted the value of seed broadcasting in restoring south Australian Eucalypt forests (Roche et al. 1997), South African fynbos shrublands (Holmes 2001) and North American prairie grasslands (Wali 1999). Seeds are generally collected within the mine lease area (Pamela Grout 2007, Iluka Resources Ltd, personal communication) that covers all substrate-types included here, plus some winter-wet depressions not included in our study. It is evident from our results and the literature (Lamont et al. 1977; Hnatiuk and Hopkins 1981; Griffin et al. 1983) that some species not represented in the pre-mined vegetation (which was dune and swales) were also obtained from limestone and laterite areas, as well as winter-wet locations, possibly to ensure that the species richness target was achieved and uncertainty about which species would succeed.

The rehabilitated sites had lower evenness than natural sites mostly because of two dominant species, *Acacia blakelyi* and *Melaleuca leuropoma*. Following topsoil replacement, *Acacia blakelyi* was historically mixed with cover crop seeds to provide stabilization of tailings and to produce a quick scattered cover of native shrubs to tide over the period between cover crop decay and full native regeneration (Black 1979). However, due to its prolific reproductive capacity and ability to dominate disturbed sites (personal observation), it survives, reproduces and remains dominant in older rehabilitated stands. Similarly, the highly fecund *Melaleuca leuropoma*, which usually possesses an order of magnitude more viable seeds per unit area in all substrate types than other serotinous species (Enright et al. 2007), dominates rehabilitated sites after the mulching process.

In view of the above, why the youngest stand (R8) had less species and lower H’, D’ and J’ than other rehabilitated sites requires an explanation. R8 received topsoil that had been stockpiled for 10 years, creating an ideal environment for *Acacia blakelyi* to flourish and contribute to the soil seedbank before spreading (Bob Muir 2007, Iluka Resources Ltd, personal communication). As a result, *Acacia blakelyi* covered 36 % of the site post-restoration (more than other rehabilitated sites) and may have competitively displaced other species (Lamont et al. 1989). In addition, it is the only site on a slope and may have meant it dried out faster than other sites and led to differential survival, as soil moisture is critical for establishment here (Enright and Lamont 1992a). A lower ratio of soil to canopy-stored species was also observed at R8 compared with other sites likely due to its topsoil being stockpiled for such a long period which may have affected the seed viability of soil-stored species. Several studies have shown areas that receive fresh topsoil yield more species and/or individuals than areas receiving stockpiled topsoil (Ward et al. 1996), and the longer the stockpile is stored, the lower the species diversity (Bellairs and Bell 1993).

Vegetation densities were lower in rehabilitated sites possibly due to seeds and/or seedlings being added at lower (than required) densities, or the seeds and/or seedlings experiencing greater mortality levels associated with the adverse substrate-type (lack of sand over poorly penetrable silt-sand). After simulating mining conditions, Holmes (2001) reported lower densities and survival of fynbos shrublands (Cape floristic kingdom, South Africa) in subsoil versus topsoiled plots. Griffins and Hopkins (1985) suggested that the principal influence on species distribution on the Eneabba sandplain was related to the depth of sand over clay having a consequential effect on water relations and root penetration. Shallow or no sand over clay may prevent taproots from accessing groundwater (Enright and Lamont 1992a), intensifying the effect of drought, and over winter, occasionally creating waterlogged soils.

Rehablitated sites were most similar to each other (averaging 45%; except R8) due to similar selections of species being rehabilitated at most sites. Among the natural sites, the Swale and the Dunes (High and Low) were most similar to the rehabilitated sites (about 30% similarity) possibly due to their close proximity as a source of seeds. The distinctly different substrate type and distance away of the Lim, and to a lesser extent the Lat site, contributed to their reduced similarity with other sites. This also reflects restoration efforts primarily aimed in restoring the original swale and dunal communities of the local sandplain. However, some species that are usually restricted to limestone (e.g., *Labichea cassioides*) or laterite (e.g., *Hakea stenocarpa*, *Conostylis androstemma*) were also present in some rehabilitated sites indicating that collections may have occurred over broader areas than the adjacent swales and dunes. Rehabilitated sites were also more similar to each other in the composition plus cover ordination probably again due to the outcome of similar restoration efforts. Of all natural sites, the Dunes were most similar to the rehabilitated sites, both having some species with taller and broader growth forms (e.g., *Adenanthos cygnorum*, *Banksia attenuata*).

The rehabilitated sites had growth form distributions most similar to those of the Dunes with some tall shrubs. The reduced competition associated with low plant densities may have allowed many species to grow large in rehabilitated sites. The addition of unknown quantities of fertilizers into rehabilitated sites may have also contributed to larger plant size (Phil Scott 2007, Iluka Resources Ltd, personal communication). Percentage growth form cover varied in the rehabilitated sites due to variable cover values of a few species in the shrub layers, such as *Acacia blakelyi*, *Banksia leptophylla* and *Adenanthos cygnorom*.

All sites had more resprouter (~70% average) than nonsprouter species. It is important to restore similar regeneration mode proportions to maximise the vegetation's resilience to future disturbances such as fire, drought, and herbivory (Bellairs and Bell 1993). However, the relative richness, density and cover of resprouters were lower in rehabilitated sites than nearby natural sites. The most common species were also mostly nonsprouters in rehabilitated sites and resprouters in natural sites. Similarly, Grant and Loneragan (1999) recorded lower densities of resprouters in rehabilitated versus natural sites in eucalypt forests of SW Australia. This poorer representation of resprouters in post-mined lands reflects the difficulty in restoring resprouters. They generally produce fewer seeds than nonsprouter species (Lamont and Wiens 2003) so their seed densities are lower in the added seed mix, mulch and topsoil.

Only a few resprouter species were well represented in rehabilitated sites (e.g., *Melaleuca leuropoma*, *Jacksonia floribunda*) and these produce moderate numbers of seeds in natural areas (Enright et al. 2007). In rehabilitated sites, they show even faster growth and higher seed set and storage than do the same species in natural areas (personal observation). The longer-term consequences of these growth responses (i.e., type of recruitment after fire returns to restoration sites) is worthy of investigation.

Although the ordination revealed a strong separation of rehabilitated sites from natural sites, the only variables that were significant in separating them were total iron and soil hardness. This results from mining the finer iron-bearing particles of the B horizon and bringing them to the surface during the soil replacement process. This has no doubt contributed to some differences in species composition and structure between rehabilitated and natural areas (Enright and Lamont 1992a). Subsequent erosion of topsoil appeared partly responsible for the lack of sand over the fines in rehabilitated sites, thus increasing soil hardness at just 5 cm depth.

Conclusions and recommendations

Although the flora of rehabilitated blocks was more similar to each other than to any nearby natural plant communities, there were some compositional, structural and functional affinities with the nearest Swale and Dune substrate sites. Growth form structure was most similar to High. Most soil nutrients were low and similar to those of the Swale and Dunes while extractable iron and soil hardness were much higher in rehabilitated sites. Species composition was most similar to that of Swale and Dunes, albeit only in the order of 30% species in common. Other studies in the region have also shown low similarity among samples between shrubland communities, even for similar substrate types (Griffin et al. 1983; Hnatiuk and Hopkins 1981; Lamont 1976), highlighting the floristic complexity of the region. Similar restoration difficulties appear in species-rich fynbos shrublands in South Africa (Holmes and Richardson 1999; Holmes 2001).

To further improve similarity to swale or dune shrublands, recent protocols (since Iluka Resources Ltd acquired the Eneabba operations in 1999) have formulated swale, dune, laterite and wetland seed species mixes that are not combined during the restoration process. This should produce species richness levels closer to those of the natural community types. Topsoil needs to be stockpiled for as short a period as possible to minimize the loss of viability of soil-stored species. The practice of growing *Acacia blakelyi* to stabilize tailings has ceased but it is still a major component of newly rehabilitated pits and remains a problem. It may be possible to manually clip or chainsaw the main stems of weedy species (e.g., *Acacia blakelyi*) at early stages of restoration before the onset of seed production (2 to 3 years, personal observation). However, sufficient native vegetation cover is necessary to buffer against topsoil erosion.

To increase restored vegetation densities, seeds and seedlings may need to be added at higher densities (including subsequent years post rehabilitation) and/or greater efforts need to be made to mimic the deeper sands of the natural substrates (Enright and Lamont 1992a). Topsoil should always be stripped in natural areas ahead of mining fronts, and spread over the restored subsoil. Subsoil should always be ripped to reduce compaction and improve root and water penetration (Holmes 2001).

The low density of resprouters is of particular concern as fire is inevitable in the long term, and may kill individuals with inadequate rootstock development (personal observation). The paucity of seeds of some resprouter species makes their adequate collection via soil or mulch difficult, so that targeting seed collection and infill planting is required. Given the fast growth and high seed set of some resprouter species within rehabilitated blocks (personal observation), it may also be possible to harvest seeds from plants of these species within the restoration area in order to increase their abundance in seed mixes used elsewhere on site. It would have assisted interpretation of our data if initial site preparation treatments and composition of seeding mixtures and nursery stock were better documented, so that good data storage and management procedures are also important to the long-term success of restoration programs.

Implications for practice

- Tailings should be returned in such a way as to mimic the range of substrates (in this case dune and swale) present pre-mining and to create a variety of habitat-types.
- Collections/mixes of topsoil, mulch and seeds for minesite restoration should be restricted to the pre-mined substrate type, to achieve similar composition and abundance to that present prior to mining.
- Highly fecund species that dominate disturbed lands (e.g., *Acacia blakelyi*, *Melaleuca leuropoma*) should be identified and managed to prevent competitive exclusion of other native species. This involves minimizing their presence in seed applications and avoiding mulching in thickets containing these species.
- Increased densities, especially of the poorly represented resprouters, may be achieved by supplementary sowing seeds and/or interplanting nursery stock in subsequent years until resprouters are adequately represented. The seeds could be

harvested from plants within rehabilitated areas, which are more fecund than those in surrounding natural areas.

- Completion criteria in restoration projects should consider including a degree of compositional and regeneration mode similarity to that of the desired 'local community', in addition to species richness and vegetation density goals.

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CHAPTER 3.0

Impact of fire of plant species persistence in post-mine rehabilitated and natural shrubland communities in southwestern Australia

(Submitted manuscript)

3.1 Abstract

In many parts of the world, it is inevitable that fire will return as a natural disturbance factor to vegetation restored on anthropogenically-disturbed lands. Therefore, assessment of the ultimate success of restoration programs should include the ways in which these ecosystems respond to such natural disturbances. I compared the response of vegetation to experimental fires on post-mine restored and nearby natural shrubland communities in a Mediterranean-climate region of Australia. Pre and post-fire perennial species composition was assessed in 40 × 40 m plots at three sites rehabilitated 8–24 years ago after sand-mining, and at five natural communities from 8–24 years since last fire. Quadrats were monitored for seedling survival over the first summer after fire.

Species richness fell by 22–41% after fire in rehabilitated sites, but increased by 4–29% in natural sites. Species present before fire were reduced by 40–56% in rehabilitated sites and 4–12% in natural sites. Only 42–66% of resprouting species recovered in rehabilitated sites, whereas 96–100% recovered in natural sites. Nonsprouting species recruitment was also lower in rehabilitated (18–57%) than natural (67–85%) sites. Seedling mortality over the first summer after fire was higher in rehabilitated sites (59–86% death of individuals) than in natural sites (14-60%). PCoA ordination showed that fire altered the floristic composition of rehabilitated sites much more than of natural sites, and their vegetation diverged further from the targeted properties of natural communities. Our study highlights the importance of including the ability of post-anthropogenically-altered lands to recover from natural disturbances in determining the success of restoration programs.

Keywords: disturbance, ecosystem function, mineral sand-mining, nonsprouter, restoration, resprouter, shrubland, species richness

3.2 Introduction

Disturbance can cause major changes in plant communities depending on the nature, intensity, extent, frequency, seasonality of disturbance events, and the resilience properties of the component species (Levin and Paine 1974; Connell 1978; Huston 1994; Grime 2001; Ross et al. 2004). Fire is a common form of endogenous disturbance in

Mediterranean-climate regions of Australia, South Africa, Europe and North America, and, at intermediate levels of frequency and intensity, is usually credited with helping maintain plant species diversity (Gill 1981; Bell et al. 1984; Fox and Fox 1986a), playing key roles in natural resource management (Keeley et al. 2003; Huang et al. 2007; Yang et al. 2008). Strip mining is an intense anthropogenic (exogenous) disturbance involving complete removal of vegetation and disruption of ecological processes (Hobbs and Hopkins 1990), impacting substantial areas of forest and shrubland in Australia (Bellairs and Bell 1993; Grant and Loneragan 1999; Brewer and Whelan 2003; Ross et al. 2004) and elsewhere (Levitt 1997; van Aarde et al. 1998; Booth et al. 1999; Clewell 1999; Cooper and MacDonald 2000). Return of indigenous species to these mined lands is now commonplace with restoration targets including specific values for ecosystem composition, richness and structure relative to those present in natural analogues (Jefferies et al. 1991; EPA 2006; Gardner and Bell 2007; Grant and Loneragan 2003). As biomass accumulates following restoration of plant communities, the probability of disturbance by fire increases. The question then arises as to whether vegetation restored following mining will move more or less strongly towards the natural analogue state following the re-establishment of such 'natural' disturbance regimes (Ross et al. 2004).

Few studies have investigated the functional property responses of rehabilitated mine-sites to disturbance by fire, and, in Australia, these have been restricted to eucalypt forests and woodlands (Smith et al. 2000; Grant and Loneragan 2001; Comino et al. 2004; Ross et al. 2004; Grant et al. 2007). They indicate that burning restored vegetation can positively influence ecosystem properties, driving them towards those characteristic of natural analogues (Hooper 1985; Grant and Loneragan 1999; 2001; Ross et al. 2004). Fire begins to have such beneficial effects for restored eucalypt forests >8 years old in southwestern Australia (Smith et al. 2000; Smith 2001) and >10 years in New South Wales (Ross et al. 2004), with responses likely to be related to the life histories of the constituent species and their reproductive strategies (Noble and Slatyer 1980; Fox et al. 1996; Benwell 1998; Ross et al. 2004). For example, nonsprouters recover through the fire-cued germination of seeds accumulated in a soil or canopy seedbank, while resprouters recover vegetatively after fire from buds protected beneath bark, or from underground perennating organs such as lignotubers, rhizomes or bulbs (Bell et al. 1984). As a result, population persistence of most perennial plant species is possible if sufficient time is allowed for nonsprouting species to generate a substantial seedbank (Enright et al.

1996) and juvenile resprouting species to develop sufficient fire tolerance (Lamont and van Leeuwen 1988).

A fundamental measure of the success of restoration programs should be the extent to which the response to natural disturbances such as fire mimics that of natural analogue sites (EPA 1995). Our study sought to compare the short-term impact of introducing fire in highly biodiverse Mediterranean-climate type shrublands of three post-mining rehabilitated (8 to 24 years since restoration) and five natural (8 to 24 years since fire) shrubland sites near Eneabba, 275 km north of Perth, Western Australia. The following questions were addressed: Are the mean and range of post-fire responses of species richness, regeneration mode distribution (resprouter/nonsprouter), and first year seedling survival the same for rehabilitated and natural sites? Are the post-fire responses similar to those reported for restored eucalypt forests elsewhere in Australia? What are the implications for management of the restored shrublands?

3.3 Methods

Pre-and post-fire vegetation and substrate properties

Mature vegetation at three sites rehabilitated 8, 15 and 24 years ago by Iluka Resources Ltd (and their predecessors) and five surrounding natural (8, 14, 15, 19, and 24 years since last fire) sites representing typical plant community/substrate types were examined near the town of Eneabba, 275 km north of Perth, Western Australia (29° 49' S, 115° 16' E). Most species found in the natural sites also occurred in the rehabilitated sites (Herath et al. in press).

Presence/absence of pre-fire perennial plant species was surveyed within a single 40 × 40 m plot at each site in autumn 2005, and individuals were identified to species where possible using a field herbarium based on material in the Western Australian Herbarium, in the Florabase on-line herbarium (Western Australian Herbarium 1998+), our own field knowledge and that of A. Tinker, C. van den Bergh. The plots were all burned in late autumn in different years (2005-7) with assistance from Iluka Resources, the Bushfires Board of Western Australia, and Department of Environment and Conservation fire management staff. All burns were of high intensity with no post-fire evidence of litter and foliage in the crowns of plants at all sites, except at R8 where 3 subplots did not burn well (i.e., not surveyed) and therefore 3 new subplots were staked at R8 (adjacent to the 40 × 40 m plot). Following these fires, 90 random locations were selected within each plot, and at each location, two 25 × 25 cm quadrats were systematically established 20 cm south, and 180 cm east of the point, giving n = 180 quadrats per site with a total sample area of 11.25 m^2. Within each quadrat, seedling emergence (species and frequency of occurrence, annuals excluded) was recorded in the first spring post-fire (~6 months) and seedling survival (over first summer) was recorded at the same random quadrats in the first autumn (~12 months) following fire. In addition, at 2 years post-fire, a presence/absence species composition survey was conducted for each of the 40 × 40 m plots (when the likelihood of survivorship to maturity would be greater than at 1 year). Individuals of resprouter species were recorded as live or dead by examining evidence for post-fire regrowth of foliage. The mean post-fire responses of rehabilitated vs. natural sites were compared using two-tailed t-tests (SPSS Inc. 2004), after testing for normality (if not normal – data transformed – square root/arcsine) and equality of variances between treatments (if not equal – t-test assuming unequal variances).

Soil total nitrate, ammonium, phosphate, potassium, sulphur, organic carbon, reactive iron, conductivity, pH (in $CaCl_2$ solution), pH (in H_2O) and soil penetrability (using a penetrometer) were measured for surface soil samples (0-5 cm) using the techniques described in Herath et al. (in press).

Floristic composition

Patterns of species composition among pre- and post-fire samples based on species presence-absence were investigated using Principal Coordinate Analysis (PCoA). PCoA is a metric multidimensional scaling (ordination) method that employs a dissimilarity matrix eigenvalue analysis (Gower 1966). It was run with SYN-TAX 2000 software (Podani 2001) using Sorensen's distance metric.

3.4 Results

General attributes

Of all soil factors measured pre-fire, only reactive iron levels, soil penetrability and depth of sand to hardpan were significantly different between rehabilitated and natural sites (Table 3.1). The soil surface was more than twice as hard (lower penetrability) in rehabilitated sites and with much shallower depth of sand over the hardpan. Reactive iron levels were more than double that of natural sites. Other ecological attributes, such as plant density, total plant cover, time since last disturbance and total rainfall 12 months post-fire, were matched between rehabilitated and natural sites.

Post-fire species persistence

237 species in 31 families were recorded in rehabilitated sites pre-fire and 266 species belonging to 39 families were recorded in natural sites (Herath et al. in press). After fire, rehabilitated sites supported 158 species in 29 families, and natural sites supported 310 species in 38 families (Appendix B). No perennial weed species were observed. The most species-rich genera in the natural sites before burning – *Leucopogon* (Ericaceae) and *Dryandra* (Proteaceae) – were superseded by *Conostylis* (Haemodoraceae) and *Calytrix* (Myrtaceae) after fire (Table 3.2). In the rehabilitated sites, *Verticordia* (Myrtaceae), *Leucopogon* and *Petrophile* (Proteaceae) were superseded by *Hibbertia* (Dilleniaceae) and *Melaleuca* (Myrtaceae) after fire. Species richness of the most strongly represented genera increased after fire in natural sites but decreased after fire in rehabilitated sites.

Although pre-fire mean species richness was not significantly different between rehabilitated and natural sites, post-fire mean (and their range) species richness in rehabilitated sites was much lower with two-thirds that in natural sites (Table 3.3). All rehabilitated sites showed a decrease in plant species richness after fire whereas natural sites showed an increase in species richness following fire. The post-fire appearance of new species was generally similar in natural and rehabilitated sites: 14–28% of post-fire species were not observed pre-fire.

Table 3.1. General attributes in post-mine restored and natural shrubland sites near Eneabba. Significantly different means ($P < 0.05$) are given in bold, [1] = P-value after square root transformation, [2] = t-test assuming unequal variances.

Attributes	Rehabilitated		Natural		P-value
	Mean	Min-Max	Mean	Min-Max	(2-tailed t-test)
Depth of sand to hardpan (cm)	10	0-20	236	10-800	**0.0314**[1]
pH	6.0	6.0-6.1	6.2	6.1-6.7	0.1644
N (mg/kg)	2.1	1.4-3.7	2.0	1.1-5.0	0.8610[1]
P (mg/kg)	2.4	2.0-2.8	3.3	2.0-4.0	0.0912
NH4 (mg/kg)	3.6	2.5-5.3	2.8	2.4-3.4	0.2244
K (mg/kg)	29	24-34	39	22-56	0.1956
Fe (mg/kg)	228	183-316	97	53-193	**0.0114**
S (mg/kg)	3.4	2.9-4.4	4.3	2.0-6.5	0.3393
Organic C (mg/kg)	0.6	0.5-0.8	0.8	0.5-1.1	0.2268
Soil penetrability (cm)	16	15-18	33	26-39	**0.0034**
Time since last disturbance (years)	16	8-24	17	8-24	0.9539
12 month rainfall following fire (mm) (BoM 2008)	393	353-472	433	353-472	0.4315[2]
Plant density (per m^2)	3.95	3.9-4.0	10.7	6.6-15.2	0.0894[1]
Total plant cover (%)	110	90-129	111	68-171	0.9284

A greater fraction of resprouter species failed to recover following experimental fire – as seedlings or resprouts – in the rehabilitated sites relative to the natural ones (Table 3.3). Only 43% of the resprouter species present in the rehabilitated sites before fire survived as resprouting individuals after fire. A small number of resprouter species produced seedlings so that 54% of pre-fire resprouter species returned after fire in the rehabilitated sites (Table 3.3). In contrast, just 2–4% of resprouter species failed to resprout in the natural sites, while additional seedling recruitment meant that almost all pre-fire resprouter species returned following fire. Seedling recruitment was observed in most pre-fire non-sprouting species in natural sites, significantly more than was observed in rehabilitated sites.

Densities of post-fire seedlings at both site types were greatest in the first spring following fire, and then decreased (in the first autumn) due to mortality over the first summer (Table 3.3). The mean (and range) of seedling survival over the first summer in rehabilitated sites was less than half that in natural sites. This difference in mortality contributed to a disproportionate loss of species surviving as seedlings. That is, while 83% of seedling species were still present in the natural sites after 12 months post-fire, only half of post-fire seedling species survived to this stage in the rehabilitated sites.

Table 3.2. The five most speciose genera in post-mine restored and natural shrubland sites, pre and post-fire.

five most speciose genera (# spp in brackets)			
Rehabilitated sites pre-fire	Rehabilitated sites post-fire	Natural sites pre-fire	Natural sites post-fire
Hakea (13)	*Hakea* (8)	*Hakea* (11)	*Verticordia* (13)
Leucopogon (12)	*Lepidosperma* (6)	*Leucopogon* (10)	*Stylidium* (13)
Lepidosperma (8)	*Dryandra* (6)	*Dryandra* (10)	*Conostylis* (12)
Dryandra (9)	*Hibbertia* (5)	*Stylidium* (10)	*Hakea* (11)
Acacia, *Banksia*, *Conostylis*, *Petrophile*, *Verticordia* (6)	*Conostylis*, *Banksia*, *Melaleuca* (5)	*Verticordia* (9)	*Calytrix* (10)

Table 3.3: Overall species persistence in post-mine rehabilitated and natural shrubland sites in SW Australia. RS = resprouter, NS = nonsprouter, * = unknown species/regeneration modes excluded, attributes with significantly different means ($P < 0.05$) are given in bold, [1] = P-value after square-root transformation, [2] = t-test assuming unequal variances, [3] P-value after arcsine transformation (fractional data).

Measure	Rehabilitated sites Mean (range) Pre-fire	 Post-fire	Natural sites Mean (range) Pre-fire	 Post-fire	Difference (2-tailed *t*-test *P*-value) Pre-fire	 Post-fire
Species richness	118 (83-144)	80 (49-112)	99 (78-113)	116 (101-129)	0.2602	**0.0404**[1]
% new spp after fire		25 (23-27)		21 (14-28)		0.2802[3]
% RS species*	72 (71-74)	71 (63-76)	77 (67-88)	77 (71-88)	0.3980[3]	0.2123[3]
% spp persisting (seedlings+resprouts)		50 (44-60)		91 (88-96)		**0.0001**[3]
% RS spp persisting (seedlings+resprouts) after fire*		54 (42-66)		98 (96-100)		**0.0002**[3]
% RS spp resprouting after fire*		43 (31-50)		97 (96-98)		**0.0000**[3]
% RS spp resprouting and producing seedlings*		13 (7-19)		23 (22-26)		**0.0100**[3]
% of NS spp recruiting after fire*		42 (18-57)		73 (67-85)		**0.0174**[3]
Seedling densities in first spring after fire (per m^2)		10 (6-17)		13 (5-26)		0.6654[1]
Seedling densities in first autumn after fire (per m^2)		3 (1-7)		7 (6-16)		0.2000[2]
% seedlings surviving first summer after fire		24 (14-41)		58 (40-86)		**0.0481**[3]
% spp surviving first summer as seedlings after fire		51 (45-58)		83 (65-97)		**0.0145**[3]

Floristic composition

The PCoA ordination (Figure 3.1) shows that fire did not change the relationships among sites as all sites post-fire remained more similar to their pre-fire state than to the composition of other sites. Fire did lead to a change in community composition that was consistent in direction across all sites, but was more pronounced in the rehabilitated sites, leading to a greater degree of differentiation among natural and rehabilitated sites.

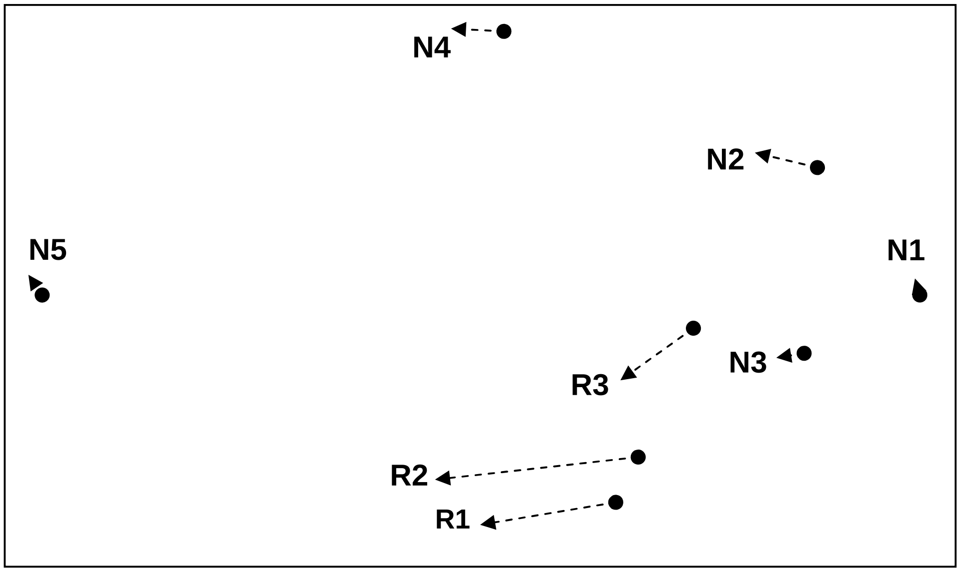

Figure 3.1: PCoA of pre- and post-fire species composition based on presence-absence using Sorensen's distance. Movement of site positions are shown by dashed lines indicating the trajectory from pre-fire (start: circle) to post-fire (end: arrowhead) composition. N1 to N5 site locations are in order of increasing distance away from the rehabilitated sites. R1 to R3 are in order of time since restoration. Horizontal axis = 19% of total variance, vertical axis = 15% of total variance

3.5 Discussion

Post-fire species persistence

Burning of shrubland vegetation restored on previously mined lands had strong negative effects on perennial plant species diversity. All natural sites showed an increase in native species richness (of 4–29%) after fire whereas rehabilitated sites showed a substantial decline in richness (of 22–41%) after fire. Most community studies in natural areas of Mediteranean-type shrublands have reported an increase in native species richness immediately after fire (e.g., in Australia: Bell et al. 1984; Fox and Fox 1986b; Fensham 1990, North America: Keeley and Keeley 1987; Parker 1987; Tyler 1996, South Africa: Kruger and Bigalke 1984; Le Maitre and Midgley 1992) as do those in post-mine vegetation elsewhere (Ward et al. 1990; Grant and Loneragan 1999; Smith et al. 2000; Ross et al. 2004). This is because fire plays an important role in providing opportunities for recruitment of annuals (not analysed in this study, but see Grant and Loneragan 1999; Smith et al. 2000; Brooks 2002; Ross et al. 2004) and perennial nonsprouting species (Smith et al. 2000; Grant and Loneragan 1999; Ross et al. 2004; Keeley et al. 2005), that may have completed their life cycle before fire, to re-establish through fire-stimulated germination of seeds. While the percentage of new species was similar (around 20%) in the natural and rehabilitated areas in our study, it was the failure of extant species to re-establish in the rehabilitated sites that accounted for most of the decline in total species there.

Much of the reduction in species richness can be attributed to loss of resprouting species. Several reasons could account for the lower recovery of resprouters in rehabilitated sites. First, strip mining involves complete removal of the former vegetation so that resprouters must be re-introduced as a new population cohort in the same way as nonsprouter species (Bellairs and Bell 1993). As a result, populations of resprouters in rehabilitated sites were younger (a single cohort of no more than 24 years) than those in natural sites that may have passed through many fire cycles – recurring with a mean interval of 13 years in the study area over the last 40 years (Miller et al. 2007). These restored resprouters may be individually more fire-sensitive as they have had insufficient time for many individuals to develop a large enough lignotuber to reach fire-tolerance (Bradstock and Myerscough 1988;

Lamont and van Leeuwen 1988; Auld 1990). Second, growing conditions on the reconstituted mine site substrates (with less penetrable soils) may not be conducive to lignotuber development. Plants of many resprouter species have much larger crowns and higher fecundity on the rehabilitated sites than in natural sites of similar age post-mining and post-fire (Herath et. al unpubl.), and investment in above-ground growth and fecundity may have been at the expense of fire-tolerant features such as the lignotuber. Third, recruitment from seeds in resprouter species is low after fire anyway since they generally produce fewer viable seeds than nonsprouters (Lamont and Wiens 2003).

Despite substantial seed production in some restored resprouter species (Herath et al. unpub.), seedling recruitment only raised the number of resprouting species persisting after fire from a mean of 43 to 54% compared with almost 100% in the natural areas. The exact cause of lower resprouter recovery on rehabilitated sites here is worthy of further investigation since the results differ from fire impact studies in restored eucalypt forests in higher rainfall regions of southwestern Australia (Grant et al. 1997; Grant and Loneragan 1999, 2001; Smith et al. 2000) and New South Wales (Ross et al. 2004). Some of these studies have also indicated that burning rehabilitated areas (at intermediate age) increases the abundance of resprouter species with established plants resprouting and seedlings observed in close proximity to the parents, whereas little seedling recruitment of resprouters was observed in natural sites after fire (Smith 2001). The discrepancy in post-fire results between our study and other fire impact studies in mined areas of Australia could be related to differing vegetation resilience properties to fires, restoration methods (e.g., depth of topsoil and subsoil profile), time to reproductive maturity of lignotubers and seedbanks, and/or rainfall patterns, with greater impact of summer drought on resprouter recovery and seedling recruitment after fire here.

Many more nonsprouting species failed to recruit after fire in rehabilitated sites (43–82% lost) than natural sites (15–33%). This seedling recruitment failure could be due to the low penetrability of soils on rehabilitated sites (Enright and Lamont 1992a). Our results indicate that rehabilitated site soils were twice as ‘hard’ as natural sites on average. The compact soils of rehabilitated areas may result from mining the finer iron-bearing particles of the B horizon and bringing them to the surface during the soil replacement process where they subsequently bake hard over summer and become water-repellent. This prevents roots from penetrating deeply into the soil in the critical first growing season, making young plants especially vulnerable to the summer drought (Enright and Lamont 1992a). Conversely, those individuals that do manage to penetrate the hard sub-soil benefit from access to its greater soil moisture storage and thereby, increasing their later year survivorship and growth.

Floristic composition

The PCoA ordination showed that fire shifted the species composition of rehabilitated sites relatively further away from that of natural sites – a result of the loss of many common species (particularly resprouters), the appearance of new species that may previously have existed on site but completed their life cycle prior to the initial assessment (e.g., *Anigozanthus humilis*, *Thysanotus patersonii*), and the emergence or immigration of species as seeds but not yet present as plants until fire triggered their germination. The natural sites were more stable in composition after fire due to the better persistence of both resprouting and nonsprouting species (Table 3.3).

Conclusions

While other studies have reported beneficial effects on species diversity after burning ≥8 year old post-mine restored eucalypt forests (Smith 2001; Grant and Loneragan 1999; Ross et al. 2004), our study in restored shrublands under lower rainfall conditions recorded negative impacts after burning 8 to 24 year old stands. This range encapsulates the average, and much of the range of variability, of modern fire return intervals observed in nearby natural vegetation (Miller et al. 2007). Species richness declined after fire at all rehabilitated sites with a major loss of resprouting species (which failed to resprout), and nonsprouting species (which failed to recruit), despite the accession of many (nonsprouting) perennial species not recorded previously. Seedling mortality over summer was also greater in the rehabilitated sites, highlighting the need for continuous monitoring following fires as well as assessing their immediate impact. Without further human intervention, it appears that these restored communities will take different successional trajectories from that of their natural analogues, at least in the short to medium term.

Few studies have attempted to examine the functional properties of restored communities in relation to natural disturbance regimes. In Mediterranean regions, fires are frequent, and a true measure of restoration success should concern the ways in which these restored communities respond to fires. Although the task of restoring highly diverse ecosystems may be difficult in itself, I recommend that completion criteria for mined lands in Mediterranean regions should also include the ability of the system to recover from fire, to ensure that an appropriate, self-sustaining analogue ecosystem has been established. In the case studied here, species richness was reduced by 22–41%, and new management approaches would be required to maintain species richness, including subsequent seeding, fill-ins, and possibly prescribing low intensity burns (on unburnt sites), to counter the potential negative effects of subsequent fires on important ecosystem properties.

Acknowledgements

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CHAPTER 4.0

Post-fire persistence of resprouter species in post-mine rehabilitated and natural shrubland communities in southwestern Australia

(Submitted manuscript)

4.1 Abstract

This study compared post-fire persistence of individuals in ten lignotuberous shrub species between three sites rehabilitated 8–24 years ago after mineral-sand mining and three surrounding natural shrubland communities (ranging 9–25 years since previous fire) near Eneabba, southwestern Australia. For the ten species, overall persistence of individuals was 11–93% in rehabilitated sites with a mean of 52%, and 79–100% in natural sites with a mean of 96%. Persistence increased with time since rehabilitation for five species with <25% individuals of three species persisting in the youngest stand. For equivalent crown size, lignotuber circumferences were on average 50% smaller at rehabilitated sites and likely accounted for their higher post-fire mortality. Apart from differences in the age of plant, rehabilitated sites had lower soil penetrability than natural sites which may have restricted rootstock development. A tradeoff favoring a higher crown volume to lignotuber size ratio was apparent in nine of the ten species with greater crown volumes (by 37%) and smaller lignotubers (by 36%) in rehabilitated sites. Two resprouting species for which canopy seed store was quantified had much higher fecundity (accompanied by greater crown volume) in rehabilitated sites. This indicates seedling recruitment may have a greater role in post-fire recovery in rehabilitated sites. Nevertheless, fire drove some species, usually considered stable in the surrounding natural communities, close to local extinction with <20 adult individuals remaining per ha.

Fires reduced resprouter persistence in rehabilitated sites due to poor development of lignotubers in rehabilitated soils. Management after fires is required to enhance long term persistence of resprouter species in rehabilitated sites: prescribing low-intensity fires, resprouter seed application, restoring a deeper sandy profile to provide a suitable medium for lignotuber development, and transplanting adult resprouters from natural areas ahead of mining fronts.

Keywords: disturbance, fire ecology, function, lignotuber, mineral sand-mining, resprouter, restoration

4.2 Introduction

In Mediterranean-climate regions, resprouting allows for the persistence of many perennial plant species after fire (Bell et al. 1984; Bond and van Wilgen 1996; Bond and Midgley 2001). In shrubland communities of southwestern Australia, fires have recurred at mean intervals of 7–16 years (Enright et al. 1998; Lamont et al. 2003; Miller et al. 2007) and 60–80% of species resprout after fire from protected buds on stems, lignotubers, rhizomes or corms (Bell et al. 1984; Enright et al. 2007). Those species that do not resprout (nonsprouters) are usually prolific seed producers and recover from fires via seedling recruitment.

Most resprouting species in Australian shrubland communities regenerate via swollen rootstocks called lignotubers. Lignotubers are mostly buried under soil so that the extreme heat of fire cannot reach and kill the buds stored there. Carbohydrates and mineral nutrients are stored in lignotubers (Dell et al. 1985; Cruz and Moreno 2001a) and are mobilized during resprouting, acting as the main supply of carbon for regrowth at the early stages of recovery (Bowen and Pate 1993; Van der Heyden and Stock 1996). Lignotuber development is a function of plant age (Enright and Lamont 1992b): many lignotuberous species require at least 10, and as much as 15–30 years before they routinely survive fire (Lamont and van Leeuwen 1988; Enright et al. 1998). Independent of age, plants with larger lignotubers survive better and regenerate biomass more vigorously after disturbance (Bradstock and Myerscough 1988; Auld, 1990; Moreno and Oechel 1991). Optimal partitioning theory predicts that there should be proportionately larger lignotubers in less productive, nutrient- and water-deficient environments (Bloom et al. 1985), but this trend has not been supported in other studies (Cruz and Moreno 2001b).

Restoration of resprouter species on previously mined lands to densities similar to natural populations has proved a difficult task for rehabilitation managers in southwestern Australia (Bellairs and Bell 1993; Koch and Ward 1994; Grant and Loneragan 1999). Strip mining involves complete removal of former vegetation so that resprouting is eliminated as a source of recruitment during restoration (Bellairs and Bell 1993). Colonisation from surrounding natural areas is likely to be a slow process due to low rates of seed production, and so to accelerate it, managers apply seeds and seedlings of resprouting species into restored topsoil. However, only limited numbers of seeds are available to harvest from resprouters as they generally produce fewer seeds with lower viability than nonsprouters (Bellairs and Bell 1993; Lamont and Wiens 2003).

Since the 1970s, Iluka Resources Ltd (and their predecessors) has been restoring vegetation on land previously mined for heavy minerals near Eneabba, southwestern Australia. A recent survey in this restored vegetation reported lower resprouter densities (4 individuals per m^2) compared with surrounding natural areas (7–15 per m^2) (Herath et al. in press.). A subsequent post-fire survey at the same sites recorded lower persistence of resprouting species at the community level in rehabilitated sites (42–66% of resprouter species persisted, including as seedlings, and 31–50% by resprouting only) compared with natural sites (96–100% and 96–98%, respectively) (Herath et al. unpub.). It therefore became important to determine what factors caused such a marked decline and how this plant functional group might be restored in adequate densities in the long term. It is possible that resprouters in the rehabilitated sites had smaller lignotubers as they are much younger, on average, than those in the surrounding vegetation where most resprouter individuals have likely passed through many fire cycles and only a few are likely to be recruits following the most recent fire. Also, woody species are much larger in the rehabilitated sites (Herath et al. unpub.) suggesting that their crowns may have grown at the expense of lignotuber development. I examined post-fire persistence of individuals for ten major species common to both rehabilitated and nearby natural sites to confirm patterns for the vegetation generally, and to determine if higher mortality in restored resprouters could be attributable to smaller lignotuber sizes. I also investigated whether shoot growth and fecundity has occurred at the expense of lignotuber growth (resprouting potential). Based on these results, recommendations are given for the long-term conservation of resprouting species in rehabilitated sites. Specifically, the following directional hypotheses were tested:

1. Post-fire persistence of resprouter individuals is lower in rehabilitated sites;
2. Lignotuber size of resprouters is smaller in rehabilitated sites;
3. For a given plant size, lignotubers of resprouters are smaller in rehabilitated sites;

4. Resprouters have greater crown size in rehabilitated sites; and
5. Resprouters are more fecund in rehabilitated sites.

4.3 Methods

This study was conducted near the town of Eneabba (29° 49' S, 115° 16' E), 280 km north of Perth, southwestern Australia. The area is known as the Eneabba Sandplain (Commander 1978) and is a low-lying basin composed of a series of early Pleistocene (or Late Tertiary) shoreline, lagoon and dune deposits having locally high concentrations of heavy minerals (Playford et al. 1976). The region experiences a dry Mediterannean-type climate of hot, dry summers (daytime temperatures regularly exceeding 40 ºC), and mild, wet winters, with average annual rainfall at Eneabba of 504 mm (BoM 2008). The vegetation of the sandplain is classified as shrubland (Beard 1984) and is extremely diverse with many species endemic to the region (Lamont et al. 1984). It is rich in Proteaceae, Myrtaceae, Cyperaceae, Papilionaceae and Ericaceae (Hnatiuk and Hopkins 1981; Enright et al. 2007). The shrubland communities occur on soils with sandy surface horizons, and vegetation composition is largely determined by the depth of the sand (Griffins and Hopkins 1985; Enright and Lamont 1992a) and the nature of the underlying horizons, whether laterite, limestone, silt-clay or bedrock (Beard 1984; Enright et al. 2007). Three mined sites rehabilitated 8 (R8), 15 (R15) and 24 (R24) years ago and three nearby natural shrubland communities on swale (25 years since previous fire) and dune (9 and 19 years since previous fire) substrates were examined in the study area (Table 4.1).

Ten locally well-represented resprouting (lignotuberous) shrub species were studied: *Hakea incrassata*, *Dryandra shuttleworthiana*, *Conospermum wycherleyi*, and *Isopogon tridens* – all Proteaceae, *Jacksonia floribunda* (Febaceae), *Allocasuarina humilis* (Casuarinaceae), and *Calothamnus hirsutus*, *Eremaea beaufortioides, Leptospermum spinescens* and *Melaleuca leuropoma* – all Myrtaceae (nomenclature follows Western Australian Herbarium 1998+). All species are woody, sprawling to erect, sclerophyllous shrubs to 1.5 m tall and occurring on sandy soils, sometimes over laterite. Not all species occurred in all sites with the number of sites occupied per species ranging from a minimum of 4 to a maximum of 6 (i.e. all) sites.

Prior to experimental fires at study sites, numbered metal tags were placed adjacent to study plants so that burnt species could be identified after fire. Late autumn experimental fires were conducted at all sites with assistance from Iluka Resources Ltd, Bushfires Board and Department of Environment and Conservation fire department staff. The fires covered 1–2.5 ha per site, were of high intensity at all sites except in some small patches in R8. Nevertheless, plants that did not burn well were easily distinguishable (i.e., by observing degree of burn in stems and crown foliage, any surrounding litter) and were not studied. Sixty individuals of each species per site were tagged with the intention that at least 50 individuals of each species would be relocatable post-fire and burnt succesfully. However, this target was not reached at some sites with sample size averaging 48 (range 30–55). Burnt individuals were assessed alive or dead one year after fire by examining if any fresh re-growth of stems or leaves had occurred. Percentage of individuals alive (at each site) was calcultated.

Before fires, the number of individuals of all species was recorded in ten 15 × 15 m quadrats at each site (total 2250 m^2 per site). These values were converted to number of individuals per ha. With records of percentage of individual's alive post-fire at each site, it was possible to estimate the number of post-fire individuals alive per ha.

As an index of lignotuber size, circumference around the base of the numerous branches arising at ground level (or rootstock if it was visible) of the study species was taken adjacent the three rehabilitated and natural sites assessed before fire. A random starting point was chosen and then sequential plants were measured as they were encountered in random walks. Preliminary observations indicated that outer branches arose around the edge of the living lignotuber, and could be used to indicate its areal extent. Plant height and crown size (in two – longest and perpendicular – dimensions) were recorded to determine if any relationship existed between crown volume (calculated as an ellipsoid) and lignotuber circumference. The samples for each species were pooled from the three natural sites (averaging 46 individuals per species, range 30–62), and the rehabilitated sites (averaging 45 individuals per species, range 34–52). Combined totals were used since the resprouters in natural sites were of unknown age, and equal numbers of individuals of all species were not available at each rehabilitated block.

To investigate if similar-sized plants (equal crown volume) of each species had smaller lignotuber sizes (circumference) in rehabilitated sites than in natural sites, I haveplotted log crown volume of all individuals against their lignotuber circumference. A line of best fit was constructed for the natural plants (r^2 x-y, *P* x-y) and a horizontal line drawn from the mean volume (and circumference) of the restored plants to intersect the line of best fit for the natural plants. A vertical line was dropped from the intercept to determine the equivalent lignotuber circumference for the natural sites. Mean lignotuber sizes and crown volumes were compared by Wilcoxon signed rank test.

Seed production data were obtained for *M. leuropoma* and *C. hirsutus* as part of a larger study on resprouters and nonsprouters at Eneabba (Herath et al. unpub.). Sites were chosen to cover as wide a range as possible of time since last fire ages for natural sites, and time since rehabilitation for post-mine sites. At each site, the numbers of mature fruit clusters (*M. leuropoma*) or fruits (*C. hirsutus*) were recorded on 50 plants. Since *C. hirsutus* is only weakly serotinous, releasing seeds after only a few years, the numbers of fruits on the ground were recorded as well, and pooled with those on the plant. A random point was selected, and the nearest five individuals were sampled before moving to another point at least 50 m away. For the first two individuals at each point, 15 mature clusters (*M. leuropoma*) or 50 fruits (*C. hirsutus*) were picked and placed in a paper bag. Fruits /clusters from 20 individuals of each species at each site, were oven-dried at 55ºC for two days to force seed release. Twenty seeds from each of these 20 individuals from each site were placed into Petri dishes on moist filter paper, and a dissecting microscope was used to record seeds that had germinated each day for 30 days. The following calculations were then made to estimate the germinable seeds per individual at each site:

1. Mean no. of seeds per cluster/fruit = no. of seeds in bag ÷ 15 (total clusters of *M. leuropoma*) or 50 (total fruits of *C. hirsutus*)
2. Total no. of seeds per individual = mean no. seeds per cluster/fruit × no. of clusters/fruits on plant

3. Total germinable seeds per individual = average germination rate (fraction) × total no. seeds per individual

At each site, a 40 × 40 m plot was established and within it, 90 soil samples (14 × 14 cm area, 0–5 cm depth) were collected for nutrient analysis from random points. Each soil sample was dried at 40^{o}C for 48 h, mixed thoroughly and a 20 mL subsample was analysed for total nitrate, ammonium, phosphate, potassium, sulphur, organic carbon, reactive iron, conductivity, and pH (in $CaCl_2$ solution and in H_2O) (analyses performed by CSBP Wesfarmers, Perth). To measure soil hardness, an Australian standard sand penetrometer (AS 1289 6.3.3) with a 16 mm diameter flat-ended rod driven by a 9 kg mass dropping 600 mm was used. After one hammer blow, the distance the rod penetrated the ground was measured. Short penetration distances indicate less penetrable/harder soil. Soil penetrability was measured at 20 random measurements along the edges of each plot. Significant differences in soil factors between rehabilitated and natural sites were tested by 1-way ANOVA, and post-hoc tests were applied on significant variables, using SPPS 13.0 (2004).

Table 4.1: Selected study sites sampled in natural and restored shrubland near Eneabba, southwestern Australia. High = High dune, Low = Low dune.

Sites	Location	Substrate type	Age (years)
High	S29°52'23.6" E115°15'02.4'	6–10 m of sand over silt and clay	19 since previous fire
Low	S29°42'41.4" E115°13'28.1"	2–3 m of sand over silt and clay	8 since previous fire
Swale	S29°51'54.8" E115°15'12.5"	50 cm of sand over silt and clay	24 since previous fire
R8	S29°55'48.6" E115°17'15.6"	0–30 cm sand over silt and clay	8 since restoration
R15	S29°53'44.6" E115°17'02.9"	0–10 cm sand over silt and clay	15 since restoration
R24	S29º55'58.9" E115º16'27.8"	0–20 cm sand over silt and clay	24 since restoration

4.4 Results

For all ten species, post-fire persistence was lower in rehabilitated than natural sites (Figure 4.1). Overall, 79–100% of individuals of all species survived fire in the natural sites with a mean of 96% across all sites, while 11–93% survived fire in the rehabilitated sites with a mean of 52%. *C. wycherleyi*, *E. beaufortioides*, *J. floribunda*, and *I. tridens* were the most adversely affected with less than 50% of individuals regenerating. *H.incrassata* was the least affected with 96% alive in the natural site and a mean of 87% in the rehabilitated sites. Persistence in rehabilitated sites was lowest at the youngest rehabilitated site (R8) for five species, especially *M. leuropoma*, *J. floribunda*, *C. hirsutus*, and *L. spinescens* with <25% alive. Persistence was highest at the oldest site (R24) for *C. wycherleyi*, *J. floribunda*, and *L. spinescens*, while five species had equal highest persistence at the two older sites (R16 and R24). Poor persistence in rehabilitated sites made little difference to relative abundances for several very abundant species as they remained in their thousands (*M. leuropoma*) or hundreds (*C. wycherleyi*, *E. beaufortioides*) per ha, but some less abundant species (with densities <100 pre-fire) fell to very low levels (<20 post-fire: *D. shuttleworthiana*, *I. tridens*, *L. spinescens*, *A. humilis*).

Lignotuber circumferences were 19% lower overall in rehabilitated sites than in natural sites (Wilcoxon ranked test, $P<0.05$) (Table 4.2). The lignotubers of rehabilitated *C. wycherleyi*, *E. beaufortioides* and *J. floribunda* were less than half the size of those in natural sites, although *I. tridens* and *H. incrassata* lignotubers were larger in rehabilitated sites. At a crown volume equivalent to the mean at rehabilitated sites, overall lignotuber circumference of all species was 50% smaller in rehabilitated sites than in natural sites ($P<0.05$), though *H. incrassata* was slightly larger (Table 4.2). Five species had lignotuber circumferences three or more times those in rehabilitated sites. Average crown volume of resprouter species was 40% greater in rehabilitated sites than in natural sites ($P<0.05$), though *E. beaufortioides* was smaller (Table 4.2). In rehabilitated sites, six species had crown volumes double or more those in natural sites.

Seed production of *M. leuropoma* and *C. hirsutus* in rehabilitated sites was considerably higher than in natural sites at almost all ages analysed (Figure 4.2), for instance: *M. leuropoma* germinable seed production in the 8-year old rehabilitated site was more than double that of the equivalent aged (time since fire) natural sites. Similarly, after 15 years since restoration/fire, *C. hirsutus* produced more than twice as many seeds in rehabilitated sites than in natural sites.

Of the ten soil traits assessed only reactive iron and soil hardness differed significantly between natural and rehabilitated sites ($P < 0.05$, 1-way ANOVA; Table 4.3). Reactive iron levels in rehabilitated soils were double those in natural soils and penetrability was half.

Table 4.2: Lignotuber circumference (cm) at a plant volume equivalent to their mean crown volume in restored stands, mean lignotuber circumference (cm) and mean crown volume (m²) of 10 resprouter species – in natural and restored shrubland communities near Eneabba, WA. Bold values indicate values significantly differing (greater values in bold) between rehabilitated and natural sites. Lig. circ = Lignotuber circumference, Cv = coefficient of variation.

Species	Lig. circ. at equal crown size		Mean lig. circ. (cm)				Mean crown volume (m²)			
	Natural	Restored	Natural	Cv	Restored	Cv	Natural	Cv	Restored	Cv
Hakea incrassata	64	**72**	33	0.55	**77**	0.43	0.21	0.93	**0.61**	0.73
Allocasuarina humilis	**64**	37	**64**	0.94	36	0.96	0.35	0.81	**0.44**	0.67
Leptospermum spinescens	**68**	19	**27**	0.92	18	0.54	0.03	0.90	**0.1**	1.05
Melaleuca leuropoma	**85**	30	**50**	0.58	30	0.45	0.06	0.65	**0.16**	0.98
Dryandra shuttleworthiana	**116**	65	**60**	0.5	54	0.37	0.17	0.64	**0.4**	0.57
Calothamnus hirsutus	**101**	62	**80**	0.51	62	0.6	0.23	0.98	**0.31**	0.55
Conospermum wycherleyi	**69**	19	**44**	0.62	20	0.72	0.07	0.96	**0.16**	0.88
Eremaea beaufortioides	**53**	17	**75**	0.59	16	0.75	**0.37**	1.03	0.25	0.82
Jacksonia floribunda	**124**	32	**80**	0.81	32	0.75	0.25	1.08	**0.48**	0.83
Isopogon tridens	**30**	18	25	0.81	**30**	0.90	0.22	0.87	**0.36**	0.84

Table 4.3: Mean values of soil properties (before fire) in natural and restored shrubland communities near Eneabba, WA. Bold numbers indicate soil characteristics significantly differing (post-hoc testing of means, higher values in bold) between natural and rehabilitated sites. Cv = coefficient of variation.

	High		Low		Swale		R8		R15		R24	
Soil properties	Mean	Cv	Mean	Cv	Mean	Cv	Mean	Cv	Mean	Cv	Mean	Cv
Nitrate (mg/kg)	1.1	0.37	1.3	0.36	1.1	0.31	1.8	0.77	1.4	0.59	3.7	0.91
Ammonium (mg/kg)	2.4	0.38	2.7	0.51	3.4	0.42	3.5	0.94	2.5	0.75	5.3	0.76
Phosphorus colwell (mg/kg)	2.9	0.27	2	0.00	3.7	0.46	2.2	0.35	2	0.00	2.8	0.43
Potassium colwell (mg/kg)	31	0.29	21	0.30	37	0.23	24	0.33	25	0.37	31	0.41
Sulphur (mg/kg)	4	0.35	2	0.40	4.4	0.38	3	0.54	2.9	0.54	3.4	0.57
Organic carbon (%)	0.72	0.31	0.5	0.26	0.77	0.28	0.61	0.39	0.53	0.50	0.8	0.46
Reactive Iron (mg/kg)	81	0.19	72	0.18	53	0.35	**208**	0.36	**317**	0.11	**183**	0.36
Conductivity (dS/m)	0.025	0.32	0.023	0.37	0.028	0.30	0.028	0.33	0.03	0.46	0.035	0.44
pH (CaCl2)	5.1	0.06	5.2	0.04	5.1	0.06	5.1	0.04	5	0.04	4.9	0.06
pH (H2O)	6.2	0.04	6	0.02	6.1	0.04	6	0.03	6	0.03	6	0.03
Penetrometer depth (cm)	**38.9**	0.16	**39.1**	0.16	**26.9**	0.09	16.2	0.18	15.1	0.13	17.9	0.22

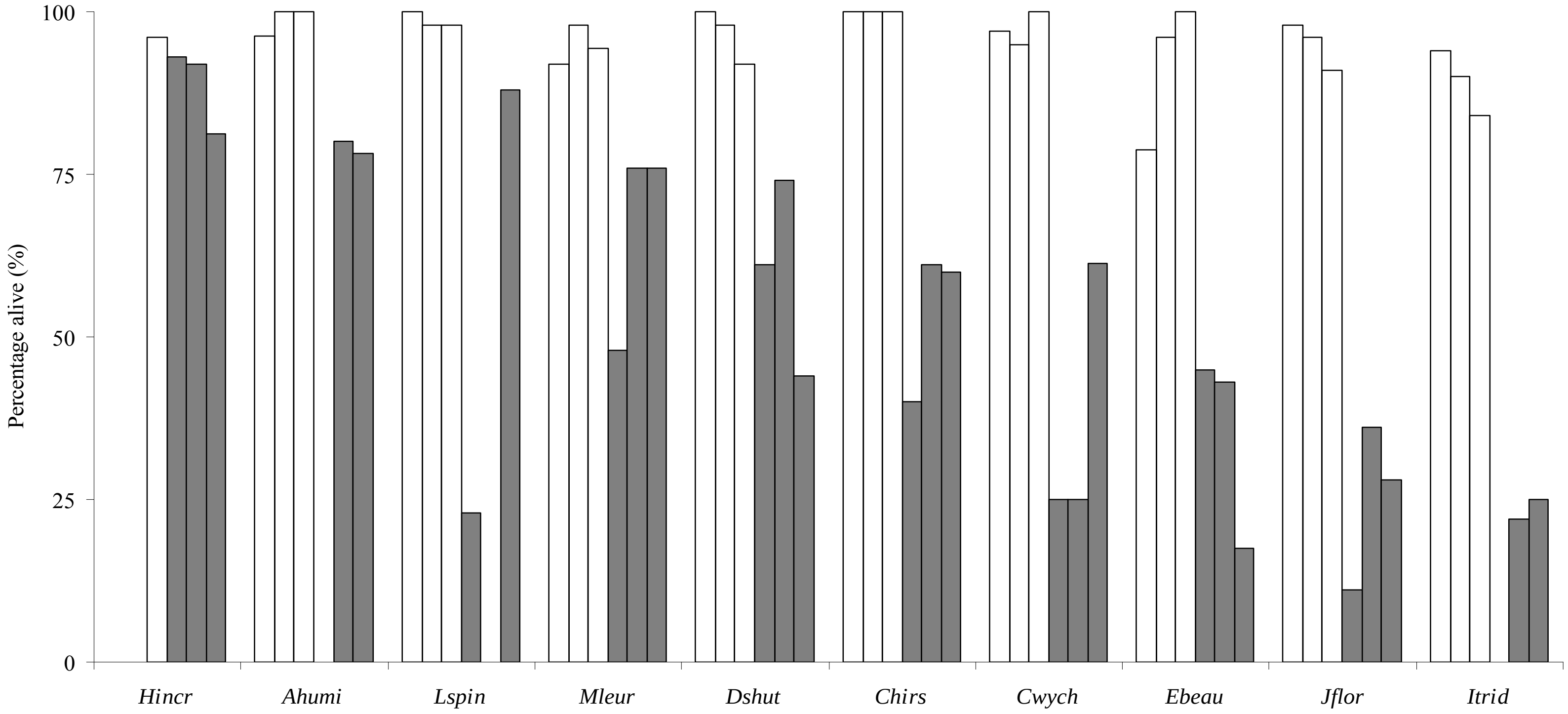

Figure 4.1: Postfire persistence (% alive) of 10 resprouting species in natural (white) and restored (grey) shrubland communities near Eneabba, WA. Natural sites = 1st column: High, 2nd: Low, 3rd: Swale. Rehabilitated sites = 1st column: R8, 2nd: R15, 3rd: R24. *Hincr* = *H. incrassata*, *Ahumi* = *A. humilis*, *Lspin* = *L. spinescens*, *Mleur* = *M. leuropoma*, *Dshut* = *D. shuttleworthiana*, *Chirs* = *C. hirsutus*, *Cwych* = *C. wycherleyi*, *Ebeau* = *E. beaufortioides*, *Jflor* = *J. floribunda*. Note: not all species were present at all sites.

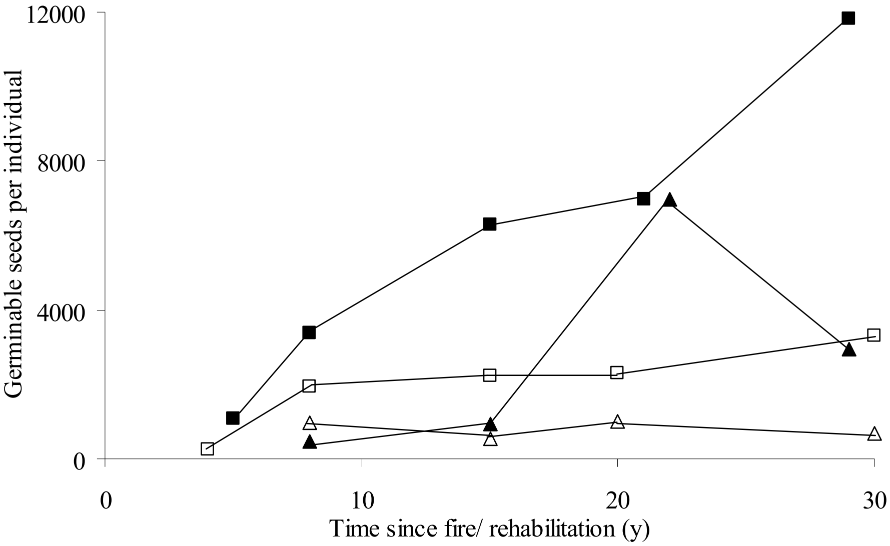

Figure 4.2: Seed production by *M. leuropoma* (squares) and *C. hirsutus* (triangles) in natural (unfilled) and restored (filled) shrubland communities near Eneabba, WA.

4.5 Discussion

Burning shrubland restored on previously mined lands near Eneabba, southwestern Australia resulted in the death of far more individuals (mean of 52%) of ten resprouting species than occurred in nearby natural sites (4%). The most likely cause of this disparity in persistence after fire was smaller lignotuber size (19% lower mean circumference) recorded in rehabilitated sites. Several studies have shown that resprouting capacity is dependent on lignotuber size, whereby plants with larger lignotubers survive better than those with smaller ones (Rundel et al. 1987; Auld 1990; Moreno and Oechel 1991; Enright et al. 1998) since they are better insulated against fire heat and contain greater numbers of dormant buds that are able to sprout following fire (Cruz et al. 2002). The restored resprouters were at most 24 years of age whereas resprouter populations in the natural sites were multi-aged, containing many individuals that may have survived (and grown) through multiple fire-cycles, which for example at Eneabba have been at mean intervals of 13 years over the last 40 years (Miller et al. 2007). This at least partly explains smaller lignotuber size in the rehabilitated sites, and also explains the tendency for species to survive better in the oldest rehabilitated stands.

While it was not possible to standardize for plant age between rehabilitated and natural sites, it was possible to standardize for crown size (developmentally matched) which showed that lignotuber circumference at the natural sites was on average 50% greater. This confirms that growth forms were different between rehabilitated and natural sites. One factor that may have contributed to smaller lignotubers in rehabilitated sites is their lower soil penetrability than natural sites, due to the high concentrations of silt-clay at the surface that would only be present at depths >50 cm in natural sites (Table 4.1). Consequently, the dense substrate may have restricted lignotuber growth and allocated the extra energy reserves used for above-ground growth, as all studied species (except *E. beaufortioides*) had greater crown volumes in rehabilitated sites. It is also possible that the lower plant density in the rehabilitated sites (Herath et al. in press) reduced competition for resources and promoted rapid shoot growth rather than lignotuber development (Bloom et al. 1985; Iwasa and Kubo 1997; Bellingham and Sparrow 2000).

Smaller lignotuber size may not be the full explanation for greater mortality in the rehabilitated sites as there was no simple correlation between them at the species level. Mortality among *I. tridens* individuals was much greater in the rehabilitated sites although their lignotuber size was similar to that of natural areas. This may be because shoot growth occurred at the expense of bud storage – this species is rarely multistemmed indicating little capacity for bud production and most resprouts occur as single-stemmed root suckers: perhaps the lateral roots were not sufficiently well developed to support suckers. Persistence of *H. incrassata* was high in both locations (> 85%) even though lignotubers in the rehabilitated sites were twice the size of those in natural sites. It is possible that the critical size for persistence for this species was already satisfied at the natural sites. Plants are also more likely to die in more intense fires (Burrows 1985).

The two species assessed for fecundity, *M. leuropoma* and *C. hirsutus*, also had much higher seed production on rehabilitated than natural sites at matched times since disturbance. Similarly, an earlier study at R24, High and Swale recorded up to 10 times more seed production in *E. beaufortioides*, *L. spinescens* and *H. incrassata* at the rehabilitated site (McClaren 2005). This indicates that the lower resprouting capacity observed in rehabilitated sites was at least partly offset by a potential increase in seedling recruitment. Studies examining seed production in the other five resprouting species included in our study may confirm this trade-off, since the same lignotuber-crown volume trends were recorded in all species except *H. incrassata*. Thus, there is an apparently plastic trait response in rehabilitated-site resprouters trending towards that of nonsprouters, whereby they allocate more energy and nutrients to shoot and seed production and rely less on resprouting via lignotuberous buds in response to fire. This process has also been described by Bond and van Wilgen (1996), and Cruz and Moreno (2001b). A post-fire survey at the same study sites (Herath et al. unpub.) supports this idea, with 46–72% of resprouting species producing seedlings after fire in rehabilitated sites but only 49–51% in natural sites. However, this tradeoff was far from sufficient in our study to match death of pre-fire plants: higher seedling mortality was recorded over summer in rehabilitated than natural sites as the impenetrable substrate intensified the effects of summer drought (Enright and Lamont 1992a). The net effect was a marked reduction in population size of resprouting species after fire in rehabilitated sites, contrasting with population stability in the natural communities.

Management recommendations

There may be a case for delaying the introduction of fire into restored shrubland until sites are >24 years old to provide sufficient time for substantial lignotuber development in resprouters. However, the dilemma for managers is that such long fire intervals may threaten the persistence of some nonsprouting species that senesce sooner than this, recruit poorly interfire and have limited seed storage (Herath et al. unpub.). It is also likely that natural fires will recur within 24 years since mean fire intervals in the surrounding natural areas are currently at 13 years (Miller et al. 2007). It is possible that fires could be introduced at an earlier age if they are of low-intensity and patchy so that persistence of many resprouter individuals (including unburnt ones) would buffer sites against rapid decline in resprouter numbers (Noble 1984). If economically viable, assisted rehabilitation methods, such as sowing of resprouter seeds immediately after fire, may be necessary. Species in especially low abundance after fire should be targeted. Restoration of a deeper sand profile directly beneath the topsoil would also provide a more suitable medium for lignotubers to fast-track their development and improve their persistence. This will likely have other benefits on vegetation dynamics in rehabilitated sites such as reduced mortality of seedlings and adults associated with summer drought (Enright and Lamont 1992a, Herath et al. unpub.). Stabilizing returned topsoil against erosion would assist the same ends. A further option is to transplant mature resprouter individuals from natural areas ahead of the mining front directly into sites ready to be rehabilitated: the technology now exists for grasstrees (Lamont et al. 2004) which are conspicuously absent from rehabilitated lands at present. This would ensure a diversity of ages and hence survival abilities after fire if the mechanical procedures do not significantly affect the vitality of lignotuberous plants.

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CHAPTER 5.0

Demography of selected shrub species in post-mine rehabilitated *versus* natural shrubland sites and optimum return intervals for fire

(Submitted manuscript)

5.1 Abstract

Due to large scale disruption of ecological processes and environmental conditions during mining, restored vegetation may develop at a different rate from that of natural analogues, affecting demographic parameters including time to reproductive maturity, rates of biomass accumulation, seed production and storage. This study compares the demography (size, fecundity, longevity) of six selected woody species with different regeneration (resprouter vs. nonsprouter) and seed production/storage (soil vs. canopy) attributes for post-mine restored and adjacent fire-prone shrublands of similar age (years since restoration vs. years since last fire: 3-30 years (y)) near Eneabba, southwestern Australia. Using these results, I seek to identify at what stand age fire would best be re-introduced to restored vegetation to maximize the chances of population recruitment (via seedling recruitment) in the selected species, and whether these optimum fire intervals differ from those for the same species in natural shrublands.

At matched years since restoration or last fire, nonsprouter species in rehabilitated sites grew larger (× 1.1 to 4.7) and produced/stored more viable seeds per plant (× 1.1 to 10.9). Nonsprouter species reached reproductive maturity 1 (*Beaufortia elegans*) to 4 (*Petrophile drummondii*) years earlier in rehabilitated sites (except *Acacia blakelyi*: 3 y in both). Demographic attributes for resprouters were not directly comparable (at equivalent years since restoration/last fire) because individuals in natural sites comprised overlapping generations of individuals which had regrown vegetatively after one or more past fires, while individuals in rehabilitated sites had grown from seed at the time of restoration. Despite this, resprouters were larger (1.1 to 3.6 times - except *Jacksonia floribunda*), produced/stored more viable seeds (1.1 to 6.9 times), and showed inter-fire recruitment at a younger stand age (except *Melaleuca leuropoma*) in rehabilitated sites.

Although (overall) growth and seed production rates were greater in rehabilitated sites, the estimated optimum fire return interval based on maximum seed production was generally similar for both treatments - except for *Beaufortia elegans* where it was 8 y in rehabilitated but 15 y in natural sites. Optimum fire intervals were longer for resprouter (≥22 y) than for nonsprouter (≤16 y) species in both treatments. Mean fire intervals typical of surrounding natural vegetation near the Eneabba area (13 y over last 40 y: Miller et al. 2007) may not be suitable for rehabilitated minesites at Eneabba, where longer initial fire intervals (20-30 y) would better ensure persistence in resprouter species.

Keywords: disturbance, fecundity, longevity, nonsprouters, rehabilitation, resprouters, restoration, shrubland.

5.2 Introduction

Natural disturbances such as fire play a key role in the functioning of many Mediterranean-type plant communities (Christensen 1985; Kelly and Parker 1990; Whelan 1995; Bond and Van Wilgen 1996). In the shrublands of Western Australia (kwongan), North America (chaparral) and South Africa (fynbos), resprouting, and seed stores protected from fire (serotinous fruits in the plant canopy/hard-seed coats for soil stored seeds) are common functional adaptations in plants which facilitate persistence after fire (Kruger 1983; Bell et al. 1984; Bond 1984; van der Moezel et al. 1987; Kelly and Parker 1990; Keeley 1992; Enright et al. 1996). Resprouter species recover after fire by regrowing from protected buds beneath bark or tightly packed leaf bases above ground, or from lignotubers, rhizomes or bulbs below ground (Bell et al. 1984). Most resprouters can also recruit from seed, although levels of seed production and rates of recruitment in resprouters may be low (Groom et al. 2001). Nonsprouter species are killed by fire and depend solely on seed production for their persistence (Bell et al. 1984), generally producing more viable seeds than resprouter species (Lamont and Wiens 2003). Seeds are typically stored either in a canopy (serotinous) or soil seed bank, with germination cued to fire, because post-fire conditions of increased access to light, moisture and nutrients favour seedling establishment (Enright et al. 1996). Serotinous species store their seeds inside protective woody fruits in the plant canopy (serotiny) from several (weakly serotinous) to many (strongly serotinous) years (Cowling and Lamont 1985; Lamont et al. 1991: Enright et al. 1996). Serotiny is especially prominent in the flora of SW Australia in areas where fires are frequent, where species have shorter life spans, and where granivory is more common (Lamont et al. 1991; Lamont and Enright 2000). Adaptations for soil storage of dormant seeds in fire prone environments include hard-seededness (Keeley 1991; Bell and Williams 1998), with the heat of fire required to crack the seed coat, and smoke-induced germination, where the chemical signature in smoke breaks dormancy (Roche et al. 1997; Rokich et al. 2002).

Effective fire management of species in Mediterranean-type ecosystems requires an understanding of plant demography, particularly concerning critical life history stages and levels of seed-stores necessary for persistence in relation to the return interval between successive fires (Bradstock and Myerscough 1981; Bell et al. 1984; Kelly and Parker 1990; Witkowski et al. 1991). A sufficient seedbank is required to ensure population persistence post-fire. If fires occur too early, i.e., before the onset of reproductive maturity or before a substantial seedbank or budbank is accumulated, then a population may be threatened with local extinction (Enright et al. 1996). If fires occur too late, plants may have senesced and their seed stores declined (Enright et al. 1996; Witkowski 1991). The chance of inter-fire recruits contributing to future generations is also low (especially for nonsprouters) due to strong competition from mature vegetation which reduces seedling survivorship (Cowling and Lamont 1987; Lamont and Barker 1988), and the high probability of fire recurrence before such recruits reach reproductive maturity.

Most mine-site restoration programs focus on matching species composition and community structure of rehabilitated sites to those of desired natural analogue sites. However, few studies have reported on the relative demographic performance (growth, fecundity and longevity) of species in rehabilitated and natural sites, or on the potential impacts of re-introduced natural disturbances, such as fire. Since mining is one of the most radical forms of human disturbance in the landscape (Hobbs and Hopkins 1990), restoration is often difficult and may not adequately restore pre-mining environmental conditions (including nutrient levels, soil physical properties, mycorrhizal associates, etc). Therefore, it is important to examine how plant species behave demographically in post-mined rehabilitated sites, and what implications this may hold for the sustainability of these reconstructed systems. This is particularly important where landscape scale disturbances such as fire are an integral part of the natural system dynamics. This study compares the demography (size – a surrogate index of growth, fecundity, and longevity) of six common woody species with different life history attributes associated with persistence (resprouter vs. nonsprouter) and seed storage (soil vs. canopy), for post-mine restored and nearby natural shrubland sites of similar ages (years since restoration vs. years since last fire) (Table 5.1). On the basis of these results, I seek to predict fire intervals for the selected species which maximize their chances of persistence after fire. Specifically, the following questions were posed for the six selected species:

Are there differences between restored and natural shrubland sites, and/or between species with differing sets of life history attributes (nonsprouter/resprouter, canopy/soil seed storage), among the selected species in:

a) mean plant size (since sites are matched by age, size represents total growth over comparable time intervals);
b) mean viable seed production/storage per plant;
c) mean seed viability rates;
d) median time (years) to maturity;
e) years to maximum viable seed production per plant;
f) years to onset of inter-fire recruitment and extent of inter-fire recruitment; and
g) mean plant longevities.

2. Is there an identifiable optimum fire interval for each species, does it differ between rehabilitated and natural sites in a consistent manner, and how might it affect the sustainability of the restored shrublands?

5.3 Methods

Study species and sites

Six plant species were chosen to represent widespread and abundant components of the Eneabba shrublands region spanning a variety of regeneration and seed storage modes (Table 5.1). *Acacia blakelyi* (Mimosaceae) is a tall nonsprouter shrub growing to about 3 m height on yellow, red or white sands, or lateritic soils. Moderate sized, hard seeds (6mm, plus eliasome) are dropped from pods upon ripening, are dispersed by ants, and accumulate in a soil seed bank. *Petrophile drummondii* (Proteaceae) is a multi-branched nonsprouter shrub which grows to about 1.2 m on sandy laterite, grey or yellow sand. Seeds are moderate in size (4 mm) and are held between woody bracts in weakly to moderately serotinous fruits. *Beaufortia elegans* (Myrtaceae) is a small, nonsprouter shrub found on white, yellow or grey sand, often over laterite. It rarely exceeds 1 m in height (Delfs et al. 1987). Small seeds (1 mm) are held in moderately serotinous capsules. *Jacksonia floribunda* (Papilionaceae) is a multibranched resprouter shrub growing to about a maximum of 3 m height. Moderate sized seeds (4-6 mm) are released from pods open ripening and are accumulated in the soil seedbank. *Calothamnus hirsutus* (Myrtaceae) is a compact multi-branched (resprouter) spreading shrub which grows to about 1.5 m on yellow/grey sand, clay, sandy clay, loam, gravel, weathering sandstone or granite. Small seeds (1-2 mm) are held in weakly serotinous fruits which fall from the plant within a few years. *Melaleauca leuropoma* (Myrtaceae) is a resprouter shrub which grows to about 1 m on well drained, white or brown sands over laterite. Small seeds (1 mm) are held in moderately serotinous capsules in the plant canopy

Study sites were all near the town of Eneabba (29° 49' S Latitude, 115° 16' E Longitude), 280 km north of Perth, southwestern Australia. Sites were chosen to cover as wide a range as possible of times since last fire (years) for natural sites and times since restoration (years) for post-mined sites (see Tables 5.4 and 5.5 for years sampled). A total of five sites per treatment for each species were sampled, except for *Calothamnus hirsutus* and *Petrophile drummondii*: four sites per treatment. Perfect matching of ages (years since restoration vs. years since last fire) was not always possible since selection of sites depended on the presence of selected species at the available sites, and the youngest stand for each site type was selected to capture the onset of reproductive maturity (defined as age by which >50% individuals showed evidence of flowering or fruiting). For sites >15 years, if exactly matched ages (i.e., for species, between treatments) were not available but the difference was ≤2 years, then they were grouped as the same age since the difference in demographic attributes would be considered negligible relative to the differences for sites with ages either much younger or older.

Data collection and analysis

At each site, plant size (height, average width of crown – N to S and E to W) and numbers of mature fruit clusters (for *M. leuropoma*, *B. elegans*), fruits (*C. hirsutus*), cones (*P. drummondii*) or ripe pods (*A. blakelyi*: late December, *J. floribunda*: May) were recorded for the first 50 plants of each species encountered. For all species, a random start point was selected, and the nearest five individuals were sampled for fruit numbers and plant dimensions, and the nearest 25 individuals were classified as alive or dead to estimate longevity rates (% alive), before moving to another point at least 50 m away, until a total of 50 plants had been sampled for fruiting and 250 for survivorship. For the first two individuals at each point, a target of 15 mature clusters (*M. leuropoma*, *B. elegans*), 50 fruits (*C. hirsutus*), and 5 cones (*P. drummondii*) were picked (excluding ground seeds) and placed in labelled paper bags. The soil storage species, *A. blakelyi*, held seeds in mature pods of the canopy only briefly prior to seed shedding (in December), and so was sampled just before pods opened. 50 pods per plant (total of 50 plants per site) were placed in labelled paper bags and released firm seeds (ripe embryo and endosperm) were counted in the laboratory.

The fruits collected from a total of 20 individuals per species per site were oven-dried at 55ºC for two days to force seed-release. Released seeds of *P. drummondii* were identified as viable by presence of a plump embryo and endosperm. For the other species, twenty seeds (excluding aborted, predated) from each of the 20 individuals per site were placed into Petri dishes on moist filter paper, and a dissecting microscope was used to search for seeds that had germinated each day for 30 days. The following calculations were then made to estimate the viable seed store per individual at each site:

1. Mean no. of seeds per cluster/fruit/cone = no. of seeds in bag ÷ 15 (total clusters of *M. leuropoma*, *B. elegans*) or 50 (total fruits of *C. hirsutus*) or 5 (total cones of *P. drummondii*)
2. Total no. of seeds per individual = mean no. seeds per cluster/fruit/ cone × no. of clusters/fruits/cones on plant
3. Total viable seeds per individual = average germination/viability rate (fraction) × total no. seeds per individual.

Total seed store was examined using different methods for each species since they varied in serotiny levels. For the strongly (*B. elegans*, *M. leuropoma*) and weakly serotinous species (*P. drummondii*, *C. hirsutus*), it was assumed that the total seed store per plant represented the total number of seeds accumulated on the plant since their fall of seeds on the ground floor would be considered negligible – only a minute proportion were observed to either establish as seedlings (as inter-fire recruits – in which a very small fraction would reach reproductive maturity within the years sampled ~0–30 years) and their seeds would rarely be viable after a year and so most ground floor seeds would likely be incinerated after a fire (personal observation). Since the seeds of non serotinous *A. blakelyi* are hard, and persist for many years in the soil seed bank (personal observation), total seed store would be a cumulative function of the pattern of annual seed production. Therefore, to estimate (mean) total viable seed store available per plant at a given year (on plant and soil), the annual seed store per plant (i.e., on plant) was added to the previous year's annual seed production (i.e., on soil floor) from 3–30 years (3 years = when seed production began). Mean annual seed store per plant for years not sampled between 0–30 years was estimated by interpolating (smooth line function in Microsoft Excel® 2000) the data points for mean annual seed production on the y-axis with the years since restoration/fire on the x-axis. A seed decay rate function was then applied to the cumulated seeds per plant at a given year, by multiplying it by a decay rate fraction for a given age: at age x = exp (-0.0375 × age x) – based on Holmes and Newton (2004) decay rate equation for *Acacia saligna*, a very similar legume species which coexists with *A. blakeyi* on the Eneabba sandplains. *J. floribunda* released seeds from pods at different times of the year in different sites and so the number of fallen pods (down to 2 cm below litter) was counted and pooled with those on the plant (if any). Based on observations for ripe pods not opened (on the plant), it was assumed that fallen pods likely contained one viable seed per pod. Fallen aborted pods were clearly distinguishable from mature pods and were not counted.

T-tests were performed on demographic attributes to detect significant ($P<0.05$) differences between rehabilitated and natural sites using SPSS 13.0 (2004) statistical software. Data from most sites displayed a normal distribution with equal variances, except sites where interfire recruitment occurred; t-tests assuming unequal variances were performed in those cases.

Table 5.1: Regeneration (nonsprouter/resprouter) and seed storage (soil/canopy) modes for study species. n = nonsprouter, r = resprouter, ws = weakly serotinous, ss = strongly serotinous, h=hard-seeded.

Species	Regeneration mode	Seed store
Acacia blakelyi	Nonsprouter (n)	Soil – hard seeded (h)
Petrophile drummondii	Nonsprouter (n)	Canopy - weakly serotinous (ws)
Beaufortia elegans	Nonsprouter (n)	Canopy - strongly serotinous (ss)
Jacksonia floribunda	Resprouter (r)	Soil – hard-seeded (h)
Calothamnus hirsutus	Resprouter (r)	Canopy - weakly serotinous (ws)
Melaleuca leuropoma	Resprouter (r)	Canopy - strongly serotinous (ss)

5.4 Results

Nonsprouter species had faster growth rates, i.e., were taller and had greater mean biovolumes at equivalent years since restoration vs. last fire ($P<0.05$): P. drummondii (2.2 to 4.7 times more at all matched ages), *B. elegans* (1.2 to 3.2 times more) and *A. blakelyi* (3.7 to 5.7 times more at sites up to 8 years) (Figure 5.1). The onset of reproductive maturity was reached faster in rehabilitated sites: 1 (*B. elegans*) to 4 years (*P. drummondii*) earlier, except for *A. blakelyi* (at 3 years old in both treatments) (Table 5.2). Larger viable seed stores were recorded in rehabilitated sites (at matched years) ($P<0.05$) for *P. drummondii* (1.5 to 10.7 times more at all matched ages), *B. elegans* (1.1 to 10.8 times more), and *A. blakelyi* (4.3 to 5.9 times more at sites up to 8 years) (Figure 5.1). Peak seed production/storage occurred at similar plant ages in both treatments, except for *B. elegans* which reached its maximum at 8 years in rehabilitated sites and 15 years in natural sites (Table 5.2). Maximum seed production/storage for all nonsprouter species was reached in the range 8–16 y in both treatments. Seed viability was similar across treatments and ages ($P>0.05$), except for *A. blakelyi* which showed a decline in viability both with time since restoration (57–9%) and time since last fire (100–45%) (Table 5.4). Interfire recruitment in *A. blakelyi* occurred much earlier in rehabilitated sites (16 vs.

30 y) whereas it occurred at similar ages for *P. drummondii* (16 y at each) and was not observed (up to stand ages of 30 y) for *B. elegans* (Table 5.3, Figure 5.3). Plant senescence increased with time since restoration and time since last fire, but rates did not differ between rehabilitated and natural sites (P>0.05), except for *A. blakelyi* with more plant senescence in sites >20 years since last fire (P<0.05) (Table 5.4).

Despite resprouter individuals in rehabilitated sites being younger on average than those in natural sites (since they all grew from seed at the time of restoration site establishment), individuals in rehabilitated sites had greater mean biovolumes (C. hirsutus 1.1 to 1.9 times more, and *M. leuropoma* 1.5 to 3.6 times more, respectively) at equivalent years since disturbance, except *J. floribunda* (Figure 5.2). More viable seeds per plant were also recorded in rehabilitated sites at most matched years (P<0.05): *J. floribunda* 1.1 to 17.4 times more at 5 y, C. hirsutus 2.9 to 7.7 times more at 8 y, *M. leuropoma* 1.7 to 4.3 times more. Peak seed production/storage occurred at similar site age in both treatments for all resprouter species (all ≥22 y) (Table 5.2) and seed viability did not differ consistently between rehabilitated and natural sites (Table 5.5). Plant senescence increased with time since disturbances, but only up to 12%, and it did not differ between rehabilitated and natural sites (P>0.05) (Table 5.5).

Resprouter species took more than twice the time required to reach reproductive maturity in rehabilitated *vs.* natural sites (Table 5.2), although more accurately, while data for rehabilitated sites represent an estimate of time to grow to maturity, that for natural sites is an estimate of the recovery time of previously mature resprouting individuals to maturity once again (what might be referred to as length of the secondary juvenile phase). Most pertinent in this comparison is that resprouter species in the rehabilitated sites took little or no longer to reach maturity than did nonsprouter species (range 3–6 y for nonsprouters, 3–7 y for resprouters)

Interfire recruitment occurred earlier in rehabilitated sites for *C. hirsutus* (22 y vs. indeterminate – for up to 30 y sampled) but was not recorded for other resprouter species for sites up to 30 y (Table 5.3). There was more evidence of interfire recruitment (both treatments) in soil storage species (e.g., *A. blakelyi*, *J. floribunda*) compared with strongly serotinous canopy storage species (e.g., *M. leuropoma*, *B.*

elegans) (Table 5.3, Figure 5.3). There was also evidence of interfire recruitment in weakly serotinous species (e.g., *P. drummondii*). In all cases where inter-fire recruitment was observed, it occurred earlier and in greater abundance in rehabilitated sites than in natural sites.

Table 5.2: Time (years since last fire/ rehabilitation) to the onset of reproductive maturity (flowering and/ fruiting by >50% of individuals in a population), and age (years) at maximum seed production for selected species from post-mine restored and natural shrublands. n = nonsprouter, r = resprouter, ws = weakly serotinous, ss = strongly serotinous, h=hard-seeded.

	Years to >50 % plants flowering/fruiting		Years to maximum seed production (per plant)	
Species	Rehabilitated	Natural	Rehabilitated	Natural
Acacia blakelyi n, h	3	3	8	8
Petrophile drummondii n, ws	6	10	16	16
Beaufortia elegans n, ss	4	5	8	15
Jacksonia floribunda r, h	7	3*	30	30*
Calothamnus hirsutus r, ws	6	2*	22	22*
Melaleuca leuropoma r, ss	3	1*	30	30*

* Natural resprouters were of unknown age representing overlapping cohorts of individuals recruited after different fires, so that most returned quickly to reproductive stage via regrowth from well-established below-ground parts.

Table 5.3: Time (years since restoration/ last fire) to the onset of, or evidence for, inter-fire recruitment for selected species from post-mine rehabilitated and natural shrubland sites near Eneabba, southwestern Australia. Indeterminate = no evidence of interfire recruitment up to 30 years. n = nonsprouter, r = resprouter, ws = weakly serotinous, ss = strongly serotinous, h=hard-seeded.

	Years to inter-fire recruitment (≥ 5% of individuals as seedlings (≤20 cm))	
Species	Rehabilitated	Natural
Acacia blakelyi n, ns	16	30
Petrophile drummondii n, ws	16	16
Beaufortia elegans n, ss	indeterminate	indeterminate
Jacksonia floribunda r, ns	24	30
Calothamnus hirsutus r, ws	22	indeterminate
Melaleuca leuropoma r, ss	indeterminate	indeterminate

Table 5.4: Mean seed viability and % individuals alive for nonsprouter species a) *Acacia blakelyi*, b) *Petrophile drummondii*, and c) *Beaufortia elegans* in natural and post-mine restored shrublands. Bold values indicate significantly (*t*-test, p<0.05) higher values between matched years since restoration *vs.* last fire, ± value = standard error, n = nonsprouter, r = resprouter, ws = weakly serotinous, ss = strongly serotinous, h=hard-seeded.

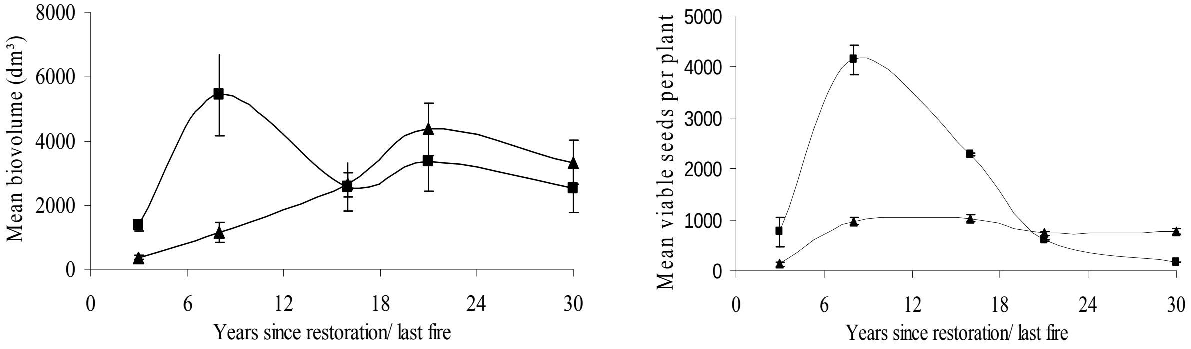
a) *Acacia blakelyi* (n, h)				
Years since fire/ rehabilitation	Seed viability (%)		% individuals alive*	
	Natural	Rehab	Natural	Rehab
3	**67** ± 4	57 ± 5	100 ± 0	100 ± 0
8	42 ± 7	**70** ± 4	**89** ± 2	80 ± 3
16	26 ± 5	22 ± 6	63 ± 3	**85** ± 3
21	19 ± 3	22 ± 5	24 ± 9	**91** ± 2
30	**26** ± 6	15 ± 3	45 ± 5	**71** ± 5
b) *Petrophile drummondii* (n, ws)				
9	**18** ± 3	7 ± 2	**99** ± 1	93 ± 3
16	22 ± 2	20 ± 2	88 ± 3	86 ± 3
22	18 ± 2	19 ± 2	**78** ± 6	73 ± 2
30	19 ± 4	**24** ± 2	63 ± 4	**70** ± 3
c) *Beaufortia elegans* (n, ss)				
5	**99** ± 0	89 ± 2	100 ± 0	100 ± 0
8	**98** ± 0	93 ± 2	**99** ± 0	88 ± 4
15	**99** ± 0	93 ± 1	90 ± 4	90 ± 3
21	**97** ± 1	93 ± 1	75 ± 8	79 ± 6
30	**98** ± 1	94 ± 2	70 ± 9	67 ± 12

* Interfire recruits not included

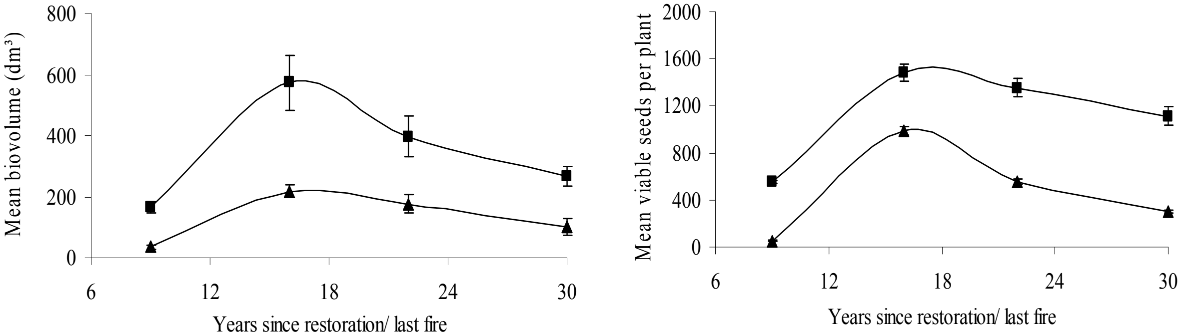
Table 5.5: Mean seed viability and % individuals alive for resprouter species a) *Jacksonia floribunda*, b) *Calothamnus hirsutus*, and c) *Melaleuca leuropoma* in natural and post-mine restored shrubland sites. Bold values indicate significantly (*t*-test, p<0.05) higher values between matched years since restoration *vs*. last fire, ± value = standard error, n = nonsprouter, r = resprouter, ws = weakly serotinous, ss = strongly serotinous, h=hard-seeded.

Years since fire/ rehabilitation	Seed viability (%)		% individuals alive	
	Natural	Rehab	Natural	Rehab
a) *Jacksonia floribunda* (r, h)				
5	-	-	100 ± 0	100 ± 0
10	-	-	100 ± 0	100 ± 0
17	-	-	**100** ± 0	98 ± 0
24	-	-	**98** ± 0	92 ± 2
30	-	-	**97** ± 1	88 ± 2
b) *Calothamnus hirsutus* (r, ws)				
8	**80** ± 4	66 ± 7	100 ± 0	100 ± 0
15	**72** ± 5	54 ± 4	**100** ± 0	98 ± 0
22	65 ± 5	63 ± 6	**100** ± 0	96 ± 1
30	**72** ± 4	48 ± 6	89 ± 2	**94** ± 3
c) *Melaleuca leuropoma* (r, ss)				
4	75 ± 2	**84** ± 3	100 ± 0	100 ± 0
8	**96** ± 1	70 ± 3	**100** ± 0	99 ± 0
15	70 ± 4	73 ± 1	91 ± 1	**96** ± 1
21	73 ± 5	77 ± 4	92 ± 2	**96** ± 1
30	**83** ± 2	78 ± 2	93 ± 1	92 ± 2

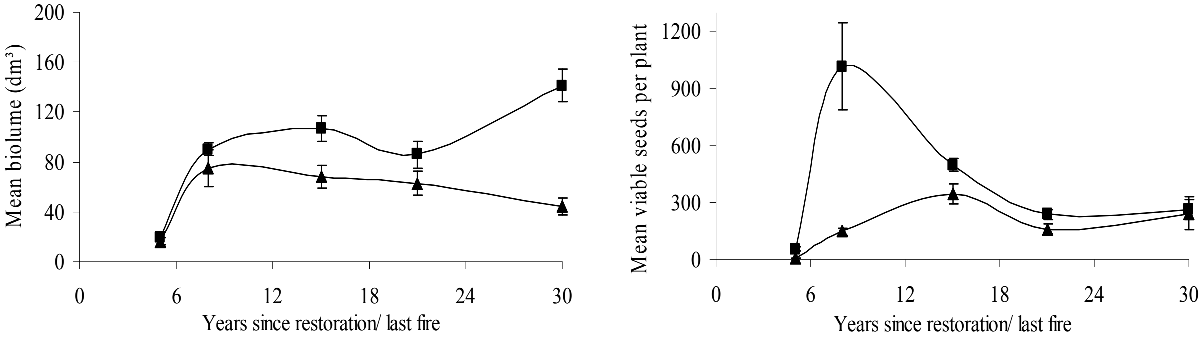
* interfire recruits not included



a) *Acacia blakelyi*

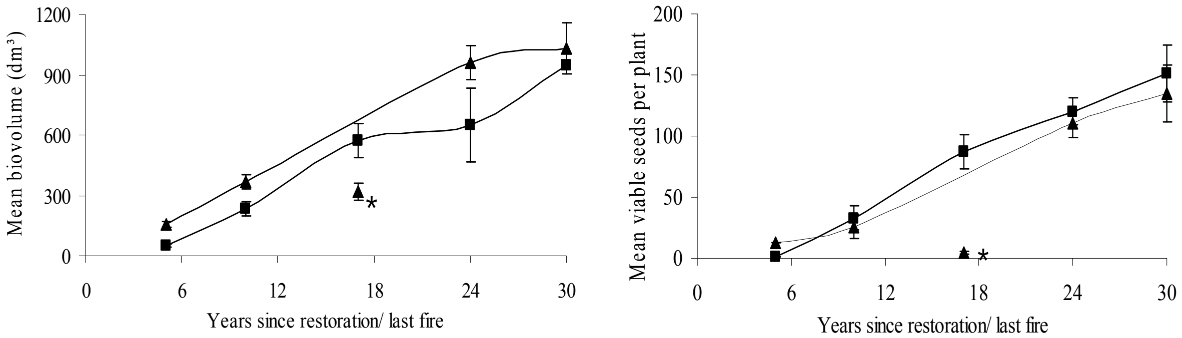


b) *Petrophile drummondii*

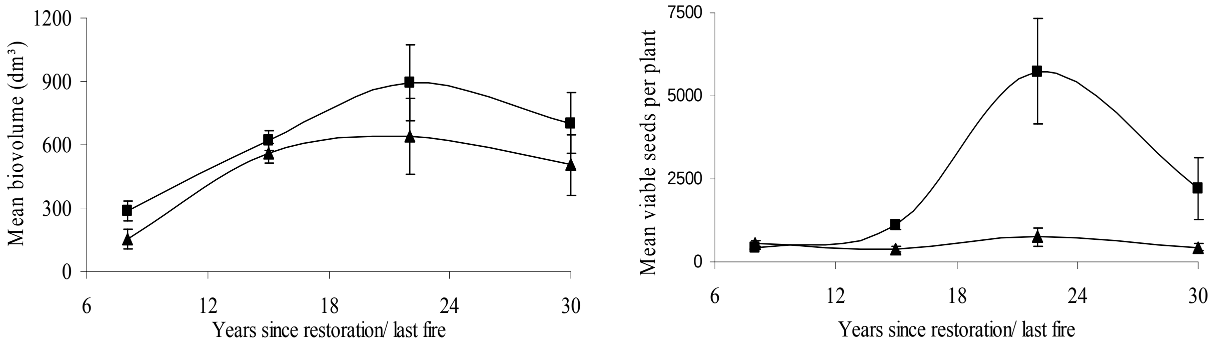


c) *Beaufortia elegans*

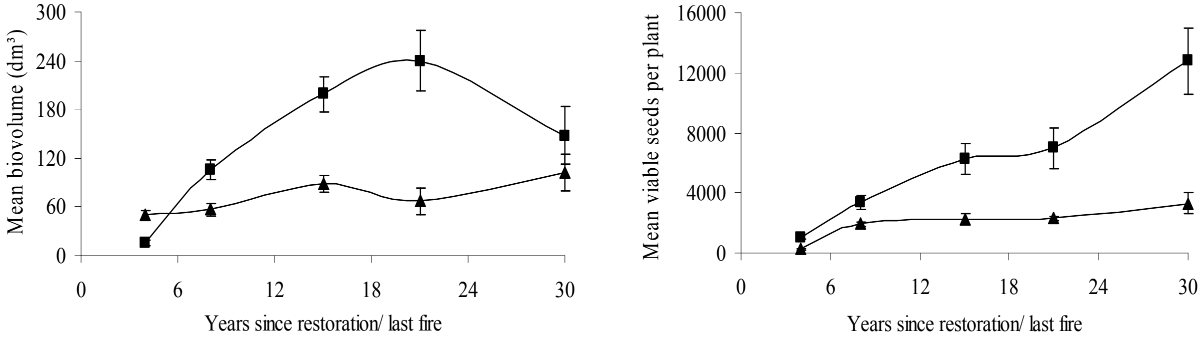
Figure 5.1: Mean biovolume (dm³) and viable seeds per plant for nonsprouter species a) *Acacia blakelyi* (n, h), b) *Petrophile drummondii* (n, ws), and c) *Beaufortia elegans* (n, ss) in post-mine restored and natural shrubland sites. A smooth line curve option in Microsoft Excel® was used to connect the data points on figures. n = nonsprouter, r = resprouter, ws = weakly serotinous, ss = strongly serotinous, h=hard-seeded.



a) *Jacksonia floribunda*

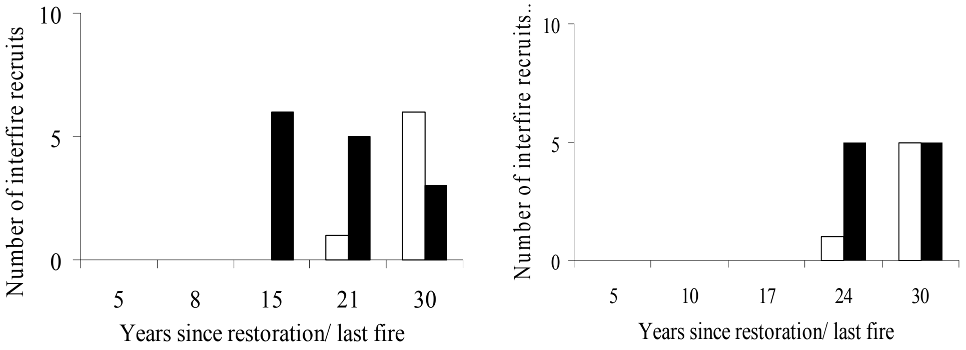


b) *Calothamnus hirsutus*

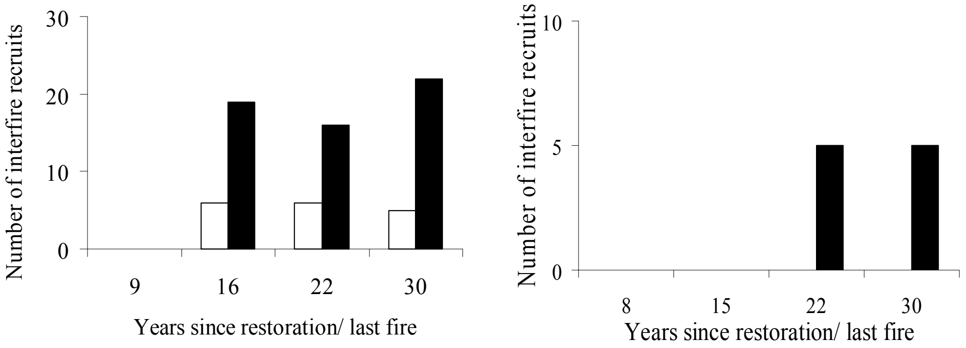


c) *Melaleuca leuropoma*

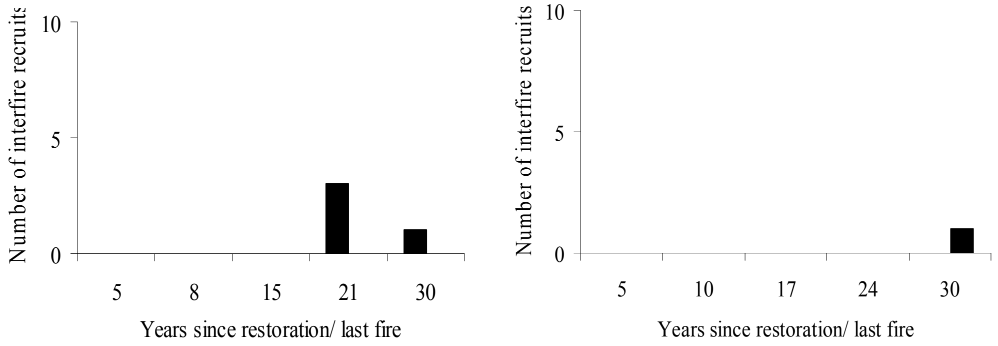
Figure 5.2: Mean biovolume (dm³) and viable seeds per plant for resprouter species a) *Jacksonia floribunda* (r, h), b) *Calothamnus hirsutus* (n, ws), and c) *Melaleuca leuropoma* (n, ss) in natural and post-mine restored shrubland sites. A smooth line curve option in Microsoft Excel ® 2000 was used to connect the data points on figures. * = outlier: preliminary observations over several years revealed something unusual in fruiting rates sampled during this year. n = nonsprouter, r = resprouter, ws = weakly serotinous, ss = strongly serotinous, h=hard-seeded.



a) *Acacia blakelyi*　　b) *Jacksonia floribunda*



c) *Petrophile drummondii*　　d) *Calothamnus hirsutus*



e) *Beaufortia elegans*　　f) *Melaleuca leuropoma*

Figure 5.3: Number of interfire recruits (<20 cm height – out of a total of 50 random individuals assessed per site) of non serotinous species: a) *Acacia blakelyi* (n, h), and b) *Jacksonia floribunda* (r, h); weakly serotinous species: c) *Petrophile drummondii* (n, ws), and d) *Beaufortia elegans* (n, ss); and strongly serotinous species: e) *Calothamnus hirsutus* (n, ws), and f) *Melaleuca leuropoma* (n, ss) in post-mine

rehabilitated (filled) and natural (unfilled) shrubland sites. n = nonsprouter, r = resprouter, ws = weakly serotinous, ss = strongly serotinous, h=hard-seeded.

5.5 Discussion

At matched years since restoration vs. last fire, restored nonsprouter species generally grew larger (1.1 to 4.7 times), produced/stored more viable seeds (1.1 to 10.9 times), and the weakly serotinous *A. blakelyi* displayed evidence of inter-fire recruitment at a much earlier age (rehabilitated 8 y vs. natural 15 y). Nonsprouter species also reached reproductive maturity 1 to 4 years earlier in rehabilitated sites (except *A. blakelyi*: 3 y in both). The overall more rapid growth and higher fecundity in rehabilitated sites may be a result of several factors. First, synthetic fertilizers (Croprich: Nitrogen, Phosphorus, Sulphur, Copper, Zinc – suppliers Summit fertilizers) were added into restored topsoil at a rate of 70–100 kg per ha to fast-track the initial growth process (B. Wynne, Iluka Resources Ltd., personal communication 2008). However, a previous study which examined soil factors in relatively old (8–24 y since restoration) restored stands on the same mine and surrounding natural stands (14–24 y since last fire) only found higher reactive iron levels and lower soil penetrability in rehabilitated sites (Herath et al. in press.). Other soil factors, including nitrate, ammonium, phosphorus, potassium, sulphur, organic carbon, conductivity and pH were consistently low in both rehabilitated and natural sites. However, it is possible that most of the soil nutrients may have already been accumulated by plants in sites rehabilitated ≥8 y ago, and therefore excess nutrients in younger stands may have encouraged faster growth and seed production in the early restoration phase (O'Connell and Grove 1996). Second, Herath et al. (in press.) recorded lower plant densities in rehabilitated sites than in natural analogues (half or less) and this may have reduced competition for resources and promoted rapid shoot growth and seed production (e.g., Bloom et al. 1985; Iwasa and Kubo 1997; Bellingham and Sparrow 2000). Third, while the hard, clayey substrate returned in rehabilitated sites may have impeded root growth in some species, it retains more soil water than do the near-surface sands of the natural analogue sites (Enright and Lamont 1992a) and this may bolster above-ground plant growth (and seed production) for species that can access it (Herath et al. unpub.).

Rehabilitated site resprouters took more than twice the time to reach reproductive maturity than found in natural site resprouters. However, this is because resprouters in natural areas were multi-aged, representing overlapping cohorts of individuals recruited after different fires, which for the Eneabba sandplain region have been at mean fire interval of 13 years over the last 40 years (Miller et al. 2007). Most resprouters in natural vegetation return to reproductive stage quickly after fire (1-2 years) through regrowth from well-established below-ground lignotubers (Gill 1981; Pyke 1983; Keeley and Keeley 1984). This also explains why resprouters *J. floribunda* and *C. hirsutus* both had greater seed production at the youngest natural stand (multi-aged cohorts) vs. youngest restored stand (single-age cohorts), but after 8 y, fecundity for restored individuals was much greater than that found in natural sites (Table 5.5). This means that despite the presence of younger resprouter individuals at matched years since restoration vs. last fire, resprouters in rehabilitated sites had faster growth rates and fecundity than that of secondary resprouters in natural sites. Restored resprouters generally grew larger (1.1 to 3.6 times, except *J. floribunda*), produced/stored more viable seeds (1.1 to 6.9 times), and had earlier inter-fire recruitment (except *M. leuropoma*). This suggests that primary growth rates and seed production in restored resprouters are much faster than for primary resprouters in natural sites, likely due to the same reasons discussed above for restored nonsprouter - excess nutrients associated with fertilizer additions, lower plant densities reducing competition, and the clayey substrate which retains water longer into summer.

There was more evidence of inter-fire recruitment for soil-storage (hard-seeded) species *A. blakelyi* and *J. floribunda*, especially in rehabilitated sites, than for the strongly serotinous species *B. elegans* and *M. leuropoma*. Soil-storage species release their seeds onto the ground surface annually upon ripening so that seeds are appropriately positioned for recruitment if seed coat damage or weathering breaks the enforced dormancy – which is increasingly likely as seed age increases. Inter-fire recruitment was also evident for the weakly serotinous *P. drummondii* and *C. hirsutus* in rehabilitated sites. Since their seeds are only stored within the canopies for a few years, previous seed crops are continuously released to the soil surface once plants have been mature for several years, and some of these will germinate and establish. There was no evidence of inter-fire recruitment for the strongly serotinous, prolific seed producing *B. elegans* (Bell et al. 1987) and *M. leuropoma* (Enright et al. 2007) because they both hold their seeds in the canopy for many years so that little seed is available for germination except immediately after fire. However, inter-fire recruitment in these species is possible when individuals die and drying fruits spontaneously release their seeds (Enright et al. 1996; Witkowski 1991). Nevertheless, only a few *B. elegans* (and no *M. leuropoma*) seedlings were recorded in unburned 22 and 30 y rehabilitated sites.

A. blakelyi grows much faster and produces more seeds in rehabilitated vs. natural sites, but this pattern only holds true for the first 8 years (or 20 years for seed stores), after which seeds accumulated in the soil decay rapidly, and inter-fire recruitment begins after 8 years in rehabilitated sites, dramatically reducing the overall estimates of mean plant size and seed set. The lower levels of seed viability for seed crops on older (>16 years) plants may be a result of a reduction in plant nutrients at older rehabilitated sites as the effects of initial fertilizer additions decline. Plant senescence also naturally occurs early in this species with shorter lifetime (Black 1979), rapid onset of reproductive maturity (3 y), early peak in seed production (8 y), and in interfire recruitment (between 8 and 16 y in rehabilitated sites). Galls were also observed on the stems of individuals once the plants reached about 8 years, possibly affecting their survival due to insect damage. Nevertheless, *A. blakelyi* remains abundant and is a major problem in rehabilitated sites at Eneabba since it creates competition with other native species. Management of this species requires removal before it reaches reproductive maturity and before it contributes seeds into the seedbank (i.e. before 3 y). This may involve manually cutting the main stems at early stages of restoration before the onset of maturity.

Optimum fire intervals

Despite faster growth and seed production/storage rates in rehabilitated (vs. natural) sites, optimum fire age based on maximum seed production levels generally occurred at similar stand age in both natural and rehabilitated treatments, except for the nonsprouter *B. elegans* where optimum seed production was recorded earlier in rehabilitated sites (8 y vs. 15 y). There was no one fire interval optimal for all species studied: nonsprouter species would benefit most from intermediate fire intervals (8–16 y), whereas resprouters would benefit most from longer fire intervals (≥22 y). However, the main criterion for selecting an optimum fire interval was based on the notion that maximum levels of recruitment are most likely at intervals when seed store is also at its maximum. This theory holds true for nonsprouters (e.g., Auld 1987; Lamont and van Leeuwen 1988), but resprouters have an additional reproductive strategy: vegetative regrowth after fires (Gill 1981; Bell et al. 1984).

An increase in fire frequency in natural sites has historically favored resprouter (over nonsprouter) species (Biswell 1974: Zedler et al. 1983; Lloret et al. 1999) since existing individuals can re-establish vigorously (after fire) via well-established below-ground parts. However, strip mining involves complete removal of the former vegetation and so woody resprouters are eliminated from the site (Bellairs and Bell 1993) and must be re-introduced as a new population cohort from seed, in the same way as nonsprouters. In Chapter 4, it was found that post-fire persistence of resprouters in rehabilitated sites among ten lignotuberous shrub species (including the three resprouter species of this study) was significantly lower (mean of 52%) than in natural sites (96%) because restored resprouters were all younger plants with lower regenerative capacity – i.e., correlated with smaller lignotuber size (e.g., Moreno and Oechel 1991; Enright et al. 1998; Cruz et al. 2002). The lower lignotuber recovery rates for restored resprouters highlight the importance of the seedling recruitment strategy and the benefits of higher seed set in resprouters of rehabilitated sites where higher plant mortality due to fire must be offset by seedling recruitment if populations are to persist. If resprouters do not recruit via seeds in rehabilitated sites, they may eventually become locally extinct after single or multiple fires – unless lignotubers become much more resilient with time.

Frequent fires typical of natural shrublands in the Eneabba area – mean interval of 13 y over last 40 y (Miller et al. 2007) – play an important role in maintaining plant species diversity. However, this fire regime may not be appropriate for rehabilitated sites, at least in relation to the first occurrence of fire following mine-site restoration, since it disadvantages resprouter species due to lower seed stores (than at ≥22 y) and higher mortality of existing plants at younger rehabilitated sites (Chapter 4). However, restored nonsprouter species may benefit from the surrounding natural fire regime since they accumulate large seed stores by ages 8–16 y. Thus, the selection of an appropriate fire interval for rehabilitated sites may require a compromise between possible deleterious effects on resprouters and nonsprouters. Nevertheless, resprouter species may be given priority due to the difficulties associated with restoring these recalcitrant species in rehabilitated sites (Bellairs and Bell 1993; Koch 2007a) and the fact that the three studied nonsprouter species were still producing sufficient quantities of seeds in older rehabilitated stands. Therefore, it may be most parsimonious to use a longer than average (20–30 y) fire interval on the introduction of the first-fire following restoration of shrublands at Eneabba, in order to promote higher persistence of resprouters via both lignotuber regrowth and seedling recruitment. Since senescence for nonsprouters measured here was only around 30% by 30 y for rehabilitated sites, the majority of their seed stores may likely remain sufficient for recruitment after fire (Davis et al. 1988; Zedler and Zammit 1989; Keeley 1992). Subsequent fires may be feasible at shorter intervals as populations of resprouter species increase in age, and size and viability of their lignotubers, eventually converging on the regional mean interval of 13 y. Since the sites experimentally treated with fire in this study are the only restoration sites burned in this restoration area, it is recommended that they continue to be monitored, and treated with fire again in around 13 years time, so that the best informed plans for reintroduction of fire after completion of mining can be established.

CHAPTER 6.0

Synthesis and Recommendations

This chapter reviews the ecological relationships identified in the preceding chapters and their relevance to management. Limitations of the study and topics requiring further research are also discussed.

6.1 Pre-fire ecology and the management of restored shrublands

Among the natural sites, presence-absence ordinations revealed that the local swales and dunes were floristically most similar to those of the adjacent post-mined shrublands. Although this similarity was only within the order of 30%, earlier studies in the region have also shown low similarity among samples between natural shrubland communities, even for similar substrate types (Lamont 1976; Hnatiuk and Hopkins 1981; Griffin et al. 1983). This highlights the floristic complexity of the region and thus the difficulties associated with restoring composition of areas with such high species richness and turnover. Similar mining restoration (of composition) difficulties appear in species-rich fynbos shrublands of South Africa (Holmes and Richardson 1999; Holmes 2001), in coal mine restoration (up to 36 years post-mining) of hardwood forests of eastern USA (Holl and Cairns 1994; Holl 2002), and in previously mined (up to 26 years) south-eastern eucalypt (Ross et al. 2004) and south-western eucalypt (Norman et al. 2006; Koch 2007b) forests. The studies by Alcoa in south-western eucalypt forests recorded a mean of 34% floristic similarity between rehabilitated sites (pre 1988) and intact forests, using Sorensen's similarity measure (the same method as in my study), and their restoration efforts are considered among the most successful in the world (particularly over the last decade: Bell and Hobbs 2007; Gardner and Bell 2007). My study revealed a similar degree of floristic similarity between natural and the restored shrublands after mineral-sand mining near Eneabba. The fact that no perennial weeds were recorded in this study is also encouraging, as other studies (although including annual weeds which were rarely found after fire in our sites, personal observation) have found that exotic weeds are a major problem in rehabilitated minesites and require management (Partridge 1992; Grant and Koch 1997; Grant and Loneragan 1999; Ross et al. 2004).

The return of species richness was an important milestone. Mattiske (2004), who was commissioned to monitor the restored vegetation against completion targets, found that over 90% of the Eneabba rehabilitated sites (including our study sites) satisfy the minimum requirement of 6 species per m² or 70 species per 10 ha. Although the Mattiske (2004) report included annuals in their survey, perennial plant species richness in my study sites was well above the targets with more than 70 species recorded in just 0.16 ha of restored shrubland. However, it seems likely that some of the high species richness in restored shrublands (about 140 (except R8) *vs.* about 100 species in natural sites) was achieved by seed collecting over broader areas than the adjacent swales and dunes. For instance, some species that are usually restricted to specific substrate types, for example, *Labichea cassioides* on limestone and *Hakea stenocarpa* on laterite, were recorded in some rehabilitated sites. To improve floristic similarity to that of surrounding swales/dunes, I recommend that collections of seeds, mulch and topsoil should be restricted to local provenance areas (Lubke and Avis 1999; Krauss and Koch 2004; McKay et al. 1995; Koch 2007a). The collections should also occur in an equivalent spatial extent to that being rehabilitated. For example, one hectare of topsoil collected should be applied to only one hectare of rehabilitated ground. This would improve floristic similarity and species richness levels closer to that of the specific natural analogue. Tailings should also be returned in such a way as to mimic the range of substrates (in this case dune and swale) present pre-mining and to create a variety of habitat-types, since rehabilitated sites were recorded to be more similar (floristically) to each other (about 45%) than to the surrounding natural areas.

Fire-resprouter densities were under-represented in rehabilitated sites – nearly half (of high dune, limestone) to less than half (swale, laterite) of natural sites. The most common species were mostly nonsprouters in rehabilitated sites whereas they were mostly resprouters in natural sites. Similarly, studies in restored bauxite-mined eucalypt forests of SW Australia have also recorded lower densities of resprouters in restored *versus* natural sites (Grant and Loneragan 1999; Smith 2001; Koch 2007a;

Koch and Hobbs 2007). The relative imbalance of resprouters to nonsprouters reflects the difficulties associated with restoring resprouters. For instance, regenerative organs (e.g. lignotubers, rhizomes, bulbs) of resprouter species are often destroyed or die during the mining process (Bellairs and Bell 1993; Fox et al. 1996), and therefore they can only re-establish via seedlings after rehabilitation. However, the additional problem is that resprouters generally produce fewer seeds than nonsprouter species (Bell et al. 1993; Lamont and Wiens 2003; Koch 2007a) so their seed densities are much lower in the added seed mix, mulch and topsoil. Spreading from surrounding natural areas (Vigilante 1997) is likely to be a slow process and it may take centuries to develop sufficient densities, therefore further manipulation is necessary. Transplanting mature resprouter individuals from natural areas (ahead of mining fronts) into rehabilitation sites may be an option. For example, the transplant technology now exists for grasstrees (Lamont et al. 2004) that are conspicuously absent from rehabilitated sites. Further research regarding tissue culture/artificial planting schemes may also help restore resprouters to pre-mining levels.

Although no perennial weeds were recorded in this study, two native species were observed to be displaying ‘invasive’ properties in rehabilitated sites: the fire-killed *Acacia blakelyi*, and to a lesser extent, the fire-tolerant *Melaleuca leuropoma*. In all rehabilitated sites, *A. blakelyi* (7–36% cover) and *M. leuropoma* (9–30%) were among the top three dominant species, whereas in the natural sites, *A. blakelyi* (0–0.3%) was infrequent or absent, and *M. leuropoma* (2–15%) was abundant only at the limestone and low dune sites. Following topsoil replacement, *A. blakelyi* was historically mixed with cover crop seeds to provide stabilization of tailings and to produce a quick scattered cover of native shrubs to tide over the period between cover crop decay and full native regeneration (Black 1979). However, due to its pioneer-phase properties – rapid growth rates, prolific reproductive capacity, nitrogen fixing ability (legume) in nitrogen deficient rehabilitated soils - it survives, reproduces and remains dominant in older rehabilitated stands, creating competition with other native species. For example, *A. blakelyi* at R8 had 36% ground-cover that likely contributed (in addition to the longer period of stockpiled soil at this site) to the sites lower species richness, H’, D’ and J’ diversity compared with other rehabilitated and natural sites. Similarly, in dune-mined areas of Richards Bay, South Africa, *Acacia karoo* (another native legume) dominated rehabilitated areas and its competitive abilities retarded the influx of other native species (Camp 1990; Camp and Weisser 1991; Lubke and Avis 1999). The increase in abundance of legumes on rehabilitated mine-sites has also been reported elsewhere (Koch and Davies 1993; Chambers et al. 1994; Chapman et al. 1996) likely due to the fact that the returned soil was deficient in nitrogen, making legumes much more competitive than other non-nitrogen-fixing species.

The practice of growing *A. blakelyi* as a cover crop has ceased in the restoration programme near Eneabba. However, it still remains a problem in newly rehabilitated pits due to its competitive properties. It is thought that the soil scarification process encourages the germination of this species (Black 1979). To manage its abundance, research is being conducted on brushing herbicides on the crown tops of *A. blakelyi* eventually resulting in their death (Phil Scott 2007, Iluka Resources Ltd., personal communication). However, this process assumes that *A. blakelyi* grows much faster and taller than do other native species and no herbicides percolate or contaminate the soils/other native plants. In addition, if brushing herbicides is delayed to 3 or more years (since restoration) for crown tops to be sufficiently higher than other native species, then their seeds would already be accumulated rapidly in the soil seedbank, as initial fecundity rates were observed to be high during the first few years of seed production (e.g., mean of 758 seeds per plant at 3 years leading to a soil seedbank accumulation of 4143 per plant at 8 years – Chapter 5). As a result, interfire recruitment of *A. blakelyi* seedlings would occur (recorded to begin around 16 years – Chapter 5), or after fire, a flush of seedlings may appear and dominate the nitrogen deficient rehabilitated soils – Chapter 3. A more reliable but labour intensive method may involve manually clipping or chain-sawing the main stems of the plant at early stages of restoration before the onset of seed production (2 to 3 years – Chapter 5). Thinning is also suggested as a protocol for other dominant legume species that retard other native species on minesites (e.g., Camp 1990; Camp and Weisser 1991; Chambers et al. 1994; Chapman et al. 1996; Lubke and Avis 1999). Their remains may also act as mulch material and aid in soil stabilization if thinned before seed production.

The highly fecund *Melaleuca leuropoma* may also need to be controlled in rehabilitated sites. It has been found to possess an order of magnitude more viable seeds per unit area than other serotinous species in all substrate types (Enright et al. 2007), and as a result, it dominates rehabilitated sites after the mulching process releases its seeds through desiccation. Therefore, it may be necessary to avoid mulching in thickets containing these species. However, since this resprouter comprises multiple stems arising from the lignotuber (an average of 50 cm lignotuber circumference in rehabilitated sites – Table 4.2), it assists in soil stabilization and buffers against topsoil erosion (personal observation) that is a major problem (due to strong winds at Eneabba: Bell et al. 1986) in new rehabilitated pits. In addition, *M. leuropoma* was observed to have some beneficial effects on other smaller shrub species (e.g., *Leucopogon gracillimus*, *Astroloma pallidum* etc.) that reside beneath the *M. leuropoma* crowns in conditions of lower drought stress then open/exposed areas (personal observation). For these reasons, it may be decided to leave *M. leuropoma* populations until future research can show that other native species can provide sufficient ground cover and assist in soil stabilization in rehabilitated sites.

The depth of sand over silt-clay tailings ranged 0–20 cm (topsoil) in rehabilitated sites with large areas exposed to the tailings at the surface whereas natural sites had a minimum of 20–50 cm of sand over silt-clay in swale areas to a maximum of 8–10 m of sand over silt and clay in high dunal areas (personal observation). As a result, soil penetrability levels of rehabilitated sites (recorded via penetrometer) were less than half those of natural sites. The lack of sand (of topsoil and loose subsoil) over silt-clay tailings was most likely responsible for much of the vegetation restoration problems apparent near Eneabba (also after fire – section 6.2). For example, the lower restored plant densities (about half those of the high dune and limestone sites, and about a quarter those of the swale and laterite) may have been related to the addition of seeds/plantings at lower than required densities (not documented) but more likely due to the higher mortality rates associated with the adverse substrate-type (lack of sand over poorly penetrable silt-clay), limiting roots from accessing groundwater (Enright and Lamont 1992a), intensifying the effects of summer drought (i.e., extending its duration), and over winter, occasionally creating problems with waterlogged soils. For instance, Enright and Lamont (1992a) found that despite high germination success of *Banksia* seedlings in rehabilitated sites at the same mine, high impedance of the rehabilitated soils (compared with dune soils) explained their poor root development and dependence of seedlings on soil water stored near the surface, which led to higher mortality levels. Similarly, after simulating mining conditions, Holmes (2001) reported lower densities and survival of fynbos shrublands (Cape floristic kingdom, South Africa) in subsoil *versus* topsoiled plots. These studies indicate that restoration of a soil profile with sufficiently deep rooting medium and low soil compaction is a critical stage in the restoration process (Grant 2006), and must be carefully managed.

The reconstruction of soil profiles for post-mined lands near Eneabba involves double-stripping topsoil from natural areas ahead of the mining front – the first cut being the top 5 cm of soil, the second cut being approximately 5–20 cm of soil (stripped to the soil colour change), then stockpiled or directly returned in the correct order to sites ready for rehabilitation (Bellairs 1991; Jefferies et al. 1991). The rest of the soil profile (including all of the subsoil) is removed (usually down to tens of metres deep), a small percentage is extracted for minerals, and the mixed profile is then returned and topsoil spread (Cameron Payne 2008, Iluka Resources Ltd., personal communication). As a result of mixing the subsurface soils with deep soils containing a higher fraction of minerals, minute iron-bearing particles combine with sands and clay to form silt-clay agglomerated structures, uncharacteristic of natural sandy subsoils. This compact substrate is regularly exposed at the ground surface since the strong winds at Eneabba (Bell et al. 1986) and rainfall runoff result in erosion/accumulation of topsoil deposits in contoured depressions, for example, at sites R15 and R16 (personal observation). Despite considerable efforts to retain topsoil evenly on rehabilitated ground by providing mulch and cover crops to aid in the stabilization process, the strong winds and rainfall runoff still tend to force topsoil into contoured depressions (personal observation). As a result, many plants were observed on accumulated topsoil deposits and fewer on the exposed silt-clay surfaces since roots of many plant species cannot penetrate this layer. Therefore, it is recommended that some portion of the subsoil layer (down to about 1 m) should be salvaged to buffer wind and water erosion problems of sands at the ground surface. Subsoil layers (up to 1 or 2 m) are salvaged in SW jarrah forest rehabilitation sites of Australia (Lawrie 1984; Nichols et al. 1985) since only the bauxite layer ranging over 2–8 m deep is mined. After landscaping, pre-ripping occurs and then overburden is laid over sites to be rehabilitated, and ripped again before topsoil is spread (Koch 2007b) – this technique has proved to provide sufficient rooting medium for even taller growing tree species (e.g., *Eucalyptus marginata*) with deep sinker roots (Szota et al. 2007). However, at Eneabba, the mining material starts at the ground surface to tens of metres deep, so decisions on whether to salvage subsoil down to about 1 m deep or so may depend on how valuable or how concentrated the minerals are in this layer. In addition, subsoil down to 1 m deep is not always available in natural swale areas whereby sandy soil horizons may only extend down to 20–50 cm. In this case, efficient deep ripping techniques may facilitate the access of sinker/taproots into deeper moist soil layers. For example, in America, Europe, and Australia, deep ripping has been observed to improve plant vitality, including survival and height for a range of tree species in differing soil types (Shea et al. 1979; Tacey 1979; Varelides and Kritikos 1995; Ashby 1997; Nadeau and Pluth 1997; Lacey et al. 2001; Szota et al. 2007). Ripping the subsoil with a conventional chisel-tine does occur at the rehabilitated minesites near Eneabba; however, the ripping methods in the 1980’s and 1990’s have not been well documented. Currently, ripping operations are restricted to the drier periods of the year, i.e., summer to autumn, to prevent a ‘cementing effect’ due to moist, wet conditions on the soil during winter (Mengler et al. 2006). However, more significant advances in ripping mechanisms or methods may be needed. For example, other studies in restored southwest eucalypt forests have found that the attachment of a winged-tine (1.8 m wingspan) to the shank of a conventional-chisel tine relieves mine related soil compaction, which greatly improves tillage by lifting and tilling the soil across a broad front (Croton and Ainsworth 2007). The use of chemical agents such as gypsum to improve overall soil structure has been suggested for use in conjunction with ripping (Szota et al. 2007). Ripping depth was also found to affect mining related compaction (Croton and Watson 1987).

Although it is emphasized (above) that the compact substrate type adversely affected plant establishment and survival, it was observed that for those species that can penetrate the deeper soil layers (i.e., species. or individuals which developed stronger root systems), they grow larger in rehabilitated *versus* natural sites. For instance, three common nonsprouter species were reported to be up to five times larger and produced more than 11 times more seeds than those of matched age in natural sites – Chapter 5. Three common resprouter species were also larger (up to four times) and produced seven times more seeds than those in natural sites. Therefore, it can be concluded that the compact substrate type was selective in what species establish and remain prolific (due to less competition associated with lower densities) in rehabilitated sites, thus influencing the overall floristic composition and vegetation structure that exists there. In addition, it may be possible that the heterogeneous nature of the soil structure of rehabilitated sites provides opportunities for some plant individuals to access the deeper soil reserves and grow comparatively large.

6.2 Post-fire ecology and the management of restored shrublands

The ability of a plant community to re-establish itself after environmental disturbances is one of the key ecological functions used as an indicator of ecosystem resilience (Fox et al. 1996; Smith 2001; Grant and Loneragan 1999; Ross et al. 2004). In the restored shrublands after mineral-sand mining at Eneabba, a self-sustaining plant community, that is, one that can maintain its ecological properties in the face of environmental perturbations, such as fire, has not yet developed. Experimental burning of shrubland vegetation restored 8–24 years ago had strong negative effects on perennial plant species diversity. Species richness fell by 22–41% whereas it increased by 4–29% in natural sites. Restored resprouters were severely affected by fires – only 42–66% of species recovered (as seedlings or resprouts) in rehabilitated sites whereas nearly all species recovered in natural sites. For ten common lignotuberous resprouter shrub species (described in Chapter 4), fire led to the death of far more individuals in rehabilitated sites (mean of 52% death) than occurred in nearby natural sites (4%). The lower resprouter persistence after fire in rehabilitated sites was a result of smaller lignotubers containing fewer buds that are able to sprout following fire (Rundel et al. 1987; Auld 1990; Moreno and Oechel 1991; Enright et al. 1998). For example, the ten common resprouter shrub species (noted above) also had smaller-sized lignotuber circumferences in rehabilitated sites – overall 19% smaller than in natural sites. Five of the ten species had lignotubers about a third or less their size in natural sites, three had lignotubers less than half their size in natural sites, and only one species (*H. incrassata*) had larger lignotubers in rehabilitated sites and hence high persistence levels after fire. In addition, for crown volume equivalent to the mean at rehabilitated sites, overall lignotuber circumference of all species was 50% smaller in rehabilitated sites than in natural sites, indicating differences in growth forms between these areas.

Two key factors were identified as affecting the size or development of lignotubers in rehabilitated sites. First, the initial age-structure of the natural and restored resprouter populations was different – restored resprouters commenced as seedlings at the time of rehabilitation since the mining process destroyed all former regenerative organs (Bellairs and Bell 1993; Fox et al. 1996), therefore, restored resprouters were younger (a single cohort of no more than 24 years) than those multi-aged cohorts in natural sites that may have passed through many fire cycles – recurring with a mean interval of 13 years in the study area over the last 40 years (Miller et al. 2007). As a result, restored resprouters may be, individually, more fire-sensitive since they have not previously experienced fires and may not have had sufficient time for many individuals to develop a large enough lignotuber to attain fire-tolerance (Bradstock and Myerscough 1988; Lamont and van Leeuwen 1988; Auld 1990). Second, the growing conditions on the reconstituted mine site substrates may not be conducive to lignotuber development since the lower recorded soil penetrability levels in rehabilitated sites may have restricted rootstock development (Enright and Lamont 1992a). The lack of sandy topsoil and loose subsoil (characteristic of natural areas) and replacement with a mixed silt-clay layer near the ground surface would have likely retarded lignotuber development in the compact rehabilitated soils. However, despite the lower below-ground biomass recorded in resprouters on rehabilitated sites, above-ground crown size for three common resprouter species (also assessed for lignotuber size as described above) was observed to be up to 3.6 times larger and up to 6.9 times more fecund for matched years since restoration *versus* last fire (Chapter 5). Similarly, McClaren (2005) at the same mine recorded up to 10 times more seed production in *Eremaea beaufortioides*, *Leptospermum spinescens* and *Hakea incrassata* (also assessed for post-fire persistence and lignotuber size above) at R24 (rehabilitated) *versus* the swale and high dune sites. This indicates that irrespective of plant age and lignotuber size, growth and fecundity rates were much higher in restored resprouters, and that a tradeoff may exist in restored resprouters whereby investment in above-ground growth and fecundity may have been at the expense of fire-tolerant features such as the lignotuber. The lower restored plant densities may also facilitate this tradeoff as a result of reduced competition for resources that may have promoted rapid shoot growth rather than lignotuber development (Bloom et al. 1985; Iwasa and Kubo 1997; Bellingham and Sparrow 2000).

Nonsprouter species were also disadvantaged two years after fires as more species were lost in rehabilitated sites (50–69% species died) than natural sites (15–33%). Although one would expect seedling recruitment among nonsprouters after fires to be relatively high in rehabilitated sites based on their higher (pre-fire) recorded fecundity rates (up to 11 times greater for three nonsprouter species at matched years since restoration *vs.* last fire – Chapter 5), it appears that the poor development of seedling roots on less penetrable or impenetrable (in areas directly exposed to silt-clay at the surface) soil surfaces that limited post-fire seedling establishment and survival in rehabilitated soils (Enright and Lamont 1992a). For example, seedling mortality over the first summer after fire was more than twice that of natural sites and the percentage of seedling species surviving the first summer was only half that of natural sites. It is also likely that the higher heat-holding capacity of clayey soils (especially at R15) created drought stress conditions over summer and affected seedling mortality rates. At R15, small patches were observed to be waterlogged during winter for very brief periods, and subsequently baked over summer, creating inhospitable conditions for seedlings.

The results suggest that the most important factor limiting the development of a self-sustaining plant community in the restored shrublands is the lack of a suitable rooting medium, limiting development of lignotubers among resprouters and post-fire seedling survival/recruitment among nonsprouters (and some resprouter seedlings). As a result, reconstruction of a ‘natural-type’ soil profile is probably the most critical stage in the restoration process near Eneabba, since fires were observed to exacerbate the negative effects of the compacted rehabilitated soils on vegetation resilience properties. Despite considerable efforts and investment made to restore plant diversity after mining, including topsoiling, seeding, infill planting, fertilizing, mulching etc., it seems that further management after fires is required to simulate natural recovery patterns (see below). A better and more feasible approach may involve carefully managing the replacement of a typical natural soil profile in the first place that should facilitate the development of a more resilient plant community. Recommendations are given to restoring natural sandy soil profiles in section 6.1, advising that some portion of loose sandy subsoil (down to 1 m deep if possible) should also be stripped and returned as a third cut to compensate for losses of topsoil (via wind and water erosion) at the ground surface. In that way, there is a compatible rooting medium for most plants to establish, and it would provide better access to the greater volumes of moist soil in the deeper layers, which is critical for most plants during extended periods of drought stress.

PCoA ordinations based on pre and post-fire composition showed that fire shifted the species composition of rehabilitated sites relatively further away from that of natural sites (Figure 3.1) – a result of the loss of many common species (particularly resprouters) and the emergence or immigration of species as seeds but not yet present as plants until fire triggered their germination. Without further human intervention, the existing restored shrubland communities may keep following different successional trajectories from that of their natural analogues (e.g., Adamson and Fox 1982; Friedel et al. 1990). Therefore, to simulate the recovery patterns of natural sites, it is recommended to do more in-fill planting of recalcitrant resprouters, and to a lesser extent, nonsprouters typical of the local swale-dune system after fire. If seeds are also added, they need to be placed on a suitable surface (topsoiled area) for them to establish as seedlings and develop into adults (Enright and Lamont 1989). Provided that recruitment opportunities would not be severely reduced, seeds could be harvested from plants within rehabilitated areas as these are often of higher fecundity than those of the same species from surrounding natural areas (McClaren 2005). It may also be possible to transplant resprouters into rehabilitated land from areas ahead of a mining front (Lamont et al. 2004), particularly for species of known low fecundity. This would create a diversity of resprouter ages and increase the likelihood of population recovery after fire.

Although the studied rehabilitated ages (8, 15, 24 years) were not replicated in order to identify optimum fire-age return intervals for restored shrublands at Eneabba, there is some indication that burning older rehabilitated sites may result in better species persistence after fires (supporting Smith 2001; Ross et al 2004; Koch 2007a). For instance, the oldest rehabilitated site (R24) had a 22% reduction in species richness after fire whereas the youngest ones, R15 and R8, reduced by much more – 38 and 41% respectively (Chapter 3). Individuals of eight of ten common lignotuberous shrub species (described in Chapter 4) also showed better persistence after fires in older rehabilitated stands (R15 and R24), likely due to an age-related accumulation of lignotuberous bud banks that stored more buds available to sprout after fire (Bradstock and Myerscough 1988). Three of these studied species also had maximum seed stores at older ages (≥22 years since restoration) indicating that recruitment via seedlings is most likely at older ages (Chapter 5). In contrast optimum seed reserves for three restored nonsprouter species occurred earlier at 8–16 years. Nevertheless, resprouter species should be given priority due to the difficulties associated with restoring these recalcitrant species (Bellairs and Bell 1993; Koch 2007a) and the fact that the three studied nonsprouter species were still producing sufficient quantities of seeds in older rehabilitated stands. Other studies on post-mine vegetation have reported that fires could be introduced at a much earlier age than that for this study (≤24 years) – equal to or greater than 10 years for restored south-eastern eucalypt forests (Ross et al. 2004) and 11-13 years for restored south-western eucalypt forests (Grant and Loneragan 1999). These studies have also found an increase in post-fire diversity after burning their rehabilitated sites (≤10 years) and Grant and Loneragan 1999 found that 80% of species persist after fire, unlike the results near Eneabba (studied up to 24 years). These differences could be related to differing replacement depths of topsoil and loose subsoil (affecting development of root systems), time to reproductive maturity of lignotubers and seedbanks, vegetation resilience properties (i.e., forests *vs.* shrublands) to fires, and/or rainfall patterns, with greater impacts of summer drought on resprouter recovery and seedling survival (post-fire) apparent at Eneabba.

Delaying/suppressing fires beyond what naturally occurs in surrounding shrubland areas (about 13 years: Miller et al. 2007) may be beneficial for recalcitrant resprouter species, but the dilemma for managers is that such long fire intervals may threaten the survival of some nonsprouting species that senesce sooner than this, recruit poorly interfire and have limited seed storage. For example, at 24 years since restoration, a few adult nonsprouter species (e.g., *Hakea trifurcata*) were observed senescing or dead with interfere recruits yet to produce seeds, but after fire they were eventually eliminated or poorly represented due to the fire-induced death of all plants, death of seedlings and selective kangaroo (*Macropus fuliginosus*) herbivory (Koch et al. 2004; Parsons et al. 2006). Such species longevity issues and the impacts of kangaroo herbivory after fire need to be identified, since their management might require further (targeted) seed additions and/or fencing from herbivores and placing small mesh-like bags around palatable species (Koch 2007a). Longer fire intervals may also increase the risk that weedy species already present at the site, such as *A. blakelyi*, will swamp the new generation with their seeds (accumulated in the soil seedbank) and further reduce species diversity. For example, *A. blakelyi* was much more abundant after burning the older 24 year rehabilitated site, but this may have been related to high pre-fire densities, since it was used as a cover crop at this site during that time of rehabilitation – 1980's (Black 1979). In addition, fire intensity was greatest at this site presumably stimulating greater germination (Auld and O'Connell 1991; Lamont et al. 1991) of this legume species. In that case, management needs to recognise weedy species at the outset and reduce or eliminate their representation earlier on, possibly by chainsawing the main stems before the onset of seed production, recorded to be at 3 years since restoration.

Since natural fires tend to be of high intensity/severity and are difficult to suppress (i.e., by fencing/creating lengthy firebreaks) from penetrating into rehabilitated areas, prescribing a low-intensity or patchy fire (based on weather conditions and moisture levels in fuel loads) may be an alternative option that may be less harmful to the lignotubers of resprouter individuals (Noble 1984), and thus buffer sites against rapid decline in resprouter numbers and favour a reduction in densities of the legume *A. blakeyi* (Grant et al. 1997). If this technique is employed then any future natural fires that diffuse into rehabilitated areas would also be of reduced intensity, since fuel loads are reduced from relatively recent prescribed fires. However, the low intensity fires may limit germination of seeding species as stimulation of seed release from canopy stores and germination from soil stores is a function of fire-temperature in many cases (Auld and O'Connell 1991; Lamont et al. 1991). The decision on fire dates also needs to consider the previous year's rainfall patterns that may influence survival rates post-fire (Croft et al. 2008). For instance, after a drought cycle, lignotubers are less likely to regenerate than if burnt in a period (or cycle) of relatively high rainfall.

6.3 Limitations of the study and areas requiring future research

It was not within the scope of my study to explicitly evaluate mining restoration procedures as sufficient documentation of these procedures was not available for particular sites (e.g., species and input of seed mixtures, origins and duration of topsoil stockpiling, ripping depths), partly due to the long history of transfers in mining companies that operated at Eneabba. The limited information on restoration procedures relied on a few previously published studies, annual reports, and communication among Iluka Resources Ltd. Environmental Department staff. As a result, there was potential for variation in rehabitation results since each site may have represented a unique combination of rehabilitation procedures and weather conditions that varied following different mining and fire dates. Many other mining rehabilitation studies have also acknowledged the fact that mining restoration procedures and weather conditions (including post-fire weather conditions on mined sites) would rarely be the same over the mine-life period (Fox et al. 1996; Ross et al. 2004; Koch 2007a; Koch 2007b). Significant advances and more sophisticated rehabilitation techniques have evolved since Iluka Resources Ltd. acquired the Eneabba mining operations in 1999. There has also been more attention to detail in the recording of restoration operations in annual reports.

Despite the implementation of eight experimental fires over an extensive area, the study also lacked true replication (Hurlbert 1984) or failed to cover the full range of variation at the spatial and temporal scales. The emphasis instead was on intensive quantification of apparently representative sites vegetation-wise before and after disturbance. However, this made it impossible to assign the restored sites to a single most closely related natural vegetation type, except in a general way, or to identify the optimum fire interval, except for a (representative) selection of more intensively studied species.

Further research needs to be conducted on improving mine-related compaction of rehabilitated soils, to improve development of an adequate rooting medium for both nonsprouters and resprouters, and to ensure species persistence before and after fires. Efficient ripping machinery and procedures may be required to relieve soil compaction. Research needs to be conducted on increasing the densities of recalcitrant resprouters on post-mined lands, including tissue culture/artificial planting schemes, and assessing the viability of transplanting resprouters from natural areas ahead of mining fronts. Recalcitrant species may be identified via previous studies or by evaluating the rates of seedling establishment from seeds broadcast onto rehabilitated soils. Seeding rates, tubestock planting rates, and associated establishment and mortality rates of species introduced to rehabilitated areas need to be evaluated, so that the efforts (and investment) on species restoration can be evaluated as viable or not, and further research may be required to devise a more cost-efficient method. If manually harvesting *A. blakelyi* populations is not possible due to human constraints, then research would be required on how to minimize their populations, such as by biological control.

Research on ways to create more resilient restored shrubland communities is required as the mined lands revert to the public estate and fires return. Although some inferences are made on possible fire-return intervals which may be less harmful to species diversity on restored shrublands, its relationship needs to be studied in more detail, and other fire regime characteristics such as intensity and season of burning needs to be studied as existing research has shown that they are important in determining the response of post-mine plant communities (Hobbs and Atkins 1990; Grant and Loneragan 1999; 2001; Ross et al. 2004). Longer term monitoring of post-fire responses beyond the first two years is also needed before suitable fire regimes can be identified.

6.4 Conclusions

The series of chapters presented in this thesis reveal that “complete restoration” of the pre-disturbed community has not yet been achieved in the shrubland communities restored after mineral-sand mining at Eneabba. Although it is commendable that some components of pre-fire vegetation diversity (e.g., species richness), composition, structure, and some functional attributes (e.g., nutrient levels) of natural analogue ecosystems have been restored to a marked extent, the returned ecosystem attributes were generally more similar to each other than to any other surrounding natural analogue. Furthermore, the fire responses of the restored vegetation were less resilient than that of the surrounding natural vegetation. As a result, my study indicated that the returned vegetation (to date) could be classified as "rehabilitated" or "partially restored" but not "completely restored" (Cairns 1993; EPA 1995; Lubke and Avis 1999; McDonald 2000) since complete restoration of the original/surrounding plant diversity, composition, structure, function, and their fire resilience capabilities have not yet been achieved. It may not be possible/realistic to achieve "complete restoration" since mining is such a disruptive disturbance type (Fox et al. 1996; Fox 1988; Ross et al. 2004; Koch and Hobbs 2007) that some complex ecological attributes may take centuries to recover. However, it is recommended that further refinement of restoration processes be undertaken to improve the overall success of restoration efforts – at least to 'rehabilitate' it to a state in which species diversity after fires can be retained or a self-sustaining community is achieved. Five key factors were identified that would contribute to the development of 'natural-type' shrublands on post-mined lands at Eneabba. These factors include: 1) restoration of a deeper topsoil and a looser subsoil profile; 2) collection of proportionate amounts of only local provenance species, mulch and topsoil; 3) management of highly competitive species (e.g., *Acacia blakelyi*); 4) reseeding and replanting in subsequent years after the initial restoration treatments, including after initial fires; and 5) delaying the introduction of inital fires until the restored vegetation develops sufficient fire-resilience capabilities. Finally, it is recommended that completion criteria for post-mined lands in Mediterranean regions should also include the ability of the system to recover from fire, to ensure that an appropriate, self-sustaining analogue ecosystem has been established. In the case studied here, diversity (at the alpha level) was returned to satisfy completion criteria, but it is evident that new management approaches may be required to counter the potential negative effects of subsequent fires on important ecosystem properties.

References

Adamson, D.A., and M.D. Fox. 1982. Changes to Australasian vegetation since European settlement. Pages 109-160 in J.M.B. Smith, editor. A history of Australasian vegetation, McGraw Hill, Sydney.

Allen, E.B. 1992. Evaluating community-level processes to determine reclamation success. Pages 47-58 in J.C. Chambers, and G.L. Wade, editors. Evaluating reclamation success: the ecological considerations. General Technical Report NE 164, Northeastern Forest Experimental Station, United States Department of Agriculture Forest Service, Radnor, Pennsylvania.

Anderson, A.N. 1993. Ants as indicators of restoration success at a uranium mine in tropical Australia. Restoration Ecology 1:156-167.

Aronson, J., S. Dhillion, and E. Le Floc'h. 1995. On the need to select an ecosystem of reference, however imperfect; a reply to Pickett and Parker. Restoration Ecology 3:1-3.

Aronson, J., E. Le Floc'h, C. Ovalle, and R. Pontanier. 1993. Restoration and rehabilitation of degraded ecosystems in arid and semi-arid lands. II. Case studies in southern Tunisia, central Chile and northern Cameroon. Restoration Ecology 3:168-187.

Ashby, W.C. 1997. Soil ripping and herbicides enhance tree and shrub restoration on stripmines. Restoration Ecology 5:169-177.

Atkinson, I.A.E. 1994. Guidelines to the development and monitoring of ecological restoration programmes. Department of Conservation Technical Series, no. 7, pp. 1172-6873.

Auld, T.D. 1987. Population dynamics of the shrub Acacia suaveolens (Sm.) Wild.: Survivorship throughout the life cycle, a synthesis. Australian Journal of Ecology 12:139-151.

Auld, T.D. 1990. The survival of juvenile plants of the resprouting shrub *Angophora hispida* (Myrtaceae) after a simulated low intensity fire. Australian Journal of Botany 38:255-260.

Auld, T.D., and M.A. O'Connell. 1991. Predicting patterns of post-fire germination in 35 eastern Australian Fabaceae. Australian Journal of Ecology 16:53-70.

Australian Bureau of Statistics. 2002. 1370.0 - Measuring Australia's Progress, 2002, accessed 30th June 2008, http://www.abs.gov.au/ausstats/abs@.nsf/94713ad445ff1425ca25682007192af2/6B904CFE69FE9A78CA256BDC0012240E?opendocument.

Baxter, J.L. 1972. The geology of the Eneabba Area, Western Australia. Western Australian Geological Survey Annual Report, 1971, pp 61-65.

Baxter, J.L. 1977. Heavy mineral sand deposits of Western Australia. Mineral Resources Bulletin No. 10, Geological Survey of Western Australia.

Beard, J.S. 1984. Biogeography of the Kwongan. Pages 1-26 in J.S. Beard, and J.S. Pate, editors. Kwongan: Plant Life of the Sandplain. Nedlands: University of Western Australia Press.

Bell, D.T. 1994. Seed germination ecology in Western Australia: lessons for the mining industry. Pages 5-14 in S.M. Bellairs, and D.T. Bell, editors. National workshop on Native Seed Biology for Revegetation. Australian Centre for Minesite Rehabilitation Research, Brisbane.

Bell, D.T., and D.S. Williams. 1998. Tolerance of thermal shock in seeds. Australian Journal of Botany 46:221-233.

Bell, D.T., and R.J. Hobbs. 2007. Jarrah forest ecosystem restoration: a foreword. Restoration Ecology 15:1-2.

Bell, D.T., and W.A. Loneragan. 1985. The relationship of fire and soil type to floristic patterns within heathland vegetation near Badgingarra, Western Australia. Journal of the Royal Society of Western Australia 67:98-109.

Bell, D.T., A.J.M. Hopkins, and J.S. Pate. 1984. Fire in the kwongan. Pages 178-204 in J.S. Pate, and J.S. Beard, editors. Kwongan – Plant Life of the Sandplain. University of Western Australia Press, Nedlands.

Bell, D.T., D.J. Carter, and R.E. Hetherington. 1986. Experimental assessment of wind erosion after soil stabilization treatments at Eneabba, Western Australia. Environmental Geochemistry and Health 8:99-104.

Bell, D.T., J.A. Plummer, and S.K. Taylor. 1993. Seed germination ecology in southwestern Western Australia. Botannical Review 59:24-73.

Bell, D.T., P.G. van der Moezel, J.C. Delfs, and W.A. Loneragan. 1987. Northern Sandplain Kwongan: effect of fire on *Hakea Obliqua* and *Beaufortia elegans* population structure. Journal of the Royal Society of Western Australia 69:139-143.

Bell, F.G. 1998. Environmental geology: principles and practice. John Wiley and Sons, Limited.

Bellairs, S.M. 1991. Seed biology, establishment ecology and vegetation development of northern sandplain kwongan vegetation after mineral sand mining near Eneabba, Western Australia. PhD Thesis, University of Western Australia, Perth, W.A.

Bellairs, S.M., and D.T. Bell. 1993. Seed stores for restoration of species-rich shrubland vegetation following mining in Western Australia. Restoration Ecology 1:231-240.

Bellingham, P.J., and A.D. Sparrow. 2000. Resprouting as a life history strategy in woody plant communities. Oikos 89:409-416.

Benwell, A.S. 1998. Post-fire seedling recruitment in coastal heathland in relation to regeneration strategy and habitat. Australian Journal of Botany 46:75-101.

Bettenay, E. 1984. Origin and nature of the sandplains. Pages 51-68 in J.S. Beard, and J.S. Pate, editors. Kwongan: Plant Life of the Sandplain. University of Western Australia Press, Nedlands.

Biswell, H.H. 1974. Effects of fire in chaparral. Pages 321-364 in T.T. Kozlowski, and C.E. Ahlgren, editors. Fire and Ecosystems. Academic Press, N.Y.

Black, 1979. Mineral sands rehabilitation at Eneabba, W.A. Pages 69-74 in I.R.F. Hore-Lacy, editor. Mining Rehabilitation-1979. Australian Mining Industry Council, Dickson, Australian Capital Territory.

Bloom, A.J., F.S. Chapin, and H.A. Mooney. 1985. Resource limitation in plants – an economic analogy. Annual Review of Ecological Systems 16:363-392.

BoM [Bureau of Meteorology]. 2008. *C*limate statistics for Australian locations - Eneabba. Bureau of Meteorology, Melbourne, accessed 25 June 2008, http://www.bom.gov.au/climate/averages/tables/cw_008225.shtml.

Bond, W.A. 1984. Fire survival of Cape Proteaceae-influence of fire season and seed predators. Vegetatio 56:65-74.

Bond, W.J., and B.W. van Wilgen. 1996. Fire and plants. Chapman and Hall, London.

Bond, W.J., and J.J. Midgley. 2001. Ecology of sprouting in woody plants: the persistence niche. Trends in Ecology and Evolution 16:45-51.

Booth, D.T., J.K. Gores, G.E. Schuman, and R.A. Olson. 1999. Shrub densities on pre-1985 reclaimed mine lands in Wyoming. Restoration Ecology 7:24-32.

Bowen, B.J., and J.S. Pate. 1993. The significance of root starch postfire shoot recovery of the resprouter *Stirlingia latifolia* R.Br. (Proteaceae). Annals of Botany 72:7-16.

Bowles, M., and M. Jones. 1999. Vegetation profiles and species richness indices for Chicago region graminoid plant communities described and sampled by the Illinois Natural Areas Inventory. The Morton Arboretum.

Bradstock, R.A., and P.J. Myerscough. 1981. Fire effects of seed release and the emergence and establishment of seedlings in *Banksia ericifolia* L.f. Australian Journal of Botany 29:521-531.

Bradstock, R.A., and P.J. Myerscough. 1988. The survival and population response to frequent fires of two woody resprouters *Banksia serrata* and *Isopogon anemonifolius*. Australian Journal of Botany 36:415-431.

Braun-Blanquet, J. 1932. Plant Sociology - The Study of Plant Communities. McGraw-Hill, New York, NY.

Braun-Blanquet, J. 1964. Pflanzensoziologie, Grundzüge der Vegetationskunde. 3. Auf. Springer, Wien-New York.

Brewer, I.M., and R.J. Whelan. 2003. Changes in dune vegetation over 60 years in a sand mining area of the NSW lower North Coast. Cunninghamia 8:85-92.

Brooks, D.R., T.P. Farrell, N.A. Marshman, and J. Muir-Smith. 1996. The environmental management process. Pages 77-98 in D.R. Mulligan, editor. Environmental Management in the Australian Minerals and Energy Industries-Principles and Practices, University of New South Wales Press, Sydney.

Brooks, M.L. 2002. Peak fire temperatures and effects on annual plants in the Mojave Desert. Ecological Applications 12:1088-1102.

Burrows, N.D. 1985. Reducing the abundance of *Banksia grandis* in the jarrah forest by the use of controlled fire. Australian Forestry 48:63-70.

Cairns, J. 1990. The prediction, validation, monitoring and mitigation of anthropogenic effects on natural systems. Environmental Auditor 2:19-25.

Cairns, J. 1993. Is restoration ecology practical? Restoration Ecology 1:3-7.

Camp, P. 1990. Rehabilitation after dune mining at Richards Bay Minerals. South African Mining World, October edition.

Camp, P., and P.J. Weisser. 1991. Dune rehabilitation, flora and plant succession after mining at Richards Bay, South Africa. Pages 106-123 in D.A. Evarard, and G.P. von Maltitz, editors. Dune Forest Dynamics in Relation to Land-Use Practices. Pretoria.

Chambers, J.C., J.A. MacMahon, and G.L. Wade. 1992. Differences in successional processes among biomes: importance in obtaining and evaluating reclamation success. Pages 59-72 in J.C. Chambers, and G.L. Wade, editors. Evaluating Reclamation Success: the Ecological Considerations. General Technical Report NE 164, Northeastern Forest Experimental Station, United States Department of Agriculture Forest Service, Radnor, Pennsylvania.

Chambers, J.C., R.W. Brown, and B.D. Williams. 1994. An evaluation of reclamation success on Idaho's phosphate mines. Restoration Ecology 2:4-16.

Chapman, R., J. Collins, and A. Younger. 1996. Control of legumes in a species-rich meadow re-created on land restored after opencast coal mining. Restoration Ecology 4:407-411.

Choi, Y.D. 2004. Theories for ecological restoration in the changing environment: Toward 'futuristic' restoration. Ecological Research 19:75-81.

Christensen, N.L. 1985. Shrubland fire regimes and their evolutionary consequences. Pages 85-100 in S.T.A. Pickett, and P.S. White, editors. The Ecology of Natural Disturbance and Patch Dynamics. Academic Press, New York, NY.

Clark, S.S. 1975. The effect of sand mining on coastal heath vegetation in New South Wales. Proceedings of the Ecological Society of Australia 9:1-16.

Clewell, A.F. 1999. Restoration of riverine forest at Hall Branch on phosphate-mined land, Florida. Restoration Ecology 7:1-14.

Collins, B.G., B.J. Wykes, and O. Nichols. 1985. Recolonization of restored bauxite minelands in south-western Australian forests. Pages 341-354 in A. Keast, H.F. Recher, J. Ford, and D. Saunders, editors. Birds of Eucalypt Forest and Woodlands: Ecology, Conservation and Management. Surrey Beatty and Sons, Chipping Norton.

Comino, E., B.P. Miller, and N.J. Enright. 2004. Soil seedbanks in natural and restored box ironbark forests at Stawell Gold Mine, Victoria. Pacific Conservation Biology 10:9-20.

Commander, D.P. 1978. Hydrogeology of the Eneabba bore hore line. Geological Survey of Western Australia Annual Report 1977, pp. 13-18.

Connell, J.H. 1978. Diversity in tropical rainforests and tropical reefs. Science 199:1302-1310.

Cooper, D.J., and L.H. MacDonald. 2000. Restoring the vegetation of mined peatlands in the southern rocky mountains of Colorado, U.S.A.

Corbett, E.A., R.C. Anderson, and C.S. Rodgers. 1996. Priarie revegetation of a strip mine in Illinois: Fifteen years after establishment 4:346-354.

Cowling, R.M., and B. Lamont. 1985. Variation in serotiny of three Banksia species along a climatic gradient. Australian Journal of Ecology 10:345-350.

Cowling, R.M., and B. Lamont. 1987. Post-fire recruitment of four co-occurring Banksia species. Australian Journal of Ecology 24:645-658.

Croft, P., J.T. Hunter, and N. Reid. 2008. Depletion of regenerative bud resources during cyclic drought: What are the implications for fire management? Ecological Management and Restoration 8:187-192.

Croton, J.T., and G.L. Ainsworth. 2007. Development of a winged tine to relieve mining-related soil compaction after bauxite mining in Western Australia. Restoration Ecology 15:48-53.

Croton, J.T., and G.G. Watson. 1987. Mining related compaction–a case study in the Darling Range, Western Australia. Environmental Research Bulletin No. 14.

Cruz, A., and J.M. Moreno. 2001a. Seasonal trends of total non-structural carbohydrates (TNC) in *Erica australis* and their relationship with plane water status. Oecologia 128:201-211.

Cruz, A., and J.M. Moreno. 2001b. Lignotuber size of *Erica australis* and its relationship with soil resources. Journal of Vegetation Science 12:373-384.

Cruz, A., B. Perez, J.R. Quintana, and J.M. Moreno. 2002. Resprouting in the Mediterranean-type shrub *Erica australis* affected by soil resource availability. Journal of Vegetation Science 13:641-650.

D'antonio, C.M., and P.M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle and global change. Annual Review of Ecology and Systematics 23:63-87.

Darwin, C. 1859. On the origin of species by means of natural selection or preservation of favoured races in the struggle for life. (Reprinted 1964). Cambridge (MA), Harvard University.

Davis, F.W., D.E. Hickson, D.C. Odion. 1988. Composition of maritime chaparral related to fire history and soil, Burton Mesa, Santa Barbara County, California. Madrono 35:169-195.

Delfs, J.C., J.S. Pate, and D.T. Bell. 1987. Northern Sandplain Kwongan: community biomass and selected species response to fire. Journal of the Royal Society of Western Australia 69:133-138.

Dell, B., S. Jones, and I.M. Wallace. 1985. Phosphorus accumulation by lignotubers of jarrah (*Eucalyptus marginata* Donn ex Sm.) seedlings grown in a range of soils. Plant and Soil 86:225-232.

Diels, L. 1906. Die Planzenwelt von West-Australien Sudlich des Wendekreises. A. Engler, and O. Drude. Die Vegetation der Erde, W.Engelmann, Leipzig.

Drake, J.A., H.A. Mooney, F. DiCastri, R.H. Groves, F.J. Kruger, M. Rejmanek, and M. Williamson. 1989. Biological invasions: a global perspective. Jon Wiley and Sons, Chichester.

Dublin, H.T., A.R.E. Sinclair, and J. McGlade. 1990. Elephants and fire as causes of multiple stable states in the Serengeti-Mara woodlands. Journal of Animal Ecology 59:47-64.

Elliott, P.J., J. Gardner, D. Allen, and G. Butcher. 1996. Completion criteria for Alcoa of Australia Limited’s bauxite mine rehabilitation. Pages 79-89 in 3rd International 21st Annual Minerals Council of Australia Environment Workshop, Perth, Australia. Minerals council of Australia. Canberra, ACT, Australia.

EMRC. 1996. A review of RGC mineral sands flora and fauna monitoring programme at Eneabba. Unpublished report, RGC Mineral Sands, Eneabba, Western Australia.

Enright, N.J., and B.B. Lamont. 1989. Seed banks, fire season, safe sites and seedling recruitment in five co-occurring Banksia species. Journal of Ecology 77:1111-1122.

Enright, N.J., and B.B. Lamont. 1992a. Recruitment variability in the resprouting shrub *Banksia attenuata* and non-sprouting congeners in the northern sandplain heaths of south-western Australia. Acta Oecologica 13:727-741.

Enright, N.J., and B.B. Lamont. 1992b. Survival, growth and water relations of Banksia seedlings in a sand mine rehabilitation site and adjacent scrub-heath sites. Journal of Applied Ecology 29:663-671.

Enright, N.J., B.B. Lamont, and R. Marsula. 1996. Canopy seed bank dynamics and optimum fire regime for the highly serotinous shrub, *Banksia hookeriana*. Journal of Applied Ecology 84:9-17.

Enright, N.J., R. Marsula, B.B. Lamont, and C. Wissel. 1998. The ecological significance of canopy seed storage in fire-prone environments: a model for resprouting shrubs. Journal of Ecology 86:960-973.

Enright, N.J., E. Mosner, B.P. Miller, N. Johnson, and B.B. Lamont. 2007. Soil versus canopy seed storage and plant species coexistence in species-rich shrublands of southwestern Australia. Ecology 88:2292-2304.

Environment Protection Agency, EPA. 1995. Rehabilitation and revegetation. Best Practice Environmental Management in Mining. Commonwealth of Australia, Canberra.

Environment Protection Agency, EPA. 2006. Rehabilitation of Terrestrial Ecosystems. Guidance for the Assessment of Environmental Factors. Commonwealth of Australia, Canberra.

Ewel, J.J. 1987. Restoration is the ultimate test of ecological theory. Pages 5-8 in W.R. Jordan, M.E. Gilpin, and J.D. Aber, editors. Restoration Ecology: A Synthetic Approach to Ecological Research. Cambridge University Press, Cambridge, UK.

Ewel, J.J., and S.W. Bigelow. 1996. Plant life-forms and tropical ecosystem functioning. Pages 101-126 in G.H. Orians, R. Dirzo, and J.H. Kushman, editors. Biodiversity and Ecosystem Processes in Tropical Forests. Heidelberg: Springer-Verlag.

Fensham, R.J. 1990. Interactive effects of fire frequency and site factors in tropical Eucalyptus forests. Australian Journal of Ecology 15:255-266.

Fox, M.D. 1988. Understorey changes following fire at Myall Lakes, New South Wales. Cunninghammia 2:85-96.

Fox, B.J., and M.D. Fox. 1986b. The susceptibility of natural communities to invasion. Pages 57-66 in R.H. Groves, and J.J. Burdon, editors. Ecology of Biological Invasions: an Australian Perspective, Australian Academy of Science, Canberra.

Fox, M.D., and B.J. Fox. 1986a. The effect of fire frequency on the structure and floristic composition of a woodland understorey. Austral Ecology 11:77-85.

Fox, B.J., M.D. Fox, J.E. Taylor, J.P. Jackson, J. Simpson, P. Higgs, L. Rebec, and R. Avery. 1996. Comparison of regeneration following burning, clearing or mineral sand mining at Tomago, NSW: I. Structure and growth of vegetation. Australian Journal of Ecology 21:184-199.

Friedel, M.H., B.D. Foran, and D.M. Stafford-Smith. 1990. Where the creeks run dry or ten feet high: pastoral management in arid Australia. Proceedings of the Ecological Society of Australia 16:185-194.

Froend, R.H. 1987. Investigations into species richness patterns in the northern sandplain region of Western Australia. PhD Thesis, University of Western Australia, Perth, W.A.

Frost, T.M., S.R. Carpenter, A.R. Ives, and T.K. Kratz. 1995. Species compensation and complementarity in ecosystem functioning. Pages 224-239 in C.G. Jones, and J.H. Lawton, editors. Linking species and ecosystems. New York: Chapman and Hall.

Gardner, C.A. 1944. The vegetation of Western Australia with particular reference to the climate and soils. Journal of the Royal Society of Western Australia 28:11-87.

Gardner, J.H., and D.T. Bell. 2007. Bauxite mining restoration by Alcoa World Alumina Australia in Western Australia: social, political, historical and environmental contexts. Restoration Ecology 207:3-8.

George, A.S., A.J.M. Hopkins, and N.G. Marchant. 1979. The heathlands of Western Australia. Pages 211-228 in R.L. Specht, editor. Ecosystems of the World 9A: Heathlands and Related Shrublands. Elsevier scientific publishing company, Amsterdam.

Gill, A.M. 1981. Adaptive responses of Australian vascular plant species to fire. Pages 243-272 in A.M. Gill, R.H. Groves, and I.R. Noble, editors. Fire and the Australian Biota, Australian Academy of Science, Canberra.

Gilmour, D.A., N.V. San, and T. Xiong. 2000. Rehabilitation of degraded forest ecosystems in Cambodia, Lao PDR, Thailand and Vietnam: an overview. IUCN-Asia, Cambridge, UK. pp. 1-45.

Gleason, H.A. 1926. The individualistic concept of plant association. Torreya 53:7-26.

Gower, J.C. 1966. Some distance properties of latent root and vector methods used in multivariate analysis. Biometrika 53:325-338.

Grant, C.D. 2006. State-and-transition successional model for bauxite mining rehabilitation in the jarrah forest of Western Australia. Restoration Ecology 14:28-37.

Grant, C.D., and J.M. Koch. 1997. Ecological aspects of soil seed-banks in relation to bauxite mining. Twelve year old rehabilitation mines. Australian Journal of Ecology 22:177-184.

Grant, C.D., and W.A. Loneragan. 1999. The effects of burning on the understorey composition of 11-13 year-old rehabilitated bauxite mines in Western Australia: vegetation characteristics. Plant Ecology 145:291-305.

Grant, C.D., and W.A. Loneragan. 2001. The effect of burning on the understorey composition and vegetation succession of 11-13 year-old rehabilitation in Western Australia: community changes and vegetation succession. Forest Ecology and Management 145:255-279.

Grant, C.D., and W.A. Loneragan. 2003. Using dominance-diversity curves to assess completion criteria after bauxite mining rehabilitation in Western Australia. Restoration Ecology 11:103-109.

Grant, C.D., M.A. Norman, and M.A. Smith. 2007. Fire and silvicultural management of restored bauxite mines in Western Australia. Restoration Ecology 15:127-136.

Grant, C.D., S.C. Ward, and S.C. Morley. 2007. Return of ecosystem function to restored bauxite mines in Western Australia. Restoration Ecology 15:94-103.

Grant, C.D., W.A. Loneragan, J.M. Koch, and D.T. Bell. 1997. Fire behaviour, fuel characteristics and vegetation structure of 11-15 year-old rehabilitated bauxite mines in Western Australia. Australian Forestry 60:147-157.

Griffin, E.A., and A.J.M. Hopkins. 1985. The flora and vegetation of Mt. Lesueur, Western Australia. Journal of the Royal Society of Western Australia 67:45-57.

Griffin, E.A., A.J.M. Hopkins, and R.J. Hnatiuk. 1983. Regional variation in Mediterranean-type shrublands near Eneabba, south-western Australia. Vegetatio 52:103-127.

Grime, J.P. 2001. Plant Strategies, Vegetation Processes and Ecosystem Properties, 2nd edition. Wiley, Chichester.

Groom, P.K., B.B. Lamont, and I. Wright. 2001. Lottery (stochastic) and non-lottery (biological) processes explain recruitment patterns among eight congeneric shrubs in southwestern Australia. Journal of Mediterranean Ecology 2:1-14.

Harris, J.A., P. Birch, and J.P. Palmer. 1996. Land restoration and reclamation: principles and practice. Addison Wesley Longman, Harlow, England.

He, T., S. Krauss, B.B. Lamont, B.P. Miller, and N.J. Enright. 2004. Long distance seed dispersal in a metapopulation of *Banksia hookeriana* inferred from a population allocation analysis of AFLP data. Molecular Ecology 13:1099-1109.

Henderson, R. 1999. Response to Henry Howe. Restoration Ecology 17:189-192.

Herath, D.N., B.B. Lamont, N.J. Enright, and B.P. Miller. (in press) Comparison of post-mine rehabilitated and natural shrubland communities in southwestern Australia. Restoration Ecology.

Hnatiuk, R.J., and A.J.M. Hopkins. 1981. An ecological analysis of kwongan vegetation south of Eneabba, Western Australia. Australian Journal of Ecology 6:423-438.

Hobbs, R.J., and A.J.M. Hopkins. 1990. From frontier to fragments: European impact on Australia's vegetation. Proceedings of the Ecological Society of Australia 16:93-114.

Hobbs, R.J., and D.A. Norton. 1996. Towards a conceptual framework for restoration ecology. Restoration Ecology 4:93-110.

Hobbs, R.J., and L. Atkins 1990. Fire-related dynamics of a Banksia woodland in south-western Western Australia. Australian Journal of Botany 38:97-110.

Holl, K.D. 2002. Long-term vegetation recovery on reclaimed coal surface mines in the eastern USA. Journal of Applied Ecology 39:960-970.

Holl, K.D., and Cairns Jr., J. 1994. Vegetational community development on reclaimed coal surface mines in Virginia. Bulletin of the Torrey Botanical Club. 121:327-337.

Holmes, P.M. 2001. Shrubland restoration following woody alien invasion and mining: effects of topsoil depth, seed source, and fertilizer addition. Restoration Ecology 9:71-84.

Holmes, P.M., and D.M. Richardson. 1999. Protocols based on restoration and recruitment dynamics, ecosystem structure, and ecosystem function: perspectives from South African fynbos. Restoration Ecology 7:215-230.

Holmes, P.M., and R.J. Newton. 2004. Patterns of seed persistence in South African fynbos. Plant Ecology 172:143-158.

Hooper, R.J. 1985. Management strategies, particularly use of fire, for regenerated mined lands in northern Australian savannas. Pages 329-331 in J.C. Tothill, and J.J. Mott, editors. Ecology and management of the world's savannas. The Australian Academy of Science, Canberra, Australia.

Hopkins, A.J.M., and Griffin, E.A. 1984. Floristic patterns. Pages 69-83 in J.S. Pate, and J.S. Beard, editors. Kwongan – Plant Life of the Sandplain. University of Western Australia Press, Nedlands.

Huang, C., S.E. Marsh, M.P. Mclaran, and S.R. Archer. 2007. Postfire stand structure in a semiarid savanna: cross-scale challenges estimating biomass. Ecological Applications 17:1899-1910.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs 54:187-211.

Huston, M.A. 1994. Biological Diversity – the coexistence of species in changing landscapes. Cambridge University Press, Cambridge.

Iwasa, Y., and T. Kubo. 1997. Optimal size of storage for recovery after unpredictable environments. Evolutionary Ecology 11:41-65.

Jackson, L.L., N. Lopoukhine, and D. Hillyard. 1995. Ecological restoration: a definition and comments. Restoration Ecology 3:71-75.

Jefferies, M.A., F.M. Nicholls, and A.E. Petersen. 1991. Rehabilitation after mining of diverse heathlands at Eneabba, Western Australia. Proceedings of the Australian Mining Industry Council Environmental Workshop, Perth W.A. October 1991, pp. 261-269.

Jones, C.G., J.H. Lawton, and M. Schachak. 1994. Organisms as ecosystem engineers. Oikos 86:373-386.

Jordan, W.R., R.L. Peters, and E.B. Allen. 1988. Ecological restoration as a strategy for conserving biological diversity. Environmental Management 12:55-72.

Keeley, J.E. 1991. Seed germination and life-history syndromes in the California chaparral. The Botanical Review 57:81-116.

Keeley, J.E. 1992. Demographic structure of California chaparral in the long-term absence of fire. Journal of Vegetation Science 3:79-90

Keeley, J.E., and S.C. Keeley. 1984. Energy allocation patterns of sprouting and non-sprouting species of Arctostaphylos in the Californian chaparral. The American Midland Naturalist 111:105-117.

Keeley, J.E., C.S. Fotheringham, and M. Baer-Keeley. 2005. Determinants of postfire recovery and succession in Mediterranean-climate shrublands of southwestern Australia. Ecological Applications 15:1515-1534.

Keeley, J.E., D. Lubin, and C.J. Fotheringham. 2003. Fire and grazing impacts on plant diversity in the southern Sierra Nevada. Ecological Applications 13:1355-1374.

Keeley, J.E., and S.C. Keeley. 1984. Energy allocation patterns of sprouting and non-sprouting species of Arctostaphylos in the Californian chaparral. The American Midland Naturalist 111:105-117.

Keeley, S.A., and S.C. Keeley. 1987. The role of fire in the germination of chaparral herbs and suffrutescents. Madrono 34:240-249.

Kelly, V.R., and V.T. Parker. 1990. Seed bank survival and dynamics in sprouting and nonsprouting Arctostaphylos species. American Midland Naturalist 124:114-123.

Kindscher, K., and L.L. Tieszen. 1998. Floristic and soil organic matter changes after five and thirty-five years of native tallgrass prairie restoration. Restoration Ecology 6:181-196.

Koch, J.M. 2007a. Restoring a jarrah forest understorey vegetation after bauxite mining in Western Australia. Restoration Ecology 15:17-25.

Koch, J.M. 2007b. Alcoa’s mining and restoration process in southwestern Australia. Restoration Ecology 15:11-16.

Koch, J.M., and S. Davies. 1993. The effect of a tall dense Acacia understorey on small shrub and herb species native to the jarrah forest. Alcoa of Australia, Environmental Research Bulletin No. 21.

Koch, J.M., and S.C. Ward. 1994. Establishment of understorey vegetation for rehabilitation of bauxite-mined areas in the jarrah forest of Western Australia. Journal of Environmental Management 4:1-15.

Koch, J.M., and R.J. Hobbs. 2007. Synthesis: is Alcoa succesfuly restoring a jarrah forest ecosystem after bauxite mining in Western Australia? Restoration Ecology 15:137-144.

Koch, J.M., J.R. Richardson, and B.B. Lamont. 2004. Grazing by kangaroos limits the establishment of the grass trees *Xanthorrhoea gracilis* and *X. preissii* in restored bauxite mines in eucalypt forest of southwestern Australia. Restoration Ecology 12:297-305.

Koch, J.M., S.C. Ward, C.D. Grant, and G.L. Ainsworth. 1996. Effects of bauxite mine restoration operations on topsoil seed reserves in the jarrah forests of Western Australia. Restoration Ecology 4:501-512.

Krauss, S.L., and J.M. Koch. 2004. Rapid genetic delineation of provenance for plant community restoration. Journal of Applied Ecology 41:1162-1173.

Kruger, F.J. 1983. Plant community diversity and dynamics in relation to fire. Pages 446-472 in F.J. Kruger, D.T. Mitchell, and J.U.M. Jarvis, editors. Mediterranean-type Ecosystems: The Role of Nutrients. Springer, Berlin.

Kruger, F.J., and R.C. Bigalke. 1984. Fire in fynbos. Pages 67-114 in P.deV. Booysen, and N.M. Tainton, editors. Ecological Effects of Fire in South African Ecosystems. Springer-Verlag, Berlin.

Kruse, B.S., and J.W. Groninger. 2003. Vegetative characteristics of recently reforested bottomland sites in the Cache River Watershed, Illinois. Restoration Ecology 11:273-280.

Lacey, S.T., P.D. Brennan, and J. Parekh. 2001. Deep may not be meaningful: cost and effectiveness of various ripping tine configurations in a plantations cultivation trial in eastern Australia. New Forests 21:231-248.

Lamont, B.B. 1976. Report on a biological survey and recommendations for rehabilitating a portion of reserve 31030 to be mined for heavy minerals during 1975-81. WAIT–AID Ltd Perth, Western Australia. Unpublished report for Allied Eneabba Pty Ltd.

Lamont, B.B. 1995a. Mineral nutrient relations in mediterranean regions of California, Chile and Australia. Pages 211-235 in M.T. Arroya, P.H. Zedler, and M.D. Fox, editors. Ecology and Biogeography of Mediterranean Ecosystems in Chile, California and Australia. Springer Verlag, New York.

Lamont, B.B. 1995b. Testing the effect of ecosystem composition/structure on its functioning. Oikos 74:283-295.

Lamont, B.B., and D. Wiens. 2003. Is seed set and speciation always low among species that resprout after fire, and why? Evolutionary Ecology 17:277-292.

Lamont, B.B., and M.J. Barker. 1988. Seed bank dynamics of a serotinous, non sprouting Banksia species. Australian Journal of Botany 36:193-203.

Lamont, B.B., and N.J. Enright. 2000. Adaptive advantages of aerial seed banks. Plant Species Biology 15:157-166.

Lamont, B.B., and S.J. van Leeuwen. 1988. Seed production and mortality in a rare Banksia species. Journal of Applied Ecology 25:551-559.

Lamont, B.B., and S.M. Bergl. 1991. Water relations, shoot and root architecture, and phenology of three co-occurring Banksia species: no evidence for niche differentation in the pattern of water use. Oikos 60:291-298.

Lamont, B.B., A.J.M. Hopkins, and R.J. Hnatiuk. 1984. The Flora – composition, diversity and origins. Pages 27-50 in J.S. Pate, and J.S. Beard, editors. Kwongan – Plant Life of the Sandplain. University of Western Australia Press, Nedlands.

Lamont, B.B., N.J. Enright, and S.M. Bergl. 1989. Coexistence and competitive exclusion of *Banksia hookeriana* in the presence of congeneric seedlings along a topographic gradient. Oikos 56:39-42.

Lamont, B.B., R. Wittkuhn, and D. Korczynskyj. 2004. Ecology and ecophysiology of grasstrees. Australian Journal of Botany 52:561-582.

Lamont, B.B., S. Downes, and J.E.D. Fox. 1977. Importance-value curves and diversity indices applied to a species-rich heathland in Western Australia. Nature 265:438-441.

Lamont, B.B., S.W. Connell, and S.M. Bergl. 1991. Seed bank and population dynamics of *Banksia cuneata*: the role of time, fire and moisture. Botanical Gazette 152:114-122.

Lamont, B.B., D.C. Le Maitre, R.M. Cowling, N.J. Enright. 1991. Canopy seed storage in woody plants. Botanical Review 57:277-317.

Lamont, B.B., T. He, N.J. Enright, S.L. Krauss, and B.P. Miller. 2003. Anthropogenic disturbance promotes hybridization between Banksia species by altering their biology. Journal of Evolutionary Biology 16:551-557.

Lawrie, J. 1984. Regeneration at Weipa bauxite. Mining Magazine 150:206-213.

Le Maitre, D.C., and J.J. Midgley. 1992. Plant reproductive ecology. Pages 135-174 in R.M. Cowling, editor. Ecology of Fynbos Nutrients, Fire and Diversity. Oxford University Press, Cape Town.

Levin, S.A., and R.T. Payne. 1974. Disturbance, patch information, and community structure. Proceedings of the National Academy of Science of the United States of America 71:2744-2747.

Levitt, M. 1997. Putting fynbos back: the rehabilitation of Du Toitskloof. Veld and Flora 83:81-83.

Lloret, F., M. Verdu, N.F. Flores-Hernandez, and A. Valiente-Banuet. 1999. Fire and resprouting in Mediterranean ecosystems: Insights from an internal biogeographical region, the Mexical shrubland. American Journal of Botany 86:1655-1661.

Lockwood, J.L., and S.L. Pimm. 1999. When does restoration succeed? Pages 338-362 in E. Weiher, and P.A. Keddy, editors. Ecological Assembly Rules: Perspectives, Advances, Retreats. Cambridge University Press. NY. USA.

Low, A.B., and B. Lamont. 1985. Nutrient allocation in native heath and rehabilitated communities. Proceedings of a seminar on the plant ecology of the Eneabba heathlands, Western Australian Institute of Technology, School of Biology Bulletin 10, pp. 12-19.

Lowry, D.C. 1974. Geological survey of Western Australia. 1:250 000 Geological Series – Explanatory Notes. Dongara – Hill River, Western Australia (Sheet SH/50-5,9). Australian Government Publishing Service, Canberra. pp. 33.

Lubke, R.A., and A.M. Avis. 1999. A review of the concepts and application of rehabilitation following heavy mineral dune mining. Marine Pollution Bulletin 37:546-557.

Majer, J.D., M. Sartori, R. Stone, and W.S. Perriman. 1982. Recolonisation by ants and other invertebrates in rehabilitated mineral sand mines near Eneabba, Western Australia. Reclamation and Revegetation Research 1:63-81.

Marchant, N.G. 1973. Species diversity in the southwestern flora. Journal of the Royal Society of Western Australia 56:23-30.

Mattiske Consulting Pty Ltd (2004) Assesment of rehabilitated mining areas – Iluka Resources Ltd, Eneabba operations. Unpublished report prepared for Iluka Resources Ltd.

McClaren, H. 2005. Is there a relationship between seed demography and spatial pattern, and what does this mean for the restoration of mined kwongan? B.Sc. Honours thesis. The University of Melbourne.

McCune, B., and M.J. Mefford. 1999. PC-ORD Multivariate Analysis of Ecological Data, Version 5, MjM Software Design, Oregon, USA.

McDonald, T. 2000. When is the management 'restoration' and restoration, 'management'? Ecological Management and Restoration 1:i.

McKay, J.K., C.E. Christian, S. Harrison, and K.W. Dixon. 1995. *In vitro* propagation of Western Australian rushes (Restionaceae and related families) by embryo culture. Part 1. In vitro embryo culture. Plant Cell, Tissue and Organ Culture 41:107-113.

Mengler, F.C., G.A. Kew, R.J. Gilkes, and J.M. Koch. 2006. Using instrumented bulldozers to map spatial variation in the strength of regolith for bauxite mine floor rehabilitation. Soil and Tillage Research 90:126-144.

Mentis, M.T., and W.N. Ellery. 1994. Post-mining rehabilitation of dunes on the north-east coast of South Africa. South African Journal of Science 90:69-74.

Microsoft® Excel (2000) Microsoft Corporation

Miller, B.P., N.J. Enright, and BB. Lamont. 2007. Record error and range contraction, real and imagined, in the restricted shrub *Banksia hookeriana* in south-western Australia. Diversity and Distributions 13:406-417.

Moreno, J.M., and W.C. Oechel. 1991. Fire intensity and herbivory effects on postfire resprouting of *Adenostoma fasciculatum* in southern California chaparral. Oecologia 85:429-433.

Morley, I.W. 1981. Black sands: A History of the Mineral Sand Mining Industry in Eastern Australia. University of Queensland Press, St Lucia.

Nadeau, L.B., and D.J. Pluth. 1997. Spatial distribution of lodgepole pine and white spruce seedling roots 10 years after deep tillage of a gray luvisol. Canadian Journal of Botany 27:1606-1613.

Naiman, R.J., G. Pinay, C.A. Johnston, and J. Pastor. 1994. Beaver influences on the long-term biogeochemical characteristics of boreal forest drainage networks. Ecology 75:905-921.

Nash, R. 1990. The rights of nature: history of environmental ethics. Primavera Press, Sydney.

Nichols, O.G. 2006. Developing completion criteria for Native ecosystem reconstruction–a challenge for the mining industry. Pages 61-74 in A. Fourie, and M. Tibbett, editors. Mine closure 2006. Proceedings of the first international seminar on mine closure. Australian centre for geomechanics, Perth, Australia.

Nichols, O.G., and D.V. Michaelsen. 1986. Successional trends in bauxite mines rehabilitated using three topsoil return techniques. Forest Ecology and Management 14:163-175.

Nichols, O.G., and F.M. Nichols. 2003. Long term trends in faunal recolonization after bauxite mining in the jarrah forest of south-western Australia. Restoration Ecology 11:261-272.

Nichols, O.G., B.A. Carbon, I.J. Colquhoun, J.T. Croton, and N.J. Murray. 1985. Rehabilitation after bauxite mining in south-western Australia. Landscape Planning 12:75-92.

Noble, I.R. 1984. Mortality of lignotuberous seedlings of Eucalyptus species after an intense fire in montane forest. Australian Journal of Ecology 9:47-50.

Noble, I.R., and R.O. Slatyer. 1980. The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbances. Vegetatio 43:5-21.

Norman, M.A., J.M. Koch, C.D. Grant, T.K. Morald, and S.C. Ward. 2006. Vegetation succession after bauxite mining in Western Australia. Restoration Ecology 14:278-288.

Novacek, M.J., and E.E. Cleland. 2001. The current biodiversity extinction event: scenarios for mitigation and recovery. Proceedings of the National Academy of Science 98 10:5466-5470.

O'Connell, A.M., and T.S. Grove. 1996. Biomass production, nutrient uptake and nutrient cycling in the Jarrah (*Eucalyptus marginata*) and Karri (Eucalyptus diversicolor) forests of south-western Australia. Pages 155-189 in P.M. Attiwill, and M.A. Adams, editors. Nutrition of the Eucalypts. CSIRO Publishing, Melbourne.

Odum, H.T. 1996. Environmental accounting: Emergy and environmental decision making. Jon Wiley and Sons, New York, pp. 384.

Oldeman, L.R. 1994. The global extent of soil degradation. Pages 99-118 in D.J. Greenland, and I. Szabolcs, editors. Soil resilience and Sustainable Land Use. CAB International Publishers, Wallingford, Oxford, UK.

Paczkowska, G., and A.R. Chapman. 2000. The Western Australian Flora, A Descriptive Catalogue. The Wildflower Society of Western Australia (inc.), the Western Australian Herbarium, CALM and the Botanic Gardens & Parks Authority.

Parker, V.T. 1987. Effects of wet-season management burns on chaparral vegetation: implications for rare species. Pages 233-237 in T.S. Elias, editor. Conservation and Management of Rare and Endangered plants. California Native Plant Society, Sacramento, California, USA.

Parrotta, J.A., and O.H. Knowles. 1999. Restoration of tropical moist forest on bauxite-mined lands in the Brazillian Amazon. Restoration Ecology 7:103-116.

Parsons, M.H., J.M. Koch, B.B. Lamont, S. Vlahos, and M. Fairbanks. 2006. Planting density effects and selective herbivory by kangaroos on species used in restoring forest communities. Forest Ecology and Management 229:39-49.

Parsons, R.F., and D.G. Cameron. 1974. Maximum plant species diversity in terrestrial communities. Biotropica 6:202-203.

Partridge, T.R. 1992. Vegetation recovery following sand mining on coastal dunes at Kaiforete Spit, Canterbury, New Zealand. Biological Conservation 61:59-71.

Peterson, G., C.R. Allen, and C.S. Holling. 1998. Ecological resilience, biodiversity and scale. Ecosystems 1:6-18.

Pickett, S.T.A., and V.T. Parker. 1994 Avoiding the old pitfalls: opportunities in a new discipline. Restoration Ecology 2:75-79.

Pielou, E.C. 1966. The measurement of diversity in different types of biological collections. Journal of Theoretical Biology 13:131-144.

Playford, P.E., A.E. Cockbain, and G.H. Low. 1976. Geology of the Perth Basin, Western Australia. Geological Survey of Western Australia Bulletin 124:311.

Podani, J. 2001. SYN-TAX 2000. Computer Programs for Data Analysis in Ecology and Systematics. User's Manual. Scientia Publishing, Budapest.

Pyke, G.H. 1983. Relationship between time since last fire and flowering in *Telopea speciosissima* R. Br. and *Lambertia formosa* Sm. Australian Journal of Botany 31:293-296.

Reay, S.D., and D.A. Norton. 1999. Assessing the success of restoration plantings in a temperate New Zealand forest. Restoration Ecology 9:298-308.

Rice, B., and M. Westoby. 1983. Species richness in vascular vegetation of the West head, N.S.W. Australian Journal of Ecology 8:163-168.

Roche, S., J.M. Koch, and K.W. Dixon. 1997. Smoke enhanced seed germination for mine rehabilitation in the southwest of western Australia. Restoration Ecology 5:191-203.

Rokich, D.P., K.W. Dixon, K. Sivasithamparam, and K.A. Meney. 2002. Smoke, mulch, and seed broadcasting effects on woodland restoration in Western Australia. Restoration Ecology 10:185-194.

Ross, K.A., J.E. Taylor, M.D. Fox, and B.J. Fox. 2004. Interaction of multiple disturbances: importance of disturbance interval in the effects of fire on rehabilitating mined areas. Austral Ecology 29:508-529.

Ruiz-Jaen, M.C., and T.M. Aide. 2005. Restoration success: how is it being measured? Restoration Ecology 13:569-577.

Rundel, P.W., G.A. Baker, D.J. Parsons, and T.J. Stohlgren. 1987. Postfire demography of resprouting and seedling establishment by *Adenostoma fasciculatum* in the California chaparral. Pages 555-596 in J.D. Tenhunen, F.M. Catarino, O.L. Lange, and W.C. Oechel, editors. Plant-Responses to Stress-Functional analysis in Mediterannean Ecosystems. Springer-Verlag, Berlin.

Salinas, M.J., and J. Guirado. 2002. Riparian plant restoration in summer-dry riverbeds of southeastern Spain. Restoration Ecology 10:695-702.

Schindler, D.W. 1990. Experimental perturbations of whole lakes as tests of hypotheses concerning ecosystem structure and function. Oikos 57:25-41.

SER (Society for Ecological Restoration International Science and Policy Working Group). 2004. The SER International Primer of Ecological Restoration (available from http//www.ser.org), accessed July 2008. Society for Ecological Restoration International, Tucson, Arizona.

Shannon, C.E., and W. Weaver 1949. The mathematical theory of communication. University of Illinois Press, Urbana. pp. 117.

Shaw, P.J.A. 1996. Role of seedbank substrates in the revegetation of fly ash and gypsum in the United Kingdom. Restoration Ecology 4:61-70.

Shea, S.R., J. McCormick, and C.C. Portlock. 1979. The effect of fires on regeneration of leguminous species in the northern Jarrah (*Eucalyptus marginata* Sm.) forest of Western Australia. Australian Journal of Ecology 4:195-205.

Shiklomanov, I.A. 2000. Appraisal and assessment of world water resources. Water International 25:11-32.

Simpson, E.H. 1949. Measurement of diversity. Nature 163:688.

Smith, M.A. 2001. Vegetation response to fire as an indicator of restoration success after bauxite mining in the jarrah forest of Western Australia. PhD Thesis. The University of Western Australia, Department of Botany.

Smith, M.A., W.A. Loneragan, C.D. Grant, and J.M. Koch. 2000. Effect of fire on the topsoil seedbanks of rehabilitated bauxite mine sites in the jarrah forest of Western Australia. Ecological Management and Restoration 1:50-60.

SPSS, Inc. 2004. SPSS Base 13.0 for Windows User's Guide. SPSS Inc., Chicago, Illinois.

Szota, C., E.J. Veneklaas, J.M. Koch, and H. Lambers. 2007. Root architecture of jarrah (*Eucalyptus marginata*) trees in relation to postmine deep ripping in Western Australia. Restoration Ecology 15:65-73.

Tacey, W.H., and B.L. Glossop 1980. Assessment of topsoil handling techniques for rehabilitation of sites mined for bauxite within the jarrah forest of Western Australia. Journal of Applied Ecology 17:195-201.

Tilman, D. 1996. Biodiversity: population versus ecological stability. Ecology 77:350-63.

Toth, L.A., D.A. Arrington, M.A. Brady, and D.A. Muszick. 1995. Conceptual evaluation of factors potentially affecting restoration of habitat structure within the channelized Kissimmee River ecosystem. Restoration Ecology 3:160-180.

Tyler, C.M. 1996. Relative importance of factors contributing to postfire seedling establishment in maritime chaparral. Ecology 77:2182-2195.

Urbanska, K.M., N.R. Webb, and P.J. Edwards. 1997. Why restoration? Pages 3-7 in K.M. Urbanska, N.R. Webb, and P.J. Edwards, editors. Restoration ecology and sustainable development. University Press, Cambridge, United Kingdom.

van Aarde, R.J., A.M. Smit, and A.S. Claassens. 1998. Soil characteristics of rehabilitating and unmined coastal dunes at Richards Bay, KwaZulu-Natal, South Africa. Restoration Ecology 6:102-110.

van Aarde, R.J., S.M. Ferreira, J.J. Kritzinger, P.J. van Dyk, M. Vogt, and T.D. Wassenaar. 1996. An evaluation of habitat rehabilitation of coastal dune forests in northern Kwazulu Natal, South Africa. Restoration Ecology 4:334-345.

Van der Heyden, F., and W.D. Stock. 1996. Regrowth of a semiarid shrub following simulated browsing: the role of reserve carbon. Functional Ecology 10:647-653.

Van der Moezel, P.G., W.A. Loneregan, and D.T. Bell. 1987. Northern Sandplain Kwongan: Regeneration following fire, juvenile period and flowering phenology. Journal of the Royal Society of Western Australia 69:123-132.

Varelides, C., and T. Kritikos. 1995. Effect of site preparation intensity and fertilisation on *Pinus pinaster* survival and height growth on three sites in northern Greece. Forest Ecology and Management 73:111-115.

Vigilante, T. 1997. Seed dispersal and grazing by kangaroos along habitat boundaries of rehabilitated bauxite pits. Pages 56-69 in 22nd Annual Environmental Workshop. Minerals Council of Australia, Canberra.

Vitousek, P.M. 1990. Biological invasions and ecosystem processes: towards an integration of population ecology and ecosystem studies. Oikos 57:7-13.

Vitousek, P.M., H.A. Mooney, J. Lubchenco, and J.M. Mellilo. 1997. Human domination of earth's ecosystems. Science 277:494-499.

Wali, M.K. 1999. Ecological succession and the rehabilitation of disturbed terrestrial ecosystems. Plant and Soil 213:195-220.

Ward, S.C., J.M. Koch, and G.L. Ainsworth. 1996. The effect of timing of rehabilitation procedures on the establishment of a jarrah forest after bauxite mining. Restoration Ecology 4:19-24.

Ward, S.C., J.M. Koch, and O.G. Nichols. 1990. Bauxite mine rehabilitation in the Darling Range, Western Australia. Proceedings of the Ecological Society of Australia 16:557-565.

Western Australian Herbarium. 1998+. FloraBase — The Western Australian Flora. Department of Environment and Conservation. URL http://florabase.calm.wa.gov.au/.

Whelan, R.J. 1995. The Ecology of Fire. Cambridge University Press, Cambridge.

Whittaker, R.H., W.A. Niering, and M.D. Crisp. 1979. Structure, pattern and diversity of a mallee community in New South Wales. Vegetatio 39:65-76.

Wilkins, S., D.A. Keith, and P. Adam. 2003. Measuring success: evaluating the restoration of a grassy Eucalypt woodland on the Cumberland Plain, Sydney, Australia. Restoration Ecology 11:489-503.

Witkowski, E.T.F. 1991. Effects of invasive alien acacias on nutrient cycling in the coastal lowlands of the Cape fynbos. Journal of Applied Ecology 28:1-15.

Witkowski, E.T.F., B.B. Lamont, and S.J. Connell. 1991. Seed bank dynamics of three co-occurring Banksias in south coastal Western Australia: the role of plant age, cockatoos, senescence and interfire establishment. Australian Journal of Ecology 39:385-397.

Woodward, A., K.J. Jenkins, and E.G. Schreiner. 1999. The role of ecological theory in long-term ecological monitoring: report on a workshop. Natural Areas Journal 19:223-233.

Yang, J., S. He. Hong, and S.R. Shifley. 2008. Spatial controls of occurrence and spread of wildfires in the Missouri Ozark highlands. Ecological Applications 18:1212-1225.

Zedler, P.H., and C. Zammit. 1989. A population-based critique of concepts of change in the chaparral. Pages 73-83 in S.C. Keeley, editor. The California Chaparral: Paradigms Reexamined. Natural History museum of Los Angeles County, Los Angeles. C.A.

Zedler, J.B., and J.C. Callaway. 1999. Tracking wetland restoration: Do mitigation sites follow desired trajectories? Restoration Ecology 7:69-73.

Zedler, P.H., R.G. Clayton, and S. McMaster. 1983. Vegetation response to extreme events: the effect of a short interval between fires in California chaparral and coastal scrub. Ecology 64:809-818.

Zimov, S.A., V.I. Chuprynin, A.P. Oreshko, I.F.S. Chapin, J.F. Reynolds, and M.C. Chapin. 1995. Steppe-tundra transition: a herbivore driven biome shift at the end of Pleistocene. The American Naturalist 146:765-794.

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APPENDICES

Appendix A: Pre-fire master list (actual cover values) for Chapters 2 and 3. Species nomenclature follows Paczkowska and Chapman 2000. WA Wildflower Society, Perth. R = Regeneration mode, G = Growth form, SB = seedbank storage mode, r = resprouter, n = nonsprouter, sh = shrub – woody plant plus grasstrees 1–2.5 m tall on average , ss = sub-shrub – woody plant <1 m tall on average, gm = graminoid or grasslike plant, hb = herb, p = parasite, S = soil seedbank storage, C = canopy seedbank storage, blank space = 0, ? = unknown. Description of how cover values were calculated is given in chapter 2 (under heading 'Methods').

Species	**Family**	**High**	**Low**	**Swale**	**Lat**	**Lim**	**R8**	**R15**	**R16**	**R24**	**R**	**G**	**SB**
Acacia auronitens	Mimosaceae							0.41	0.01		r	ss	S
Acacia barbinervis subsp. *borealis*	Mimosaceae	0.04	0.63	0.10					0.04	0.08	r	ss	S
Acacia blakelyi	Mimosaceae		0.25				36.44	6.75	12.16	10.18	n	sh	S
Acacia fagonioides	Mimosaceae		0.03	0.16	0.03				1.50		r	ss	S
Acacia pulchella	Mimosaceae						0.31	1.28	0.21	5.63	n	ss	S
Acacia rostellifera	Mimosaceae							0.10	0.22		r	sh	S
Acacia saligna	Mimosaceae						0.05				r	sh	S
Acacia spathulifolia	Mimosaceae					10.49					n	sh	S
Acanthocarpus canaliculatus	Dasypogonaceae							0.29	0.05	0.57	r	gm	S
Acanthocarpus preissii	Dasypogonaceae				0.00	0.30					r	gm	S
Actinostrobus acuminatus	Cupressaceae	1.68		0.09					0.00		r	ss	C
Actinostrobus arenarius	Cupressaceae								0.00		n	ss	C
Adenanthos cygnorum subsp. *cygnorum*	Proteaceae	8.96					4.81	2.75	11.01		n	sh	S
Alexgeorgea nitens	Restionaceae	0.56	0.06	0.33							r	gm	S
Allocasuarina huegeliana	Casuarinaceae						2.00				r	sh	C
Allocasuarina humilis	Casuarinaceae	0.55	1.16	0.35	0.08	0.05	0.53	0.44	0.37	2.69	r	ss	C
Allocasuarina microstachya	Casuarinaceae			0.96	3.86			0.10	0.05	0.40	r	ss	C
Amphipogon aff. *strictus*	Poaceae					0.23					r	gm	S

Amphipogon 'fine'	Poaceae	0.05									r	gm	S
Amphipogon 'hairy'	Poaceae				0.07						r	gm	S
Amphipogon sp.	Poaceae					0.03					r	gm	S
Amphipogon strictus	Poaceae	0.61		0.13	0.02			0.10	0.03	0.01	r	gm	S
Amphipogon turbinatus	Poaceae		0.06	0.02	0.66			0.05	0.24	0.39	r	gm	S
Anarthria polyphylla	Restionaceae			0.00							r	gm	S
Andersonia heterophylla	Epacridaceae	2.31	0.41	0.36				0.05	0.04		r	ss	S
Anigozanthos humilis	Haemodoraceae	0.00				0.02					r	gm	S
Arnocrinum gracillimum	Anthericaceae							0.10	0.05		r	hb	S
Astroloma ciliatum	Epacridaceae							0.10	0.00		n	ss	S
Astroloma microdonta	Epacridaceae		0.03			0.03		0.05	0.02	0.19	r	ss	S
Astroloma pallidum	Epacridaceae		0.03		0.01			0.10	0.19	0.01	r	ss	S
Astroloma xerophyllum	Epacridaceae	0.02	0.63	0.85			0.03	0.05			n	ss	S
Austrostipa compressa	Poaceae							0.05	0.09	0.06	n	gm	S
Baeckea camphorosmae	Myrtaceae			0.05	1.92						r	ss	S
Banksia attenuata	Proteaceae	13.19	15.63				6.28		0.37		r	sh	C
Banksia candolleana	Proteaceae	3.75	10.97				0.22	0.10			r	ss	C
Banksia grossa	Proteaceae						2.00			0.21	r	sh	C
Banksia hookeriana	Proteaceae	15.19							3.95	1.37	n	sh	C
Banksia lanata	Proteaceae	0.20					6.75				n	ss	C
Banksia leptophylla var. *melletica*	Proteaceae					8.77	6.28	16.50	14.04		n	ss	C
Banksia menziesii	Proteaceae	1.67	0.03								r	sh	C
Beaufortia elegans	Myrtaceae	4.12	2.75	4.50			0.44	1.30	1.38	2.77	n	ss	C
Boronia cymosa	Rutaceae				0.15						r	ss	S
Boronia ramosa subsp. *anethifolia*	Rutaceae					0.00			0.01	0.06	r	ss	S
Boronia sp. hairy	Rutaceae									0.00	n	ss	S
Bossiaea eriocarpa	Papilionaceae					0.57					n	ss	S
Burchardia umbellata	Colchicaceae			0.00						0.00	r	hb	S
Calectasia narragara	Dasypogonaceae	0.18	0.63	0.21	0.02						r	ss	S
Calothamnus hirsutus	Myrtaceae	0.11	1.75	0.64	0.75		0.63	1.75	1.37	2.19	r	ss	C
Calothamnus longissimus	Myrtaceae				4.10						r	ss	C
Calothamnus quadrifidus	Myrtaceae					5.54		0.10	1.13		r	ss	C

Calothamnus torulosus	Myrtaceae			0.27				1.03	0.08	0.17	r	ss	C
Calytrix brevifolia	Myrtaceae		0.22								r	ss	S
Calytrix depressa	Myrtaceae			1.65	0.45						r	ss	S
Calytrix flavescens	Myrtaceae	0.50					0.03	1.22	0.17		n	sh	S
Calytrix fraseri	Myrtaceae	0.12						0.05	0.00		r	ss	S
Calytrix sapphirina	Myrtaceae	0.16									n	ss	S
Calytrix sp.	Myrtaceae					0.17					n	ss	S
Calytrix sp.1	Myrtaceae		0.22								n	ss	S
Calytrix superba	Myrtaceae	0.61	0.04	1.03							r	ss	S
Cassytha aurea	Lauraceae		0.25	0.39	0.05	0.02		0.10	0.19		n	p	S
Cassytha flava	Lauraceae			0.60							n	p	S
Cassytha glabella	Lauraceae		0.10	0.90	0.56	0.99		0.10	0.27	0.45	n	p	S
Cassytha 'hairy'	Lauraceae						0.39				n	p	S
Cassytha pubescens	Lauraceae		8.20				3.70	8.50	0.38	7.77	n	p	S
Cassytha sp.	Lauraceae	0.14									n	p	S
Caustis dioica	Cyperaceae		0.97	2.64	0.92			0.15	0.09	2.10	r	gm	S
Comesperma confertum	Polygalaceae			0.02							n	ss	S
Comesperma sp. (blue)	Polygalaceae					0.04					n	ss	S
Commersonia pulchella	Tremandraceae				0.08						r	ss	S
Conospermum incurvum	Proteaceae	0.06	4.31					0.05	0.04	0.03	n	ss	S
Conospermum sp.1	Proteaceae							0.10			r	ss	S
Conospermum stoechedis	Proteaceae					1.52					r	ss	S
Conospermum wycherleyi	Proteaceae	2.54	8.47	0.53	0.02		1.03	1.44	0.75	4.95	r	ss	S
Conostephium magnum	Epacridaceae									0.08	r	ss	S
Conostephium pendulum	Epacridaceae		0.22					0.05	0.00	0.46	r	ss	S
Conostylis aculeata	Haemodoraceae								0.09		n	gm	S
Conostylis androstemma	Haemodoraceae				0.15					0.05	r	gm	S
Conostylis aurea	Haemodoraceae	0.78	2.16	0.09	0.01		0.03	0.79	0.10	0.02	r	gm	S
Conostylis candicans	Haemodoraceae		0.22			3.71					r	gm	S
Conostylis crassinervia subsp *absens*	Haemodoraceae	0.36	2.16	0.17				0.05	0.02		n	gm	S
Conostylis neocymosa	Haemodoraceae							0.78	3.38	1.62	n	gm	S
Conostylis setigera	Haemodoraceae	0.08	0.06	0.01				0.44		0.01	n	gm	S

Conostylis sp.4 (hairy)	Haemodoraceae	0.18	2.16	0.20	0.01				0.01	0.04	n	gm	S
Conothamnus trinervis	Myrtaceae			2.07			0.31		0.01		r	ss	C
Corynotheca micrantha	Anthericaceae	0.16									r	hb	S
Cristonia biloba	Papilionaceae				0.20				0.00	0.01	r	ss	S
Cryptandra myriantha	Rhamnaceae				0.00					0.02	r	ss	S
Cryptandra sp. linear	Rhamnaceae								0.01		r	ss	S
Cryptandra sp. long	Rhamnaceae								0.00		r	ss	S
Cryptandra sp.1	Rhamnaceae					0.29					r	ss	S
Cyperaceae sp.1	Cyperaceae				2.20				0.00		r	gm	S
Cyperaceae sp.2	Cyperaceae				0.03						r	gm	S
Dampiera carinata	Goodeniaceae		0.06	0.03	0.69			0.10	0.00	0.06	r	hb	S
Dampiera spicigera	Goodeniaceae			0.32	0.17				0.03		r	hb	S
Danthonia sp.	Poaceae						0.05	0.05		0.05	n	gm	S
Darwinia neildiana	Myrtaceae	0.71	0.69	0.88		0.21	0.05	0.10	0.24	0.20	r	ss	S
Darwinia speciosa	Myrtaceae	0.04	1.53	0.18	0.10					0.01	r	ss	S
Davesia triflora	Papilionaceae		1.38								r	ss	S
Daviesia decurrens	Papilionaceae				0.11					0.09	r	ss	S
Daviesia divaricata	Papilionaceae	1.19	0.03				0.41	0.20	0.00	0.01	r	ss	S
Daviesia nudiflora	Papilionaceae	0.20	2.25	1.30						0.11	r	ss	S
Daviesia pedunculata	Papilionaceae			1.12							r	ss	S
Daviesia quadrilatera	Papilionaceae	0.32							0.15	0.52	r	ss	S
Daviesia triflora	Papilionaceae	1.69	1.38					0.15	0.10		r	ss	S
Desmocladus 'flat'	Restionaceae								0.05		r	ss	S
Desmocladus parthenicus	Restionaceae								0.02		r	ss	S
Desmocladus semiplanus	Restionaceae	1.69	10.44	0.08		4.93	0.15	0.10	0.12	0.28	r	gm	S
Dianella revoluta var. *divaricata*	Phormiaceae					0.03					r	gm	S
Diplolaena angustifolia	Rutaceae									0.17	r	ss	S
Diplopeltis huegelii subsp. *subintegra*	Sapindaceae					1.06					r	ss	S
Drosera menziesii	Droseraceae	0.03									r	hb	S
Drosera sp.	Droseraceae			0.02							r	hb	S
Dryandra bipinnatifida subsp. *multifida*	Proteaceae			0.04							r	ss	C
Dryandra carlinoides	Proteaceae			0.29	1.32		0.05	4.00	0.02	0.11	n	ss	C

Dryandra falcata	Proteaceae						9.16				r	sh	C
Dryandra fraseri	Proteaceae				1.33						r	ss	C
Dryandra lindleyana subsp. Media	Proteaceae		1.75			0.18	0.29	0.74	0.02		r	ss	C
Dryandra nobilis	Proteaceae	0.01			0.09						r	ss	C
Dryandra polycephala	Proteaceae						0.05				n	ss	C
Dryandra sessilis var. *cygnorum*	Proteaceae					1.43				0.39	n	ss	C
Dryandra shuttleworthiana	Proteaceae	0.14	2.66	3.30	1.56		1.96	3.25	1.02	0.31	r	ss	C
Dryandra stenoprion	Proteaceae	0.02		0.42				0.10	0.10	0.03	r	ss	C
Dryandra tortifolia	Proteaceae	0.36	0.08						0.00	0.05	r	ss	C
Dryandra tridentata	Proteaceae			0.68			0.10	0.29		0.04	r	ss	C
Ecdeiocolea monostachya	Ecdeiocoleaceae			5.92	11.30			0.25	0.15	18.10	r	gm	S
Eremaea acutifolia	Myrtaceae							0.25		0.03	r	ss	C
Eremaea beaufortioides	Myrtaceae	5.56	4.00	0.75	0.78		0.05	2.84	1.42	3.52	r	ss	C
Eremaea ebracteata var. *ebracteata*	Myrtaceae	1.00		0.34				0.05	0.06	0.15	r	ss	C
Eremaea pauciflora	Myrtaceae		0.03				0.05				r	ss	C
Eremaea violacea subsp. *violacea*	Myrtaceae	0.03	0.74	1.20			0.05	0.72	0.21	1.87	r	ss	C
Eucalyptus megacarpa	Myrtaceae							0.10	0.25		r	sh	C
Eucalyptus tetragona	Myrtaceae						3.38	0.20	0.78	0.54	r	sh	C
Eucalyptus todtiana	Myrtaceae		2.94							0.21	r	sh	C
Gastrolobium acutum	Proteaceae				0.03				0.34		r	ss	S
Gastrolobium capitatum	Proteaceae			0.00	0.41	1.40		0.10	0.13	2.07	r	ss	S
Geleznowia verrucosa	Rutaceae								0.00	0.01	n	ss	S
Georgeantha hexandra	Ecdeiocoleaceae			0.80	5.69					0.03	r	gm	S
Gompholobium knightianum	Papilionaceae							0.97		0.10	n	ss	S
Gompholobium purpureum	Papilionaceae								0.00		n	ss	S
Gompholobium shuttleworthii	Papilionaceae	0.01									n	ss	S
Gompholobium tomentosum	Papilionaceae					3.82		0.20	0.88	0.31	n	ss	S
Goodenia filiformis	Goodeniaceae		0.03	0.13				0.15			r	hb	S
Grevillea eriostachya	Proteaceae	0.12	0.78				0.06	0.10	0.77	0.54	r	ss	S
Grevillea integrifolia	Proteaceae								0.41	0.00	r	ss	S
Grevillea thelemanniana	Proteaceae					0.61					r	ss	S
Grevillea vestita subsp. *isopogoides*	Proteaceae					0.07					n	ss	S

Haemodorum simplex	Haemodoraceae			0.04	0.05						r	gm	S
Haemodorum spicatum	Haemodoraceae			0.02							r	gm	S
Hakea auriculata	Proteaceae							0.10		0.04	r	ss	C
Hakea candolleana	Proteaceae			0.13	0.01		0.10	0.09	0.11		r	ss	C
Hakea conchifolia	Proteaceae						0.53				r	sh	C
Hakea costata	Proteaceae			0.27		1.99	0.25	1.22	0.69	0.19	r	ss	C
Hakea eneabba	Proteaceae	0.48	0.10	0.39					0.04	0.01	r	ss	C
Hakea flabellifolia	Proteaceae						0.05	0.03	0.47		r	ss	C
Hakea incrassata	Proteaceae			0.49	2.18	0.26	0.05	0.10	0.33	1.74	r	ss	C
Hakea lissocarpha	Proteaceae				0.34	0.21					r	ss	C
Hakea polyanthema	Proteaceae	0.45	10.81	0.59						0.15	n	sh	C
Hakea prostrata	Proteaceae			0.07		0.02	0.10	0.15	0.05	0.08	r	ss	C
Hakea psilorrhyncha	Proteaceae	1.34							0.02	0.04	n	sh	C
Hakea smilacifolia	Proteaceae						0.20				n	sh	C
Hakea spathulata	Proteaceae				0.56						r	ss	C
Hakea stenocarpa	Proteaceae				0.05		0.05				r	ss	C
Hakea trifurcata	Proteaceae					5.68	0.05	0.10	0.60		n	sh	C
Hemiandra pungens	Lamiaceae	0.22	1.03	0.07					0.02	0.00	r	ss	S
Hemigenia sp.	Lamiaceae						0.98				r	ss	S
Hibbertia crassifolia	Dilleniaceae	0.73	1.09	1.23	2.71		0.10	1.41	0.27	1.23	r	ss	S
Hibbertia huegelii	Dilleniaceae							0.88	0.02	0.52	r	ss	S
Hibbertia aff. *hypericoides*	Dilleniaceae	10.09	1.77	2.85	3.50	14.73	0.44	0.98	0.33	2.25	r	ss	S
Hibbertia sp.3	Dilleniaceae				0.20						r	ss	S
Hibbertia spicata subsp. *spicata*	Dilleniaceae		0.47		0.45	4.34			0.00		r	ss	S
Hibbertia subvaginata	Dilleniaceae						0.81	0.05	0.13	1.06	n	ss	S
Hovea pungens	Papilionaceae				0.08						n	ss	S
Hybanthus calycinus	Violaceae					0.04					r	hb	S
Hybanthus floribundus subsp. Hill River	Violaceae				0.01						n	hb	S
Hypocalymma xanthopetalum	Myrtaceae	0.09	0.91	0.19	0.04			0.10	0.00	0.05	r	ss	S
Isopogon adenanthoides	Proteaceae								0.01	0.05	n	ss	C
Isopogon divergens	Proteaceae				1.88						r	ss	C
Isopogon tridens	Proteaceae	0.15	2.06	1.17			0.05	0.39	0.61	0.42	r	ss	C

Isotropis cuneatus	Papilionaceae		0.02			0.02					r	ss	C
Isotropis cuneifolia subsp. *cuneifolia*	Papilionaceae					0.06					r	ss	S
Jacksonia sp.1	Papilionaceae							3.52			r	ss	S
Jacksonia calcicola	Papilionaceae								0.00			ss	S
Jacksonia fasciculata	Papilionaceae					6.99					n	sh	S
Jacksonia floribunda	Papilionaceae	1.45	3.59	0.04			1.56	2.44	1.33	4.08	r	ss	S
Jacksonia hakeoidies	Papilionaceae		0.38								r	ss	S
Jacksonia restioides	Papilionaceae	0.01		0.21	0.01						r	ss	S
Johnsonia pubescens	Anthericaceae	0.13								0.00	n	hb	S
Kennedia prostrata	Papilionaceae									0.00	n	ss	S
Labichea cassioides	Caesalpiniaceae					8.70		0.06			r	ss	S
Lachnostachys ferruginea	Lamiaceae.		0.10					0.10		0.10	r	ss	S
Lambertia multiflora	Proteaceae	0.00					4.00		0.51	2.28	r	ss	C
Lasiopetalum drummondii	Sterculiaceae	2.20		0.02			0.05			0.10	r	ss	S
Laxmannia omnifertilis	Anthericaceae	0.02									n	hb	S
Laxmannia sessiliflora subsp. *drummondii*	Anthericaceae					0.26		0.20	0.03	0.01	n	hb	S
Lechenaultia linarioides	Goodeniaceae					0.40					r	hb	S
Lepidobolus chaetocephalus	Restionaceae								0.24	0.23	r	gm	S
Lepidobolus preissianus subsp. *preissianus*	Restionaceae	4.68		5.96	0.40		0.05		0.08	0.04	r	gm	S
Lepidobolus sp.	Restionaceae					1.19					r	gm	S
Lepidosperma ‘flat’	Cyperaceae					0.16					r	gm	S
Lepidosperma 2	Cyperaceae	0.68									r	gm	S
Lepidosperma 'branched'	Cyperaceae						0.20	0.15			r	gm	S
Lepidosperma 'cluster'	Cyperaceae		1.31				0.05	0.03			r	gm	S
Lepidosperma 'fine'	Cyperaceae		0.84				0.05	0.72	0.03		r	gm	S
Lepidosperma 'flat'	Cyperaceae				0.09			1.06	1.01	0.27	r	gm	S
Lepidosperma 'hairy tuff'	Cyperaceae			0.50							r	gm	S
Lepidosperma 'round'	Cyperaceae		0.06		0.05	1.23	0.10		0.31	0.73	r	gm	S
Lepidosperma scabrum	Cyperaceae	0.90		1.25					0.04		r	gm	S
Lepidosperma sp.2	Cyperaceae							0.15			r	gm	S
Lepidosperma sp.3	Cyperaceae			1.72							r	gm	S
Lepidosperma 'spikey'	Cyperaceae						0.20	0.05	0.18		r	gm	S

Lepidosperma 'spiney'	Cyperaceae							0.29			r	gm	S
Lepidosperma 'wirey'	Cyperaceae			0.50							r	gm	S
Leptospermum oliganrum	Myrtaceae			0.38	0.03		0.15	0.10	0.24	2.19	r	ss	S
Leptospermum spinescens	Myrtaceae	0.12	1.44	0.15	0.01		0.64	0.10	0.07	0.49	r	ss	C
Leucopogon sp.6	Epacridaceae	0.02									n	ss	S
Leucopogon aff. *hispidus*	Epacridaceae	1.09									n	ss	S
Leucopogon aff. *tenuis*	Epacridaceae	0.01									r	ss	S
Leucopogon biflor	Epacridaceae								0.07	0.04	n	ss	S
Leucopogon ciliatum	Epacridaceae								0.03		n	ss	S
Leucopogon conostephioides	Epacridaceae	0.71	0.05	0.11				0.15			n	ss	S
Leucopogon gracillimus (recurved ericoid)	Epacridaceae	0.79	0.22	0.18		0.45		0.39	0.07	0.21	n	ss	S
Leucopogon hispidus	Epacridaceae			0.02						0.03	n	ss	S
Leucopogon oxycedrus	Epacridaceae					0.09					n	ss	S
Leucopogon 'small'	Epacridaceae									0.08	n	ss	S
Leucopogon sp.	Epacridaceae		1.57								n	ss	S
Leucopogon sp.3	Epacridaceae					0.28					n	ss	S
Leucopogon sp.5	Epacridaceae					0.02					n	ss	S
Lobelia gibbosa	Lobeliaceae							0.10			n	hb	S
Lobelia sp.	Lobeliaceae									0.00	n	hb	S
Lomandra hastilis	Dasypogonaceae	0.04	0.22	0.21		0.00				0.03	r	gm	S
Lomandra micrantha	Dasypogonaceae	0.14		0.48	0.00	0.26			0.02	0.05	r	gm	S
Lomandra sp.3	Dasypogonaceae			0.01	0.02						r	gm	S
Lomandra sp.4	Dasypogonaceae				0.00						r	gm	S
Lomandra 'spiral'	Dasypogonaceae									0.00	r	gm	S
Lyginia barbata	Restionaceae	3.16	0.05	0.93							r	gm	S
Lyginia imberbis	Restionaceae		0.15		0.05				0.60	0.15	r	gm	S
Lyginia sp.1	Restionaceae							0.10			r	gm	S
Lysinema ciliatum	Epacridaceae	0.17	2.28	0.02			0.29		0.02	0.06	n	ss	S
Melaleuca aff. *leuropoma*	Myrtaceae						1.50	1.00	2.00	1.00	n	sh	C
Melaleuca leuropoma	Myrtaceae	4.74	15.50	4.11	2.09	9.40	30.00	24.40	8.94	12.45	r	ss	C
Melaleuca 'ovate'	Myrtaceae									0.04	r	sh	C
Melaleuca scabra	Myrtaceae				3.27			0.05	0.00	0.11	r	ss	C

Melaleuca trichophylla	Myrtaceae			1.30	2.04		1.91	3.25	0.51	1.96	r	ss	C
Melaleuca uncinata	Myrtaceae								0.04		r	sh	C
Melaleuca viminea	Myrtaceae								0.79		r	ss	C
Mesomelaena pseudostygia	Cyperaceae	2.59	1.97		0.99	0.61	0.20	0.29	0.26	3.09	r	gm	S
Mesomelaena stygia subsp. *deflexa*	Cyperaceae	0.35	0.22	4.90				0.25	0.01	0.21	r	gm	S
Mesomelaena tetragona	Cyperaceae		2.16	1.16	0.15			0.25	0.02	1.18	r	gm	S
Mirbelia sp.	Papilionaceae				0.00			0.15	0.00		r	ss	S
Monotaxis bracteata	Euphorbiaceae	0.03		0.01						0.02	r	hb	S
Nemcia capitata	Papilionaceae		0.41			1.30	0.10	1.41			r	ss	S
Neurachne alopecuroidea	Poaceae					0.01					r	hb	S
Nuytsia floribunda	Loranthaceae								0.04		r	sh	NS
Olax benthamiana	Olacaceae		1.27								n	ss	S
Opercularia sp.	Rubiaceae		0.09								r	hb	S
Opercularia spermacocea	Rubiaceae					0.15					r	hb	S
Opercularia vaginata	Rubiaceae				0.25	0.15					r	hb	S
Patersonia juncea	Iridaceae								0.00	0.00	n	gm	S
Patersonia occidentalis	Iridaceae					0.18					r	ss	C
Patersonia sp.	Iridaceae				0.02				0.00		r	gm	S
Pericalymma erubescens	Myrtaceae		0.98								r	sh	S
Persoonia ‘flat’	Proteaceae						0.49		0.00		r	ss	S
Persoonia sp. Twisty	Proteaceae									0.00	r	ss	S
Petrophile brevifolia	Proteaceae	0.29	0.69	1.00	0.28	0.01	0.29	5.97	2.62	0.78	r	ss	C
Petrophile drummondii	Proteaceae	4.37	3.00	0.05			7.81	3.53	1.62	0.40	n	ss	C
Petrophile linearis	Proteaceae		0.10	0.07			0.09				r	ss	C
Petrophile macrostachya	Proteaceae	0.46	3.14	0.46	0.08		0.05			0.29	r	ss	C
Petrophile rigida	Proteaceae				0.93						r	ss	C
Petrophile scabriuscula	Proteaceae			0.51				0.10		0.07	n	ss	C
Petrophile serruriae	Proteaceae		0.10			0.11	0.05	0.05			n	ss	C
Philotheca spicata	Rutaceae						0.05				n	ss	S
Phyllanthus sp.	Euphorbiaceae				0.03						r	ss	S
Phymatocarpus porphyrocephalus	Myrtaceae	0.55						0.25	0.21	0.21	r	ss	C
Pileanthus filifolius	Myrtaceae	4.24	0.10	2.39	0.01		0.10	0.05		0.02	r	ss	S

Pimelea leucantha	Thymeliaceae	0.02	0.10	0.04	0.05						r	ss	S
Pimelea sulphurea	Thymeliaceae			0.38							r	ss	S
Pityrodia bartlingii	Lamiaceae	0.02									r	ss	S
Pterostylis nana	Orchidaceae	0.01									n	gm	S
Ptilotus sp.	Amaranthaceae					0.15					n	ss	S
Restio sp. (white)	Restionaceae			0.27							r	gm	S
Restio sp.1	Cyperaceae	0.39		0.38						0.01	r	gm	S
Restio 'spike'	Restionaceae								0.18	0.07	r	gm	S
Scaevola canescens	Goodeniaceae	0.00	1.47		0.18	2.61		0.10	0.01	0.01	r	hb	S
Scaevola eneabba	Goodeniaceae					1.20					n	ss	S
Schoenus sp.1	Cyperaceae	0.02	0.39	0.31	1.10	0.99	0.05	0.25	0.02	0.71	r	gm	S
Schoenus subbulbosus	Cyperaceae	0.01		0.26	0.04			0.10	0.00	0.01	r	gm	S
Scholtzia capitata	Myrtaceae		0.59	0.10			0.05	0.05	1.05	0.14	r	ss	S
Scholtzia involucrata	Myrtaceae	0.83	2.09	0.95	2.39		0.13	0.98	0.02		r	ss	S
Scholtzia laxiflora	Myrtaceae								0.55	2.13	r	ss	S
Scholtzia umbellifera	Myrtaceae					1.37					r	ss	S
Sphaerolobium sp.	Euphorbiaceae									0.00	n	ss	S
Stachystemon axillaris	Euphorbiaceae	0.01							0.02	0.00	r	ss	S
Stenanthemum notiale subsp. *chamelum*	Rhamnaceae					0.18			0.00		n	ss	S
Stenanthemum pomaderroides	Rhamnaceae	0.01									n	ss	S
Sterculiaceae sp.1	Sterculiaceae				0.02						r	ss	S
Stirlingia latifolia	Proteaceae	0.55	1.77					0.10	0.14		r	ss	S
Stylidium adpressum	Stylidiaceae			0.01							n	hb	S
Stylidium brunonianum	Stylidiaceae					0.03			0.00		r	hb	S
Stylidium ciliatum aff. *piliferum*	Stylidiaceae				0.00						n	hb	S
Stylidium crossocephalum	Stylidiaceae	0.18	0.44	0.10	0.00	0.02	0.10	0.10	0.00	0.00	n	hb	S
Stylidium flagellum	Stylidiaceae		0.25			0.46		0.10			n	hb	S
Stylidium junceum	Stylidiaceae	0.01									n	hb	S
Stylidium kalbarriense	Stylidiaceae					0.00					n	hb	S
Stylidium repens	Stylidiaceae	0.04		0.05	0.01				0.07	0.02	n	hb	S
Stylidium sp. hairy	Stylidiaceae				0.00						n	hb	S
Stylidium sp. rosette	Stylidiaceae				0.00						n	hb	S

Synaphea spinulosa	Proteaceae		0.34			0.01			0.03	0.03	r	ss	S
Tetraria octandra	Cyperaceae					2.02		0.05	0.00	0.09	r	gm	S
Thryptomene sp.	Myrtaceae				0.36						r	ss	S
Thysanotus fastigiatus	Anthericaceae	0.27	0.05	0.52	0.41		0.10				r	ss	S
Thysanotus sparteus	Anthericaceae	0.24	0.15	0.11	0.07	0.07				0.03	r	ss	S
Thysanotus triandrus	Anthericaceae			0.01						0.00	r	ss	S
Trachymene coerulea subsp *leucopetala*	Apiaceae					0.00					n	hb	S
Trachymene leucopetala	Apiaceae							0.29			n	hb	S
Tricoryne elatior	Anthericaceae	0.01		0.05	0.03	0.20				0.02	r	hb	S
Tricoryne-like	Anthericaceae									0.00	r	hb	S
Triodia longipalea	Poaceae					0.16					r	gm	S
Tripterococcus brunonis	Stackhousiaceae									0.00	n	hb	S
Unknown R15	?							0.34					
Unknown R16	?							0.05					
Unknown R17	?							0.05					
Unknown R18	?							0.05					
Unknown R19	?							0.10					
Unknown R20	?							0.05					
Unknown sp R1	?									0.04			
Unknown sp R2	?									0.13			
Unknown sp R3	?									0.23			
Unknown sp R4	?									0.04			
Unknown sp.1	?	0.00									?	hb	?
Unknown sp.3	?	0.00									n	?	?
Unknown sp.4	?	0.00									?	?	?
Unknown sp.5	?	0.00									n	?	?
Unknown sp.8	?		0.39								?	?	?
Verreauxia reinwardtii	Goodeniaceae							0.05	0.49		r	sh	S
Verticordia 7	Myrtaceae	0.01									n	ss	S
Verticordia argentea	Myrtaceae	0.13					0.05	0.44			n	sh	S
Verticordia aurea	Myrtaceae							0.34	0.18	0.11	n	ss	S
Verticordia 'bushy'	Myrtaceae									0.13	?	sh	S

Verticordia chrysanthella	Myrtaceae				0.20						r	ss	S
Verticordia densiflora subsp *caespitosa*	Myrtaceae	0.09	1.69	1.15	0.08		0.05	0.10	0.10	0.14	r	ss	S
Verticordia grandis	Myrtaceae	2.37	1.69				0.15	0.05			r	ss	S
Verticordia monodelpha	Myrtaceae	0.01									r	ss	S
Verticordia nobilis	Myrtaceae	0.01	0.20				0.05				r	ss	S
Verticordia ovalifolia	Myrtaceae	0.13									n	ss	S
Verticordia pennigera	Myrtaceae				0.27						r	ss	S
Xanthorrhoea acanthostachya	Xanthorrhoeaceae	0.07		0.90	0.27			0.10	0.01		r	ss	C
Xanthosia huegelii	Apiaceae	0.01		0.02		0.08				0.09	n	hb	S
Xylomelum angustifolium	Proteaceae		1.22								r	sh	C
348 spp.	Total cover	125	169	79	68	115	143	122	90	120			

Appendix B: Post-fire master list (presence = 1, absence = blank) for Chapter 3. R = Regeneration mode, SB = seedbank storage mode, r = resprouter, n = nonsprouter, S = soil seedbank storage, C = canopy seedbank storage, ? = unknown. N1 = High dune, N2 = Swale, N3 = Low dune, N4 = Laterite, N5 = Limestone, R1 = R8, R2 = R15, R3 = R24.

Species	**Family**	**N1**	**N2**	**N3**	**N4**	**N5**	**R1**	**R2**	**R3**	**R**	**SB**
Acacia barbinervis subsp. *borealis*	Mimosaceae	1	1	1					1	r	S
Acacia blakelyi	Mimosaceae			1			1	1	1	n	S
Acacia fagonioides	Mimosaceae		1		1				1	r	S
Acacia pulchella	Mimosaceae							1	1	n	S
Acacia spathulifolia	Mimosaceae					1				n	S
Acanthocarpus canaliculatus	Dasypogonaceae								1	r	S
Acanthocarpus preissii	Dasypogonaceae				1	1				r	S
Actinostrobus acuminatus	Cupressaceae	1	1		1					r	C
Adenanthos cygnorum subsp. *cygnorum*	Proteaceae	1					1		1	n	S
Alexgeorgea nitens	Restionaceae	1	1	1						r	S
Allocasuarina humilis	Casuarinaceae	1	1	1	1	1	1		1	r	C
Allocasuarina microstachya	Casuarinaceae		1		1					r	C
Amphipogon aff. *strictus*	Poaceae					1				r	S
Amphipogon 'fine'	Poaceae	1								r	S
Amphipogon 'hairy'	Poaceae				1					r	S
Amphipogon sp.	Poaceae					1				r	S
Amphipogon strictus	Poaceae	1	1	1	1			1	1	r	S
Amphipogon turbinatus	Poaceae	1	1	1	1			1	1	r	S
Anarthria polyphylla	Restionaceae		1							r	S
Andersonia heterophylla	Epacridaceae	1	1	1	1	1		1		r	S
Anigozanthos humilis	Haemodoraceae	1		1		1		1	1	r	S
Anigozanthus manglesii	Haemodoraceae						1			r	S
Astroloma ciliatum	Epacridaceae							1		n	S
Astroloma microdonta	Epacridaceae		1	1	1	1				r	S
Astroloma pallidum	Epacridaceae	1	1	1	1				1	r	S
Astroloma xerophyllum	Epacridaceae		1	1	1				1	n	S

Austrostipa compressa	Poaceae						1			n	S
Baeckea camphorosmae	Myrtaceae		1		1					r	S
Banksia attenuata	Proteaceae	1		1			1			r	C
Banksia candolleana	Proteaceae	1		1			1			r	C
Banksia grossa	Proteaceae						1			r	C
Banksia hookeriana	Proteaceae	1		1					1	n	C
Banksia lanata	Proteaceae	1								n	C
Banksia leptophylla var. *melletica*	Proteaceae					1		1		n	C
Banksia menziesii	Proteaceae	1		1						r	C
Beaufortia elegans	Myrtaceae	1	1	1					1	n	C
Boronia cymosa	Rutaceae				1					r	S
Boronia ramosa subsp. *anethifolia*	Rutaceae			1		1			1	r	S
Bossiaea eriocarpa	Papilionaceae					1				n	S
Bracteantha sp.	Colchicaceae					1				r	S
Briza maxima	Poaceae						1			n	S
Burchardia congesta	Colchicaceae				1	1			1	r	S
Burchardia umbellata	Colchicaceae		1						1	r	S
Byblis lamellata	Byblidaceae		1							n	S
Caladenia sp.	Orchidaceae						1	1	1	r	S
Caladenia flava	Orchidaceae						1	1	1	r	S
Calectasia narragara	Dasypogonaceae	1	1	1	1					r	S
Calothamnus hirsutus	Myrtaceae	1	1	1					1	r	C
Calothamnus longissimus	Myrtaceae				1					r	C
Calothamnus quadrifidus	Myrtaceae					1		1		r	C
Calothamnus sanguineus	Myrtaceae		1		1	1				r	C
Calothamnus torulosus	Myrtaceae		1						1	r	C
Calytrix brevifolia	Myrtaceae			1						r	S
Calytrix depressa	Myrtaceae		1		1					r	S
Calytrix eneabbensis	Myrtaceae		1							r	S
Calytrix flavescens	Myrtaceae	1						1		n	S
Calytrix fraseri	Myrtaceae	1								r	S
Calytrix sapphirina	Myrtaceae	1								n	S
Calytrix sp.	Myrtaceae					1				n	S
Calytrix sp.1	Myrtaceae		1							n	S

Calytrix sp.3	Myrtaceae					1				?	S
Calytrix superba	Myrtaceae	1	1							r	S
Carpobrotus sp.	Aizoaceae						1	1	1	n	S
Cassytha aurea	Lauraceae		1					1		n	S
Cassytha glabella	Lauraceae		1							n	S
Cassytha 'noby'	Lauraceae								1	n	S
Caustis dioica	Cyperaceae		1	1	1				1	r	S
Comesperma drummondii	Polygalaceae		1							n	S
Comesperma sp.	Polygalaceae				1					n	S
Commersonia pulchella	Tremandraceae				1					r	S
Conospermum incurvum	Proteaceae	1		1						n	S
Conospermum stoechedis	Proteaceae					1				r	S
Conospermum wycherleyi	Proteaceae	1	1	1	1		1	1	1	r	S
Conostephium pendulum	Epacridaceae			1					1	r	S
Conostylis androstemma	Haemodoraceae				1					r	S
Conostylis aurea	Haemodoraceae	1	1	1	1		1	1	1	r	S
Conostylis candicans	Haemodoraceae	1		1		1				r	S
Conostylis crassinervia subsp *absens*	Haemodoraceae	1		1				1	1	n	S
Conostylis sp.4 (hairy)	Haemodoraceae			1						n	S
Conostylis neocymosa	Haemodoraceae							1	1	n	S
Conostylis setigera	Haemodoraceae	1	1	1				1	1	n	S
Conostylis sp.	Haemodoraceae			1						n	S
Conostylis sp.4 (hairy)	Haemodoraceae	1	1		1					n	S
Conostylis sp.R1	Haemodoraceae							1		n	S
Conostylis sp.1	Haemodoraceae		1		1					n	S
Conostylis sp.2	Haemodoraceae		1							n	S
Conostylis sp.3	Haemodoraceae					1				n	S
Conostylis sp.5	Haemodoraceae					1				n	S
Conothamnus trinervis	Myrtaceae		1			1	1			r	C
Corynotheca micrantha	Anthericaceae	1								r	S
Cristonia biloba	Papilionaceae				1					r	S
Cryptandra myriantha	Rhamnaceae				1	1		1	1	r	S
Cryptandra sp.1	Rhamnaceae					1				r	S
Cyperaceae sp.1	Cyperaceae				1					r	S

Cyperaceae sp.2	Cyperaceae				1					r	S
Dampiera carinata	Goodeniaceae		1	1	1			1	1	r	S
Dampiera sp.	Goodeniaceae								1	r	S
Dampiera spicigera	Goodeniaceae		1		1				1	r	S
Danthonia sp.	Poaceae								1	n	S
Darwinia neildiana	Myrtaceae	1	1	1		1		1	1	r	S
Darwinia speciosa	Myrtaceae	1	1	1	1				1	r	S
Davesia quadrilatera	Papilionaceae	1						1	1	r	S
Davesia triflora	Papilionaceae			1						r	S
Daviesia decurrens	Papilionaceae				1				1	r	S
Daviesia divaricata	Papilionaceae	1		1						r	S
Daviesia nudiflora	Papilionaceae	1	1	1						r	S
Daviesia pedunculata	Papilionaceae		1							r	S
Daviesia triflora	Papilionaceae	1		1						r	S
Desmocladus semiplanus	Restionaceae	1								r	S
Desmocladus semiplanus	Restionaceae		1	1		1	1		1	r	S
Dianella revoluta var. *divaricata*	Phormiaceae					1				r	S
Diplolaena angustifolia	Rutaceae								1	r	S
Diplopeltis huegelii subsp. *subintegra*	Sapindaceae					1				r	S
Dryandra bipinnatifida subsp. *multifida*	Proteaceae		1							r	C
Dryandra carlinoides	Proteaceae		1		1			1	1	n	C
Dryandra falcata	Proteaceae						1			r	C
Dryandra fraseri	Proteaceae				1					r	C
Dryandra lindleyana subsp. *Media*	Proteaceae		1	1	1	1	1	1		r	C
Dryandra nobilis	Proteaceae	1			1					r	C
Dryandra sessilis var. *cygnorum*	Proteaceae					1				n	C
Dryandra shuttleworthiana	Proteaceae	1	1	1	1		1	1	1	r	C
Dryandra stenoprion	Proteaceae	1	1						1	r	C
Dryandra torifolia	Proteaceae	1		1						r	C
Dryandra tridentata	Proteaceae		1						1	r	C
Ecdeiocolea monostachya	Ecdeiocoleaceae		1		1			1	1	r	S
Eremaea beaufortioides	Myrtaceae	1	1	1	1		1	1	1	r	C
Eremaea ebracteata var. *ebracteata*	Myrtaceae	1	1							r	C
Eremaea violacea subsp. *violacea*	Myrtaceae	1	1	1					1	r	C

Eucalyptus megacarpa	Myrtaceae						1	1		r	C
Eucalyptus tetragona	Myrtaceae						1	1	1	r	C
Eucalyptus todtiana	Myrtaceae			1					1	r	C
Gastrolobium capitatum	Proteaceae		1		1	1		1	1	r	S
Geleznowia verrucosa	Rutaceae							1	1	n	S
Georgeantha hexandra	Ecdeiocoleaceae				1					r	S
Gompholobium knightianum	Papilionaceae							1	1	n	S
Gompholobium shuttleworthii	Papilionaceae	1								n	S
Gompholobium tomentosum	Papilionaceae					1		1	1	n	S
Goodenia filiformis	Goodeniaceae	1	1	1						r	S
Grevillea eriostachya	Proteaceae	1		1					1	r	S
Grevillea integrifolia	Proteaceae								1	r	S
Grevillea thelemanniana	Proteaceae					1				r	S
Gyrostemon racemiger	Gyrostemonaceae					1				n	S
Haemodorum simplex	Haemodoraceae	1	1		1					r	S
Haemodorum spicatum	Haemodoraceae		1							r	S
Hakea auriculata	Proteaceae								1	r	C
Hakea candolleana	Proteaceae		1		1		1			r	C
Hakea conchifolia	Proteaceae						1			r	C
Hakea costata	Proteaceae		1			1		1		r	C
Hakea eneabba	Proteaceae	1	1	1						r	C
Hakea flabellifolia	Proteaceae						1			r	C
Hakea incrassata	Proteaceae		1		1	1	1	1	1	r	C
Hakea lissocarpha	Proteaceae				1	1				r	C
Hakea polyanthema	Proteaceae	1	1	1						n	C
Hakea prostrata	Proteaceae		1			1			1	r	C
Hakea psilorrhyncha	Proteaceae	1								n	C
Hakea spathulata	Proteaceae				1					r	C
Hakea stenocarpa	Proteaceae				1		1			r	C
Hakea trifurcata	Proteaceae					1				n	C
Hemiandra pungens	Lamiaceae	1	1	1						r	S
Hemigenia sp.	Lamiaceae						1			r	S
Hibbertia crassifolia	Dilleniaceae	1	1	1					1	r	S
Hibbertia huegelii	Dilleniaceae							1	1	r	S

Hibbertia aff. *hypericoides*	Dilleniaceae	1	1	1	1	1	1		1	r	S
Hibbertia sp.3	Dilleniaceae				1					r	S
Hibbertia spicata subsp. *spicata*	Dilleniaceae		1	1	1	1		1		r	S
Hibbertia subvaginata	Dilleniaceae						1	1	1	n	S
Hovea pungens	Papilionaceae				1					n	S
Hybanthus calycinus	Violaceae					1				r	S
Hypocalymma xanthopetalum	Myrtaceae	1	1	1	1				1	r	S
Isopogon divergens	Proteaceae				1					r	C
Isopogon tridens	Proteaceae	1	1	1				1		r	C
Isotropis cuneatus	Papilionaceae		1	1		1			1	r	C
Isotropis cuneifolia subsp. *cuneifolia*	Papilionaceae				1	1		1		r	S
Jacksonia sp.1	Papilionaceae							1		r	S
Jacksonia fasciculata	Papilionaceae					1				n	S
Jacksonia floribunda	Papilionaceae	1		1			1	1	1	r	S
Jacksonia hakeoidies	Papilionaceae			1				1		r	S
Jacksonia restioides	Papilionaceae	1	1		1					r	S
Jacksonia sp.	Papilionaceae					1				r	S
Johnsonia pubescens	Anthericaceae	1								n	S
Kennedia prostrata	Papilionaceae						1		1	n	S
Lachnostachys ferruginea	Lamiaceae.			1				1		r	S
Lambertia multiflora	Proteaceae	1					1		1	r	C
Lasiopetalum drummondii	Sterculiaceae	1	1						1	r	S
Laxmannia omnifertilis	Anthericaceae	1	1					1	1	n	S
Laxmannia sessiliflora subsp. *drummondii*	Anthericaceae	1	1	1		1	1	1	1	n	S
Laxmannia sp.	Anthericaceae				1					n	S
Lechenaultia linarioides	Goodeniaceae					1		1		r	S
Lepidobolus chaetocephalus	Restionaceae								1	r	S
Lepidobolus preissianus subsp. *preissianus*	Restionaceae	1	1		1				1	r	S
Lepidobolus sp.	Restionaceae					1				r	S
Lepidosperma "flat"	Cyperaceae					1				r	S
Lepidosperma 2	Cyperaceae	1								r	S
Lepidosperma 'branched'	Cyperaceae							1		r	S
Lepidosperma 'cluster'	Cyperaceae			1				1		r	S
Lepidosperma 'fine'	Cyperaceae			1				1		r	S

Lepidosperma 'flat'	Cyperaceae				1			1	1	r	S
Lepidosperma 'round'	Cyperaceae			1	1	1	1			r	S
Lepidosperma scabrum	Cyperaceae	1	1							r	S
Lepidosperma sp.2	Cyperaceae							1		r	S
Lepidosperma sp.3	Cyperaceae		1							r	S
Leporella fimbriata	Orchidaceae				1					n	S
"Lepto cono"	Cyperaceae				1					r	S
Leptospermum "hairy" sp.	Myrtaceae								1	r	C
Leptospermum oliganrum	Myrtaceae		1		1			1	1	r	S
Leptospermum spinescens	Myrtaceae	1	1	1	1		1		1	r	C
Leucopogon aff. *hispidus*	Epacridaceae	1								n	S
Leucopogon conostephioides	Epacridaceae	1	1	1	1			1		n	S
Leucopogon gracillimus (recurved ericoid)	Epacridaceae	1	1	1	1	1		1	1	n	S
Leucopogon hispidus	Epacridaceae		1							n	S
Leucopogon oxycedrus	Epacridaceae					1				n	S
Leucopogon 'small'	Epacridaceae								1	n	S
Leucopogon sp.	Epacridaceae			1						n	S
Leucopogon sp.3	Epacridaceae					1				n	S
Leucopogon sp.5	Epacridaceae	1								n	S
Leucopogon 'tall'	Epacridaceae				1					n	S
Lobelia sp.	Lobeliaceae			1						n	S
Lomandra hastilis	Dasypogonaceae	1	1	1		1				r	S
Lomandra micrantha	Dasypogonaceae	1	1		1	1				r	S
Lomandra sp.3	Dasypogonaceae		1		1					r	S
Lyginia barbata	Restionaceae	1	1	1						r	S
Lyginia imberbis	Restionaceae	1		1	1					n	S
Lysinema ciliatum	Epacridaceae		1							n	S
Macarthuria australis	Molluginaceae						1	1		n	S
Melaleuca aff. *leuropoma*	Myrtaceae						1	1		r	C
Melaleuca leuropoma	Myrtaceae	1	1	1	1	1	1	1	1	r	C
Melaleuca 'ovate'	Myrtaceae								1	r	C
Melaleuca scabra	Myrtaceae				1				1	r	C
Melaleuca trichophylla	Myrtaceae		1		1		1	1	1	r	C
Mesomelaena pseudostygia	Cyperaceae	1		1	1	1		1	1	r	S

Mesomelaena stygia subsp. *deflexa*	Cyperaceae	1	1	1						r	S
Mesomelaena tetragona	Cyperaceae		1	1	1				1	r	S
Mirbelia sp.	Papilionaceae				1					r	S
Mirbelia spinosa	Papilionaceae		1						1	r	S
Monotaxis bracteata	Euphorbiaceae	1	1						1	r	S
Monotaxis sp.	Euphorbiaceae				1					n	S
Myrtaceae sp.2	Myrtaceae	1								?	?
Myrtaceae sp.3	Myrtaceae	1								?	?
Nemcia capitata	Papilionaceae			1	1	1		1	1	r	S
Neurachne alopecuroidea	Poaceae					1				r	S
Olax benthamiana	?			1						n	S
Opercularia sp.	Rubiaceae			1						r	S
Opercularia sp.1	Rubiaceae					1				r	S
Opercularia spermacocea	Rubiaceae				1	1				r	S
Opercularia vaginata	Rubiaceae				1	1				r	S
Patersonia occidentalis	Iridaceae					1				r	C
Patersonia sp.	Iridaceae				1					r	S
Pericalymma erubescens	Myrtaceae			1						r	S
Petrophile brevifolia	Proteaceae	1	1	1	1	1	1		1	r	C
Petrophile drummondii	Proteaceae	1		1			1	1	1	n	C
Petrophile linearis	Proteaceae		1							r	C
Petrophile macrostachya	Proteaceae	1	1	1	1		1		1	r	C
Petrophile rigida	Proteaceae				1					r	C
Petrophile scabriuscula	Proteaceae		1							n	C
Petrophile serruriae	Proteaceae		1			1		1		n	C
Phyllanthus sp.	Euphorbiaceae				1					r	S
Phymatocarpus porphyrocephalus	Myrtaceae	1								r	C
Pileanthus filifolius	Myrtaceae	1	1	1	1					r	S
Pimelea leucantha	Thymeliaceae	1		1						r	S
Pimelea species	Thymeleaceae					1				r	S
Pimelea suaveolens	Thymeleaceae							1		n	S
Pimelea sulphurea	Thymeleaceae		1						1	r	S
Pityrodia bartlingii	Lamiaceae	1								r	S
Pterostylis nana	Orchidaceae				1		1	1	1	n	S

Restio sp. (white)	Restionaceae		1							r	S
Restio sp.1	Cyperaceae	1	1							r	S
Scaevola canescens	Goodeniaceae	1	1	1	1	1		1	1	r	S
Scaevola eneabba	Goodeniaceae					1				n	S
Scaevola paludosa	Goodeniaceae			1	1	1			1	r	S
Scaevola species	Goodeniaceae					1				r	S
Schoenus sp.1	Cyperaceae	1	1	1	1	1	1	1	1	r	S
Schoenus subbulbosus	Cyperaceae	1	1						1	r	S
Scholtzia capitata	Myrtaceae	1		1				1	1	r	S
Scholtzia involucrata	Myrtaceae	1	1	1	1	1	1	1	1	r	S
Scholtzia laxiflora	Myrtaceae								1	r	S
Scholtzia umbellifera	Myrtaceae					1		1		r	S
Sowerbaea laxiflora	Anthericaceae	1	1	1		1		1	1	r	S
Sphaerolobium sp.	Euphorbiaceae							1		n	S
Spiky unknown	?					1				?	?
Stachystemon axillaris	Euphorbiaceae	1								r	S
Stackhousia sp.	Stackhousiaceae							1		n	S
Sterculiaceae sp.1	Sterculiaceae				1					r	S
Stipa elegantissima	Poaceae					1				r	S
Stirlingia latifolia	Proteaceae	1		1						r	S
Stylidium "leucopogon"	Stylidiaceae					1				n	S
Stylidium adpressum	Stylidiaceae		1							n	S
Stylidium blue	Stylidiaceae				1	1				n	S
Stylidium brunonianum	Stylidiaceae	1		1		1			1	r	S
Stylidium ciliatum aff. *piliferum*	Stylidiaceae	1			1					n	S
Stylidium crossocephalum	Stylidiaceae	1	1	1	1	1	1		1	n	S
Stylidium flagellum	Stylidiaceae			1	1	1				n	S
Stylidium kalbarriense	Stylidiaceae				1	1				n	S
Stylidium repens	Stylidiaceae	1	1	1	1			1		n	S
Stylidium short	Stylidiaceae		1		1	1				n	S
Stylidium sp. rosette	Stylidiaceae				1					n	S
Stylidium sp.1	Stylidiaceae		1		1	1				n	S
Stylidium sp.2	Stylidiaceae					1				n	S
Synaphea spinulosa	Proteaceae			1		1				r	S

Tersonia brevipes	Gyrostemonaceae					1				n	S
Tetraria octandra	Cyperaceae					1				r	S
Thryptomene sp.	Myrtaceae				1					r	S
Thysanotus fastigiatus	Anthericaceae	1	1	1	1					r	S
Thysanotus patersonii	Anthericaceae		1	1	1	1	1		1	r	S
Thysanotus sparteus	Anthericaceae	1	1	1	1	1				r	S
Thysanotus species	Anthericaceae				1	1				r	S
Thysanotus triandrus	Anthericaceae	1	1					1	1	r	S
Trachymene leucopetala	?							1		n	S
Tricoryne elatior	Anthericaceae	1	1		1	1			1	r	S
Tricoryne excelsior	Anthericaceae								1	r	S
Triodia longipalea	Poaceae					1				r	S
Tripterococcus brunonis	Stackhousiaceae		1							n	S
Unknown C1	?	1								?	?
Unknown C2	?	1								?	?
Unknown C9	?					1				?	?
Unknown dicot	?	1								?	?
Unknown grass	Poaceae				1					?	?
Unknown myrtaceous	Myrtaceae				1					?	?
Unknown sp.R2	?								1	?	?
Unknown sp.R4	?								1	?	?
Unknown sp.7	?							1		?	?
Unknown sp.n1	?			1						?	?
Unknown sp.n3	?			1						?	?
Unknown sp.n5	?			1						?	?
Unknown sp.9	?				1					?	?
Unknown sp.10	?					1				?	?
Unknown sp.11	?		1							?	?
Unknown sp.12	?				1					?	?
Unknown sp.13	?				1					?	?
Unknown sp.14	?				1					?	?
Unknown sp.15	?					1				?	?
Unknown sp.16	?				1					?	?
Unknown sp.17	?				1					?	?

Unknown sp.18	?				1					?	?
Unknown sp.19	?				1					?	?
Unknown sp.20	?				1					?	?
Unknown sp.21	?				1					?	?
Unknown sp.22	?				1					?	?
Unknown sp.23	?					1				?	?
Unknown sp.24	?					1				?	?
Unknown sp.25	?		1							?	?
Unknown suckering *Scaevola*	Goodeniaceae								1	r	?
Unknown *Verticordia*	Myrtaceae				1					?	?
Verticordia "darwinia like"	Myrtaceae		1							?	?
Verticordia 7	Myrtaceae	1								n	S
Verticordia aurea	Myrtaceae							1	1	n	S
Verticordia chrysanthella	Myrtaceae				1					r	S
Verticordia densiflora subsp *caespitosa*	Myrtaceae	1	1	1	1					r	S
Verticordia grandis	Myrtaceae	1	1	1			1			r	S
Verticordia *long*	Myrtaceae				1	1				?	?
Verticordia nobilis	Myrtaceae	1		1						r	S
Verticordia ovalifolia	Myrtaceae	1								n	S
Verticordia pennigera	Myrtaceae		1		1					r	S
Verticordia sp.1	Myrtaceae		1							?	?
Verticordia sp.2	Myrtaceae		1		1					?	?
Verticordia sp.3	Myrtaceae			1						?	?
Verticordia very small	Myrtaceae		1							?	?
Xanthorrhoea acanthostachya	Xanthorrhoeaceae	1			1					r	C
Xanthosia huegelii	Apiaceae	1	1			1			1	n	S
Xylomelum angustifolium	Proteaceae	1		1						r	C
359 spp.	Total spp.	117	128	106	129	101	49	80	112		