Ecological restoration of land with particular reference to the mining of metals and industrial minerals: A review of theory and practice

J.A. Cooke and M.S. Johnson

Abstract: Mining causes the destruction of natural ecosystems through removal of soil and vegetation and burial beneath waste disposal sites. The restoration of mined land in practice can largely be considered as ecosystem reconstruction — the reestablishment of the capability of the land to capture and retain fundamental resources. In restoration planning, it is imperative that goals, objectives, and success criteria are clearly established to allow the restoration to be undertaken in a systematic way, while realizing that these may require some modification later in light of the direction of the restoration succession. A restoration planning model is presented where the presence or absence of topsoil conserved on the site has been given the status of the primary practical issue for consideration in ecological restoration in mining. Examples and case studies are used to explore the important problems and solutions in the practice of restoration in the mining of metals and minerals. Even though ecological theory lacks general laws with universal applicability at the ecosystem level of organization, ecological knowledge does have high heuristic power and applicability to site-specific ecological restoration goals. However, monitoring and management are essential, as the uncertainties in restoration planning can never be overcome. The concept of adaptive management and the notion that a restored site be regarded as a long-term experiment is a sensible perspective. Unfortunately, in practice, the lack of post-restoration monitoring and research has meant few opportunities to improve the theory and practice of ecological restoration in mining.

Key words: restoration, rehabilitation, revegetation, mining, succession, ecological theory.

Résumé : L’exploitation minière entraîne la destruction d’écosystèmes naturels par l’élimination du sol et de la végétation et l’établissement de sites d’enfouissement pour les déchets. En pratique, la restauration des terrains miniers peut largement être considérée comme une reconstruction d’écosystème avec la remise en place de la capacité du terrain à capter et à retenir les ressources fondamentales. Dans la planification de la restauration, il est impératif que les buts, les objectifs et les critères de succès soient clairement établis pour s’assurer que la restauration soit entreprise de façon systématique, tout en réalisant que ces paramètres peuvent nécessiter quelques modifications plus tard, à la lumière de la direction dans laquelle la succession de la restauration suit son cours. Les auteurs présentent un modèle de planification selon lequel la présence ou l’absence de sol fertile conservé sur le site reçoit le statut de problématique pratique primaire en vue de la restauration

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écologique des terrains miniers. Ils utilisent des exemples et des études de cas pour explorer des problèmes importants ainsi que des solutions pour la pratique de la restauration liée à l’exploitation des métaux et minéraux. Bien que la théorie écologique manque de lois générales avec des applications universelles au niveau d’organisation de l’écosystème, la connaissance écologique a un grand potentiel heuristique et une forte applicabilité pour des objectifs de restauration écologique sur des sites spécifiques. Cependant, le suivi et l’am énagement sont essentiels, puisque les incertitudes dans la panification de la restauration ne peuvent jamais être totalement surmontées. Le concept d’aménagement adaptatif avec la notion qu’un site restauré doive être considéré comme une expérimentation à long terme est une perspective raisonnable. Malheureusement, en pratique, le manque de suivi et de recherche post-restauration offre peu d’occasions d’améliorer la théorie et la pratique de la restauration des terrains miniers.

Mots clés : restauration, réhabilitation, végétalisation, exploitation minière, succession, théorie écologique.

Introduction

The scale of human activities has become such that most of the ecosystems of the earth have been disturbed in some way (Ehrlich 1993). More than 40% of the terrestrial vegetated surface of the earth has been directly disturbed (Daily 1995) and its natural productive capacity diverted, reduced, or destroyed (Vitousek et al. 1986). This is largely through overgrazing, deforestation, agriculture, over-exploitation for fuelwood, and urban and industrial use (Daily 1995). The area of land directly altered by mining industries is still relatively low in terms of the global inventory of degradation, but can represent considerable quantities on an individual country basis. Examples of estimates on a country basis include, for the U.S.A. 3.7 Mha (Dudka and Adriano 1997), for China 2 Mha (Guo et al. 1989), and for South Africa about 0.2 Mha (Fairbanks et al. 2000). Further, the scale of mining is increasing and the impacts are generally more severe than most other kinds of disturbance (Walker and Willig 1999). In this context of increasing land degradation, both the ecological and economic imperatives demand that restoration of land be prioritized even if restoration ecologists are “doomed to fight an uphill battle” (Ehrlich 1993). This review discusses some of the main issues, both theoretical and practical, concerning ecological restoration of land in the particular context of metal and industrial mineral mining.

The mining context

The focus of this review is the mining industries, whose products are the metals and minerals on which many national economies depend (Table 1). There has been a progressive increase in the production of most of these metals and minerals over the last 50 years (Table 2). The mining of metals and minerals represents a large range of activities, which include primary (extraction) and secondary (milling, processing, refining, and waste disposal) phases (Barbour 1994). Table 3 summarizes the main methods of underground or surface extraction and the nature of the associated land disturbances. In many of the forms of mining it is the waste production and disposal that can cause the most extensive and long-lasting disturbance to land. The disposal of rock and overburden, the construction of impoundments (dams) for the fine tailings ($<$0.1 mm) produced from the milling operations, and the disposal of slags from the smelting and refining stages can involve large areas of land.

The history of modern mining is one of globalization (Mining Annual Review 1995), together with the economic development of lower grade ore bodies with larger mines, increased waste production and consequently greater potential environmental impacts, and land disturbance. The history of modern copper production illustrates this (Mining Annual Review 1985), where the average ore grade has decreased from 4% in 1900 to 0.5% in 1975, with a considerable increase in the tailings produced, going from approximately 17 to 290 Mt a^{-1} worldwide over the same period (Williamson et al. 1982). In terms of land disturbance this can translate into the production of both massive open pit workings (e.g., $>$3 km wide and 0.5 km deep) and individual tailings dams of over 2000 ha.

Table 1. Major metals and industrial minerals produced by the mining industry.

Type	Main minerals
Energy mineral	Uranium
Precious metals and minerals	Gold, silver, platinum group, diamonds
Ferrous (steel industry) metals	Iron, manganese, nickel, chromium, cobalt, molybdenum
Major non-ferrous metals	Aluminum, lead, zinc, copper, tin
Speciality metals	Titanium, cadmium
Industrial minerals	Asbestos, gypsum, phosphate, potash, kaolin, salt, perlite, vermiculite, fluorspar, barytes, graphite, sulphur

Table 2. World production of some metals and minerals (Mt a^{-1}).

	1945	1961	1984	1995
Silver	0.005	0.007	0.009	0.015
Iron	157	499	804	1018
Manganese	4.0	13.8	24.9	21.5
Aluminum	3.9	13.7	69.3	114.4
Copper	2.1	4.2	6.3	8.2
Chromium	1.1	4.2	9.2	12.1
Lead	1.2	2.4	2.3	2.7
Nickel	0.15	0.37	0.7	1.0
Tin	0.09	0.16	0.16	0.18
Zinc	1.6	3.3	5.0	6.7
Fluorspar	2.2	2.7	4.5	4.0

Note: These data are comparative and represent mine production (ore or metal in ore). Data from Johnson and Putwain 1981; Mining Annual Review 1985; Metals and Minerals Annual Review 1996; and Anon 1997.

The direct impacts of mining disturbance to land surfaces are usually severe with the destruction of natural ecosystems, either through the removal of all previous soils, plants, and animals or their burial beneath waste disposal facilities. Phosphate mining on the small Pacific island of Nauru provides an example of both severe mining disturbance, with 80% of the islands soil and vegetation lost, and extreme ecological restoration challenges (Anderson 1992; Gowdy and McDaniel 1999). Structural faults and major failures of mining waste impoundments have caused gross pollution of the wider environment and threatened agricultural and sensitive natural ecosystems. For example, the Ok Tedi gold and copper mine in Papua, New Guinea, discharged 60 Mt of tailings per year into the Fly river and the Gulf of Papua over many years from an original dam failure in 1984 (Allan 1995; Anderson 1996), and the Los Frailes open-pit pyrite mine in southern Spain released 4.5 x 10^6 m^3 of acidic and metalliferous slurry into the Agrio and Guadiamar rivers, which covered some 4000 ha of land and threatened the largest conservation area in Europe, the Doñana National Park (Sassoon 1998; Cabrara et al. 1999). Further widespread geographical impacts are possible through the pollution of air (e.g., sulphur dioxide and metals, including mercury (Hutchinson and Whitby 1974; Ripley et al. 1996; Dudka and Adriano 1997; He et al. 1998)), surface waters (e.g., acid mine drainage (Ferguson and Erickson 1988), mercury (Veiga and Meech 1995)), and ground water (Sengupta 1993). Other indirect or multiplier effects can

Table 3. Main types of metal and industrial minerals mining and the associated land disturbance (adapted from Cooke (1999)).

Mining method	Brief description	Land disturbance
Shallow underground mining	Seams up to 60 m deep.	Surface spoil heaps; subsidence and collapsed old workings often left derelict
Deep underground mining	Seams deep underground, accessed through shafts	Subsidence; surface waste disposal — spoil heaps, tailings, and slurry lagoons usually remain after closure
Strip mining (opencast, opencut)	Horizontal or sloping seams usually up to 60 m below surface, taken out from surface.	Removal of vegetation and stockpiles of overburden-rock, subsoil, and topsoil. Temporary if using progressive backfilling; mineral processing and waste disposal facilities may be left after closure
Dredge mining	Alluvial and mineral deposits throughout bulk of mined materials; mining ponds created with floating dredger and concentrator	Removal of vegetation and stockpiling of topsoil and tailings from concentrator, usually progressively backfilled
Open-pit mining	Ore body near surface usually steeply dipping seam or pipe. Ore taken out by blasting or hydraulically	Little waste left to backfill; steep faces and pit floor left; also rockdumps and waste disposal facilities such as tailings lagoons remain after closure

occur through the fragmentation of the original natural ecosystems and the alteration of surface and groundwater drainage patterns.

There is also a probability of recurrence of impacts after mining (even after site reclamation) where mining wastes are regarded as secondary ore bodies, and old spoil heaps and tailings dams are reworked for metals and gangue minerals. The recent accident at the Aural gold and silver producing plant at Baia Mare in Romania clearly showed the environmental consequences of reprocessing wastes activities when 50–100 t of cyanide were released into the Danube river system. The cyanide plume travelled to the Black Sea some 2000 km from the source of the spill, causing mortality of river biota (UNEP/OCHA 2000).

Mining is a temporary landuse because, in any one place, the mineral deposit is finite and eventually exhausted. The social and legislative context of mining in many parts of the world today means that some form of landuse goals will be set prior to the granting of planning permission for a new mine. Reclamation considerations will be incorporated into the mine planning such that it becomes a major governing factor in the mining operations, waste disposal, and site closure (Johnson et al. 1994). However, it should be emphasized that there is a considerable past legacy of poor reclamation practice that, at best, has not provided any successful ecosystem development and, at worse, has allowed continual environmental damage (Berger 1990). It may be concluded that it is the reclaimed land surface that remains indefinitely and is required to meet the major goal of sustainability, which is the maintenance of the land use options of future generations (Haigh 1993). In this context, ecological restoration of mined land represents the best approach to promote both sustainability and the maintenance of biodiversity.

What is ecological restoration?

There is currently no agreed terminology in restoration ecology and in many ways this reflects a theoretical debate, not just about restoration, but also about the ecologist's view of nature. The term reclamation describes the general process whereby the land surface is returned to some form of beneficial use. Where reclamation is guided by ecological principles and promotes the recovery of ecological integrity (SER 1996) the term restoration is used.

A more detailed set of definitions has been proposed to reflect the goals (or success?) of the reclamation process. Here, restoration refers to reinstatement of the original (pre-mining) ecosystem in all its structural and functional aspects, rehabilitation is the term used for the progression towards the reinstatement of the original ecosystem, and replacement is the creation of an alternative ecosystem to the original (Bradshaw 1984, 1990). It is likely, given the nature of the science of ecology, that these definitions make the use of the word "restoration" redundant and only used by the optimistic or naive ecologist. Obvious caution, when dealing with the restoration of complex ecosystems, may lead to the overuse of the word "rehabilitation" in official documents such as Environmental Impact Assessments of proposed mining operations (e.g., Lubke et al. 1993). In practice, the terms are still used interchangeably and restoration, rehabilitation, and replacement (or habitat creation in a nature conservation context (Anderson 1995)) may be described as the resetting of an ecological clock (Cairns 1991). All of these terms are included in the one general term restoration in this review, and this reflects both the earlier literature (e.g., Johnson and Bradshaw 1979) and recent discussions of the topic (e.g. Wyant et al. 1995; Hobbs and Norton 1996; Pastorok et al. 1997).

As well as accepting the broader view of ecological restoration, it should be emphasized that it is a "process", driven by ecological knowledge and research and not just the means of producing a "product", e.g., the (fully?) restored pre-mining ecosystem. From this perspective ecological restoration is about a broad set of activities (enhancing, repairing, or reconstructing degraded ecosystems (Fig. 1)) appropriate to the specific types (or severity) of disturbances and not the outcomes per se of such activities (Hobbs and Hopkins 1990; Cairns 1991; Hobbs and Norton 1996). Within this broad overview of restoration activities, the restoration of mined land can largely be considered as ecosystem reconstruction. Here it is usually a question of the re-establishment of the capability of the land to capture and retain fundamental resources (energy, water, nutrients, and species). In nearly all cases, the resilience of the pre-mining ecosystem has been compromised (more precisely the amplitude of the system has been exceeded (Westman 1991)) and the recovery process, when left to natural succession, is often too slow (Bradshaw 1990; Hobbs and Norton 1996).

This view of restoration as a process is regarded as more appropriate to current thinking of succession: that it is not governed by any holistic or universal law (Miles 1987) and recognizes multi-directional pathways for any given succession, driven by stochastic processes without assumption of either long-term stability or any one equilibrium endpoint (Glenn-Lewin and van der Maarel 1992). Further, our view of nature has changed from one of its balance (homeostasis) to acknowledging its dynamic flux (homeorhesis), especially at the larger spatial, temporal, and organizational scale of natural communities in landscapes, i.e., 100 species, 100 years, and areas larger than 100 ha (Pimm 1991; Cairns 2000).

It is at this scale of nature that a forward-looking perspective of ecological restoration of mined land must operate. However, from this theoretical view of nature the decision as to what to restore to becomes something of a moving target (Jackson et al. 1995; Webb 1996). Should it be an exact replica of the immediate pre-mining ecosystem, an ecologically superior (more pristine?) and perhaps historical standard, or even a future state, which is the condition that natural succession may have produced if no disturbance had occurred (Cairns 1989; Cairns 1991; Westman 1991; Clewell 2000)? Further, with a view of restoration as a process, the essential role of monitoring and management is emphasized as the uncertainties in restoration planning can never be overcome.

An overview of restoration planning

Ecological restoration must involve an orderly set of considerations that promote successful procedures and practices. Often these practices, although based on similar general considerations, will need to be innovative because of the unique set of circumstances each area and ecosystem to be restored

Fig. 1. The continuum of general ecological restoration effort from enhancement to reconstruction (adapted from Hobbs and Norton (1996)).

Enhancing Conservation Value in Disturbed Landscapes

e.g. tackling overgrazing, alien species invasion, pollution, and decreasing landscape fragmentation; the introduction of semi-natural areas (patches and corridors) within agricultural or commercial forestry landscapes

↓

Repairing Degraded Land

e.g. improving the productivity and biodiversity of land with soil erosion or salinization problems

↓

Reconstruction of Highly Degraded Sites

e.g. amelioration of substrates where the original soil is lost and introducing vegetation where it is absent

represents. The important stages in restoration planning are shown in Fig. 2. This restoration planning model recognizes that, for most mine reclamation programmes over the last 30 years, an over-riding consideration has been whether the topsoil has been retained or lost (Johnson and Bradshaw 1979; Bradshaw 1983, 1984, 1990). This will in all probability determine how quickly a pre-mining ecosystem can be restored and whether such a restoration goal is ecologically possible and sustainable.

Although it has been emphasized that restoration should be conceptualized as a process, the product is still important. It must be the starting point for any restoration project (i.e., restoring to what?) and it is essential to define the product in a time frame (i.e., restored by when?). This restoration goal must then be translated into specific process attributes — the restoration objectives (Fig. 2).

The restoration objectives must be formulated from a detailed knowledge of the basic structural and functional characteristics of natural ecosystems (Table 4). Ecological restoration may implicitly want all attributes to be achieved (e.g., to claim close correspondence to the pre-mining ecosystem), but the practical context of any site restoration demands that the following are considered: speed of attainment, economics (or cost-benefit), achievability, and long-term stability with on-going management at reasonable (low) cost (Bradshaw 1990). Such practical considerations are necessary for, without them, unrealistic objectives both in ecological and economic terms can be set.

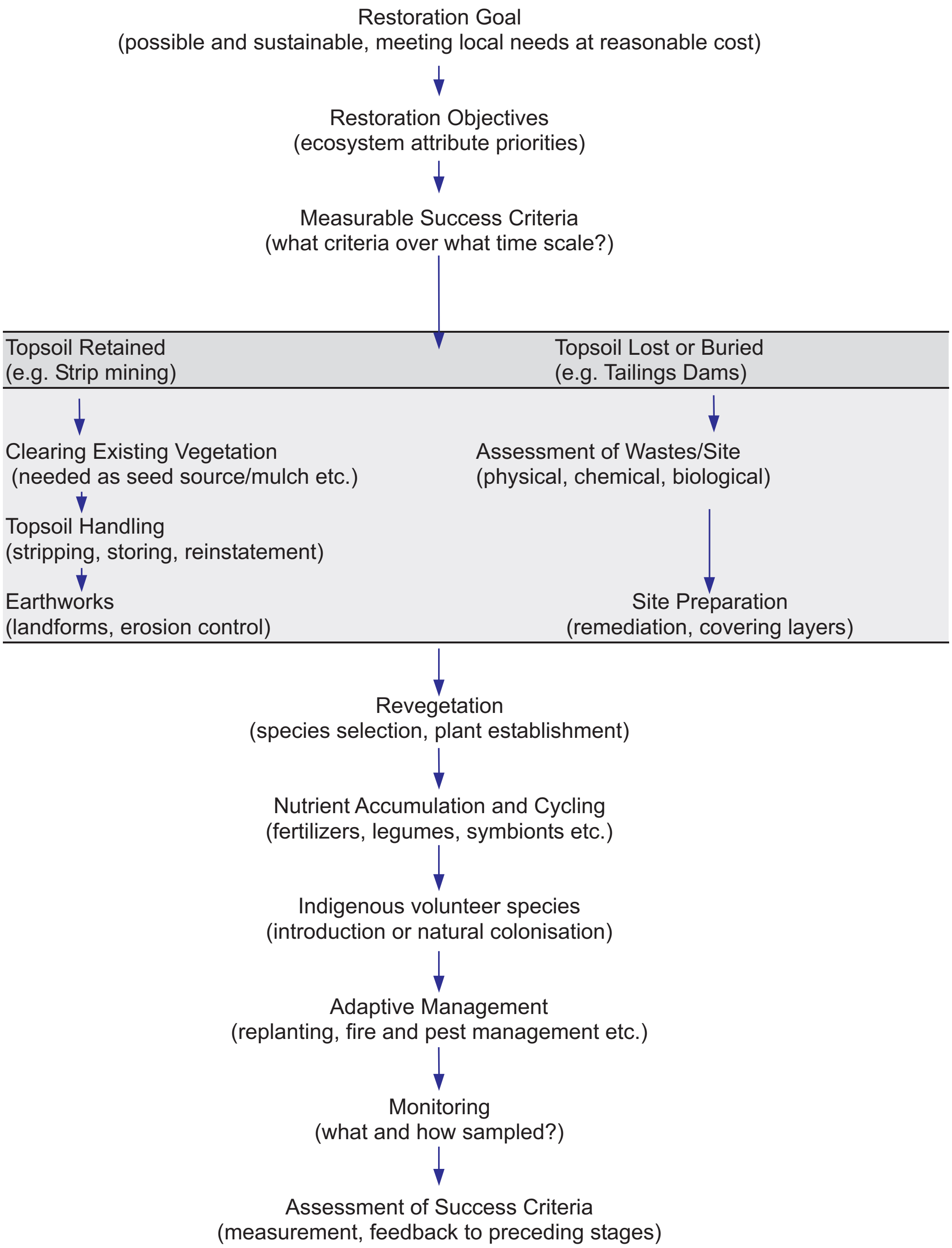
The development of measurable criteria derived from the particular desired community and ecosystem characteristics (as restoration objectives) for judging restoration success, have proved difficult to establish (Johnson and Putwain 1981; Chambers and Wade 1992; Hobbs and Norton 1996). Cairns (1993) provides three general success guidelines that the restored ecosystem should attain:

(i) self-regulation for some set period of time, where self-regulation means the structural and functional attributes persist in the absence of whatever “subsidies” (fertilizer, seeding etc.) may have been necessary during the initial phases of implementation;

(ii) the design criteria (restoration goal and objectives) established before restoration was undertaken;

(iii) no observable adverse effects in the larger ecological landscape.

Fig. 2. The stages in the conceptual planning of the ecological restoration of mined land (adapted in part from Johnson et al. (1994), and Australian Environment Protection Agency (1995)).



From these, it can be seen that it is absolutely necessary to have restoration objectives that have unambiguous operational definitions (technically feasible), which are ecologically sound (scientifically valid) and socially relevant, and that are accessible to measurement and prediction (Wyant et al. 1995; Pastorok et al. 1997; Cairns 2000). The ecosystem characteristics usually measured are those related to the composition, structure, and pattern of the vegetation (1–3 in. (1 in. = 2.54 cm) Table 4 and Allen

Table 4. Ecosystem characteristics for consideration as ecological restoration objectives (adapted from Hobbs (1999)).

1.	Composition: species presence and their relative abundance
2.	Structure: vertical arrangement of vegetation and soil components
3.	Pattern: horizontal arrangement of system components
4.	Heterogeneity: a variable composing of characteristics 1–3
5.	Function: performance of basic ecosystem processes (energy capture, water retention, nutrient cycling)
6.	Species Interactions, e.g., pollination, seed dispersal etc.
7.	Dynamics and resilience: succession and state-transition processes, ability to recover from normal episodic disturbance events (e.g., floods, drought, fire)

Table 5. Structural vegetation measurements commonly used to monitor restoration and the related functional characteristics that are implied (adapted from Allen (1992)).

The structural measurement	Functional characteristic needed
Biomass ($g\,m^{-2}$)	Productivity ($g\ m^{-1}\ a^{-1}$)
Species density	Species turnover (mortality, reproduction)
Species richness	Functional loss caused by missing species
Life form spectra	Functional loss caused by absence of life-forms
Indices of diversity/similarity	Species interactions which promote ecosystem functioning

1992). An assumption is that if the structure of vegetation is similar to the desired ranges set for those variables (whether derived from a reference area or not) then the “related” functional characteristics will also be satisfactory (Table 5). The cost and length of time over which measurements need to be made usually mean that measurements of function are not used (Chambers et al. 1992). However, it should be recognized that some important structural measurements are also not usually made (Chambers et al. 1994). In particular, measurements concerning plant root architecture including mycorrhizae, the structure of the soil biotic community, and animal species numbers are not usually made, even though they can often provide important indications of long-term productivity and successional pathways (Chambers and Wade 1992).

The ecological considerations needed for practical restoration planning, summarized in Fig. 2, must be considered in some detail in relation to situations where topsoil has been lost or retained within the mining and waste disposal operations.[1] In the restoration of sites where the topsoil has been lost, the major ecological challenges are still concerned with plant species–substrate interactions, i.e., revegetation. Restoration practice where topsoil has been retained focuses less on vegetation establishment and more on the spatial and temporal factors affecting species colonization and establishment, the criteria for monitoring and assessing success, particularly in the longer term, and the restoration of natural indigenous ecosystems on mined land. In the following sections the examples, case studies, and discussion reflect these different perspectives. However, it should always be remembered that adaptive restoration planning is site specific and the ecological knowledge base plays a continually active and critical role. Thus, any of the ideas presented and discussed below may be found to have relevance in preventing the failure of any particular restoration project.

[1] Smelter-damaged land is more of a special case, however, where the impacts on surface soils are severe, such as topsoil erosion or soils strongly acidified and (or) metal-contaminated, then these can be included in the “topsoil lost” category.

Restoration in practice: where topsoil has been lost

Mining substrates

Compared to normal soils, mining substrates derived from deep in the earth or wastes produced from the processing of the minerals can present extreme challenges to the colonization by plants and the formation of any kind of self-sustaining ecosystem. The physical and chemical nature of the substrates from mining operations is such that a fundamental basis for restoration, that of establishing vegetation, can be extremely difficult in the absence of either soil as a cover material or of some other suitable alternative material for capping, amending, or dilution of mining substrates. Physical texture may be very coarse as in some rock wastes, intermediate as in sand wastes, or very fine as in milled tailings. Fine texture and no organic matter may lead to high bulk densities, extreme compaction, low water infiltration rates, and surface waterlogging (Johnson et al. 1994).

Nearly all mine substrates have very low levels of macronutrients (especially nitrogen (N), phosphorus (P), and potassium (K)). Low pH is a particularly intransigent problem in wastes that contain iron pyrites, which, on weathering, will generate sulphuric acid and (if there is no acidic neutralising capacity in the waste) induce pH values of <2.0. Toxicity, especially of aluminium, zinc, and other metals in acidic wastes, can be a significant problem for plant growth. High salinity can be caused by natural weathering, acid neutralization by carbonates, additions of chemicals in the milling and concentration processes, and by evaporation from the surface in warm climates. These constraints on plant growth have been reviewed extensively (Bradshaw and Chadwick 1980; Williamson et al. 1982; Johnson et al. 1994; and Tordoff et al. 2000).

Mining substrates do vary considerably in their physical and chemical nature, but they are likely to inhibit natural colonization by most plant species for many years. However, often a few plant species (which may be particularly tolerant or have tolerant ecotypes or populations) may form an open vegetation cover representing an arrested succession prevented from further development because of the toxicity of the metalliferous mine spoils, the infertility, or the extreme acidity of pyritic mine wastes and tailings. For example, centuries of mining of metalliferous ores and their processing have created many derelict areas throughout the world with substrates characterized by high concentrations of metals.

These old mining areas have often developed plant communities, through natural colonization, with distinctive metallophyte or pseudo-metallophyte species (Antonovics et al. 1971), which have been described by metal type and soil metal concentration. Examples include the well-described floras of the copper–cobalt areas of south-central Africa (Wild 1978; Brooks and Malaisse 1985), and the lead–zinc areas of Britain (Baker and Proctor 1990; Department of the Environment 1994) and western Europe (Simon 1978). Many old mining sites have become of high conservation value both because of endemic plant species or populations and as refugia for both rare plant and animal species whose natural habitats have been considerably reduced (Johnson 1978; Johnson et al. 1978; Box 1992; Spalding and Haes 1995).

Recovery

At its simplest, ecological restoration may equate with primary succession or the recovery (Cairns 1980, 1991) of mined land when it is largely left to natural processes after disturbance. Studies of abandoned mining areas have enabled the investigation of the issues concerning the development of ecosystems, e.g., the role of nitrogen accumulation (Walker 1993; Bradshaw 1999) and so have helped inform the practice of modern ecosystem restoration. The presence of populations of plant species in a particular site will depend on the ability of propagules to be transported to the site and to germinate and of the young plants to survive and reproduce. The time scales involved are often long and the initial colonization phase, in particular, can show a considerable lag depending on the harshness of the site and substrate conditions (Ash et al. 1994). For example, in the study of a chronosequence on calcareous lead zinc mines in SW Wisconsin there was little colonization for 20 years and about 40% cover was obtained after 50 years (Kimmerer 1984). The vegetation that developed was a mixture of bare areas and well-vegetated patches with grasses, sedges, and perennial forbs. This is typical of many mine sites that have been abandoned to recover naturally. On metalliferous sites plant colonization success depends upon avoidance of the high soil metal areas, where substrate heterogeneity exists or metal tolerance is established either through natural selection of tolerant populations, or because it is constitutional to a particular species, e.g., a metallophyte species (Baker and Proctor 1990). Other soil factors influencing natural plant colonization of metalliferous sites include pH, nutrient levels especially calcium and phosphorus, and the amount of organic matter and the presence of mycorrhizal fungi (Alvarez et al. 1974; Williams et al. 1977; Johnson et al. 1977; Simon 1978; Smith and Bradshaw 1979; Morrey et al. 1988; Baker and Proctor 1990; Hetrick et al. 1994).

An example of recovery is demonstrated by attitudes to land restoration at Mount Lyell, a substantive copper mine complex on the King River catchment in western Tasmania. Slow natural succession has been promoted as a better option for the "lunar landscape", partly on the grounds that it leads to an ecosystem more in harmony with the surrounding area but also because the bare landscape is a stark contrast to the adjacent land and is actually a tourist attraction (McBride 1997). However, completely non-interventive or "passive" restoration is conceptually difficult and usually politically unacceptable in an era when "closure planning", an active process, is becoming an everyday mining term. Moreover, recovery on bare mine waste and tailings will usually involve a colonization sequence and plagioclimax communities that are, in fact, very different from the original or surrounding vegetation, again because of the physical and chemical properties of the substrates being so different from those of the original soils.

Ameliorative and adaptive approaches

Despite the wide ranging constraints of mining sites and substrates there have been some important success stories in the direct restoration of metal and other mineral mining wastes. However, where the original topsoil has been lost, faithful restoration of original ecosystems is rare. An overview of the range of approaches is given in Table 6 and is based upon the degree of toxicity, salinity, and acidity of the waste material or site. These approaches combine, on a site-specific basis, the ameliorative approach (improving the physical and chemical nature of the site) with the adaptive approach (the careful selection of species, cultivars, or ecotypes) in a way which seeks to achieve the ecological restoration goal of establishing ecosystem structure and function (Jeffrey et al. 1975; Johnson et al. 1994).

It is perhaps not surprising that the most successful restoration case histories where there has been no reliance on soil coverings have come in the industrial minerals sector of the industry where the wastes have fewer chemical toxicity problems. Pioneer work on ecosystem restoration on exhausted quarries was conducted as long ago as 1970 on the coral limestones and Jurassic shales on the Kenya coast (Sassoon 1996). Ecologically sustainable restoration has revolved around direct establishment of 26 tree species and grasses, centred around tall Casuarina (*Casuarina equisetifolia*), now indigenous to the East African coast. This salt-tolerant nitrogen-fixing species acted as the catalyst for ecosystem development, perhaps rather surprisingly by facilitating the expansion of a millipede (*Epilobus pulchripes*) population. Millipedes feed voraciously on dry needles of *Casuarina*, which when digested initiate humus formation and secondary colonization by plants and animals. Over 250 plant species now inhabit the quarry habitat, an achievement that was largely made without fertilizers or ameliorant substrates.

In more temperate settings, one of the best examples of effective ecosystem restoration without topsoil is of the western European Atlantic heaths (dominated by *Calluna vulgaris* and *Erica* spp.) on sand, mica wastes, overburden, and rock waste produced by the mining of china clay in south-west England. Many European countries have lost up to 70 % of their heathland area this century so the creation and management of replacement dry heaths in the semi-lowlands of Britain is an important restoration strategy for old china clay workings. Successful techniques have been established, although their long-term sustainability has still to be confirmed (Department of the Environment 1993). In heathland restoration,

Table 6. Promotion of ecosystem restoration for metalliferous wastes based upon relative toxicity, acidity, and salinity (adapted from Johnson et al. 1994).

Waste characteristics	Restoration technique	Problems
Low metal content and toxicity. No major acidity or alkalinity problems	Amelioration by applying lime, fertilizer, and organic matter as required. Seed with commercial or wild seed. Use turf or thin layer of native soil as an inoculum if available. Plant indigenous trees.	Probable commitment to long-term maintenance and irrigation may be necessary in arid climates. Metals and other trace elements may reach toxic levels in vegetation and grazing should be monitored and controlled.
Medium metal content and toxicity, or medium salinity or acidity	Ameliorate and adaptive approach by applying lime, fertilizer, and organic matter as required. Sow metal-tolerant commercial varieties or seed of wild metallophytes and/or acid or salinity tolerant seed.	Few species/ varieties available. Cost of collecting wild seed may be very high. Possible commitment to regular fertilizer additions. Swards have low trampling resistance. Grazing of animals not advisable and wildlife may be affected by high levels of metals in food.
High metal content and toxicity	Treat surface with 10–50 cm of innocuous material such as overburden. Lime, fertilize, and seed with indigenous species.	Regression may occur through upward movement of soluble toxins.
Very high metal content, extreme acidity, toxicity, or salinity	Cover surface with 30–100 cm barrier layer such as unmineralised rock and cover with suitable rooting medium.	High cost and may get regression through drought or root penetration through barrier layer.

because of the absence of commercial seed, the key steps are vacuuming or cutting of top growth from a natural area of heather moorland such that this donor area survives to regrow but yields propagules for restoration purposes elsewhere (Pywell et al. 1996). This collection of seed capsules and ground surface litter is then redeployed after careful preparation of the receptor area, namely the mine waste. This involves rotovating the litter into the waste surface sowing of seed capsules and nurse grasses and the judicious use of fertilizers (Box et al. 1998). It is possible to spread the seed and litter over a larger area than that from which it was stripped, perhaps up to 10 times. However, this still puts a high demand on natural donor areas.

The restoration of the Sudbury "barrens", some 17 000 ha around the three main metal smelters, which lacked vegetation in 1970, provides an example of minimal amelioration of severely smelter-damaged land for facilitating the long-term restoration of natural ecosystems. The barrens were created largely through the destruction of the original vegetation by high atmospheric SO_2 prior to 1972 and re-colonization by metal-intolerant plants was inhibited by soil acidification (pH 3–4.5), high levels of available soil copper and nickel from aerial deposition, and high aluminum released from clay minerals (Winterhalder 1995*a*). However, it was found that a thin surface application by hand of ground dolomitic limestone led to the establishment of woody plants such as white birch, trembling aspen, and willows from local seed sources. Simple liming can be regarded as a "trigger factor" because it allowed the establishment of some tree species because the stony mantle that covered the eroded soils of the slopes trapped the limestone particles and seeds. The establishing trees then through, inter alia, the production of leaf litter and transporting of bases to the soil surface, improved the prospects for further plant colonization, especially by ameliorating the microclimate and reducing the effects of drought and frost heaving (Winterhalder 1995 *b*).

Ecological restoration without topsoil usually depends on careful selection of adapted species and includes their suitability for ground stabilization, the value of the species as wildlife habitat (and as forage for wild animals), and then also for achieving aesthetic goals. Indigenous species available as propagules do not always satisfy these criteria, in which case native but not locally indigenous species can be sown as a supplement, usually in a way that provides a rapid solution to short-term problems such as erosion, but one which enables colonization by local volunteer species and thus facilitates succession to eventually restore the native ecosystem. This reasonable compromise has been the approach used for copper tailings berms and surfaces in the arid and testing conditions of southern Arizona (Bengson 1995).

The use of metal tolerant ecotypes, in particular of the temperate grasses *Agrostis capillaris* and *Festuca rubra*, is a proven reclamation technology of 20 years standing for lead, zinc, and copper mine tailings (Smith and Bradshaw 1979; Morrey 1995; Tordoff et al. 2000). These ecotypes have metal tolerance as a genetically heritable character, and some have been bred on to cultivar status (e.g., *F.rubra* cv. "Merlin"), they are known to partially restrict metals from root absorption (e.g., Bergholm and Steen 1989) and to be able to exclude that fraction that is absorbed from active metabolic sites. However, this approach carries with it certain land use restrictions, for example grazing and intensive recreational use are usually precluded (Table 6). Moreover, because direct seeding is used, the initial economic attractions are somewhat tempered by the higher maintenance costs of regular applications of fertilizer and, sometimes, the need for "patch seeding" to deal with areas where initial failure occurs. It is also important to recognize that the metal tolerance mechanisms in plant species are specific. With some exceptions, e.g., *Deschampsia cespitosa* (Cox and Hutchinson 1980), cross tolerance between metals is not known, so an ecotype is unlikely to thrive on mine waste containing toxic quantities of metals other than those that it is adapted to based upon its site of origin. Despite these limitations, direct seeding of tolerant cultivars is still a promising area of further development, with candidate species from tropical areas including *Chloris gayana*, *Eragrostis curvula*, and *Cynodon dactylon*.

The use of inherent tolerance to particular chemical constraints is not restricted entirely to trace metals. A recent successful direct revegetation programme on lead mine tailings in the west of Ireland was based upon the use of salt-tolerant grasses (e.g., *Agrostis stolonifera* cv. "Seaside"). In this case chemical weathering within the tailings, through the interaction of sulphuric acid, the breakdown product of iron pyrites, with magnesian limestone ($MgCO_3 \bullet CaCO_3$), made the high osmotic pressures caused by magnesium sulphate the main constraint to growth (Brady 1993).

More recently, a new technology has been promoted whereby the tolerance to metals of some plants is used in a different way. Some species, particularly members of the Brassicaceae (e.g., *Thlaspi caerulescens* and *Cardaminopsis halleri*) and Caryophyllaceae (e.g., *Minuartia verna*), are described as "hyperaccumulators" in recognition of their ability to accumulate elements that are usually present in trace concentrations in plants. In extreme cases accumulation may be to an extent that the metal is no longer a trace element but a major inorganic constituent, with concentrations exceeding 10 000 mg kg^{-1} (1% w/w) or 10–20% on an ash weight basis (Reeves et al. 1995).

For the highly toxic metal mine wastes, it has been suggested that such species could be manipulated for phytomining or phytoextraction of some elements as a commercial enterprise or be used to farm metals by making hay from biomass. The hay product would then be subjected to combustion processes to recover the metals as an ash concentrate or to reduce the volume of material for landfill disposal (Chaney et al. 1995). Hyperaccumulator species could thus be used to bioremediate soils and at the same time both stabilize and reclaim land for other purposes (Salt et al. 1998). Innovative though this may be, this “clean-up” technique is unproven on a commercial scale. Trials are underway in temperate zones (Baker et al. 1995*a*) and under the more exacting climatic conditions of the Sierra Nevada in the U.S.A. (Nicks and Chambers 1995) and Chile (Ginocchio 1998). However, before this approach, which may combine restoration with land remediation, can be considered viable, major problems of species rarity, low productivity, gene manipulation into more productive species, suitable harvesting methods, and final disposal of the biomass as waste must be addressed.

Ecotoxicology

In addition to the chemical constraints upon the establishment of plants, the levels of potentially toxic residual chemicals in some mine wastes, which may include radiobiological hazards (Ritcey 1989), can be toxic to animals and soil microorganisms. The toxic elements most commonly found are arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), mercury (Hg), nickel (Ni), selenium (Se), silver (Ag), and zinc (Zn) (Pierzynski et al. 1994). This important ecotoxicological aspect of ecological restoration is illustrated by extensive work on one site used for the disposal of metalliferous fluorspar tailings at Cavendish Mill in the Peak District of Derbyshire, England. Elevated levels of metals (Pb, Zn, Cd) and fluorides (F) in the tailings did not prevent the successful direct establishment of grassland (using a hydroseeding technique to establish a commercial grass–legume mix) and trees on the tailings dam in 1974 (Johnson et al. 1976). A survey of the dam in 1977 listed 69 higher plant species, 6 breeding birds, 30 species of visiting bird, 9 mammals, and a wide range of invertebrate species with most of these species being volunteers from surrounding areas.

In 1979 it was shown that wild small mammal species were found on the dam surface in numbers that were similar to areas of grassland–scrub habitat nearby (Johnson 1980; Johnson and Putwain 1981). At various times since the initial restoration work, major assessments of the distribution of Pb, Zn, Cd, and F have been undertaken and included surface soil, live standing plant crop, plant leaf litter, invertebrates, and small mammals (Andrews et al. 1984; Andrews et al. 1989*a*; Andrews et al. 1989 *b*; Andrews et al. 1989*c*; Cooke et al. 1990; and Boulton et al. 1994). Data in Tables 7 and 8 are derived from these publications.

A summary of some of these data for Cd and F is given in Table 7. The vegetation and plant litter have high levels of both elements compared to an uncontaminated reference site. This is reflected in the earthworm (*Lumbricus*) data shown in Table 7 where, in general, the levels for Cd and F are 10 to 25 times normal. The concentrations in the field vole (*Microtus agrestis*) and the shrew (*Sorex araneus*) also show significant increases in F (in bone) and in Cd (kidney in the field vole and kidney and liver in the shrew). The remarkable affinity of the shrew for Cd in its food items is clearly shown, with extremely high concentrations in the kidney and the liver. Cadmium-induced lesions in kidney and liver tissue in shrews occurred, and field voles showed minor symptoms of dental fluorosis (excessive ware of the incisor teeth caused by F that can prevent feeding). Thus, through the movement of toxins from soil to plant, and in food webs, accumulation can occur in wildlife colonizing the newly restored and developing ecosystem. This remobilization of residual toxins can have ecotoxicological impacts on individual species and also possible consequences for ecosystem function; for example, the reduction in decomposition rates if the soil biota are affected.

In practical terms, if judgements are made that the continual toxic stress is of a magnitude to prevent the establishment of the desired species (introduced or volunteer), then this should “feedback” into the original restoration planning. This may mean a revision of restoration goals or modification of the restoration strategy adopted. For example, should cover layers or a barrier layer be used to isolate the waste from the rooting zone and thus reduce plant uptake and translocation of toxins? However, successful direct establishment of vegetation, of itself, can reduce the hazards of toxicity: for example, through the organic enrichment of the surface horizons of the wastes. An example of this, from the Cavendish Mill case study, was the level of fluoride in standing live grass on the fluorspar dam. This declined considerably in the years after restoration (Table 8). The most likely explanation of this is the

Table 7. Concentrations of cadmium and fluoride (mg kg^{-1}) in soil, grass, litter, earthworms (*Lumbricus*), and kidney, liver, and bone tissue of *Microtus agrestis* (Ma; field vole) and *Sorex araneus* (Sa; common shrew). At least 8 samples were collected from both the restored tailings dam (TD), Cavendish Mill, Derbyshire, and an uncontaminated Reference (R) Site in 1980.

	Cadmium		Fluoride	
	TD	R	TD	R
Soil	23	0.5	98876	131
Grass	5	1.5	579	15
Litter	15.6	1.9	5610	75
Lumbricus	75	7.1	3736	14
Ma Kidney	5.2	1.8	15	8.4
Liver	1.9	1.1	11	6.5
Bone	0.7	0.4	1106	91
Sa Kidney	158	4.1	15	12
Liver	236	2.9	12	8.5
Bone	1.9	1.0	1553	524

Note: see text for sources of data.

Table 8. Fluoride concentration (mg kg^{-1}) mean (standard error) in vegetation from the restored tailings dam at Cavendish Mill, Derbyshire, at various sampling dates.

Sampling date	Composite standing (green) vegetation
May 1976	4260 (810)
April 1980	1043 (412)
July 1983	290 (40)
April 1985	187 (34)
July 1990	74 (7)

Note: see text for sources of data.

reduced availability of F in the organic-enriched surface layers of the developing soil where most of the plant roots proliferated (Table 9). However, it must be noted that further decomposition of metal- or fluoride-enriched organic matter could lead to the release of these elements in plant available form (Geiger et al. 1993).

Progressive improvements in mineral processing technologies have, in some respects, worked in favour of metalliferous mine waste restoration. Whilst the metal levels in waste material always reflect “cut-off” grades, metal prices, and other non-technical economic factors, there is a trend towards lower levels of metals in mine tailings. In some cases, for example at Outokumpu-Tara’s zinc–lead mine in Ireland, residual metal levels are now so low that when disposed of within limestone-dominated tailings, not only is direct seeding of agricultural grasses possible, but so is a low output agricultural enterprise on the tailings surface without the use of topsoil and without any significant toxic hazard to the grazing livestock (Brady 1993; Crilly et al. 1998). Agricultural uses for mine tailings remain very rare, and there is little pressure to develop such objectives except insofar as they may demonstrate flexibility in land

Table 9. Soil profile variables of the restored fluorspar tailings dam at Cavendish Mill, Derbyshire sampled in 1983.

Depth (cm)	% loss on ignition	pH	Available fluoride (mg kg^{-1})
0–2	43	7.1	35
2–4	6	7.8	42
8–10	1	8.2	89
20	2	7.8	118
40	1	8.2	196

Note: available fluoride measured using a resin extraction method (Larsen and Widdowson 1971), data from Carrick (1984).

Table 10. Considerations and practice in topsoil conservation.

Soil characteristics	1. Depth and horizonation in relation to the need to handle topsoil, subsoil, and overburden separately. 2. Texture class, plastic limit, structural status, and bulk density in relation to loss of porosity and other physical and biological changes during soil moving.
Soil movement	Lifting, transport, storage, and re-instatement to avoid compaction, killing of soil animals, and the release of dormancy of buried seed bank. Avoid spreading over dissimilar underlying material to prevent hydraulic discontinuity and slope instability.
Changes during storage	1. Physical: loss of organic matter and the alteration of binding of soil particles, loss of aggregate stability, soil compaction. 2. Chemical: Centre of storage mound anaerobic conditions develop, ammonium nitrogen increases, redox potential, and pH changes with an increase in organic acids, availability of metals such as Mn, Cu, and Zn can increase. Metal sulphides can increase. 3. Biological: initial increase in bacterial populations in response to dead fungal biomass, soil animals, and plant roots. Sharp declines in soil invertebrates especially earthworms.

use planning and in some respects, lower environmental risk. An example would be the growing of rice, vegetables, and sugar cane on gold tailings in the Philippines, although no details of residual metals in these crops have been published (Walker 1992).

Restoration in practice: where topsoil has been retained

Topsoil as the strategic restoration resource

The modern context of restoration as part of the total mining process involves carefully planned decommissioning rather than the common past practice of simple abandonment. Topsoil, other surface materials, and overburden from mine sites are today viewed as strategic restoration resources that should be conserved if at all possible. Thus their removal, storage, and replacement have received much technical research in recent times. The main reason for this is to protect their physical and chemical properties and biological processes (Table 10; see also Harris et al. 1996).

The following discussion of restoration practice where topsoil has been retained focuses less on revegetation and more on the ecological, spatial, and temporal factors affecting species colonization and establishment. The criteria for monitoring and assessing success, particularly in the longer term, for the restoration of natural indigenous ecosystems on mined land are also specifically considered. This discussion is based around two case studies of current mining operations that have restoration histories going back to the 1970s: dredge mining of coastal sand for heavy minerals in South Africa, and area strip-mining for bauxite in Australia. As noted in Table 3 the land disturbance of the primary extraction phases of both strip and dredge mining can be progressively restored with the retention and reinstatement of topsoil. Reasonable restoration success can be expected if based on ecological considerations. Restoration approaches differed between these two examples, but both sought to restore the pre-mining indigenous forest and both represent interesting case studies because extensive post-restoration research and monitoring has been undertaken and this has been published in the peer-reviewed scientific literature.

Dredge mining in South Africa

On the northeast coast of South Africa in Zululand near Richards Bay, dredge mining for heavy minerals such as rutile, ilmenite, and zircon in coastal dunes has taken place since 1977 (Camp 1990). Ecosystem restoration is a fundamental part of the mining operation. Mining entails the removal of the dune forest in a prescribed mining path through the dunes. Topsoil is then stored and an artificial pond created in the ore-bearing dune and the sand is mined using a floating dredger that pumps the sand in slurry to a gravity separator. Here the heavy minerals are separated from the sand. The mined sand is then pumped back as tailings behind the mining path, dewatered, and stacked to resemble the topography prior to mining. Over 90% of the sand bulk is returned after mineral extraction.

The restoration process has relied heavily on the initiating of succession and then leaving the system to develop naturally. The success of the approach was thought to be likely as, over ecological and evolutionary time scales, the dunes have been built, vegetated, and destroyed many times along the south east African coast (Mentis and Ellery 1998). Aerial photographs taken in 1937 showed that highly degraded coastal dune vegetation had, because of subsequent human depopulation, seemingly recovered through natural succession. This indicated that a minimal interventionist approach could be successful. To promote the establishment of indigenous dune forest, the topsoil is re-spread on the non-toxic tailings to a depth of about 10 cm. Then, artificial windbreaks (shade cloth fences) are erected and the area is sown with a mixture of seeds consisting mainly of fast-germinating species (e.g., *Helianthus annuus*, Sorghum spp., *Pennisetum americanum*, *Crotalaria juncea*). This mix acts as a nurse crop that protects the slower germinating indigenous species present in the seed bank from the high surface temperatures and winds. After this nurse crop has died off the vegetation is dominated by *Acacia karoo*, from the reinstated soil seed bank, which is regarded as the major pioneer species in the succession in this area (Camp and Weisser 1991).

Over 400 ha have been reclaimed in this way since 1978, providing a chronosequence of restored mined dunes. The chronosequence approach requires the assumption that spatially separate sites investigated at one time can be equated to the temporal response at any one site. While care must be taken with this underlying assumption, chronosequences can provide valid insights into the basic patterns of successional change (Twigg et al. 1989; Wali 1999; Foster and Tilman 2000). A summary of vegetational differences and numbers of some animal taxa for the restored dunes at Richards Bay are given in Tables 11 and 12. A reference or benchmark site of a mature dune forest (assumed to be climax) without mining disturbance is given for comparison. The mined dunes appeared to follow a succession that appears to be leading to dune forest typical of the area.

van Aarde et al. (1996) used Bray and Curtis (1957) similarity coefficients for comparing presence and abundance of different taxa in the restored stands of different ages and the reference stand. The analysis of patterns of similarity for nearly all of the studied taxa (woody plants, millipedes, beetles, birds, and rodents) showed an increase in similarity to the reference with increasing age since restoration (Table 13). However, this approach raises the questions, similar to what and how similar is similar enough, given that complete similarity will never be attainable? In comparing restored areas with one or two “pristine”

Table 11. Ecosystem restoration after dune mining in Zululand, South Africa: a comparison of three stands of different ages with a mature forest reference stand for vegetation (adapted from van Aarde et al. 1996).

5 to < 8 years	8 to < 11	11 to 16 years	Reference stand
Acacia karoo (sweet thorn) scrub 1.5–3.0 m high, sparse middle layer of *Vepris lanceolata* (white ironwood) and *Brachylaena discolor* (coast silver oak). Herb layer mainly grasses *Panicum maxima* and *Digitaria diversinerva.*	*Acacia karoo* woodland 3–8m high, dense canopy dominated by *Acacia karoo* with some *Brachylaena discolor* and *Rhus nebulosa* (sand currant). Herb layer mainly *Digitaria diversinerva*.	*Acacia karoo* 9–12 m high with some secondary dune forest species such as: *Trichilia emetica* (Natal mahogany), *Trema orientalis* (pigeon wood), *Mimusops caffra* (coastal red milkwood), and *Celtis africana* (white stinkwood). Herb layer mainly *Digitaria diversinerva.*	Secondary dune forest with canopy 12–15m. or higher with main species: *Celtis africana*, *Mimusops caffra*, *Allophylus natalensis* (dune false currant), *Teclea gerrardii* (Zulu cherry-orange), and *Ochna natalitia* (Micky Mouse bush). Herb and Shrub layer dominated by a shrub, *Isoglossa woodii* and the fern *Microsorium scolopendrium*.

Table 12. Ecosystem restoration after dune mining in Zululand, South Africa: a comparison of three stands of different ages with a mature forest reference stand for the number of species of different animal taxa (adapted from van Aarde et al. 1996).

	5 to < 8 years	8 to < 11 years	11 to 16 years	Ref. stand
Millipedes	3	4	6	11
Beetles caught by				
i) sweep net	54	–	80	116
ii) flight inter-cept	74	–	107	188
Rodents	1	3	3	4

reference areas, sampling efficiency is low (insufficient replicates). However, even where extensive pre-disturbance sampling has been carried out there will be “noise” (i.e., intrinsic variation among sites) due to habitat heterogeneity and stochastic elements influencing ecosystem history (Westman 1991). Perhaps 80% similarity in plant species can be taken as similar enough, but then what is a satisfactory timescale to reach this level of resemblance?

Time-associated changes were investigated at Richards Bay using different approaches in a separate study of the vegetation of these restored dunes by Mentis and Ellery (1994,1998). They sampled 0.1 ha plots from the chronosequence of restored (mined) plots together with plots from unmined land. These unmined plots were all in the same coastal dune area and would be expected to develop into dune forest, but most had been disturbed in some way other than by mining, and the time since the last

Table 13. Stand-specific similarity indices[1] for different taxon groups recorded during the summer months for different aged restored dune areas with an unmined mature coastal dune forest reference stand, modified from van Aarde et al. 1996.

	Age (years) of restored stand		
	5 to < 8	8 to < 11	11 to < 16
Trees <2m	26	28	39
Millepedes	36	40	59
Beetles (pitfalls)	28	–	32
Birds	61	82	90
Rodents	34	63	87

[1]Bray and Curtis (1957).

major disturbance was known from local records. The unmined plots sampled included three that were thought to represent climax dune forest. These authors sought to investigate whether succession was occurring and if it differed on mined land when compared to unmined disturbed land. They used a series of approaches (Mentis and Ellery 1994): correspondence analysis, euclidean distances of each sample plot from an average mature reference (climax) plot, and multiple regression. A regression of euclidean distance against time was highly significant (Fig. 3), and there were no significant differences between mined and unmined areas in terms of the observed convergence (trend to lower values of euclidean distance and thus greater similarity) of their vegetative state on the local climax dune forest. This suggests that the successional trajectory on the mined land would be likely to lead to the achievement of the restoration objectives. Further, the data for plant species richness shows a saturation curve with an initial rise and levelling off with time (Fig. 4). This research indicates that the self-restorative capacity of these dune systems can overcome both mining and non-mining disturbance. Following further sampling of unmined and mined plots, Mentis and Ellery (1998) were able to predict from the time of initial restoration that the upper asymptote of the species richness curve would be reached in 28–40 years and that convergence on forest climax plant species composition would occur in 54–70 years.

These studies showed that species richness increased with time as the even-aged pioneer tree species (predominantly *Acacia karoo)* were replaced by tropical and sub-tropical broad-leaved shrubs and trees (Table 11; Mentis and Ellery 1994; van Aarde et al. 1996). This was facilitated, functionally, by the nitrogen-fixing capability of *Acacia karoo,* which promotes an increase of soil nitrogen and soil development (Lubke et al. 1993). The changes in vegetation structure were accompanied by increases in the number of animal taxa (Table 12). Through the death of older trees and the creation of small-scale gaps in the dense *Acacia karoo* stands, the colonization by secondary tree-canopy species was facilitated. However, it is very likely that many of these secondary successional tree species were not recruited from the seed banks of the original replaced topsoil, but dispersed into the older restored areas by animals — particularly fruit-eating birds and vervet monkeys (Foord et al. 1994). Recalcitrant seeds have a high water content and short life span after shedding. The importance of animal seed dispersal in the tropics and sub-tropics because, inter alia, there is a high level of recalcitrant-seeded tree species, needs to recognized. They will not survive storage in soil although it is possible that they could germinate rapidly and become a “seedling bank” waiting for canopy gaps to appear (Pammenter and Berjak 2000).

The efficacy of animal seed dispersal to restored sites will be limited by the absence of the range of animal seed dispersers; the highly specialized dispersal some seed may have, e.g., the very large-seeded species; the degree of isolation from the seed source including the nature of any intervening habitat; and the attraction of the seed dispersers to the restored areas (Wunderle 1997). This latter point introduces a paradox and may mean the planting of “attractant plant species” such as fleshy-fruited native species

Fig. 3. A plot of Euclidean distance (from the average climax state) of the known-aged unmined and mined sample plots against years since disturbance. The line is the least squares regression (redrawn from Mentis and Ellery (1994)).

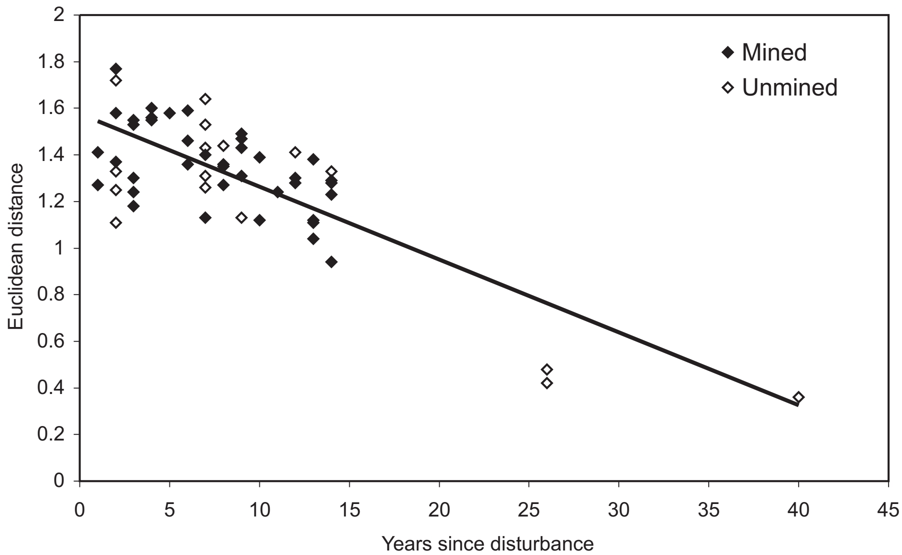
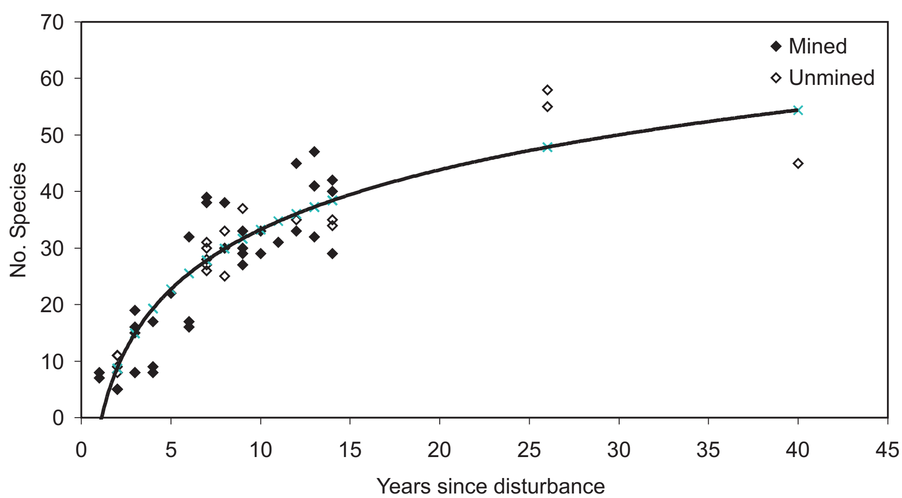


Fig. 4. Species richness of known-aged unmined and mined sample plots against years since disturbance. Line is the log normal regression y = 15.25 Ln(x)- 1.85 (redrawn from data in Mentis and Ellery (1994)).



from different stages of the normal forest succession (Tucker and Murphy 1997; Wunderle 1997). For the restoration ecologist the presence of nearby pristine habitat and vegetation, even as small strips or as an "archipelago" of small islands, could be a key aspect of restoration success. The size of these areas of undisturbed natural ecosystems in relation to their capacity to be refugia of viable animal populations, and the spatial relationship of such patches to the mined area, are often neglected facets of restoration planning that must be addressed before mining starts. Similar arguments and solutions could be made for the restoration of other important animal species to ecosystem development and function, such as pollinators (Neal 1998).

Bauxite mining in western Australia

Bauxite mining in the northern Jarrah forest in Western Australia currently requires the restoration of 450 ha of forest per year (Baker et al. 1995 *b*) and, as it is currently practiced, provides some interesting contrasts in the approach to restoration of the South African dune forests. In particular, a much more clearly defined set of success criteria has been combined with a greater interventionist approach to reach the desired restoration objectives within specific time limits. This has required a considerable knowledge of the ecosystem structure and function before mining commenced. However, this knowledge has been acquired over 30 years as a result of " trial and error, planned research, lucky discovery, standardisation, and fine tuning" (Baker et al. 1995 *b*).

The vegetation is dominated by the relatively slow-growing *Eucalyptus marginata* (jarrah) that attains a height of 30–40 m. Other eucalypt tree species present include *E. megacarpa* (bullich), *E. patens* (blackbutt), and *E. calophylla* (marri) and an understory of many species including the Cycad *Macrozamia riedlei*. The ore is mined from numerous discrete shallow deposits of 3–6 m deep in small discontinuous pods of 1–100 ha in area. The soils are typically nutrient-poor laterite gravels, 0.5 m deep, that overlie a pallid zone of kaolinitic clay.

The basic mineral extraction is technically simple. The forest is cleared with some timber kept to provide animal habitat on the restored area. The top 5–15 cm of topsoil is stripped separately to maintain the seed bank (350–1500 germinable seeds per metre square). If possible the topsoil should not be stockpiled (but used directly for an area awaiting restoration), as this can reduce the number of germinable seeds, cause the loss of nutrients, and reduce aerobic microbial activity (Tracey and Glossop 1980). The Jarrah forest is a nutrient-poor ecosystem and most of the native species have vesicular-arbuscular mycorrhizae or ectomycorrhizae, and these are rapidly lost in the stockpiling of topsoil. This leads to low levels of re-infection in the early years after restoration (Jasper et al. 1987; Baker et al. 1995*b*). Overburden above the caprock (about 40 cm depth) is then removed and stockpiled. The caprock is blasted and the ore removed down to the pallid zone of the original laterite soil profile.

Ore extraction leaves shallow pits with 2–5 m high vertical faces and compacted clay floors. The entire pit surface is then deep-ripped along the contours, breaking up the clay floor to allow root penetration and prevent waterlogging. The pit walls are battered down and reshaped and the overburden and topsoil replaced in the correct sequence. Further surface-ripping then occurs to improve drainage as waterlogging is very undesirable because it encourages the soil-borne fungus *Phytophthora cinnamomi*, which causes die-back in jarrah and some understory species. In fact, much emphasis is placed on the prevention of spread of dieback, such that infected soils are stripped and stored separately and vehicles are cleaned if they move from an infected to an uninfected area (Baker et al. 1995 *b*).

As in the South African dune forest restoration, the topsoil seed bank is the major source of seed for the developing ecosystem, with an estimated 75% of the native species becoming established from the soil seed bank. However, in the restoration of the Jarrah forest this is now considerably augmented by a seed mix that is sown by hand, together with fertilizer, across the entire restored pit surface. This mix includes more than 60 different native species, including the important understory legumes (especially *Acacia* spp.), eucalypt species, and the Cycad species. Sowing immediately after ripping of the topsoil has increased successful plant establishment. This allows seed to become lodged in microsites before surface crusts develop. The crusts prevent good seed–soil water contact. Koch and Ward (1994) carried out a comparative study of recently (after 9 months) restored sites using both topsoil transfer and direct seeding. These restored sites were compared with either the floristics of the site prior to mining or with nearby unmined areas. In terms of plant reestablishment three main groups of species were recognized: 129 plant species were from the replaced soil seed bank and early volunteer species being dispersed from surrounding unmined land; 39 species had established from the seed mix; and a further group of 48 species that did not occur in the restored areas were mainly from the Orchidaceae, Liliaceae, Epacridaceae, and Restionaceae (Koch and Ward 1994).

In a fire-regulated system such as the Jarrah forest, the main restoration objective is to establish as many of the most common plant species as soon as possible. This includes having a minimum success criteria for plant establishment of 2000 eucalypt seedlings per hectare and two legume understory species per square metre after 9 months. These restoration objectives have required specific research into the reproductive biology of many plant species, for example, vegetative propagation of species with recalcitrant seeds or species that produce very little seed but which usually respond to disturbance by re-sprouting (Koch and Ward 1994; Baker et al. 1995*b*). Monitoring and evaluation of the recently restored areas have shown satisfactory ecosystem development in relatively short time periods compared to the unmined areas of Jarrah forest. The development of understory shrub species through hand seeding and vegetative propagation appears to be crucial, not only to establishing plant species richness, structural diversity, and nutrient build-up but also to the colonization by animals. The likely time scales for the achievement of various success criteria are plant species richness (<2 years), litter biomass (4–10 years), nitrogen capital (15 years), 80% of animal groups including birds, mammals, reptiles, frogs, ants, and other invertebrates (4–6 years) (Nichols and Bamford 1985; Nichols et al. 1989; Baker et al. 1995*b*; Majer and Nichols 1998; Armstrong and Nichols 2000).

Conclusions and the future prospects: integrating theory and practice

Ecological theory, uncertainty, and adaptive management

Ecological knowledge and theory must support the art and practice of land restoration. Because of the problems associated with the nature of ecological theory, particularly the lack of general laws with universal applicability at the community level of organization, it is unrealistic to expect a high order of predictability. However, ecological knowledge does have high heuristic power and applicability to environmental problem-solving in general and site specific ecological restoration goals in particular. Thus, ecological knowledge can be used to considerable effect in the context of its application within an inductive, case-specific methodology and must not be rejected because it does not have a hypothetico-deductive paradigm generating universal laws applicable to all ecological restoration situations (Shrader-Frechette and McCoy 1994).

Thus, the nature of ecological theory and its general lack of predictability does affect both the interaction of ecology with the other disciplines involved in overall restoration planning and the interpretation of restoration plans. It is important that industry, the regulators, and the general public understand the fundamental nature of the science of restoration ecology and what it can and cannot achieve. It is important that restoration not be seen as a new technological panacea that can cure all man's mining disturbance and degradation. Although the restoration genie is out of the bottle, the task is to see that restoration is used properly and not used, of itself, to legitimize new degradation (Berger 1990). Rather, ecological restoration must go hand in hand with the conservation of pristine ecological resources and be part of holistic environmental management at the landscape scale, for example, where the restored ecosystems offer alternatives to the overuse or development of pristine areas.

Equally, it is vitally important that ecological restoration not be perceived as having no value because it cannot guarantee absolute replication of pre-mined ecosystems. The nature of all science, including the science of restoration ecology, is probabilistic, and statements are not certainties but expert guesses. The language and processes of risk analysis can be informative in the context of land restoration. For each case (i.e., specific site restoration) the challenge is to distinguish the risks (where the odds are known), uncertainties (where system parameters are known but not the probabilities), ignorance (where we do not know what is not known), and indeterminacies (where outcomes are outside the parameters of science) in a way that can facilitate secure judgements within the restoration process (Stonehouse and Mumford 1994; Kruger et al. 1997). Because there are indeterminacies within any restoration caused by the social, political, and economic context within which it occurs, science can make only judgements concerning risk, uncertainty, and ignorance. Table 14 puts these ideas in the context of the ecological restoration of a mine site.

Table 14. Definitions of risk, uncertainty, ignorance, and indeterminacies in the context of an ecological restoration.

Risk — probability that some kind of vegetation and ecosystem (e.g., forest and grassland) will be established using basic ecological knowledge of predictors (basic driving variables such as climate and soil quality or development) that can be quantified and modelled. Predictability at this level is possible.
Uncertainty — the exact type of ecosystem is uncertain given that chance events determine successional trajectory; thus important processes and variables of system may be known but their quantification in space and time is not, for example, the source and colonization of secondary successional species or the rates of nutrient build up and retention. Predictability at this level is not possible.
Ignorance — by definition escapes recognition so examples are difficult to provide but would include the fact that "complete" knowledge (e.g., the biology and even the presence of every species) of the pre-mining ecosystem cannot be known even though research can be carried out and new knowledge is discoverable.
Indeterminacies — these are uncertainties outside the realm of scientific assessment and may include contingencies, for example, caused by local political interference, social pressures during the restoration process, for example, illegal cattle grazing, poor contractor competence leading to major mistakes in implementing the restoration plan.

In view of the above, efforts to reduce ignorance and uncertainty (through ecological research and experimentation) are necessary and admirable but must be accompanied by design and management for uncertainty. In ecosystem restoration this means building in ecosystem variability to provide and maintain resilience (Levy et al. 2000). It also focuses on the need for adaptive management to take advantage of surprise elements and convert problems into opportunities (Holling 1978). Further, the essential role of monitoring and management is emphasized as the uncertainties in restoration planning can never be overcome. The concept of adaptive management (Holling 1978) and the notion that a restored site be regarded as a long term experiment (Walters and Holling 1990) gives, a sensible perspective for the restoration paradigm proposed here and by others (Pastorok et al. 1997; Thom 2000). A well-designed restoration plan should incorporate alternative pathways and endpoints and so inform stakeholders as to potential problems and possible costs (Hackney 2000; Thom 2000). It is important that failures can be recognized when the success criteria indicate unsatisfactory progress. As Woodwell (1992) observed "although our knowledge is imperfect we know enough to be sure that when succession fails...extraordinary efforts at restoration are appropriate." Unfortunately, in practice, the lack of post restoration monitoring has meant that failures have gone unnoticed or have been ignored and few lessons have been learned to improve practice (Hackney 2000). This is what makes the scientific studies such as those being conducted following heavy mineral mining and bauxite mining discussed above so valuable to the future of ecological restoration in both theory and practice.

Resilience and restoration success

Resilience is an important concept in restoration and is defined as the ability of an ecosystem to recover following disturbance (Hobbs 1999; Walker 1999). It is difficult to measure ecosystem resilience except after some kind of disturbance has actually occurred such as the ability to "snap back" or recover from natural episodic effects such as floods, drought, or fire (Cairns 1993). The concept is useful, however, in thinking about different ecosystems types and their relative resilience and thus may help

Table 15. Ecological considerations for the assessment of ecosystem restoration success.

Greater success	⟷	Lower success
Soil similar to original		No soil. Physical, chemical, or biological constraints
Soil seed bank largely intact		Soil seed bank lost
Ability to establish soil nitrogen capital and mineralization quickly		Difficult to establish any N inputs through legumes, fertilizers, or organic wastes
Low level of site disturbance		High disturbance
Small-scale		Large-scale
Locally common ecosystem types		Rare ecosystem type or types
Low complexity of ecosystem structure and function		High complexity
Low biodiversity of genes, species, and habitats		High biodiversity
High potential for natural colonization of plant and animal species		Low potential for natural colonization
No rare or endangered species		Rare and endangered species present
Low species extinction risk		High extinction risk
Size greater than minimum habitat area and animal home ranges to maintain viable populations		Habitat size too small
Good knowledge of biology of "keystone" and "engineer" species		Biology of even common species not known
Knowledge and research base of pre-mining ecosystems, reference ecosystem types, and landscapes high		Knowledge and research base low

indicate, in a more absolute sense, whether a natural ecosystem can be restored or not (Hobbs 1999). Thus, it is possible that ecosystems may have low resilience such that even with costly restoration interventions they cannot be restored structurally and functionally after mining.

In this review the presence or absence of topsoil conserved on the site has been given the status of the primary practical issue for consideration in ecological restoration. However, in the initial restoration planning there are many considerations concerning the nature of the original ecosystem and the structure and functioning of the ecosystem to be restored, and these can be applicable to the evaluation of how successful a particular site-specific ecological restoration might be. Table 15 presents a simple representation of some of these considerations likely to promote greater or lesser success, should a mined site be restored to the pre-mining ecosystem(s). It is oversimplified in various respects. For example, the size of the site to be restored may be too small to provide habitat for some particular species but in the context of a fragmented landscape may be a very valuable asset in maintaining animal species such as butterflies, amphibians, and birds that demonstrate metapopulation dynamics. It will be important to select or weight these considerations on a site by site basis, and if a site contains ecosystems that are too rare, non-resilient, and impossible to restore, no mining should be undertaken. Otherwise, informed value judgements must be made from these considerations and others to give multi-criteria evaluations that inform and provide for a holistic restoration plan containing all the process elements shown in Fig. 2.

Future challenges and sustainable development

Future challenges in ecological restoration in the mining and mineral industries include the increasing scale of operations with large mining companies seeking to exploit large reserves in more remote wilderness environments, greater innovation in new technologies such as the in situ extraction of metals through leaching, the increasing need to regulate and develop environmental management in the artisanal and small mining sector, and the imperative to incorporate policies of sustainable development as far as possible.

Most of the new mining initiatives currently are in developing countries, and this will extend to mining ore deposits in more remote and fragile ecosystems, such as high altitude forest; tall canopy forest in tropics, and in the tundra; and even possibly Antarctica eventually. These developments will require considerable research and ecological knowledge. This requires the support of the formal, large company mining sector, which has to date been found to be the most innovative and creative in meeting the challenges of scientifically driven, environmentally sound mine closure, and restoration underpinned by the necessary research and development (Warhurst 1994). Most of the worlds large mining companies now know that environmentally sound practices including restoration do not add significantly to the costs of new mining projects, and innovation in environmental technologies can even provide income by being commercially exploited (Warhurst 1994). In fact environmental behaviour correlates most closely with the capacity of a company to innovate. The attitude of the large mining companies has been recently shown in the setting-up of the Mining, Minerals and Sustainable Development Project (MMSD) where 30 of the largest mining companies, who usually compete with each other, are collaborating in a global strategic initiative that is an independent analysis that the companies believe could guide the future of the industry (World Economic Forum 2000; MMSD 2001).

Artisanal and small-scale mining represents the other end of the spectrum. It is large in terms of the people directly involved in it: Indonesia 350 000 (Hollaway 1997), Brazil 300 000 “garimpeiros” (Cleary and Thornton 1994), and 200 000 in both Tanzania and Zimbabwe (van Straaten 2000). It is a growing sector particularly in southern Africa, Latin America, and south east Asia. There are an estimated 10 million people dependent on small-scale mining for gold, chrome, tin, and gemstones in southern Africa alone[2] and possibly 80 million worldwide. This mining sector provides for some of the poorest people in the world, a large proportion of whom are women and children, working small-scale, often low grade ore deposits that are not economic to large-scale mining. This sector is characterized by technological backwardness and lack of economic and environmental knowledge. It is clear that ecological restoration will have to “take its turn” and contribute to solutions for this sector that takes a holistic view of sustainable development and creating sustainable livelihoods. Problems of this sector include deforestation and removal of natural vegetation, thousands of small workings with many open pits, the extensive use of mercury in gold extraction, unplanned growth of villages and towns without clean water and sanitation, and alluvial workings which can cause extensive disturbance and damage to river systems.

The social agenda as part of sustainable development will become increasingly important to environmental management in the mining sector in general and to ecological restoration in particular. An increasing number of examples are putting into practice partnerships between the mining company and the local community for mutual benefit (Epps 1997). An unusual example concerning small-scale mining was in Las Cristinas, Venezuela, where a large Canadian company commenced gold exploration in 1992. Small-scale mining had been occurring in the area under strained relations between the small-scale miners and the government. In the event the large mining company allocated part of its concession to the small-scale miners who formed an association to regulate and manage their area. This was followed up by training in mining techniques, business management, environment, health, and safety. Thus a

[2] Private communication: Dr. D. Shoko, University of Zimbabwe, Mt. Pleasant, Harare, Zimbabwe.

legal organization that can now develop systems of environmental management including restoration has been created (Epps 1997) from a group of largely independent opportunistic small-scale miners operating illegally.

Finally, a further challenge to sustainable development is the continuing social and environmental problems associated with the enormous number of abandoned and "orphaned" mine sites. Although the case for ecological restoration of most of these sites is the same as for active mines the assignment of responsibilities is different. Non-action is usual because of non-identification of the responsible body. National approaches such as the "Superfund" arrangement in U.S.A. or national contaminated land policies elsewhere seem to be a possible answer. Again, however, it is unlikely that even strategic intervention by national governments can succeed in relatively poor developing countries without significant financial support of the private mining industry providing for the costs of restoration of past mining degradation.

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References

Allan, R.J. 1995. Impact of mining activities on the terrestrial and aquatic environment with emphasis on mitigation and remedial measures. *In* Heavy metals: problems and solutions. *Edited by* W. Salomons, U. Forstner, and P. Mader. Springer-Verlag, Berlin. pp. 119–140.

Allen, E.B. 1992. Evaluating community-level processes to determine reclamation success. *In* Evaluating reclamation success: the ecological consideration — proceedings of a symposium. *Edited by* J.C. Chambers and G.L. Wade. U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station, Radnor, Pa, pp. 47–58.

Alvarez, H., Ludwig, J.A., and Harper, K.T. 1974. Factors influencing plant colonization of mine dumps at Park City, Utah. Am. Mid. Nat. **92**: 1–11.

Anderson, I. 1992. Can Nauru clean up after the colonialists? New Scientist, 18th July. pp. 12–13.

Anderson, I. 1996. Polluter pays up in Papua. New Scientist, 22nd June, p. 9.

Anderson, P. 1995. Ecological restoration and creation: a review. Biol. J. Linn. Soc. **56**: 187–211.

Andrews, S.M., Johnson, M.S., and Cooke, J.A. 1984. Cadmium in small mammals from grassland established on metalliferous mine waste. Environ. Pollut. **33**: 153–162.

Andrews, S.M., Johnson, M.S., and Cooke, J.A. 1989*a*. Distribution of trace element pollutants in a contaminated grassland ecosystem established on metalliferous fluorspar tailings. 1. Lead. Environ. Pollut. **58**: 73–85.

Andrews, S.M., Johnson, M.S., and Cooke, J.A. 1989*b*. Distribution of trace element pollutants in a contaminated grassland ecosystem established on metalliferous fluorspar tailings. 2: Zinc. Environ. Pollut. **59**: 241–252.

Andrews, S.M., Cooke, J.A., and Johnson, M.S. 1989*c*. Distribution of trace element pollutants in a contaminated ecosystem established on metalliferous fluorspar tailings. 3: Fluoride. Environ. Pollut. **60**: 165–179.

Anon. 1997. Mining-facts and figures. UNEP Industry and Environment, October-December. pp. 4–9.

Antonovics, J., Bradshaw, A.D., and Turner, R.G. 1971. Heavy metal tolerance in plants. Adv. Ecol. Res. **7**: 1–85.

Armstrong, K.N., and Nichols, O.G. 2000. Long-term trends in avifaunal recolonization of rehabilitated bauxite mines in the jarrah forest of south-western Australia. For. Ecol. Manage. **126**: 213–225.

Ash, H.J., Gemmell, R.P., and Bradshaw, A.D. 1994. The introduction of native plant species on industrial waste heaps: a test of immigration and other factors affecting primary succession. J. Appl. Ecol. **31**: 74–84.

Australian Environment Protection Agency. 1995. Rehabilitation and Revegetation., Best Practice Environmental Management in Mining. Commonwealth of Australia, Barton, ACT.

Baker, A.J.M., and Proctor, J. 1990. The influence of cadmium, copper, lead and zinc on the distribution and evolution of metallophytes in the British Isles. Plant Syst. Evol. **173**: 91–108.

Baker, A.J.M., McGrath, S.P., Sidoli, C.M.D., and Reeves, R.D. 1995 *a*. The potential for heavy metal decontamination. Min. Environ. Manage. **3**: 12–14.

Baker, S.R., Gardener, J.H., and Ward, S.C. 1995*b*. Bauxite mining environmental management and rehabilitation practices in Western Australia. World's Best Practice in Mining and Processing Conference Sydney. Australas. Inst. Min. Metall. pp. 43–53.

Barbour, A.K. 1994. Mining non-ferrous metals. *In* Mining and its environmental impact. *Edited by* R.E. Hester and R.M. Harrison. Issues in Environmental Science and Technology, Royal Society of Chemistry, Letchworth, England, pp. 1–15.

Bengson, S.A. 1995. Stabilization of copper mine tailings: Two decades of management in the arid Southwest. Min. Environ. Manage. **3**: 14–17.

Berger, J.J. 1990. Environmental restoration science and strategies for restoring the earth. Island Press, Washington, D.C.

Bergholm, J., and Steen, E. 1989. Vegetation establishment on a deposit of zinc mine wastes. Environ. Pollut. **56**: 127–144.

Boulton, I.C., Cooke, J.A., and Johnson, M.S. 1994. Fluoride accumulation and toxicity in wild small mammals. Environ. Pollut. **85**: 161–167.

Box, J. 1992. Conservation or greening? The challenge of post – industrial landscapes. Brit. Wildlife, **4**: 273–279.

Box, J., Coppin, N., Grigg, C., and Hill, A. 1998. Restoration of Heathlands. Min. Environ. Manage. **6**: 19–22.

Bradshaw, A.D. 1983. The Reconstruction of ecosystems: Presidential address to the British Ecological Society, December 1982. J. Appl. Ecol. **20**: 1–17.

Bradshaw, A.D. 1984. Land restoration: now and in the future. Proc. R. Soc, Lond. Ser. B, **223**: 1–23.

Bradshaw, A.D. 1990. The reclamation of derelict land and the ecology of ecosystems. *In* Restoration ecology: A synthetic approach to ecological research. *Edited by* W.R. Jordan, M.E. Gilpin, and J.D. Aber. Cambridge University Press, Cambridge, U.K. pp. 53–74.

Bradshaw, A.D. 1999. The importance of nitrogen in the remediation of degraded land. *In* Remediation and management of degraded lands. *Edited by* M.H. Wong, J.W.C. Wong, and A.J.M. Baker. Lewis, Boca Raton, Fla. pp. 153–162.

Bradshaw, A.D., and Chadwick, M.J. 1980. The restoration of land: The ecology and reclamation of derelict and degraded land. Blackwell, Oxford, U.K.

Brady, E. 1993. Revegetation and environmental management of lead-zinc mine tailings. M. Phil. Thesis, Department of Environmental and Evolutionary Biology, University of Liverpool, Liverpool, U.K.

Bray, R.J., and Curtis J.T. 1957. An ordination of the upland forest communities of southern Wisconsin. Ecol. Monogr. **27**: 325–349.

Brooks, R.R., and Malaisse, F. 1985. The heavy metal-tolerant flora of Southcentral Africa. A.A. Balkema, Rotterdam.

Cabrara, F., Clemente, L., Diaz Barrientos, E., Lopez R., and Murillo, J.M. 1999. Heavy metal pollution of soils affected by the Guadiamar toxic flood. Sci. Total Environ. **242**: 117–129.

Cairns, Jr., J. 1980. The recovery process in damaged ecosystems. Ann Arbor Science Publishers, Inc. Michigan.

Cairns, Jr., J. 1989. Restoring damaged ecosystems: is predisturbance condition a viable option? Environ. Prof. **11**: 152–159.

Cairns, Jr., J. 1991. The status of the theoretical and applied science of restoration ecology. Environ. Prof. **13**: 186–194.

Cairns, Jr., J. 1993. Ecological restoration: replenishing our national and global ecological capital. *In* Nature conservation 3: Reconstruction of fragmented ecosystems —global and regional perspectives. *Edited by* D.A. Saunders, R.J. Hobbs, and P.R. Ehrlich. Surrey Beatty & Sons, Chipping Norton, New South Wales, pp. 193–208.

Cairns, Jr., J. 2000. Setting ecological goals for technical feasibility and scientific validity. Ecol. Eng. **15**: 171–180.

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

October, pp. 34–37.
Camp, P., and Weisser, P.J. 1991. Dune rehabilitation, flora, and plant succession after mining at Richards Bay, South Africa. *In* Dune forest dynamics in relation to land-use practices. *Edited by* D.A. Everard and G.P.V. Maltitz Foundation for Research Development, Pretoria, pp. 106–123.
Carrick, D.A. 1984. The pattern of fluoride accumulation in plants and soil with reference to grassland established on a fluorspar tailings dam. MSc.thesis, Department of Biology, Sunderland Polytechnic, Sunderland, U.K.
Chambers, J.C., and Wade, G.L. 1992. Evaluating reclamation success: the ecological consideration — proceedings of a symposium. General Technical Report Northeastern Forest Experiment Station, USDA Forest Service. Northeastern Forest Experiment Station, Radnor, Pa.
Chambers, J.C., MacMahon, J.C., and Wade, G.L. 1992. Differences in successional processes among biomes: importance for obtaining and evaluating reclamation success. *In* Evaluating reclamation success: the ecological consideration — proceedings of a symposium. *Edited by* J.C. Chambers and G.L. Wade. U.S.D.A Forest Service, Northeastern Forest Experiment Station, Radnor, Pa. pp. 59–72.
Chambers, J.C., Brown, R.W., and Williams, B.D. 1994. An evaluation of reclamation succession Idaho's phosphate mines. Rest. Ecol. **2**: 4–16.
Chaney, R., Brown, S., Li, Y.-M., Angle, S., Homer, F., and Green, C. 1995. Potential use of metal hyperaccumulators. Min. Environ. Manage. **3**: 9–11.
Cleary, D., and Thornton, I. 1994. The environmental impact of gold mining in the Brazilian Amazon. *In* Mining and its Environmental Impact. *Edited by* R.E. Hester and R.M. Harrison. Issues in Environmental Science and Technology, Royal Society of Chemistry, Letchworth, England, pp. 17–30.
Clewell, A.F. 2000. Restoring for Natural Authenticity. Ecol. Rest. **18**: 216–217.
Cooke, J.A. 1999. Mining. *In* Ecosystems of disturbed ground. *Edited by* L. Walker. Ecosystems of the World, Elsevier, Amsterdam, pp. 381–400.
Cooke, J.A., Andrews, S.M., and Johnson, M.S. 1990. Lead, zinc, cadmium and fluoride in small mammals from contaminated grassland established on fluorspar tailings. Water Air Soil Pollut. **51**: 43–54.
Cox R.M., and Hutchinson T.C. 1980. Multipe metal tolerances in the grass Deschampsia cespitosa (L.) Beauv. from the Sudbury smelting area. New Phytol. **84**: 631–647.
Crilly, J., Kinsella, A.J., Johnson, M.S., and Brady, E. 1998. Evaluation of blood and tissue lead as estimators of exposure in sheep. Ir. J. Agric. Food Res. **37**: 17–28.
Daily, G.C. 1995. Restoring value to the world's degraded lands. Science, **269**: 350–354.
Department of the Environment 1993. Landscaping and revegetation of China clay wastes, Main Report. HMSO, London, U.K.
Department of the Environment 1994. The reclamation and management of metalliferous mining sites. HMSO, London, U.K.
Dudka, S., and Adriano, D.C. 1997. Environmental impacts of metal ore mining and processing: A review. J. Environ. Qual. **26**: 590–602.
Ehrlich, P.R. 1993. The scale of the human enterprise. *In* Nature conservation 3: Reconstruction of fragmented ecosystems -global and regional perspectives. *Edited by* D.A. Saunders, R.J. Hobbs and P.R. Ehrlich. Surrey Beatty & Sons, Chipping Norton, New South Wales, pp. 3–8.
Epps, J.M. 1997. The social agenda in mine development. UNEP Industry and Environment, October-December. pp. 32–35.
Fairbanks, D.H.K., Thompson, M.W., Vink, D.E., Newby, T.S., van den Berg H.M., and Everard D.A. 2000, The South African Land-cover Characteristics Database: a synopsis of the landscape. S. Afr. J. Sci. **96**: 69–82.
Ferguson, K.D., and Erickson, P.M. 1988. Pre-mine prediction of acid-mine drainage. *In* Environmental management of solid waste: dredged material and mine tailings. *Edited by* W. Salomons and U. Forstner, Springer-Verlag, Berlin, pp. 24–43.
Foord, S.H., Aarde, R.J.v., and Ferreira, S.M. 1994. Seed dispersal by vervet monkeys in rehabilitating coastal dune forests at Richards Bay. S. Afr. J. Wild. Res. **24**: 56–59.
Foster, B.L., and Tilman, D. 2000. Dynamic and static views of succession: testing the descriptive power of the chronosequence approach. Plant Ecol. **146**: 1–10.
Geiger, G., Federer, P., and Sticher, H. 1993. Reclamation of heavy metal-contaminated soils: field studies and germination experiments. J. Environ. Qual. **22**: 201–207.

Ginocchio, R. 1998. Chile: Restoration challenges. Min. Environ. Manage. **6**: 7- 9.

Glenn – Lewin, D.C., and Van der Maarel, E. 1992. Patterns and processes of vegetation dynamics. *In* Plant succession: theory and prediction. *Edited by* D.C. Glenn – Lewin, R.K. Peet and T.T. Veblen . Chapman and Hall, London, pp. 11–31.

Gowdy, J.M., and McDaniel, C.N. 1999. The Physical destruction of Nauru: an example of weak sustainability. Land Econ. **75**: 333–338.

Guo, H., Wu, D., and Zhu, H. 1989. Land restoration in China. J. Appl. Ecol. **26**: 787–792.

Hackney, C.T. 2000. Restoration of coastal habitats: expectation and reality. Ecol. Eng. **15**: 165–170.

Haigh, M.J. 1993. Surface mining and the environment in Europe. Int. J. Surf. Min. Reclam. **7**: 91–104.

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

He, J., Tan, H., Sommar, J., Xiao, Z., and LIndqvist, O. 1998. Mercury pollution in a mining area of Guizhou, China: Fluxes over contaminated surfaces and concentrations in air, biological and geological samples. Toxicol. Envion. Chem. **67**: 225–236.

Hetrick, B.A.D., Wilson, G.W.T., and Figge, D.A.H. 1994. The influence of mycorrhizal symbiosis and fertilizer amendments on establishment of vegetation in heavy metal mine spoil. Environ. Pollut. **86**: 171–179.

Hobbs, R.J. 1999. Restoration of disturbed ecosystems. *In* Ecosystems of disturbed ground. *Edited by* L. Walker. Ecosystems of the World, Elsevier, Amsterdam. pp. 691–705.

Hobbs, R.J., and Hopkins, A.J.M. 1990. From frontier to fragments: European impact on Australia's vegetation. Proc. Ecol. Soc. Aust. **16**: 93–114.

Hobbs, R.J., and Norton, D.A. 1996. Towards a conceptual framework for restoration ecology. Rest. Ecol. **4**: 93–110.

Hollaway, J. 1997. Small-scale mining: how to combine development with low environmental impact. UNEP Industry and Environment, October-December. pp. 44–48.

Holling, C.S. 1978. Adaptive environmental assessment and management. John Wiley and Sons, London, U.K.

Hutchinson, T.C., and Whitby, L.M. 1974. Heavy metal pollution in the Sudbury mining and smelting region of Canada. 1. Soil and vegetation contamination by nickel, copper, and other metals. Environ. Conserv. **1**: 123–132.

Jackson, L.L., Lopoukhine, N., and Hillyard, D. 1995. Ecological restoration: A definition and comments. Rest. Ecol. **3**: 71–75.

Jasper, D.A., Robson, A.D., and Abbott, L.K. 1987. The effect of surface mining on the infectivity of vesicular-arbuscular mycorrhizal fungi. Aust. J. Bot. **35**: 641–652.

Jeffrey, D.W., Maybury, M., and Levinge, D. 1975. Ecological approach to mining waste revegetation. *In* Minerals and the environment *Edited by* M.J. Jones. The Institution of Mining and Metallurgy, London, U.K. pp. 371–386.

Johnson, M.S. 1978. Land reclamation and the botanical significance of some former mining and manufacturing sites in Britain. Environ. Conserv. **5**: 223–228.

Johnson, M.S. 1980. Revegetation and the development of wildlife interest on disused fluorspar tailings dams. Reclam. Review, **3**: 209–216.

Johnson, M.S., and Bradshaw, A.D. 1979. Ecological principles for the restoration of disturbed and degraded land. Adv. Appl. Biol. **4**: 141–200.

Johnson, M.S., and Putwain, P.D. 1981. Restoration of native biotic communities on land disturbed by metalliferous mining. Min. Environ. **3**: 67–85.

Johnson, M.S., Bradshaw, A.D., and Handley, J.F. 1976. Revegetation of metalliferous fluorspar mine tailings. Instit. Min. Metall. **85**: A32-A37.

Johnson, M.S., McNeilly, T., and Putwain, P.D. 1977. Revegetation of metalliferous mine spoil contaminated by lead and zinc. Environ. Pollut. **12**: 261–277.

Johnson, M.S., Putwain, P.D., and Holliday, R.J. 1978. Wildlife conservation value of derelict metalliferous mine workings in Wales. Biol. Conserv. **14**: 131–148.

Johnson, M.S., Cooke, J.A., and Stevenson, J.K. 1994. Revegetation of metalliferous wastes and land after metal mining. *In* Mining and its Environmental Impact. *Edited by* R.E. Hester and R.M. Harrison. Issues Environmental Science and Technology, Royal Society of Chemistry, Letchworth, England, pp. 31–48.

Kimmerer, R.W. 1984. Vegetation development on a dated series of abandoned lead and zinc mines in Southwestern Wisconsin. Am. Mid. Nat. **111**: 332–341.

Koch, J.L., and Ward, S.C. 1994. Establishment of understorey vegetation for rehabilitation of bauxite-mined areas in the Jarrah Forest of Western Australia. J. Environ. Manage. **41**: 1–15.

Kruger, F.J., Wilgen van B.W., Weaver A.v B., and Greyling T. 1997. Sustainable development and the environment: lessons from the St Lucia environmental impact assessment. S. Afr. J. Sci. **93**: 23–33.

Larsen, S., and Widdowson, A.E. 1971. Soil fluorine. J. Soil Sci. **22**: 210–221.

Levy, J.K., Hipel, K.W., and Kilgour D.M. 2000. Using environmental indicators to quantify the robustness of policy alternatives to uncertainty. Ecol. Model. **130**: 79–86.

Lubke, R.A., Moll, J.B., and Avis, A.M. 1993. Rehabilitation Ecology. *In* Environmental impact assessment, Eastern Shores of Lake St Lucia (Kingsa/Trojan Lease Area). *Edited by* C.E. Services, CSIR, Pretoria, pp. 251–302.

Majer, J.D., and Nichols, O.G. 1998. Long-term recolonization patterns of ants in Western Australian rehabilitated bauxite mines with reference to their use as indicators of restoration success. J. Appl. Ecol. **35**: 161–182.

McBride, P. 1997. Mount Lyell remediation. Min. Environ. Manage. **5**: 4–5.

Mentis, M.T., and Ellery, W.N. 1994. Post-mining rehabilitation of dunes on the north-east coast of South Africa. S. Afr. J. Sci. **90**: 69–74.

Mentis, M.T., and Ellery, W.N. 1998. Environmental effects of mining coastal dunes: conjectures and refutation. S. Afr. J. Sci. **94**: 215–221.

Metals and Minerals Annual Review 1996. Metals and Minerals Annual Review. Mining Journal Ltd, London, U.K.

Miles, J. 1987. Vegetation succession: past and present perceptions. *In* Colonization, succession and stability. *Edited by* A.J. Gray, M.J. Crawley and P.J. Edwards. Blackwell, Oxford, U.K. pp. 1–29.

Mining Annual Review. 1985. Mining Annual Review. Mining Journal Ltd London.

Mining Annual Review. 1995. Mining annual review. Mining Journal Ltd London.

MMSD. 2001. Mining Minerals and Sustainable Development Project. www.iied.org/mmsd.

Morrey, D.R., Baker, A.J.M., and Cooke, J.A. 1988. Floristic variation in plant communities on metalliferous mining residues in the northern and southern Pennines, England. Environ. Geochem. Health, **10**: 11–20.

Morrey, D.R. 1995. Using metal-tolerant plants to reclaim mining wastes. Mining Eng. March. pp. 247–249.

Neal, P.R. 1998. Pollinator restoration. Trends Ecol. Evol. **13**: 132–133.

Nichols, O., and Bamford, M.J. 1985. Reptile and frog utilization of rehabilitated bauxite minesites and dieback-affected sites in western Australia's Jarrah *Eucalyptus marginata* forest. Biol. Conserv. **34**: 227–249.

Nichols, O., Wykes, B.J., and Majer, J.D. 1989. The return of vertebrate and invertebrate fauna to bauxite mined areas in south-western Australia. *In* Animals in primary succession: the role of fauna in reclaimed lands. *Edited by* J.D. Majer,Cambridge University Press, Cambridge, pp. 397–422.

Nicks, L.J., and Chambers, M.F. 1995. Farming for metals? Min. Environ. Manage. **3**: 15–18.

Pammenter, N.W., and Berjak, P. 2000. Evolutionary and ecological aspects of recalcitrant seed biology. Seed Sci. Res. **10**: 301–306.

Pastorok, R.A., MacDonald, A., Sampson, J.R., Wilber, P., Yozzo, D.J., and Titre, J.P. 1997. An ecological decision framework for environmental restoration projects. Ecol. Eng. **9**: 89–107.

Pierzynski, G.M., Schnoor, J.L., Banks, M.K., Tracy, J.C., Licht, L.A., and Erickson, L.E. 1994. Vegetative remediation at Superfund sites. *In* Mining and its environmental impact. Issues in environmental science and technology. *Edited by* R.E. Hester and R.M. Harrison, Royal Society of Chemistry, Letchworth, England, pp. 49–69.

Pimm, S.L. 1991. The Balance of Nature? University of Chicago Press, Chicago.

Pywell, R.F., Webb, N.R., and Putwain, P.D. 1996. Harvested heather shoots as a resource for heathland restoration. Biol. Conserv. **75**: 247–254.

Reeves, R.D., Baker, A.J.M., and Brooks, R.R. 1995. Abnormal accumulation of trace metals by plants. Min. Environ. Manage. **3**: 4–8.

Ripley, E.A., Redman, R.E., and Crowder A.D. 1996. Environmental effects of mining. St. Lucie Press, Delray Beach, Florida.

Ritcey, G.M. 1989. Tailings management: problems and solutions in the mining industry. Elsevier,

Amsterdam.
Salt, D.E., Smith, R.D., and Raskin, I. 1998. Phytoremediation. Ann. Rev. Plant Physiol. Plant Mol. Biol. **49**: 643–668.
Sassoon, M. 1996. Quarry wasteland to nature trail. Min. Environ. Manage. **4**: 9–12.
Sassoon, M. 1998. Los Frailes aftermath. Min. Environ. Manage, **6**: 8–12.
Sengupta, M. 1993. Environmental impacts of mining: monitoring, restoration and control. Lewis, Boca Raton.
Shrader-Frechette, K.S., and McCoy, E.D. 1994. What ecology can do for environmental management. J. Environ. Manage. **41**: 293–307.
Simon, E. 1978. Heavy metals in soils, vegetation development and heavy metal tolerance in populations from metalliferous areas. New Phytol. **81**: 175–188.
Smith, R.A.H., and Bradshaw, A.D. 1979. The use of metal tolerant plant populations for the reclamation of metalliferous wastes. J. Appl. Ecol. **16**: 595–612.
SER 1996. Society for Ecological Restoration, Definitions 1. Ecological restoration. www.ser.org/definitions.html.
Spalding, A., and Haes, C. 1995. Contaminated land — a resource for wildlife :A review and survey of insects on metalliferous mine sites in Cornwall. Land Contam. Rehab. **3**: 25–29.
Stonehouse, J.M., and Mumford, J.D. 1994. Science, risk analysis and environmental policy decisions. Environment and Trade Series No.5. United Nations Environment Programme., Geneva.
Thom, R.M. 2000. Adaptive management of coastal ecosystems restoration projects. Ecol. Eng. **15**: 365–372.
Tordoff, G.M., Baker A.J.M., and Willis A.J. 2000. Current approaches to the revegetation and reclamation of metalliferous mine wastes. Chemosphere, **41**: 219–228.
Tracey, W.H., and Glossop, B.L. 1980. Assessment of topsoil handling techniques for rehabilitation of sites mined for bauxite within the Jarrah forest of Western Australia. J. Appl. Ecol. **17**: 195–201.
Tucker, N.I.J., and Murphy, T.M. 1997. The effects of ecological rehabilitation on vegetation recruitment: some observations from the Wet Tropics of North Queensland. For. Ecol. Manage. **99**: 133–152.
Twigg, L.E., Fox, B.J., and Jia, L. 1989. The modified primary succession following sand mining: A validation of the use of chronosequence analysis. Aust. J. Ecol. **14**: 441–447.
UNEP/OCHA 2000. Cyanide Spill at Baia Mare Romania. United Nations Environment Programme and the Office for the Co-ordination of Humanitarian Affairs Assessment Mission. UNEP. Available at http://w.w.w.mineralresourcesforum.unep.ch/BaiaMare/index.htm.
van Aarde, R.J., Ferreira, S.M., Kritzinger, J.J., van Dyk, P.J., Vogt, M., and Wassenaar, T.D. 1996. An Evaluation of Habitat Rehabilitation on Coastal Dune Forests in Northern KwaZulu-Natal, S. Afr. Rest. Ecol. **4**: 334–345.
van Straaten, P. 2000. Mercury contamination associated with small-scale gold mining in Tanzania and Zimbabwe. Sci. Total Environ. **259**: 105–113.
Veiga, M.M., and Meech, J.A. 1995. Gold mining activities in the Amazon: clean – up techniques and remedial procedures for mercury pollution. Ambio, **24**: 371–375.
Vitousek, P.M., Ehrlich, P.R., Ehrlich, A.H., and Matson, P. 1986. Human appropriation of the products of photosynthesis. Bioscience, **36**: 368–373
Wali, K. 1999. Ecological succession and the rehabilitation of disturbed terrestrial ecosystems. Plant Soil, **213**: 195–220.
Walker, S. 1992. Gold from the mine — crops from the tailing pond. Eng. Min. J. **193**: 18–20.
Walker, L.R. 1993. NItrogen fixers and species replacements in primary succession. *In* Primary succession on Land. *Edited by* J. Miles and D.W.H. Walton, Blackwell Scientific Publications, Oxford, pp. 249–272.
Walker, L.R. 1999. Ecosystems of disturbed ground. Ecosystems of the World series No 16. Elsevier, Amsterdam.
Walker, L.R., and Willig, M.R. 1999. An introduction to terrestrial disturbances. *In* Ecosystems of disturbed ground. *Edited by* L. Walker. Ecosystems of the world, Elsevier, Amsterdam. pp. 1–16.
Walters, C.J., and Holling, C.S. 1990. Large-scale management experiments and learning by doing. Ecology, **71**: 2060–2068.
Warhurst, A. 1994. Environmental best practice in metals production. Mining and its environmental impact. *Edited by* R.E. Hester and R.M. Harrison. Issues in environmental science and technology, Royal Society of Chemistry, Letchworth, England, pp. 133–159.

Webb, N.R. 1996. Restoration ecology: science, technology and society. Trends Ecol. Evol. **11**: 396–397.

Westman, W.E. 1991. Ecological restoration projects: measuring their performance. Environ. Prof. **13**: 207–215.

Wild, H. 1978. The vegetation of heavy metal and other toxic soils. *In* Biogeography and ecology of Southern Africa. *Edited by* M.J.A. Werger W. Junk, The Hague. pp. 1303–1332.

Williams, S.T., McNeilly, T., and Wellington, E.M.H. 1977. The decomposition of vegetation growing on metal mine waste. Soil Biol. Biochem. **9**: 271–275.

Williamson, N.A., Johnson, M.S., and Bradshaw, A.D. 1982. Mine wastes reclamation. The establishment of vegetation on metal mine wastes. Mining Journal Books Ltd.

Winterhalder, K. 1995*a* Early history of human activities in the Sudbury area and ecological damage to the landscape. In Restoration and recovery of an industrial region. *Edited by* J.M. Gunn Springer-Verlag, New York, N.Y. pp. 17–31.

Winterhalder, K. 1995*b* Dynamics of plant communities and soils in revegetated ecosystems-a Sudbury case study. *In* Restoration and recovery of an industrial region. *Edited by* J.M. Gunn. Springer-Verlag, New York, pp. 173–182.

Woodwell, G.M. 1992. When succession fails... *In* Ecosystem rehabilitation volume 1. *Edited by* M.K. Wali. SPC Academic Publishing, The Hague, pp. 27–35.

World Economic Forum 2000. Report of the governors meeting for the mining and metals industry. Davos Switzerland. World Economic Forum.

Wunderle, Jr., J.M. 1997 The role of animal seed dispersal in accelerating native forest regeneration on degraded tropical lands. For. Ecol. Manage. **99**: 223–235.

Wyant, J.G., Meganck, R.A., and Ham, S.H. 1995. A planning and decision – making framework for ecological restoration. Environ. Manage. **19**: 789–796.