Restoration of Degraded Landscapes: Principles and Lessons from Case Studies with Salt-affected Land and Mine Revegetation

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ABSTRACT

Land is a finite resource. Every attempt must be made to prevent land degradation. In addition restoration of the legacy of already degraded land needs to be undertaken. Restoration is still a relatively new science. It is increasingly informed by the new discipline of restoration ecology. Land restoration occurs at a range of different scales. Mine rehabilitation has developed many successful practices of land restoration at a site-specific scale. The challenge ahead is to develop effective land restoration practices at a regional or landscape scale. Most success at both scales has to date been concerned with restoring key ecosystem functions like nutrient cycling and water balance. Case studies outlining restoration of each of these functions is discussed. Mining forms a striking comparison with land degradation associated with agriculture for example. Both can result in a catastrophic change in land suitability for plant growth, and may prevent the replacement of the original vegetation. However, whereas agriculture affects vast areas of land, mining has a relatively small footprint. They differ further in that considerably more money is available per unit area for restoration of mined land than can be justified for amelioration of degraded agricultural land. Finally, whereas land degradation caused by mining is closely regulated and mining companies have contractual requirements with Governments to restore mined land to an agreed standard, generally no such obligations are mandatory for agriculture.

LAND DEGRADATION

Land degradation has generally been defined from the utilitarian perspective of agricultural, horticultural or forestry uses of land. These definitions emphasize soil properties rather than landscapes. Lal and Stewart (1992), for example, suggest that soil degradation “implies diminution of productive capacity through intensive use leading to changes in soil …. physical, chemical and biological processes”. Other authors define land degradation as a change in land quality that makes it less useful for humans. These definitions may be too restrictive in that they seem to overlook consideration of land degradation at a landscape or regional scale. Landscapes are repositories of biodiversity and the substrate for flora and fauna conservation and the maintenance of ecosystem services such as the provision of clean air, clean water, and nutrient cycling. Landscapes may also be the source of exploitable mineral resources and are used for the siting of infrastructure and human settlements. Blum (1998) takes a different approach and defines soil degradation as a loss or reduction of soil energy. Conacher and Conacher (1995) define land degradation more broadly as “alteration to all aspects of the natural (or biophysical) environment by human actions, to the detriment of vegetation, soils, landforms and water (surface and subsurface, terrestrial and marine) and ecosystems”. Their definition whilst broadly encompassing still overlooks the significant cultural or archaeological heritage contained within land and landscapes which can be degraded or compromised by inappropriate land use (Blum, 1998). Notwithstanding these differences, all definitions of land degradation include the notion of detrimental change in land or soil conditions and the actions of humans. Hence landscapes that are naturally saline, for example, cannot be said to be degraded under these definitions, even though the characteristics that define degraded saline land may be no different from those of naturally saline soils and landscapes. Similarly natural hazards such as mass movement or earthquakes are not contained within these definitions.

Land degradation that is reversible should be considered differently to that which is non-reversible. Restoration of the former is technically and economically feasible. By contrast, irreversible land degradation may involve the crossing of some threshold for productivity or ecological function that is possible to manage or treat, but not to fully reverse. An example of this is the development of soil acidity that may have minimal impact on plant productivity until pH declines below a critical level when soil solution Al^{3+} levels exceed threshold concentrations tolerated by sensitive plant species in the system. If such acidity is confined to the surface soil, then it is simply reversed by treatment with lime or some other neutralising material. By contrast when most of the acidification occurs in the sub-soil, treatment by conventional means is ineffective. When the cost of ameliorating sub-soil is considered the problem may be deemed irreversible and require management by changing the suite of plants grown to those that tolerate the elevated Al^{3+} levels.

Most authors explicitly recognise that soils are a finite and non-renewable resource (e.g. Lal and Stewart, 1992) and this also is central to the notion of land degradation. Hence the challenge for sustainable use of existing land resources embraces both the avoidance of degrading processes, and the restoration of previously degraded land. Recent monographs such as Lal, Stewart and colleagues (1990, 1992, 1998 and papers therein) have dealt in-depth with soil and land degradation processes, their measurement, impact and management. In this paper, I will focus on restoration of degraded land from a biophysical perspective. The present paper is complementary to other such as Hobbs (2002) which focus on restoration ecology and deal with function, repair and design of ecosystems at site-specific and landscape scales.

The term restoration is used in the present review as a generic term after the usage of Hobbs and Norton (1996) who suggest "that restoration occurs along a continuum and that different activities are simply different forms of restoration". Given that part of the paper concerns restoration of land degraded by mining, the term rehabilitation as defined by Aronson *et al*., (1993) is also appropriate. Restoration will usually focus on restoring ecosystem functions such as nutrient cycling, hydrological balance, and ecosystem resilience (Hobbs, 2002), although restoring the original flora may on occasions be a realistic and appropriate goal. In the present paper, I will present case studies on land rehabilitation for dryland salinity control, and on bauxite mine rehabilitation, and use these to illustrate how restoration of hydrological balance at a landscape scale, and nutrient cycling at a more site-specific scale were pursued.

LAND RESTORATION PRINCIPLES

Land restoration comprises three components: determination of end land use, determining the main limiting factors for restoration and means of alleviating them, and finally planning and implemention of the restoration programme. In this paper I will deal mostly with the first two of these components.

End land use

A clear definition of the end land use is a prerequisite for effective land restoration. This will determine the stakeholders, the scope for restoration, the key constraints that have to be alleviated, and help define the goals for determining success. End land use has a major bearing on the degree of difficulty of the task of restoration and hence its cost. Restoration of a complete ecosystem, its flora and fauna, and its function, is a markedly more complex task than restoration of a stable surface or creating a pasture for low intensity grazing (Hobbs and Norton, 1996). The former may require decades of systematic research before restoration goals can be reliably accomplished and demonstrably sustainable.

Prior land use and current use of the surrounding areas will generally determine the end land use except where the substrate or landscape is so radically altered by degrading processes as to make this impossible. On highly disturbed sites following mining, a great diversity of end land uses are possible. Apart from restoration of natural ecosystems, that is an end land use goal in many Australian mining situations (Bell, 1998; Mulligan, 1998), agriculture, forestry, housing, industry, amenity and recreation, wetlands, and waste disposal are all possible end land uses (Bradshaw, 1988; McRae, 1998). By contrast with Australia, where most mining occurs in sparsely settled areas (Mulligan, 1998), in Europe and Asia mining often occurs in areas of high population density and multiple land uses, where mining competes for access to land beside agriculture, forestry, housing, industry and amenity space. On production lands, and protected landscapes, clearly fewer choices in end land use exist. Here, economic and socio-political factors have a major bearing on end land use.

Whilst setting end land use is necessary in order to develop tangible goals for restoration, there are also merits in a somewhat flexible or adaptive rather than a prescriptive approach. An example of the former is in the bauxite mine rehabilitation in southwest Australia. Restoration of the pre-existing forest is the current end land use goal of bauxite mine rehabilitation in the dry eucalyptus forests (Koch et al., 2000). This goal has been progressively upgraded during 35 years of research and development as can be seen by comparing review papers of Koch et al., (2000) with Ward et al., (1990) and Nichols (1998). Over time, research demonstrated that increasingly more ambitious end land use goals were feasible. Initial revegetation involved simply the planting of exotic plantation timber species to achieve a stable non-eroding surface. In the 1980’s and 90’s the goal for restoration was upgraded to multiple-use forest compatible with the surrounding jarrah forest (Ward et al., 1990; Nichols, 1998). The lesson from this example is that the regulatory framework for restoration has to be set in such a way that continuous improvement in rehabilitation practice can occur, and that as practice improves new benchmarks for completion can be set. Setting targets for restoration that are too easy to achieve provides no incentive to develop improved practices. Setting targets too high may lead to frustration for both government and land managers ending with disputation and perhaps litigation. The success of the bauxite mine rehabilitation in southwest Australia is a useful model for study and can be attributed to the adaptive approach and the partnership that exists between the three main players in environmental management: industry, government and the community. The pertinent characteristics of the partnership are:

_ a requirement for community participation in the environmental assessment and review process and its willingness to become involved;
_ government policy which sets minimal environmental standards but encourages industry to exceed them and develop improved practice, and ;
_ industries which approach environmental management as a necessary part of their operations, and view mine restoration as an opportunity to raise their profile in the community by exceeding minimum standards through the development of improved practices.

There is scope for this approach to be examined as a model in other natural resource use industries such as agriculture and forestry.

Problem diagnosis and definition

Land degradation has many different forms, and hence different consequences for restoration (Lal et al., 1998). Diagnosing the main form(s) of degradation at a site or in a landscape is a prerequisite for appropriate land restoration.

Achieving slope stability is one of the fundamental requirements in land restoration. A stable non-erosive surface is usually the minimum standard for restoration on any degraded or disturbed site. Without slope stability, few of the other ecosystem restoration goals, such as plant establishment, are likely to succeed. This pre-requisite applies particularly on highly disturbed sites such as mines where the slopes created in mine pits or on waste rock/ overburden dumps greatly exceed those that are naturally stable in that environment.

Water erosion is a natural process that over geological time shapes landscapes. Accelerated erosion also occurs on virtually all forms of land but especially on sloping land that has minimal vegetation cover. The risk of water erosion is exacerbated in climates with erosive rainfall, and by soil conditions such as dispersiveness and low infiltration capacity. In less extreme cases of erosion, variable topsoil loss may result in variable levels of productivity decline but not fundamentally change land suitability for present uses. In extreme cases, complete loss of topsoil and subsoil to the weathered basement rock may occur. Such extreme loss of topsoil may cause soil conditions to cross a threshold for productivity that requires specialised treatment or management to stabilise it against continued erosion and it may exclude continuation of current land uses in favour of something that is sustainable under the new substrate conditions.

Wind erosion is mostly confined to arid and semi-arid environments. Dry exposed sandy surface soils are a pre-requisite for wind erosion together with erosive winds. In the long term wind erosion selectively removes nutrient-rich topsoil and humus and leads to a decline in soil fertility as well as the suitability of soils for plant establishment. Wind erosion is particularly hazardous for the establishment of seedlings on bare disturbed surfaces because of loss of seeds, burial of seeds, and sand blasting of the apices of newly emerged seedlings. Wind erosion tends to be episodic and hence planning for it in a land restoration programme needs to consider the long term return frequency of damaging wind events and weigh the damage they can cause against the costs of protecting vegetation against it. Paradoxically, achieving vegetative cover on a potentially erosive surface is the main form of protection against further wind erosion.

Salinisation is usually the consequence of a significant change in water balance. It is usually the result of the inadequate design and management of irrigation schemes (Gassemi et al., 1995) and is often associated with waterlogging. Less commonly salinisation occurs in dryland environments, but again it is attributable to a perturbation of the water balance. In this paper, I will discuss dryland salinity in south-west Australia and the strategies being implemented to restore water balance. Salinised land may require a complete change in the suite of species grown and the reconstruction of new ecosystems since few of the previously-occurring species are likely to tolerate a change from non-saline to saline conditions. Other forms of degradation on salt-affected land, namely sodicity and alkalinity, may occur coincidentally with salinity or as separate soil conditions.

Many soils are acidic, because acidification is a naturally occurring process. In acid soils, with a pH ($CaCl_2$) < 5, a complex of constraints may limit plant growth including combinations of low availability of nutrients such as Ca, Mg, P, Mo, and toxic levels of Al and Mn. Acidification is the process of increasing soil acidity due to land use practices that disturb nutrient cycling. Ammonium-based N fertilizer used repeatedly at moderate to high rates is well known as a cause of acidification. Atmospheric deposition of acidifying materials has caused acidification of large tracts of land in Europe and North America. Less well known is the phenomenon of acidification caused by the use of legumes in farming systems. This has become a widespread cause of land degradation in southern Australia. In these farming systems it is largely a problem of leaky nutrient cycles that allow the leaching of nitrate-N after the mineralisation of ammonium-N derived from N fixation, and the export of base cations in harvested plant products. On mine sites the exposure of sulfide ores and sulfide-rich overburden materials can generate extreme acidity with pH as low as 2. Such extreme acidity from acid mine drainage can have harmful effects on-site for revegetation, and off-site when acid water discharges into surface and groundwater. Acid sulfate soils are prone to extreme degradation from acidification if drainage exposes sulfide minerals to oxygen.

Soil structure decline is often overlooked as a land degradation phenomenon possibly because it causes quantitative changes in soil conditions that are difficult to measure and observe. Soil structure varies naturally among soils depending on texture, organic matter levels, hardsetting behaviour and dispersiveness. Cultivation can cause soil structural decline by accelerating the oxidation of organic matter. This gives rise to surface crusting, and decreased water storage and drainage. Tillage implements and trafficking of soils by vehicles, machinery or animals can cause soil compaction at the surface or at depth. Compacted soils will generally have decreased water storage, decreased water infiltration, and impeded root penetration. Substrates on mine sites commonly suffer these problems because of low organic matter levels, especially if non-soil materials are used, and because of the use of heavy machinery. Degradation in soil structure apart from decreasing productivity of plants by impeding root growth usually increases the risk of water erosion.

Declining nutrient reserves in soil and increasing nutrient levels are both potential forms and sources of land degradation. Nutrient supplies are limiting in most natural soils. Native vegetation complexes usually have specific adaptions to cope with low supply of one or more nutrients. Natural ecosystems rely on closed nutrient cycles to minimise losses of nutrients and optimise turnover within the soil-plant-animal system. Clearing or disturbing natural ecosystems generally disrupts these nutrient cycles and may result in rapid declines in plant productivity. Examples of this are found in the slash-and-burn land rotations used in many parts of the tropics where declining nutrient supply amongst other factors causes abandonment of plots after 2-3 years cultivation. In other nutrient impoverished soils, use for agriculture was not possible until specific nutrient limitations were overcome by fertilizer additions. Examples of this are in the sandplains of southern Australia where vast areas of trace element deficient soils remained unused for agriculture until the need for these nutrients was recognised. Addition of these nutrients boosted productivity manyfold. Regional and national nutriet budgets in Asia and Africa suggest that nutrient depletion is a widespread form of land degradation (Lefroy et al., 1998; Smaling and Oenema, 1998). By contrast, in many parts of the world the concern in agricultural land use is shifting from nutrient deficiency to nutrient excess. Losses of N and P in particular is becoming a widespread land degradation issue because of the negative effects these nutrient have when they accumulate in surface and groundwater.

Loss of biodiversity is both a form of land degradation and a consequence of it. Clearing of vegetation for other uses is the most obvious cause of loss in biodiversity, and is especially serious for endemic species with limited distribution. However, a range of other more insidious factors threatens biodiversity including: fragmentation of habitat, weed invasion, over-grazing, and pollution. These impacts are most often studied from the perspective of impacts on the flora and fauna, whilst effects on soil organisms are rarely considered.

Contamination and pollution of land by anthropogenic inputs of heavy metals and organic compounds is now a widespread concern. Whilst some contaminated sites may be quite localised, other sources of contamination cause widespread dispersal of the pollutant. The main sources of inputs are: atmospheric deposition from industry or the burning of fossil fuel; mining activities; the dispersal of mine refining residues; disposal of wastes such as sewage sludge, animal wastes, food processing waste, industrial wastes, ash; seepage waters and flood water; agricultural chemicals such as pesticides and fertilizer (McGrath, 2000). Specialised methods for cleaning up contaminated sites including bio-remediation and phyto-remediation are being developed but many are very expensive. The less expensive options such as *in situ* phyto-remediation are still in development and need validation (McGrath, 2000).

Mining is an extreme form of land degradation having devastating effects on flora and fauna and causing drastic changes in landform and hydrology. However, because mining is a temporary land use, the concept of restoration is now well embedded into the life-of-mine cycle. Indeed much has been learned about the concepts and practice of land restoration from mine rehabilitation. The range of factors that may limit restoration after mining is extensive and covers all of those above. In addition, the disposal of residues from ore processing generate a range of substrates for land restoration with extreme properties rarely found in natural soils (Hossner and Hons, 1992). For example, bauxite residue, known as red mud, has a pH >12 when even strongly alkaline soils rarely have pH > 10.5.

Urbanisation and the siting of infrastructure such as roads occupy increasing amounts of land. The change from another land use to urban land use represents a semi-permanent alienation of land and a significant decrease in land suitability that may be accompanied by other forms of degradation such as contamination by heavy metals and organic compounds. Major changes in hydrology result from urbanisation and infrastructure development. The consequence of land alienation for urban, industrial and infrastructure purposes depends on the quality of the land. The rapid urban development in eastern China for example is alienating large areas of formally highly productive alluvial and coastal plain soils used for high yielding rice padi and vegetable production.

Biophysical Factors Limiting Land Restoration

Climatic

Climate sets the underlying growing conditions for plants in land restoration and in a broad sense determines the range of species suited to survive and grow in a particular location. In arid and semi-arid zones, rainfall will be a serious limiting factor for restoration because evaporation exceeds rainfall for several months of the year. The beginning of reliable rainfall and its duration are all important considerations in determining when to plant. The year-to-year variability in rainfall and temperature, as well as diurnal temperature variation, need to be considered when planning land restoration. Variability determines the risk of episodic events such as erosive rainfall or erosive wind events that may damage land restoration. Similarly in areas of uncertain rainfall, the frequency of drought and its effects on success of plant establishment and survival are necessary parts of the land restoration programme. In semi-arid environments, sowing often has to occur during a narrow window of time otherwise plant establishment may fail.

Landform

The prime consideration in restoration is slope stability and effective erosion control. The important components of landform that limit restoration are slope, elevation, aspect and drainage. Restoring pre-disturbance landform is probably the best strategy but in mining increases in elevation and slope commonly occur and this necessarily increases erosion risk and alters hydrology. Engineering measures to stabilise soils, or to prevent erosive water run-off will often be necessary to achieve a stable non-eroding surface. However, re-shaping landforms to reduce slopes or improve aesthetic appeal of restored land are expensive operations and need to be well justified before being undertaken.

Hydrology

Any clearing or significant disturbance of vegetation alters hydrology. Increased run-off is usually the first response on-site, followed by erosion, but downstream consequences include altered streambed and riparian ecology, and increased flooding. Other changes in hydrology include increased profile water storage and increased groundwater recharge. In the case study below, I deal with the development of dryland salinity due to changes in water balance and strategies developed to restore pre-clearing water balance.

Significant changes in surface and groundwater hydrology occur following mining. Increases in elevation and slope created on waste rock or overburden dumps accelerate water run-off and erosion. This may necessitate an increased drainage density in revegetated mine sites to achieve hydrological stability (Bell 1990).

Voids created by mining may intercept groundwater. In many places filling of the voids has created an opportunity for the establishment of wetlands following mining. When the void exposes sulfide ores, the water body may become extremely acidic and require special treatment to ameliorate the problem or to prevent discharge of this water into downstream environments.

Substrate properties

Physical properties. Land degradation often alters the soil physical conditions in ways that decrease its suitability for plant establishment and growth, particularly by changing water storage and availability. Water erosion may strip away topsoil, decreasing soil depth, and exposing sub-soil material with different texture, lower organic matter levels and poorer soil structure. Wind erosion selectively removes clay and humus from the soil surface, increasing the prevalence of coarse materials. Sand deposits from wind erosion may bury topsoil and vegetation in the case of mobile dunes and sand sheets. This process of land degradation causes desertification in large areas of Africa and Asia. Re-establishing stable vegetation cover on mobile wind-blown sand deposits is very difficult because of low water retention and low nutrient content, especially as the areas affected are usually found in arid or semi-arid environments.

Change in soil physical properties may restrict plant recruitment in areas of disturbed native vegetation. An example of this is found in remnant Eucalyptus woodlands in southwest Australia (Yates et al., 2000a,b). Prolonged grazing by hard-hoofed animals has decreased the litter cover of soil resulting in large diurnal soil temperature fluctuations. These combined with the decline in soil structure reduce the success of seedling recruitment by producing seed bed conditions unsuited to seed germination and plant establishment. Tillage by deep ripping to alleviate compaction in these woodlands increased the success of seedling recruitment.

On mine sites, the use of non-soil materials such as overburden, waste rock or ore refining residues as substrates for plant growth results in extreme physical properties that are difficult to revegetate. Coarse residues such as those from mineral sands have low water holding capacity and are prone to wind erosion. Waste rock also has limited water retention because of the limited proportion of fines in the substrate. By contrast, finely-milled residues such as those from gold or bauxite ore have poor drainage and therefore are prone in wet climates to waterlogging. Overburden usually lacks structure and may exhibit dispersive behaviour. Erosion risk on such materials is high when the ground surface is sloping. Crusting at the surface of such dispersive material will also limit recruitment of plants from direct seeding.

Chemical properties. Nutrient levels, acidity, alkalinity, salinity, sodicity and toxic metals and organic compounds are the chemical factors in soils and substrates that can limit plant growth and therefore hamper effective land restoration. Deficiencies of nutrient elements can be diagnosed or predicted by several means (Bell, 1999). Deficiencies once identified can generally be corrected by fertilizer application, although estimation of the appropriate rates is more complicated. At extremes of pH, availability of specific nutrients is often low and fertilizer application may not be very effective. In these situations, soil amendments to change soil pH may be effective in correcting nutrient deficiencies. For example, at extreme acidity, lime application to increase pH will decrease levels of soil solution Al^{3+}, and in so doing increase availability of deficient elements such as P, Mo and Ca. Alternatively it may be necessary to limit plant species grown on these substrates to those adapted to such extreme conditions.

Salinity is a common limiting factor in semi-arid and arid environments. Irrigation is the most common cause of salinity. Even low concentrations of salt in irrigation water can cause salinity to develop if salt is allowed to accumulate in the root zone. Alleviation of salinity and restoration of land productivity requires monitoring of the quality of water, efforts to increase water use efficiency so that the amount of water used is reduced, and finally drainage systems are designed to prevent capillary rise of salt and facilitate leaching of salts.

In semi-arid and arid environments, soils and overburden materials may be naturally saline. The use of these materials for revegetation requires strategies for managing salinity. The salts in substrates may leach over time provided the soils are reasonably well drained, and there is no significant capillary rise of salts. However, since many of these substrates are also sodic, attempts to leach are counterproductive since it induces dispersion that in turn decreases hydraulic conductivity and reduces leaching potential. Addition of gypsum and organic matter are often reliable means of treatment for sodic material (Bell et al., 1995). Alternatively, growing salt tolerant species may be the only cost effective means of revegetation, even though this means a significantly different species mix to that which existed before disturbance (Ho et al., 1999).

Toxic levels of metals, apart from Al and Mn, in natural unpolluted soils are rare although not unknown. On overburden and low grade ore from base and precious metal mines, and on polluted soils, extremely high levels of heavy metals can be a major limiting factor. High levels of heavy metals is often associated with extreme acidity. Lime treatment, and adding P fertiliser or organic matter may decrease both acidity and soil solution levels of phyto-toxic heavy metal species, however, the effectiveness of these strategies is limited. Selection of plants with high tolerances of these toxicities is often the most cost-effective strategy of revegetation. However, the preferred modern practice in mining is to identify these materials and selectively place them in waste dumps where there is no contact with the root zone. The related practice of phyto-remediation involves the growing of plants that are tolerant of heavy metals and actively absorb them from the soil. In this way phyto-remediation may reduce the heavy metal loading of soils however, the technology is still relatively new and needs further testing (McGrath, 2000). Phyto-remediation is also being developed to accelerate the *in situ* bio-degradation of organic pollutants.

Biological properties Soils contain a vast diversity of organisms. This aspect of biodiversity is poorly understood. Similarly the consequences of land degradation for soil biodiversity and soil ecological function is poorly understood. Soil micro-organisms such as rhizobium respond very sensitively to heavy metal loading of soils from land application of biosolids: more sensitively than the growth of plants. Changes in farming systems such organic compared to conventional farming in southwest Australia produces measurable differences in microbial biomass and activity. Mining activities and topsoil storage have quite profound negative effects on microbial biomass and activity (Jasper et al., 199). Restoration of microbial biomass may take 7-

10 years on revegetated mine sites. However, the effects of less drastic ecosystem disturbance on soil ecological functions are not well understood.

Use of substrates for land restoration with low levels of rhizobium inoculum or mycorrhiza may limit the success of restoration by hampering nutrient accumulation and hence decreasing productivity of the vegetation and impeding effective nutrient cycling. Careful handling and re-use of topsoil can often minimise the problems associated with decreased soil biological activity after land restoration. Soil biological activity is impaired by disturbance or handling, but maximum activity is retained if topsoil is re-used immediately without a period of storage.

Plant establishment

Where any of the above substrate conditions is likely to limit plant establishment and growth, the pre-requisite for effective plant establishment is to ameliorate these constraints. In mining, the aim is to plan mine operations so that the most benign materials are placed on top of surfaces to be revegetated. This minimises the need for expensive amelioration of adverse soil conditions. In natural ecosystems, plant establishment is often triggered by disturbance caused by fire, flood, drought, storms, hurricanes, and death of old trees (e.g. Yates et al., 1994). Such disturbances allow the alleviation of resource constraints for plant establishment such as light, water or nutrients.

Plant establishment relies on three strategies, either in individually or in combination: direct seeding, transplanting of nursery-raised seedlings, spreading of seed through topsoil application or spreading cuttings of aboveground vegetation bearing seed. Topsoil will often be the most reliable strategy if it is available, because it contains seed native to the area, adds soil organisms and creates a seedbed for seed germination and establishment. However, in some plant communities, seed is stored in the plant canopy rather than the soil. For restoration of these communities, cutting branches and spreading them on the soil surface has proved effective. Nursery-raised seedlings will generally be reliable for plant establishment but relatively expensive. In addition, there are often many species that are difficult to propagate from seed in the nursery, or generally regenerate by vegetative means. Direct seeding is usually cheaper than other options, although there is still significant cost associated with collection of native seed.

After germination and emergence, establishment of seedlings especially from direct seeding is hampered by weed competition, and grazing by herbivores.

Sustainability

After plant establishment, some means of measuring success of restoration is needed. This is particularly relevant in the mining industry where bond money is often held by government regulatory agencies pending verification that restoration is complete. According to Hobbs and Norton (1996), ecosystem characteristics should be considered when setting goals for land restoration: composition; structure; pattern; heterogeneity; function; species interactions; dynamics and resilience. Presumably the same characteristics should be considered when assessing sustainability of land restoration. This raises the need for reliable low-cost indicators of success (Jasper et al., 1998). To date, most indicators have relied on composition and structure of vegetation. Microbial biomass has been advocated as an indicator of nutrient cycling functions. Indicators for pattern, heterogeneity, dynamics and resilience are less advanced in their development, in part because few land restoration activities have run for long enough to measure these characteristics let alone develop indicators for them.

CASE STUDIES

Using the above framework, and that of Hobbs (2001), two case studies of land restoration will be reviewed. The first of these is about restoration of highly disturbed, but localized, bauxite mine sites in southwest Australia. The second is concerned with restoring ecological processes over broad landscape-scale or regional areas affected by dryland salinity but has as its aims the improvement of productive capability in degraded production lands and the enhancement of nature conservation values. In both cases, brief comment is made on the relevance of these case studies to mine rehabilitation and dryland salinity in Thailand.

Restoration of Bauxite Mine Sites

Constraints for restoration

1. Climate

One of the world's major bauxite mining operations is located in the Darling Range (31-34 ^{o}S), in the southwest of Western Australia (Marshall, 1991). The annual rainfall of 850-1400 mm is restricted to May-October during the cooler months: for the rest of the year there is a prolonged hot dry period. Annual evaporation greatly exceeds annual rainfall. The strong seasonality of rainfall means that the time of sowing is most important for successful seed germination and plant survival over the first summer. Ward *et al*., (1996) concluded that ripping the pits and sowing seed in April, before the break of season rains, was optimal in this environment because it gave maximum species diversity as well as high plant density after rehabilitation.

2. Landform

Bauxite ore is rarely deeper than 5 m, so the mining operation is a shallow open-pit activity. There is no overburden or waste material produced by mining so landform constraints associated with dumps of this type of material are absent. Bauxite ore is extracted from relatively flat or gently to moderately sloping land. Nevertheless, the compacted floor of the mine pit can initiate run-off and erosion. In practice, erosion control is effected by reducing slopes on the edges of the pit, contouring the surface of the pit to conform to the surrounding topography, placing overburden and topsoil in the surface and then ripping to 1.5 m depth across the contour (Koch et al., 2000). The ripping increases infiltration and water storage by creating shallow ridges on the contour and this is generally sufficient to prevent run-off when rainfall exceeds infiltration capacity.

3. Substrate

After bauxite mining removes a 2-4 m layer of coarse sandy gravel and cemented ferricrete material, sub-soil and topsoil are placed on the floor of the mine pit that comprises a leached kaolinitic, sandy clay material. The main difference between the original substrate for plant growth and the new profile is the texture of the sub-soil and this has implications for root penetration (Pratt et al., 2000) and for hydrology.

Physical

Severe compaction occurs on the floor of the pits due to the use of heavy machinery to remove the ore (Nichols *et al*., 1985). Root growth is severely restricted and unless the floor of the mine pit is ripped compaction can be the cause of tree death. Ripping, when the soil is dry, to 1.5-1.7 m depth with a winged tyne alleviates the constraint for root growth. Perhaps the most significant of its benefits is that ripping increases access of roots to old root channels in the weathered regolith. Colonisation of old root channels is important to the longevity of the vegetation. The prolonged dry season each year means that trees need to be deep rooted and have access to water stored deep in the weathered regolith. In the undisturbed forest, roots grow to 40 m (Dell *et al.*, 1983) and presumably restored forest vegetation will eventually need to produce roots to this depth to access enough stored water to survive a sequence of dry years.

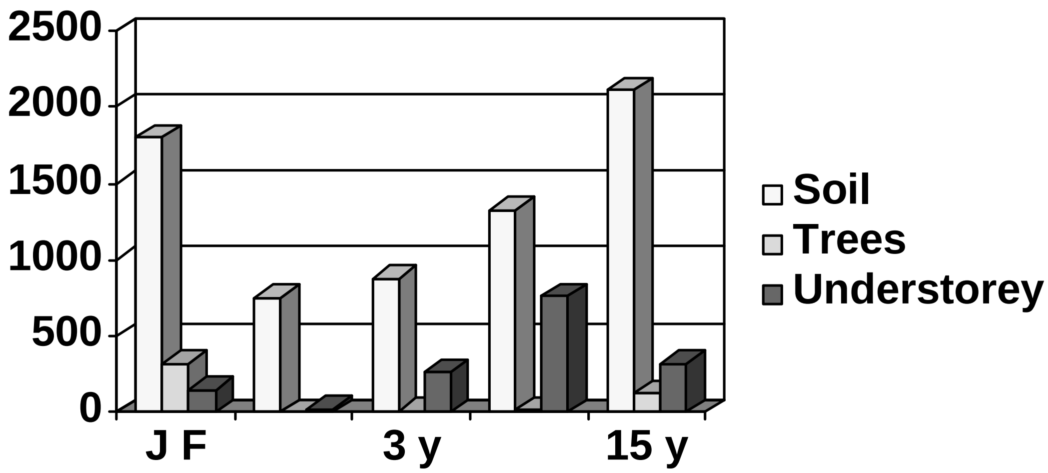
Chemical

Bauxite is the product of intense and prolonged weathering which selectively removes silica and alkali elements from the regolith. Consequently, the soils overlying the bauxite are generally low in nutrients (e.g. Hingston *et al*., 1981). The native forest vegetation that grows on areas mined for bauxite is characterised by numerous adaptations to low soil fertility. These include slow growth rates, reliance on mycorrhizal associations for nutrient uptake, development of proteoid roots in some species, efficient internal recycling of nutrients and parasitic and semi-parasitic growth habits. The highest concentration of organic matter and nutrients is found in the surface 5 cm of the soil (Hingston *et al*., 1981). Consequently the use and management of this thin layer of topsoil is critical in the retention of nutrients, and the subsequent nutrient supply for revegetation.

Revegetation of bauxite pits generally requires fertiliser addition. Fertiliser serves a dual purpose of firstly alleviating initial deficiencies to achieve satisfactory establishment growth, and secondly to replace nutrients removed from the soil-vegetation system during the mining process (see below and Fig. 1). The rate of fertiliser applied is 500 kg ammonium phosphate/ha plus trace elements broadcast after sowing (Ward, 1995).

Figure 1. A. Nitrogen (kg N/ha) and B. biomass (t/ha) distribution in 60-year old jarrah forest, and in 1, 3, 5, and 15-year old revegetated bauxite mine sites (from Ward and Koch 1996). Note: only above ground biomass was sampled; Jarrah forest soil was sampled to 1 m; revegetated soil sampled to 30 cm only.

A.



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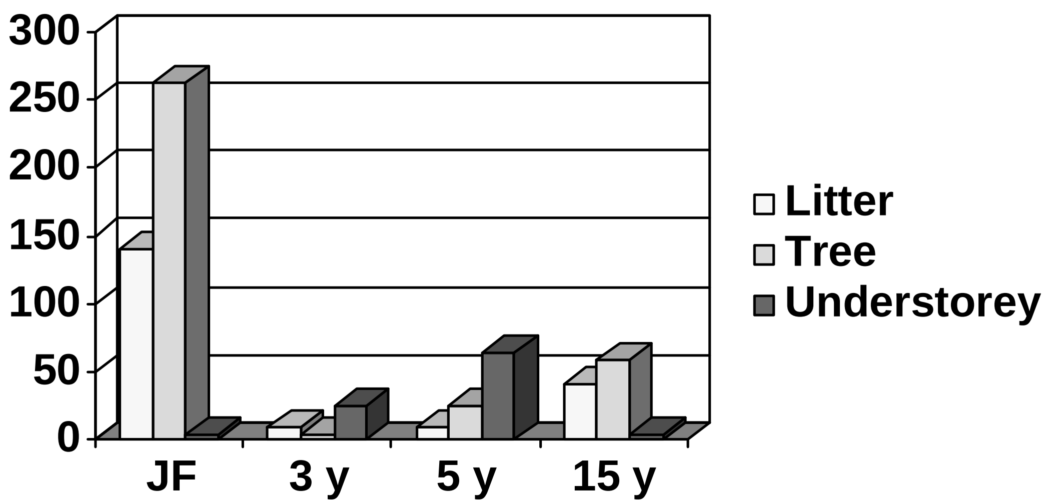

Biological

Soil biological properties are important factors for the re-establishment of forests in southwest Australia. The low inherent fertility of the soils means that nutrient cycling is necessary to optimise the turnover of nutrients locked up in the litter and soil organic matter so as to regulate nutrient supply for plant growth. Secondly, seed reserves in the soil are vitally important for regeneration of forest ecosystems that are comparable in biodiversity to the original forest (Koch et al., 1996).

Soil disturbance severely decreases populations of mycorrhiza, as does topsoil storage (Jasper *et al*. 1989). Immediate re-use of topsoil is the most effective way of retaining soil biological activity. Vesicular-arbuscular mycorrhiza infectivity is restored within 3-4 years after rehabilitation whereas ectomycorrhiza take 7 years (Gardner and Malajczuk 1988). Return of mycorrhizal infectivity is associated with the return of litter. Restoration of soil microbial activity occurs within 7-10 years after revegetation (Jasper et al., 1998).

Many of the soils where bauxite is being mined are infected with the fungus, *Phytophthora cinnamomi* which is pathogenic to many of the species of the jarrah forest, especially the understorey species of the Proteaceae. Jarrah is moderately tolerant to *P. cinnamomi* but under the right combination of moisture and temperature stress, lethal infections of the trees can occur. Hence bauxite mining strategies including topsoil handling must be arranged to minimise the risk of spreading the fungus. The primary means of containment is to map infected areas before mining and to prevent the movement of topsoil from infected to uninfected areas (Ward *et al*., 1993).

4. Plant establishment

Topsoil management is the most important on-site practise that affects plant establishment. Topsoil has diminished value for revegetation if it is stored (Tacey and Glossop, 1980). Direct transfer of topsoil retains most of the characteristics of undisturbed topsoil and achieves 53 % return of soil seed stores (Koch et al., 1996). Mining operations should be planned so that the removal of topsoil from one area before mining coincides with the replacement of topsoil at another. In this way topsoil is removed, transported and spread in a single operation. In the topsoil removal and replacement operation, a double stripping procedure is used. Topsoil from 0-15 cm is removed separately from the underlying overburden (20-80 cm) of gravelly loam. At the site to be rehabilitated, overburden is spread out first on the mine pit floor followed by topsoil. Collecting topsoil to 15 cm still involves substantial mixing of the surface 5 cm layer which contains most of the seeds with the less desirable layers underneath. Since seeds of many of the Jarrah forest species will not emerge from depths >5 cm, mixing is clearly detrimental to high seedling density and high biodiversity in the rehabilitated pits (Grant *et al*., 1996). However, as yet no technology for removal and replacement of the 0-5 cm layer of soil has been developed.

Sustainability

A considerable amount of research has gone into assessing composition and structure goals of restoration. Studies on pattern and heterogeneity are not so advanced, possibly because these are characteristics that emerge with time. The resilience of the revegetated areas to disturbance from fire is also under investigation. Functional characteristics of the revegetated mine pit including hydrology and nutrient cycling have been investigated. Based on these studies, completion criteria for bauxite mine restoration have been agreed by Alcoa and the WA State Government. In the context of the present paper, the studies on the restoration of nutrient cycling processes are of particular interest.

In the jarrah forest ecosystem, as in many other forest ecosystems (White, 1997), a high proportion of its total nutrient content is present in the litter and plant biomass (Fig. 1). Since mining removes most of litter and plant biomass, the bauxite mine pits are depleted in nutrients (e.g. Fig. 1 for biomass and nitrogen). Thus restoration of forest biomass and productivity levels comparable to those of the pre-mined forest requires a significant input of nutrients. Moreover, the accumulation of these nutrients in biomass, and their cycling in the soil-plant-animal system must be at a rate commensurate with satisfactory forest productivity (Ward et al., 1990). The strategy for doing so includes:

- direct replacement of topsoil which not only contains significant levels of

nutrients but also symbiotic micro-organisms that facilitate nutrient uptake (mycorrhiza and Rhizobium) and soil micro-organisms necessary for litter and organic matter decomposition;
- fertiliser addition in the year of revegetation;
- ensuring that legume species comprise a significant proportion of the understorey species to increase nitrogen accumulation after revegetation.

In addition, considerable research has been undertaken to assess nutrient accumulation in biomass, and verify that key processes of nutrient cycling, such as litterfall were functioning. Five years after revegetation the total store of key nutrients like N was largely restored, but the compartments in which it occurred were different from the original forest. In particular a large store of nutrients had accumulated in to the understorey vegetation largely dominated by the nitrogen fixing legumes but the stores in soil and trees were still depleted. Over the following 10 years, natural senescence of the legumes, and increased biomass accumulation by the trees resulted in a partitioning of N among the main compartments resembling closely that of the original forest.

Studies such as those by Ward and Koch (1996) are relatively expensive to carry out and considerable interest has emerged in the possibility of developing simpler indicators of success in restoration of nutrient cycling (Jasper et al., 1998). Within 4-8 years after revegetation, soil microbial biomass returns to levels comparable to that of undisturbed forest. Remotely sensed data (normalised difference vegetation index-NDVI) appeared also to be a useful indicator of plant productivity. It should also be noted that the return of fauna, particularly invertebrates like ants and spiders, may be used as indicators of nutrient cycling (Nichols, 1998).

Restoration after tin mining (Tanpibal and Sahunalu, 1989) and coal mining (Mungkorndin 1994) are carried out in Thailand but practices have not been the subject of prolonged, systematic and comprehensive research to develop best practice (EGAT, 1996). The above example from bauxite mine restoration would form a good model on which to design mine restoration programs in Thailand even though end land use is generally not a restored native ecosystem. In lignite mining at Lampang,for example, topsoil is often in short supply and overburden is the main substrate for plant growth. High fertiliser rates are needed to achieve satisfactory growth (Mongkorndin, 1994). An emphasis on planting timber and horticultural trees at Lampang will need site specific land restoration research to develop a sustainable system for those conditions.

Restoration of land affected by dryland salinity

Globally, dryland salinity is less common than irrigation salinity (Gassemi et al., 1995). However, in Southern Australia and Northeast Thailand, dryland salinity is the most prevalent form of salinity and spreading (Williamson et al., 1989; Williamson, 1998). By 1996, about 10 % of the landscape in the wheatbelt of WA was affected by dryland salinity. The area of dryland salinity is predicted to double in the next 20-40 years, and could reach 30 % of the landscape if no remedial action, or ineffective steps, are taken (Ferdowsian et al., 1996). Such a massive increase in dryland salinity will have large-scale ecological consequences for farmland productivity, for water resource quality, nature conservation in wetlands and riparian environments and other native vegetation in areas prone to groundwater discharge: infrastructure such as roads and railway lines will also suffer severe damage and flood incidence and severity is expected to increase. In Northeast Thailand, up to 30 % of land could potenitally become salt-affected (Yuvaniyama, 2001).

Dryland salinity in southwest Australia is a case study in how perturbation in water balance can have devastating consequences for landscapes, both on-site and off-site. In fact many commentators categorise dryland salinity as a water problem rather than a salt problem *per se*.

The fundamental environmental change that gave rise to dryland salinity in southwest Australia was a change in the water balance. From 1950-1980, large tracts of land were cleared of their native vegetation for agricultural land use. In some parts of the wheatbelt, <5 % of the original vegetation remains. In most areas well over 80 % of the landscape has a drastically altered water balance. Whereas the plants removed were predominantly deep rooted perennial species, those used by agriculture were predominantly shallow rooted and annual. The main consequence of the changed vegetation was a decrease in evapo-transpiration. Annual plants use water only during their main growing season from May to October, and their usage is limited by the fact that roots are generally confined to the surface 50-100 cm. Thus the extra water is distributed to increased run-off causing erosion and waterlogging; and increased recharge to groundwater (Table 1). Williamson et al., (1989) similarly concluded that dryland salinity in Northeast Thailand was induced by deforestation of the uplands to develop upland crops like kenaf.

Table 1. Changes in water balances for cleared catchments before and after clearing. From Williamson (1998).

Catchment	Year	Rainfall (mm)	Inter-cep-tion (mm)	Evapo-tran-spiration (mm)	Change in water storage (mm)	Change in ground-water sto-rage (mm)	Stream flow (mm)
Wights forested	1975	1027	130	855	-28	-11	81
Wights cleared	1985	1147	0	565	-	21	115
Lemon forested	1975	739	74	656	4	-1	5
Lemon cleared	1983	821	38	708	-	19	56

Two other attributes of the landscape have, combined with water balance, given rise to dryland salinity. Firstly much of the landscape of southwest Australia is weathered to 5-30 m depth (McArthur, 1991). Secondly, the landscape has been geologically stable for millions of years. This has allowed the accumulation of salts from rainfall accretion in the deeply weathered regolith (Williamson, 1998). Salt contents in extreme cases of up to 20,000 t/ha have been reported (Moore, 1998), most of it stored below 5 m depth (Williamson, 1998). Prior to clearing the native vegetation, plant roots in the upper 5 m of the regolith were largely separated from the salt bulge below, and the semi-permeable aquifer at the base of the regolith was often dry (Williamson, 1998). Hence under these conditions salt storage was benign. However, with increased recharge the aquifers have filled, causing water levels to rise at the rate of 0.2-1 m per year. After a 20-30 year period of groundwater rise, saline groundwater discharge is usually observed commencing generally in valley floor landforms. It is still unclear whether the salt in the discharge comes predominantly from the aquifer, or from salt stored in the regolith above, although from a practical viewpoint it probably does not matter.

In Northeast Thailand, the origin of salt is not rainfall accretion but salt stored as halite sequences in the Mesozoic sediments that underlie the Korat plateau (Williamson et al., 1989). However, replacement of deep-rooted perennial vegetation by annual crops is believed to be the main cause for perturbation of the water balance and to the rising groundwater.

Reversing dryland salinity

Ultimately, restoring the pre-clearing water balance seems to be the only complete solution to the dryland salinity problem. This requires treatments in recharge zones of landscapes to decrease recharge rates to those that existed before clearing. The species that can mimic recharge rates that existed before clearing will therefore probably need to be deep rooted and perennial. They will also have to be adapted to a variety of soil conditions and climatic regimes across the southwest of Western Australia. Finally, it is imperative that most of the species chosen to fulfil the above functions will also be economically viable within the farm enterprise.

In order to manage dryland salinity it is necessary to understand the groundwater systems as well as water balance components. With the intense winter rainfall, temporary soil water saturation is prevalent. Hence, soils where water content exceeds field capacity, and especially waterlogging-prone areas will be the main zones for recharge. Wherever deep rooted perennial plants are removed and replaced by shallow rooted annual species, recharge is likely to occur once water infiltrates below about 1 m. With deep-rooted species, water infiltrates to 6-10 m depth during and following the winter and spring rain but may still be taken up in spring and summer by roots before it becomes recharge. Hence because of the rainfall distribution, in extensively cleared landscapes recharge can occur virtually anywhere in the landscape that is not actively discharging (George et al., 1991). Computer modeling adds weight to the conclusion that the only fully effective revegetation solutions for salinity control in southwest Australia are with deep rooted perennial vegetation over the whole catchment (Clarke et al., 1998a). Even systems like agro-forestry that place a high density of woody shrubs and small trees in rows 30 m apart were insufficient in the modeling scenarios to restore water balance and achieve complete control of salinity. Continued reliance in the farming system on annual shallow rooted crops such as cereals is problematic because these crops will allow continued recharge. Loss of these species is also problematic because they are the main source of income for farmers. An experimental system that might allow continued annual crops without increasing salinity is what is known as phase farming. A period of 3-5 years of shallow roots crops allows increased water storage in the upper regolith (3-10 m). It is proposed that this is followed by a 5 year period of deep rooted trees or shrubs that are able to extract stored water in the upper regolith before it infiltrates into the aquifer.

In the meantime, until recharge control treatments start to cause decreases in discharge, treatments are also needed in the discharge areas. These may include both engineering treatments to alleviate waterlogging as well as vegetation options that cope with saline waterlogged conditions.

The land unit for water balance management is the surface catchment, usually 1st or 2 nd order. In southwest Australia, geology has a significant effect on groundwater recharge and discharge. A knowledge of the location of geological features such as faults, sediments and dykes assists in the location of recharge and discharge treatments that are effective at a paddock and catchment-scale in controlling salinity (Clarke et al., 1998b, 1999).

In Northeast Thailand there has been considerable investigation of salt tolerant species that can be grown on various classes of salt-affected soil (Yuvaniyama, 2001). There has also been significant spread of eucalyptus plantations on upland areas of Northeast Thailand. However, landscape-scale studies of revegetation and their impact on landscape water balance and dryland salinity do not appear to be available. The case study on dryland salinity in Western Australia may therefore serve as a useful model for such investigations.

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