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Setting goals and choosing appropriate reference sites for restoring mine pit lakes as aquatic ecosystems: a case study from south west Australia

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Abstract. Pit lakes may form when open cut mining leaves a pit void behind that fills with ground and surface water. Often replacing terrestrial ecosystems that existed prior to mining, the pit lake may offer an alternative ecosystem with aquatic biodiversity values that can be realised through planned restoration. Restoration theory and mine closure regulatory requirements guides us toward restoring disturbed systems towards landscapes that are of regional value and relevance. But how do we identify a restoration target for a novel aquatic habitat that did not exist prior to the new post-mining landscape? This paper presents a process of first identifying and then surveying local analogue aquatic systems to provide a direction for pit lake restoration efforts and achievement criteria for pit lake relinquishment. We illustrate this process using a case study from a sand mining operation located amongst wetlands in south-western Australia. The company mines silica sands following mechanical removal of topsoil and then extraction of the ore from below the water table by dredging. Assessment of wetland and riparian vegetation in the surrounding area was completed through the establishment and measurement of temporary monitoring transects across five natural wetlands in the Kemerton area with several more visited and observations made. Distinct zonation of vegetation was found across each wetland, although typically wetland basins were unvegetated or filled with younger woody plants with patchy distributions. Fringing riparian vegetation consisted of few species (commonly *Melaleuca rhaphiophylla* and *Lepidosperma longitudinale*) but community composition and structure were variable between wetlands. The pattern of vegetation seen across natural wetlands was best explained by topography and soil chemistry with low lying areas more likely to experience regular flooding and accumulate organic matter and nutrients. We consider that, with good planning, rehabilitation, monitoring and management interventions to achieve a restoration trajectory, these new mining pit lakes can positively contribute to regional ecological values.

Key words: Pit lake, restoration goals, wetland, riparian, vegetation, mine closure

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Introduction

Increasingly frequent, and of growing scale, open-cut/cast mining has left a legacy of many thousands of mining pit voids worldwide (Klapper and Geller, 2002). Restoration of habitats following mining has become a well-researched and standard practice that borrows from both disciplines of ecology and engineering and which often scales across entire landscapes and regions where mining is active (McCullough and Van Etten, 2011). However, this restoration typically ceases at the edge of these pit voids (Van Etten, 2011), unless the potential for backfill and/or landscaping can incorporate the pit into the terrestrial system (Lund and McCullough, 2011b). Backfill of pits is often promoted as best practice for managing pit voids at mine closure (Puhalovich and Coghill, 2011). Moreover, backfill is often not an economic or feasible option, and if the pit extends into the water table then pit lakes with aquatic ecosystems of varying value will form (Castro and Moore, 2000; McCullough et al., 2013).

Internationally, closure planning can best be described as a “land redevelopment” exercise (Jones, 2012). In most Australian states, the primary objective is to leave the site in a condition suitable for the agreed final land use, along with other goals relating to making the site safe, stable, non-polluting and maintenance free (Clark, 1999). Early consideration of the land redevelopment goals can provide clear direction to both company and stakeholders on what risks and/or beneficial end uses are expected from the post-mining landscape following closure (McCullough et al., 2009). Typically, closure outcomes (objectives) and how they will be measured (criteria) are required at least in a preliminary stage early on in a mining project development (Laurence, 2006). Leading international mine-closure guidance and practice both internationally, e.g. ICMM (2008), and domestically ,e.g. DMP/EPA (2011), is therefore to identify clear closure objectives and criteria. Mine closure planning for pit lakes specifically is no different in this sense (Jones and McCullough, 2011).

A first step in the development of a pit lake ecosystem of environmental value is to recognize an ‘Identifiable Desired State‘ (*c.f.* Grant, 2006) as a restoration goal (McCullough and Van Etten, 2011). Importantly, for the pit lake and its catchment to contribute value to the environment, this should proceed by establishing restoration targets for aquatic through to terrestrial ecosystems that are considered of ecological value and that are regionally representative (Jones and McCullough, 2011).

Setting appropriate goals and objectives is the most important stage of planning restoration projects and critical to the success of restoration. Objectives should be focussed, achievable and measureable, whereas goals should specify the desired outcomes over both the short and long term (SERI, 2004). Restoration goals typically address multiple attributes including biodiversity, aesthetics, safety, production, sustainability and social benefits, and should incorporate the concerns and expectations of stakeholders and the wider community. It is common practice in ecological restoration to select reference ecosystem(s) to use as restoration targets and to help gauge the success (or otherwise) of restoration efforts (SERI, 2004). Reference ecosystems may represent a historical or pre-disturbance state, a nearby ecosystem or a synthesis of attributes deemed desirable (Brewer and Menzel, 2009).

However selection of appropriate reference systems is fraught with difficulties and complexities, much of which stems from inherent variability of natural ecosystems over a range of temporal and spatial scales (White and Walker, 1997; Hobbs, 2007).

Even though, in a pit lake context, an aquatic ecosystem may not have been present before disturbance, development of wetland environments in the face of global (Kundzewicz et al., 2008), national (Hobday and Lough, 2011) and regional (Horwitz et al., 2008) aquatic habitat loss may be justified as preferred restoration targets, especially where there are regionally rare aquatic species or ecosystems present (Brewer and Menzel, 2009). However, restoration of a representative and functional amphibious ecotonal ecosystem is often most challenging for shallow pit lakes that may ecologically resemble wetlands, and for the riparian margins of lakes proper. The absence of riparian vegetation around new pit lakes above certain thresholds may often be unrelated to water quality (Fyson, 2000) and may more likely be a consequence of initial bank instability and/or unsuitable soils for seedling establishment (Van Etten, 2011). Further, profound variation in topography, hydrology and soil across aquatic-terrestrial ecotones needs to be taken into account (Naiman et al., 2005). Indeed, wetland margins typically experience pronounced zonation in response to a seasonal flooding regime, which is further complicated by inter-annual and longer-term variability in water levels. These ecotones of riparian zones, are therefore, noted for their acute spatial heterogeneity which results in high levels of species turnover across their perimeters (Naiman et al., 2005; Ward et al., 2002).

This study sought to determine what regional wetland ecosystems might constitute reference systems and restoration targets for a sand mining operation that was causing direct loss to natural wetland habitat, but that had potential to restore some type of wetland habitat through targeted restoration efforts. It addresses the important question of what are appropriate and realistic targets where there have been major biological and physical perturbations, an inherent part of mining. Post-mining environments are unlikely to ever completely resemble pre-mining conditions, even over the long-term, and therefore are classic novel ecosystems, *sensu* Hobbs et al. (2009). So what, under these circumstances, should we aim for and what is realistically achievable? White and Walker (1997) argue that we need to understand ecological patterns and processes in natural ecosystems across both time and space to select appropriate reference systems and that such understanding can improve restoration decision-making and practice. This study also sought to understand what environmental drivers were most important for developing representative wetland vegetation community structure and dynamics, and to assess the value of such information to help guide restoration.

Methods

Study Area

The study was conducted at the Kemerton Silica Sand (KSS) mine located within the Kemerton wetlands (33°08'S, 115°47'E), 30 km north of Bunbury and 150 km south of Perth (Van Etten et al. 2012) on the Swan Coastal Plain, Western Australia. The project area consists of an extensive aeolian sand-dune system forming a distinct Australian bioregion. Approximately 500,000 t of feldspathic silica sands are extracted annually at this mine from below the water table using dredging, both from wetland and woodland ecosystems. Once ore extraction is complete, pit lakes are formed and progressively rehabilitated. As the pit lake is essentially an expression of the groundwater, the final post-mining landforms are permanently inundated lakes Overburden and topsoil are available for sculpting and landscaping of the pit lakes and surrounding slopes. Around 10 lakes are expected at eventual mine closure of between 10–15 ha surface area and approximately 10 m deep (MBS Environmental, 2009).

Study area climate is distinctly Mediterranean with most of the average *c*.890 mm of annual rainfall falling in winter and spring. Summers are warm to hot and typically very dry (average February maximum temperature is 28^{o}C and rainfall is 13 mm), whilst winters are cool and wet (average July maximum is 17^{o}C and rainfall is 186 mm). The project operates on privately-owned land comprising and mostly surrounded by intact and relatively healthy examples of the natural ecosystems of the Swan Coastal Plain (*Eucalyptus*-*Banksia* woodland on uplands and various wetland systems of high conservation value). Shallow depth to groundwater in the inter-dunal depressions results in numerous wetland areas of palusplain, damplands, sumplands and lakes (as per the definitions of Semeniuk (1987) within the project area. The climate and shallow nature of wetlands in the Kemerton area ensure that all natural wetlands are seasonal and these wetlands become inundated from rainfall or the rising groundwater table, typically from July to November (Galeotti et al., 2010). The south west of Western Australia is regarded as a biodiversity hotspot for fauna and flora, with high levels of endemism and high numbers of threatened species (Myers et al., 2000). For example, at least eight of the ten native freshwater fish found in the south-west are endemic (Morgan et al., 1998).

Currently dredge ponds ecologically differ significantly from regional aquatic habitat analogues such as nearby Environmental Protection Policies (EPP) wetlands (Lund and McCullough, 2011a; van Etten et al., 2012). Specific closure restoration requirements of the KSS project therefore include the need to develop a closure plan with criteria to measure rehabilitation outcomes relevant to the post-mining landscape where approximately 50% of cleared land will be dredge ponds (EPA, 2012).

Riparian Vegetation Assessment

Riparian transects were used to characterise the biotic patterns of natural wetlands and to identify likely processes driving community structure. This was achieved through comparing: 1) structural attributes; 2) plant composition (using multivariate techniques such as ordination); 3) dominance and diversity patterns (Grant and Loneragan, 2003); and 4) soil and topographic features across and between transects.

Transects were placed across several natural wetlands in the Kemerton mining area (KMA) and nearby Kemerton Nature Reserve (KNR) during winter 2007. At time of survey all wetlands in the study Area were dry and three (EP4, EP5 and EP7) were surveyed in detail. Two other small wetlands (PD, PS) adjacent to EP7 were also surveyed. In addition, observations were also made at three other wetlands: EP1, EP3 and EP9. (Note: the prefix 'EP' was used because these wetlands were mapped as part of Environmental Protection (Swan Coastal Plain Lakes) Policy 1992). Each transect commenced at wetland base (lowest point in profile), traversed the fringing wetland and then finishing at the upland vegetation (if present). Each transect therefore captured the typical zonation and variation in vegetation, soil and landform of the wetland system and its fringing vegetation. To capture the variation in vegetation along transects in the most efficient manner, the relevé sampling approach was used, where a study site (or relevé) was established within each distinct vegetation type along the transect. Each relevé was positioned in vegetation representative of the wetland riparian and the cover of each plant species was then estimated within a circular area of 20 m radius. Height and cover of each vegetation strata was also recorded.

Soil and Topographic Profiling

A theodolite, GPS and tape measure were used to determine the changes in elevation and slope along the riparian transect. A sampling trench was dug in each of the different vegetation zones identified along transects and different soil horizons were identified to a maximum depth of 0.5 m. A soil sample was then collected each horizon from three different sites in each vegetation zone and then pooled for each horizon and zone. The pooled soil sample was then dried, ground and analysed for: texture, colour, nitrate-N, ammonium, phosphate, potassium, sulfur, iron, carbon, conductivity, (dS/cm) and pH (pH 1:5, $CaCl_2$ and H_2O).

Data analysis

The mean and standard error of relevé cover, density (number of plants per relevé) and richness (number of species per relevé) was calculated for each transect. Differences between transects, depth and position along transect were tested using univariate analyses of soil variables such as one- and two-way ANOVA in SPSS (2007) with a Type I error of 0.05. Prior to analysis data were $\log_{10}(x+1)$ transformed to improve normality where required and were also checked for parametric assumptions (McGuiness, 2002).

Multivariate data analyses were performed using PRIMER v6 software (Clarke, 1993; Clarke and Gorley, 2006) following a process of data transformation, graphical exploration and then statistical hypothesis testing. Two-dimensional nMDS ordinations of multivariate data were constructed for taxa frequency data using 100 iterations and based on the Bray-Curtis dissimilarity matrix. Principal Components Analysis (PCA) was used to produce ordinations of soil (environmental) data. Differences between *a priori* treatment groups were tested using the ANOSIM permutation routine with 9999 iterations (all other variables default) (Clarke and Gorley, 2006). Environmental variables and taxa most contributing to differences between wetlands were determined using the SIMilarity-PERcentages (SIMPER) routine (cut-off at 95% cumulative similarity) (Clarke, 1999). The BIO-ENV procedure was then used to determine the combination of environmental variables best rank correlating with riparian vegetation communities (Clarke and Ainsworth, 1993). Bubble Plots showing values of the environmental variables (selected using BIO-ENV) were then projected onto site positions within the ordination. Environmental – vegetation relationships were also explored using Redundancy Analysis (RDA) within CANOCO for WINDOWS version 4 (ter Braak, 1998).

Results

Topographic Profiles

Topographic profiles of the wetlands (Fig. 1) showed that wetland basins were generally flat with slopes of <0.1% and ended in relatively abrupt change of slope where dense fringing vegetation developed on slightly higher ground some 0.2 to 1 m above than the wetland basin. EP4 was slightly different as the lake basin was generally smaller in area and had a more concave profile with slopes between 0.2 to 0.4% (Fig. 1).

Soil Characteristics

The first or A-horizon of wetland riparian soil was generally sand with the highest content of organic matter (6.2±0.9% organic C) and was consequently grey to black in colour. At some sites, the A-horizon was further visually split into two layers (A1 and A2) and the A2 horizon generally had intermediate chemical characteristics compared to A1 and B horizons. The second or B-horizon was generally deep sand with low organic matter content (3.0±0.7% organic C) and generally white or yellow in colour. The A-horizon was generally thicker in the wetland basin (0.06–0.30 m) compared to fringing vegetation and uplands (Table 1; Fig. 1).

EP7 and the adjoining two small wetlands (PD and PS) had thick, dense organic matter accumulation at the surface (i.e. peat) to 0.3 m depth. Organic carbon and soil nutrients, including nitrogen and phosphorus, were substantially higher in these wetland soils compared to others (Table 1). An exception to this trend is zone 3 of EP5 which had high levels of organic matter content, organic carbon, ammonium nitrogen, potassium and phosphorus in surface- and sub-soils. This site was in a slight depression adjacent to fringing vegetation where it would be expected to inundate for greater periods than the surrounding wetland basin.

Soil nutrient concentrations (phosphorus, ammonia & nitrate) were normally higher in the A1 horizon than in the other soil horizons. Soil phosphorus levels were generally higher in soils of wetland basins compared to fringing vegetation, whereas ammonia was generally highest in fringing paperbark vegetation. Nitrogen, especially as nitrates, was particularly low in wetland basin soils, often at or below levels of detection (≤1 mg kg^{-1}). Soil phosphorus levels were very low (A1 horizon: 20.3±8.0; range 2–114 mg kg^{-1}) compared to those recorded for Perth wetlands by Davis *et al.* (1993) at a mean of

1,100±580 mg kg^{-1} (range 20–40,000 mg kg^{-1}). Phosphorus concentrations were highest in EP7 and lowest in EP4 (Table 1).

Table 1. Chemical parameters of soils collected at multiple horizons and zones across wetlands of the Kemerton Silica Sands Project area and Kemerton Nature Reserve. All units mg/kg unless otherwise stated.

Wetland		**EP4**		**EP5**				**EP7**				**PD**		**PD-PS**	**PS**
Zone		**1**	**2**	**1**	**2**	**3**	**4**	**1**	**2**	**3**	**4**	**1**	**2**	**1**	**1**
Soil Horizons (cm)	A1	0-6	0-5	0-8	0-8	0-8	0-5	0-20	0-5	0-5	0-5	0-30	0-3	0-3	0-10
	A2	6-19	-	8-25	8-15	-	5-27	-	-	-	-	-	-	-	-
	B	19-50	5-50	25-50	15-50	8-50	27-50	20-50	5-50	5-50	5-50	30-50	3-50	3-50	10-50
Nitrate-N	A1	1	1	1	1	3	3	4	2	1	1	1	7	1	5
	A2	1	-	1	1	-	3	-	-	-	-	-	-	-	-
	B	1	1	1	1	1	2	4	1	1	1	1	1	1	2
Ammonium-N	A1	1	2	1	3	19	5	1	15	7	6	2	33	5	18
	A2	1	-	1	3	-	2	-	-	-	-	-	-	-	-
	B	1	1	1	1	2	1	3	3	2	1	2	1	1	9
Phosphorus	A1	10	2	2	2	24	6	114	41	39	6	3	18	6	11
	A2	2	-	2	2	-	4	-	-	-	-	-	-	-	-
	B	2	2	2	5	5	21	17	5	2	2	5	4	4	4
Potassium	A1	364	325	121	295	790	402	436	495	374	113	57	303	110	439
	A2	73	-	66	25	-	134	-	-	-	-	-	-	-	-
	B	109	58	91	160	417	136	382	33	25	28	138	40	107	151
Sulfur	A1	240	185	22	5	32	15	202	24	17	3	52	65	7	115
	A2	112	-	66	2	-	12	-	-	-	-	-	-	-	-
	B	151	84	74	9	12	21	92	40	9	4	108	4	3	269
Organic Carbon (%)	A1	2.1	6.6	1.2	3.0	10.0	6.1	10.0	4.7	10.0	4.2	3.9	10.0	6.0	9.3
	A2	0.2	-	0.4	0.2	-	3.4	-	-	-	-	-	-	-	-
	B	0.2	0.4	0.5	0.6	3.4	1.4	4.2	3.5	2.3	2.6	6.8	3.5	2.9	9.8
Iron (g/kg)	A1	0.98	0.80	0.13	0.66	0.30	0.48	1.49	1.88	0.39	0.17	0.36	1.06	0.33	1.06
	A2	0.19	-	0.16	0.11	-	0.36	-	-	-	-	-	-	-	-
	B	0.12	0.14	0.32	0.20	0.62	0.29	0.93	0.44	0.23	0.15	0.93	0.39	0.34	0.34
Conductivity (mS/cm)	A1	1.42	2.46	0.49	0.07	0.28	0.33	2.24	3.24	0.18	0.09	0.61	0.65	0.10	2.54
	A2	1.35	-	0.88	0.05	-	0.30	-	-	-	-	-	-	-	-
	B	2.62	1.25	1.02	0.19	0.36	0.33	2.14	0.84	0.21	0.07	1.24	0.12	0.06	4.63
pH ($CaCl_2$) (no units)	A1	5.2	4.7	5.9	5.3	5.1	7.3	5.5	4.9	4.4	3.8	4.9	3.9	4.0	4.8
	A2	6.6	-	6.5	6.7	-	7.5	-	-	-	-	-	-	-	-
	B	6.6	6.4	6.8	7.7	7	7.3	5.1	5.1	4.1	3.1	5.3	4.1	3.9	5.0

Other soil parameters, such as electrical conductivity, total sulphur and iron concentrations, were generally higher in the wetland basins than in fringing zones (Table 1). Topsoil pH ($CaCl_2$) at EP7 was 5.5 in the wetland basin but declined to 3.8 in zone 4 of EP4. EP5 had the highest pH from 5.9–7.3 in zones 1 and 4 respectively. EP5 had significantly more alkaline topsoils and subsoils than other wetlands ($F=4.1$, $p=0.038$ for A1; $F=23.9$, $p<0.001$ for B). pH was generally higher in horizon B than A (Table 1).

Riparian and Wetland Vegetation Structure & Floristics

Within the KSS project area, the basins of larger wetland systems which experience regular winter-spring inundation, and which are relatively deeper, such as EP7, were largely devoid of perennial vegetation. Instead these basins mostly comprised annual herbland which grew following subsidence of the water in late spring/early summer. Smaller wetlands which do not flood to the same depth or extent, had some tree cover in wetland basin (e.g., EP4 & 5), but this vegetation was patchy. EP4 was recently colonised by *Melaleuca* trees following flooding some 5 years earlier, with counts of growth rings of cut stems confirming the age of these trees. Other observed wetlands (eg EP8, EP9) were completely in-filled with larger and presumably older *Melaleuca* trees.

Fringing the wetlands was very dense vegetation with total cover sometimes exceeding 100% (Table 2). Vegetation was most dense at the edge with little to no understorey. Further out from the wetland basin and at slightly higher elevations, the *Melaleuca* woodland was more open with an understorey of sedges and/or rushes. At slightly higher elevations, woodland dominated by eucalypts with relatively diverse understorey of shrubs, bracken, sedges and/or rushes occurs. Clear zonation of vegetation types was evident in most areas of wetlands, particular within the fringing vegetation (Table 2). Areas which are seasonally waterlogged typically had more open woodland structure with dense, diverse understorey dominated by shrubs and sedges (e.g., site EP4-3).

Table 2. Summary of vegetation at sample sites in natural wetlands areas showing cover of natives, weeds and trees, and the number of native species per site.

Wetland	Site	Description	Cover Native (%)	Cover Weeds (%)	Tree Cover (%)	Native Species Richness
EP7	1	Wetland basin with annuals	40	0	1	2
	2	Fringing *M. rhaphiophylla*	60	2	60	3
	3	Fringing *M. rhaphiophylla* with sedge	100	10	45	5
	4	Fringing Eucalypt woodland	100	1	45	13
PD	1	Wetland basin	60	0	30	4
	2	Fringing *M. rhaphiophylla* with sedge	60	0	40	4
PD-PS	1	Fringing Eucalypt woodland	40	0	25	6
PS	1	Fringing *M. rhaphiophylla*	80	0	80	2
EP5	1	Wetland basin	40	0	10	5
	2	*Melaleuca* thicket with sedge and rush	70	0	40	8
	3	Fringing *M. rhaphiophylla* with sedge	100+	0	60	10
	4	Fringing *Melaleuca* – Eucalypt Transition	100+	0	60	8
	5	Fringing mixed *Melaleuca*	80	0	60	10
	6	Fringing Eucalypt woodland	100+	0	30	13
EP4	1	Wetland basin with young *M. viminea*	45	0	45	3
	2	Fringing mixed *Melaleuca*	55	0	55	4
	3	Dampland Community – *Melaleuca* over heath	65	0	13	13

Dominant species in fringing vegetation were paperbarks such as *Melaleuca rhaphiophylla, M. priessii* and *M. viminea*. Understorey below *Melaleuca* thickets and woodland, where present, consists mainly of sedges and rushes, with *Lepidosperma longitudinale* the dominant species, and *Juncus pallidus, Baumea articulata* and *Meeboldina scariosa* common in places. *Astartea scoparia* and *Kunzea glaucescens* are also common understorey shrubs, particularly on outer edges of the fringing *Melaleuca* communities.

The fringing eucalypt woodland which surrounded fringing *Melaleuca* was dominated by *Eucalyptus rudis* (Flooded Gum), although *Corymbia calophylla* (Marri) and *E. marginata* (Jarrah) occurred on higher ground. Understorey was varied; common species included *Pteridium esculentum, Astartea scoparia, Hypocalymma angustifolium, Lepidosperma longitudinale, Pericalymma ellipticum* and *Dasypogon bromeliifolius.*

The ordination of wetland vegetation (Fig. 2) showed a general trend in plant species composition from wetland basin (e.g., 7-1, 4-1) to upland vegetation (e.g., 7-4, 5-6 and PD-PS) from right to left. There was also a separation of different fringing *Melaleuca* vegetation from top to bottom in the ordination, with *M. rhaphiophylla* dominated sites towards the top (mostly EP7), and other *Melaleuca* spp. dominants at the base (e.g., *M. viminea*, *M. teretifolia*, etc.). *M. preissiana* dominated woodlands are in the middle. Testing significant differences between wetland riparian community structure with ANOSIM show no significant difference in species composition between wetlands (Global R = 0.021; p=0.41). Indeed, changes in species composition along transects of a single wetland far exceeded overall differences between wetlands.

Environment – Vegetation Relationships

Pair-wise correlations between plant species composition and environmental variables, analysed using the BIO-ENV module, were mostly modest to weak.. Highest correlations were with depth of horizon A (A1+A2), elevation above wetland basin, horizon A potassium concentrations and horizon B pH (Table 3). Slope and iron concentration were modestly correlated to variation in plant species composition.

Table 3. Six highest Spearman rank correlations between floristic similarity and environmental variables as determined using BIO-ENV module of PRIMER.

Variable	Correlation
Depth of horizon A	0.30
Elevation above wetland base (m)	0.25
K (horizon A)	0.21
pH (horizon B)	0.21
Slope	0.15
Fe (horizon B)	0.14
Fe (horizon A)	0.12

The first two axes of the Redundancy Analysis (RDA), where the ordination was constrained by environmental variables, explained 50.6% of the variance in species composition. The RDA biplot (Fig. 3) showed the relationship between main floristic gradients (the axes), sites and environmental variables (the arrows). Specifically this biplot revealed two different complexes of environmental variables linked to differences in plant species composition across sites. The first of these environmental complexes was generally correlated with the first (horizontal) axis and revealed changes in species composition along the toposequence from wetland lowest point (left side) to upland (right side). This complex included soil fertility (N, P, etc.), conductivity, gravel content and organic carbon which all increased with height above wetland basin. Only depth of horizon A generally declined with distance along this toposequence (Fig. 3). The second complex of environmental variables was related to pH, iron concentration and texture and separated the wetland EP5 (the most alkaline) from and EP7 (the most acidic).

Many of the variables in the RDA biplot were poorly correlated to floristic gradients (shown by short arrows) and were highly correlated to other environmental variables (Fig. 3). The forward selection procedure showed that only three variables could explain a significant and unique proportion of the variance in species composition: potassium (horizon A), pH (horizon B) and gravel content (horizon B) (Table 4). These three variables explain 46% of the variance in the species-environment relationship.

Table 4. Results of forward selection (in order of selection) of environmental variables in redundancy analysis (RDA) with significance determined following Monte-Carlo testing against a random model. *Variance explained is proportion of variance in species-environment relationship.

Order	Variable	Variance Explained (%)*	P-value
1	Topsoil potassium-a	18	0.008
2	Subsoil pH (H_2O)	14	0.022
3	Subsoil gravel	14	0.024
4	Subsoil phosphorus	9	0.108
5	Subsoil iron	7	0.238
6	Slope	6	0.304

The species richness of wetlands of the KSS project area and KNR was not high relative to adjacent uplands. In fact, wetland basin and fringing vegetation were often depauperate in species with as few as 1–3 species in the dense fringing vegetation (Table 2). Fringing eucalypt woodland and winter-wet depressions were found to have the highest number of species (each had 13). As survey work occurred in early winter, these figures do not include most of the annual plant and geophytic species.

Discussion & Conclusions

Spatial Patterns of Wetland Vegetation

As is characteristic of wetlands in general, substantial differences were found in vegetation structure, species richness and species composition within wetlands with distinct zonation of vegetation occurring across the wetland profile (Naiman et al., 2005). Wetland basins were generally flat and varied from bare in terms of perennial plants through to having a variable but patchy cover of paperbark trees and shrubs. These areas were seasonally inundated for some months each year. On raised ground around the edge of the basin, where some minor flooding would be expected, dense paperbark thickets were typical on slightly raised ground. At higher elevations, flooded gum woodland and then, higher still, jarrah-marri-banksia uplands were found. This substantial change in vegetation characteristics *across* wetlands tends to mask any differences in species composition *between* wetlands. Although the major paperbark trees (*M. rhaphiophylla* and *M. preissiana*) and the understorey sedges and rush species were common to all fringing vegetation wetlands, other species of tree and shrubs varied. In addition to seasonally-flooded wetland complexes, the KSS Project Area had large expanses of seasonally waterlogged vegetation consisting of sparse *Melaleuca preissiana* and *Banksia littoralis* tree canopy over a diverse shrub and sedge/rush understorey. Only one site was located in such vegetation (site 4–3). Much of the proposed mining expansion area will occur on this vegetation type.

Wetlands of the Swan Coastal Plain (SCP) are renowned for their complexity with geomorphic, edaphic and hydrological characteristics influencing vegetation composition and structure (Balla, 1994). Wetlands of the SCP have been classified in numerous ways, based on attributes such as geomorphology, hydrology, vegetation, aquatic biota, as well as combinations of these. Wetland vegetation of the SCP has been commonly categorised at two levels: the uppermost level or ‘complex’ refers to vegetation units linked by dominant plant species and structural attributes, and the secondary level for classification, the ‘community’, based on common or typical species within the overall complex (Cresswell and Bridgewater, 1985; Pen, 1997; Semeniuk et al., 1990). The fringing vegetation around wetlands of the KSS project area resembles fringing vegetation elsewhere on the SCP in terms of structure and dominant species.

Whereas the majority of wetlands on the SCP are expressions of underlying aquifers (i.e., they are discharge areas; Balla, 1994), there is evidence that many wetlands of the Kemerton area are perched wetlands which are separated from aquifers by thick clay and other impermeable layers (e.g., ‘coffee rock’). Consequently inundation in these perched wetlands is a function of rainfall directly onto the wetland basins plus run-off from surrounding slopes. EP4 appears to be a perched wetland, primarily receiving water inflows from the surrounding wetlands which effectively act as a catchment to this wetland. It is therefore important that this catchment area is actively managed to avoid adverse impacts on inflow water quantity and quality.

Environmental Drivers of Spatial Patterns in Vegetation

Restoration of mined lands frequently uses natural ecosystem as a restoration goal (Bell, 2001). However, the environmental variables driving vegetation patterns remain unclear. This study found few strong correlates between plant species composition and environmental variables. In particular, we found that elevation (AHD) was not a good predictor of vegetation composition, although relative height above the wetland basins and slope was a reasonably reliable predictor of the main floristic differences found across wetlands. Soil variables such as thickness of horizon A (humus layer or peat), organic carbon, nutrient levels and potassium were also linked to this main floristic gradient. This general topographic-soil-vegetation relationship is also likely to be linked to hydrological regime. The higher an area is elevated above the wetland basins, the lower the duration and depth of flooding it will experience and, consequently, the lower the accumulation of organic matter (peat and so on). Both the indirect effects of inundation and soil changes which flooding promotes are likely to influence vegetation composition and structure. A clearer picture of environmental causes of vegetation patterns should emerge through more detailed studies of the hydrology of these wetland systems, with variables such as distance to groundwater and their fluctuations (for groundwater-dependent wetlands) and area of catchment (for perched wetlands), suspected to be strongly correlated to vegetation patterns.

A second floristic gradient was found to be linked to soil pH and appear to separate EP5 from the others. This is likely to be due to its proximity of EP5 to limestone formations and may explain floristic differences in vegetation between wetland systems on the south-east side of Kemerton compared to those of the north and west.

Dynamics of Wetland Vegetation

The measurements of the three wetlands and observations made at other wetlands in the KSS project area and KNR enabled a clearer picture of wetland dynamics at the KSS Project Area to emerge. Such vegetation change was most clearly demonstrated at EP4 where basin vegetation of EP4 had two zones of distinct tree age (or cohorts). The inner basin consists of *ca.* 5 years old saplings (as judged by growth rings counted on cut stems) of more-or-less the same height (1.5 m) and stem diameter (25–40 mm). This was surrounded by a ring of fringing vegetation which was 7–10 m tall and likely to be much older**Error! Reference source not found.**. The hypothesis is that a reduced incidence of flooding (through combination of groundwater and/or rainfall decline) has allowed colonisation of *M. viminea* and some *M. rhaphiophylla* in the basin of this wetland following the last major flooding event in 2001-2. Previously the wetland basin was devoid of vegetation. We anticipate that seedlings may successfully establish during drier times, but may be eliminated if and when prolonged inundation returns. It is likely that tolerance to inundation will increase with age and size of tree, so seedlings/saplings are most vulnerable to flooding in first few years. The role of wetland infill with sediment may also play a role in encouraging seedling establishment as this would decrease the depth, extent and duration of inundation. Such infill can be the response of gradual ‘natural’ accumulation, or can be enhanced through some level of vegetation/soil disturbance in surrounding area.

These hypothesised flooding and drying events at EP4 concords with rainfall records of the region. Good rains over 1998–2001 are likely to have resulted in flooding of this wetland and subsequent abundant seed crops, either in soil seed stores or in fruits retained in the canopy. This flooding may also have promoted seedling establishment on moist lake basin as the flooding subsided. Rainfall from 2001 onwards has been well below long and short term averages. Only 2005 rainfall was above the short-term average, but this year was followed by very close to the driest year on record in 2006.

This and other studies have demonstrated that wetland basin and flats in the Kemerton area can experience relatively rapid change in structure and composition. Regular and persistent flooding of these areas, where inundation occurs for a least several months each year, inhibits tree colonisation of wetland basins and persistence, and promotes accumulation of peat deposits. Alternatively, drier periods result in lower and shorter flooding events which, in turn, enable seedling establishment of *Melaleuca* and other species on the wetland floor. This woody vegetation would be expected to become denser and more resistant to flooding the longer this dry period persisted. It seems such a dry period has encouraged colonisation of EP4 by *Melaleuca* spp. between 2001–2003 with a 5 year-old cohort of such trees dominant in the centre of the wetland basins at the time of study. EP4 may well be a perched wetland, so that elements of its hydrology such as hydroperiod is more sensitive to rainfall fluctuations and changes in surface drainage compared to the more common scenario of groundwater-fed wetlands on the SCP. With a drying climate in SW Australia, drying of wetlands and colonisation of wetland basins would be expected to become more common (Malcolm et al., 2006).

Although the fringing paperbark vegetation appears to be relatively stable over recent years at Kemerton, it too is likely vulnerable to changes in flooding regime with changes in species composition and structure expected with changes in inundation frequency and duration. Fire can also dramatically affect both fringing and basin vegetation, commonly killing trees and shrubs outright, especially when burning through peat and other layers rich in organic matter (Horwitz and Smith, 2005).

Implications for Post-mining Restoration and Choosing Reference Sites

Fringing wetland vegetation of natural wetlands was found to be floristically simple and structurally complex. Such structurally complex wetlands therefore represent a mixed challenge for rehabilitation; only relatively few species need to be restored, however they need to be encouraged to develop into relatively dense vegetation formations, with distinct bands of zonation. Vegetation of fringing zones are relatively species rich (some 10–30 species per 10 m^2), but are probably not as diverse as many upland areas of Kemerton dominated by jarrah, marri and banksia. The focus on these areas should be on quick return of topsoil matched to site conditions so that high diversity will be encouraged (Van Etten et al. 2012).

Given high variability between floral communities of wetlands in the KSS project area and KNR, it is difficult to establish a single reference or analogue wetland to compare with rehabilitated mine ponds and slopes. The relationships found here between fringing flora, soil characteristics, topography and hydrology however should help improve revegetation practices and overall rehabilitation success. Specifically this information informs that rehabilitation slopes should be subtle, with varying depth to groundwater and that organic matter levels in new topsoils should be enhanced in rehabilitation attempts.

Flat or gently sloping wetland basins would be difficult to recreate in most post-mining settings and probably undesirable given their general bareness. Also, post-mining wetlands created at Kemerton will essentially be expressions of underlying groundwater with previous impervious layers such as coffee-rock removed. However, there is scope for more subtle slopes to be created near the wetland and more dramatic slope changes to be located higher in the profile (opposite to current practice in some areas where the steepest slopes are closest to the water). Also such gradual slopes would result in a greater area of fringing vegetation around mine lakes and areas which are heavily waterlogged or partly inundated by groundwater. Studies of the rehabilitation areas suggest that dense paperbark-sedge fringing vegetation is only likely to establish in the seasonally flooded zone between high and low lake water levels (van Etten et al., 2012). Areas up to 2 m above this lake level appear to be influenced by groundwater (i.e., waterlogged soil) and appear to favour dampland or seasonally-waterlogged areas in terms of vegetation. Restoration of post-mining areas is more likely to resemble such damplands (i.e., seasonally-waterlogged wetlands) given topographic profile and hydrological regime of post-mining landscapes.

The process of wetland dynamics described here for EP4 and elsewhere observed in the Kemerton area (i.e., younger, even-aged stands of *Melaleuca* spp. in the wetland basin with older *Melaleuca* spp. towards the edge) is conceptual and requires further investigation. However, the challenge for a mine in an operational phase is to retain a view to how the environment surrounding the project area changes and how closure objectives and criteria may require readdressing to meets these changes.

Development of artificial wetlands from mining of either disturbed wetlands or even disturbed uplands may not only offer opportunities to replace lost aquatic biodiversity but also to contribute greater environmental values than previous land uses (McCullough and Van Etten, 2011). Any proposed use of such disturbed lands as environmental offsets must be able to demonstrate that the regional biodiversity and the offset biodiversity are both understood and accounted for (McKenney and Kiesecker, 2010). Significant monitoring and demonstration of ecological values may still be required in order to validate this development as environmental offsets (McCullough and Van Etten, 2011).

This study has demonstrated that, although pit lakes may be able to be restored to regionally relevant wetlands, the highly altered nature of these systems prior to mining and the variability and complexity of reference systems, both spatially and temporally, means that clear restoration goals developed from robust assessment of regional wetlands are required for development of pit lakes as regional analogue aquatic ecosystems.

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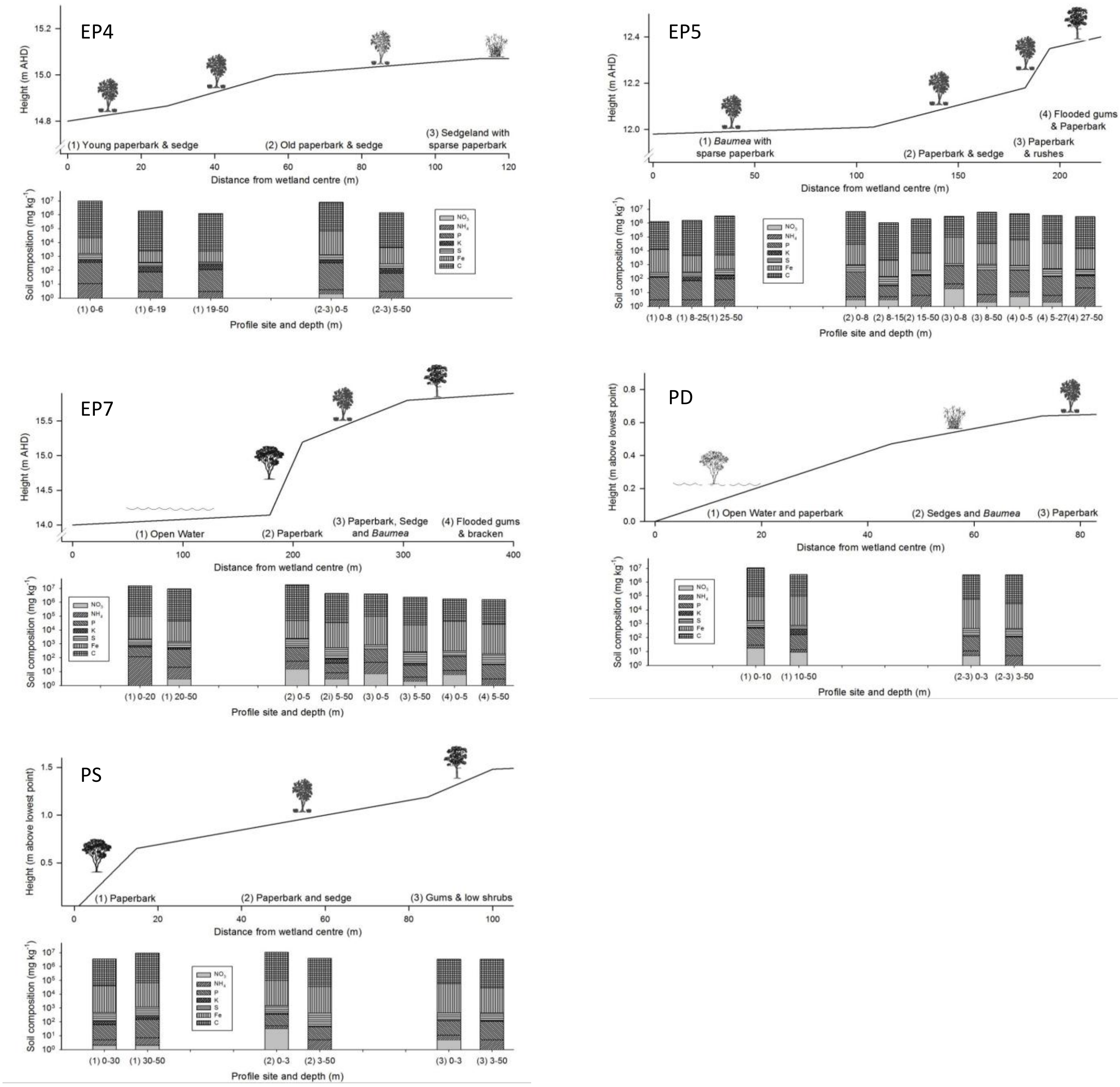

Figure 1. Wetland topographic profiles with corresponding vegetation structure and soil chemistry.

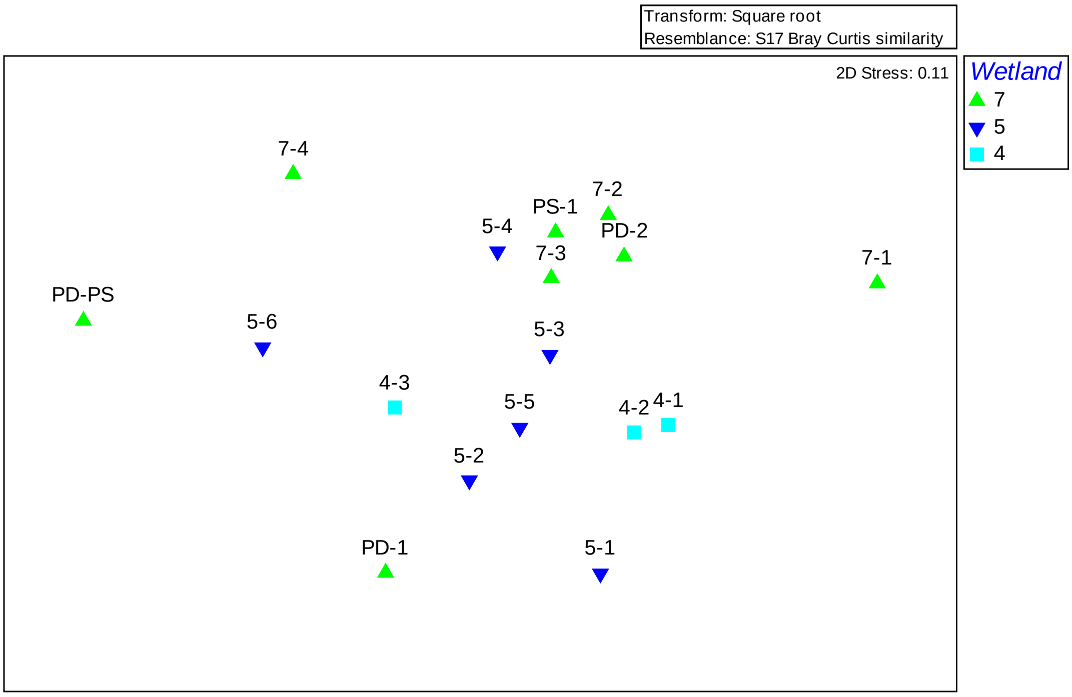

Figure 2. Non-metric multidimensional scaling of wetland sites at Kemerton based on plant species composition. Figure labels indicate wetland transect number-sampling site (e.g., 4-1 indicates first vegetation sampling site along transect through EP4).

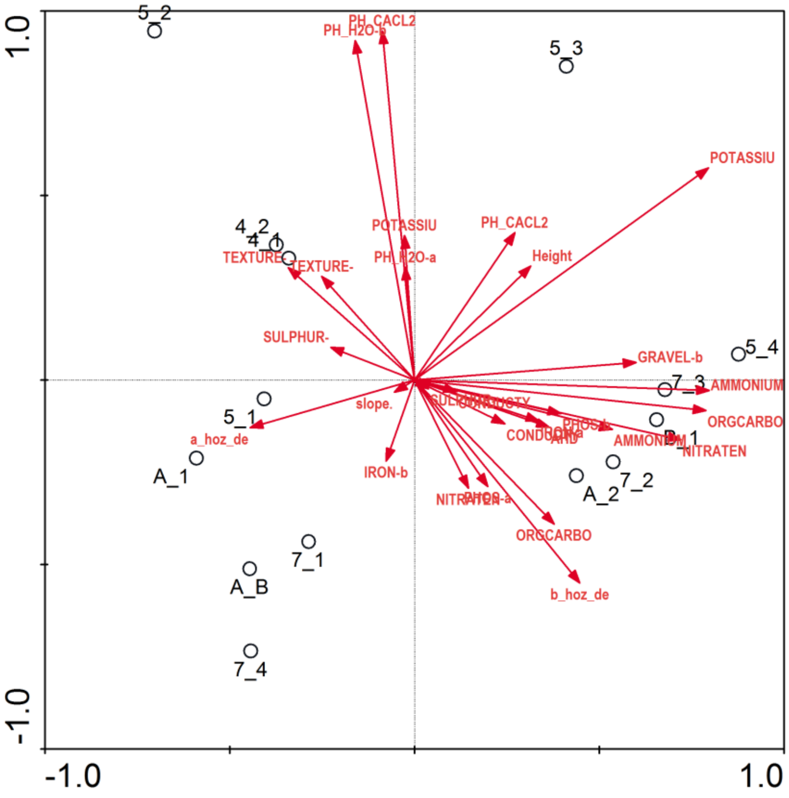

Figure 3. RDA biplot of sites using all species and all environmental variables for sites where soil was collected (14 sites). Length of arrow is proportional to strength of correlation between environmental variables and axes (major floristic gradients). Note: Site A_1 is PD_1, Site A_2 is PD_2 and Site A_B is PD-PS transition.

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