United States
Department of
Agriculture

Forest Service

**Northeastern Forest
Experiment Station**

General Technical
Report NE-164

Evaluating Reclamation Success: The Ecological Consideration—Proceedings of a Symposium

Abstract

Includes 10 papers from a symposium organized to review what is known about the ecological principles that will govern the ultimate success or failure of all reclamation efforts on drastically disturbed lands. The papers cover four general areas: soil biological properties and nutrient cycling; vegetation dynamics; animal recolonization; and landscape-scale processes.

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October 1992

Evaluating Reclamation Success: The Ecological Consideration

Based on a session of invited papers
presented at the annual meeting of the
American Society for Surface Mining
and Reclamation

April 23-26, 1990
Charleston, West Virginia

Jeanne C. Chambers and Gary L. Wade, Editors

This symposium was cosponsored by the USDA Forest Service, American Society for Surface Mining and Reclamation, and Western Regional Coordinating Committee on Revegetation and Stabilization of Deteriorated and Altered Lands.

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Introduction

Natural and anthropogenic forces lead to drastic disturbance of landscapes and ecosystems. In many cases, it is desirable to repair the damage or reclaim the areas involved. Causes of large-scale disturbance to ecosystems include landslides and debris from earth movement, floods, or tsunamis; volcanism; salinization or desertification; drastic releases of pollutants; large earth-disturbing construction projects; and surface and subsurface mining. Of these causes, disturbances due to mining are the most widespread and can be the most difficult to reclaim. However, following all disturbances, natural or anthropomorphic, the development of post-disturbance ecosystems proceeds according to basic ecological principles. In the following collection of papers these principles are discussed in the context of mined-land reclamation, but they also are applicable to other disturbances and recovery.

The reclamation of drastically disturbed lands often results in construction of ecosystems literally from bedrock up. Construction of the new ecosystem begins as soon as soils and geological materials are disturbed and replaced. It is at this point that basic characteristics of the new ecosystem are decided. Some of these characteristics are "set in stone;" others will develop or can be modified with more or less effort later in time.

The miner-reclaimer or reclamationist has the power to affect environmental changes both near and distant in space and in time. Using the energy, machinery, and geological and biological materials at hand (all of which can be considered "resources"), the reclamationist determines the new ecosystem's internal (on-site) characteristics: topography, potential soils, on-site hydrology, environmental and biological diversity, microclimate, soil quality and productivity, potential postmining land uses, and on-site aesthetic relationships. Some internal characteristics will have far ranging external characteristics (off-site effects) such as downstream water quality, stream water flow amounts and timing, products from the land, and landscape unit relationships, both financial and purely aesthetic. Some of these characteristics will have transient effects, but others will be important environmental determinants for thousands of years.

Humankind is now a factor in nearly all environments on the face of our planet. Conversely, nearly all environments have an effect on the human environment. Thus, a reclamation-constructed ecosystem can be viewed as having many homocentric characteristics: products produced for human use, potential human land uses, effects on human environments, and human-viewed aesthetic relationships. This in no way negates the importance of natural processes in structuring ecosystems or of using predisturbance ecosystems as models for reconstructed ecosystems. It merely recognizes the potential influence of humans.

With these things in mind, we must reexamine current methods of evaluating the success of land reclamation efforts. Because ecosystems are being constructed that will function in accordance with ecological rules, the success of reclamation must be evaluated holistically in terms of ecological principles.

Accordingly, we organized this symposium, part of the American Society for Surface Mining and Reclamation 1990 Annual Meeting in Charleston, West Virginia, to review what is known about the ecological principles that will govern the ultimate success or failure of all reclamation efforts on drastically disturbed lands. Because ecosystems are complex, their functional characteristics are governed by many principles. Papers produced from the Symposium "Evaluating Reclamation Success: The Ecological Consideration" were divided into four general areas.

Living organisms and populations have definite functions within the ecosystem. They drive ecosystem processes, promote ecosystem development, and lend it stability. Bacteria and fungi are important in this regard, and their roles in ecosystems are discussed in the section **Soil Biological Properties and Nutrient Cycling.** John Zak, Philip Fresquez, and Suzanne Visser address soil microbial processes and dynamics. Michael Allen and Carl Friese point out the important but often overlooked role of mycorrhizal fungi in disturbed landscapes. John Palmer then links soils, microbial populations, and vegetation together with a discussion of nutrient cycling.

New ecosystems develop stability through the process of ecological succession which is manifested most visibly in vegetation establishment and change. This is addressed in the section **Vegetation Dynamics** in which James Young covers the initial stage of plant invasion in a discussion of seeds and seed ecology. Next, Edith Bach Allen covers community-level processes and their effects on succession. Finally, successional similarities and differences among biomes are discussed by Jeanne Chambers, James MacMahon, and Gary Wade.

Animals, both vertebrate and invertebrate, play important roles in ecosystem processes. They pollinate flowers, accelerate nutrient cycles, aerate soils, consume and disseminate plant seeds throughout disturbed and recovering ecosystems, and influence ecosystem development and successional trajectories. In **Animal Recolonization,** Robert Parmenter and James MacMahon discuss animal roles in ecosystems and how these can be evaluated in constructed ecosystems. They also discuss ecosystem properties that enable the establishment of integrated and functioning animal populations.

Ultimately, the new ecosystems on reclaimed land must be integrated with the larger landscape surrounding them. This is the scale at which the external effects of constructed ecosystems become important. In the section **Landscape-**

Scale Processes, Paul Tueller reviews the place of reclaimed lands as part of the surrounding landscape, and Howard Halverson and Roy Sidle look at mining as one land use influencing the cumulative effects of all land uses in a watershed.

The term "reclamation" as commonly used actually encompasses three somewhat different goals of ecosystem construction. These were defined in 1974 by the National Academy of Sciences as follows:

- *Reclamation*: An attempt to create an ecosystem similar to the original or predisturbance ecosystem. It can include introduced species that respond similarly to the native species which they replace.
- *Rehabilitation*: Producing an alternative ecosystem, one that falls within the existing land uses of the area but replaces a native ecosystem with a pasture, park, etc.
- *Restoration*: Re-creation of conditions which are identical before and after disturbance.

Regardless of the goal, restoration, rehabilitation, or the more narrowly defined reclamation, it is understood that the goal includes a stable ecosystem. It also is assumed that characteristics of the reclaimed ecosystem are consistent with surrounding environments and land uses. In the papers that follow, "land reclamation" is used in the broader, more commonly used sense and can include any of the three goals mentioned above.

We the editors and authors hope that this proceedings will focus the attention of land reclamationists, regulators, ecologists, and land managers on the importance of ecological principles and how they relate to the art and technology of land reclamation, ecosystem construction, and progressive development of ecosystems.

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Soil Microbial Processes and Dynamics: Their Importance to Effective Reclamation

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Abstract

The dynamics and activity of the soil microflora are regulated by complex and synergistic interactions between carbon and nutrient availabilities, microfaunal grazing, and abiotic constraints. Evaluation of reclamation success must focus on measuring the occurrence and strengths of the interactions among these factors at various ecological levels. Microbial species abundance distributions and the annual variability parameter are appropriate methods for examining short and long term recovery rates and community stability. The ratio of microbial respiration (R) to microbial biomass (B), termed the metabolic quotient, links microbial activity and organic matter production, and should provide the best approximation of overall system recovery and stability.

Introduction

Disturbance events are now recognized as important factors affecting community structure, ecosystem functioning, and landscape patterns (Pickett and White 1985a; Pickett et al. 1989). In natural systems disturbances occur at various spatial and temporal scales with the effects dependent upon parameters such as intensity, frequency of occurrence, size of the event, and and lifespans of the component organisms (Pickett and White 1985b). What is common to all disturbances are the alterations of resource availability and community structure. Differences in system response to disturbance are a consequence of the degree to which the before mentioned parameters are altered. Since disturbances can affect all levels of ecological organization, which parameters are useful in assessing the responses and recovery of the biotic component will depend upon the hierarchical level at which the investigator is examining the response (Table 1). Depending upon whether one is interested in population versus ecosystem level response to disturbance will dictate which components are measured and the ability of the chosen parameter to examine short and long term perturbation effects. Pickett et al. (1989) have argued that an understanding of disturbance effects can only be revealed through an examination of the "minimal structure" of the ecological level that is being studied. By "minimal structure" they are referring to: (1) organization of entities into the observed units (e.g., individuals, populations, communities); (2) coordination of the interactions between units; (3) regulating function of the unit; (4) information flow between units (i.e., nutrients and energy); and (5) interactions between levels. With this approach, disturbance becomes a change in the minimal structure. Events which result in changes in species composition of a community but have no effect on the functioning of the observed unit are considered as configurational and do not constitute a change in the minimal structure of the unit.

Table 1. — Disturbance effects at hierarchical levels of ecological organization

Level	Component	Response
Individual	Biomass	Growth
Population	Density	Extinction
Community	Abundance distributions	Structure
Ecosystem	Functional groups	Resilience
Landscape	Fluxes of organisms	Stability

From Pickett et al. (1989).

Depending on the scale and severity of the disturbance, Odum (1985) has indicated that in most instances there is an initial negative change in the structure and composition of the component communities. At the ecosystem level nutrient cycling and efficient resource utilization is decreased, and this may result in changes in succession trajectories. The latter is more likely to occur when the disturbance is large or out of the realm of naturally occurring disturbances (e.g., surface mining). Although there is an initial decline in the relative abundances of component species, small scale disturbance usually increases system heterogeneity. Moreover, new patches for colonization are created thus increasing the overall diversity and species richness of the habitat (Denslow 1985). Species which are characterized as "r" or ruderal strategists (Grime 1979) are more likely to colonize these open patches and would not occur in a specific habitat without a disturbance regime since they are generally poor competitors (Tilman 1982). Disturbances which are viewed as catastrophic events (e.g., surface mining) impose a functional homogeneity on the system. From a surface mine reclamation perspective, the questions become: (1) at which level should the effort be made to reestablish the minimal structure of the functioning unit; (2) how does one go about reestablishing the necessary structure and associated functions; and (3) what parameters should be measured to monitor success. This paper will focus specifically on the impact of surface mining and subsequent reclamation efforts on the microbial component of the system, and will examine alternative approaches to measuring reclamation success.

Ecosystem Functioning and Microbial Activity

The processes of decomposition and subsequent nutrient cycling are intimately associated with the activity of the soil microflora (bacteria and fungi) and the microfauna (protozoa, nematodes, and microarthropods). Through their acquisition of carbon and nutrients for growth, fungal and bacterial biomass in soils constitutes an important sink and source of essential plant nutrients. The mechanisms by which nutrients immobilized in microbial biomass become available to plants are not clearly understood; however, increasing evidence indicates that microbial grazing by the microfauna is important in mineralization (e.g., Moore et al. 1988). These interactions are particularly important during litter decomposition (Coleman et al. 1988), an essential aspect of soil formation, and in the root region of plants (Ingham 1986 a,b). Several studies have shown that microbial grazing in the root region can increase plant growth (e.g., Coleman et al. 1984). These increases in plant growth are attributed to greater mineralization rates of nitrogen as a consequence of microfaunal grazing. Microfaunal grazing affects mineralization rates by: (1) altering the composition of the microbial community; (2) affecting microbial growth; and (3) altering nutrient availability (Newell 1984; Moore 1988).

The importance of microorganisms in the reclamation process has been previously stressed by Cundell (1977), Jurgensen (1978), and Parkinson (1979). Because of their ability to fix atmospheric nitrogen and their involvement in converting soil nitrogen into various forms, soil bacteria are crucial components of the nitrogen cycle (Paul and Clark 1989). Because fungi constitute the majority of the microbial biomass in terrestrial systems and are the major decomposing organisms, most studies on the effects of surface mining andd reclamation have focused only on the the fungal component of the system. However, both bacteria and fungi constitute important pools of carbon and nutrients in soilsq. and recent models have emphasized the need to include bacteria, fungi, and the mircofauna in below ground food webs (e.g., Hunt et al. 1987; Moore 1988). With respect to reclamation efforts of highly disturbed soils, simply elucidating the microbial component of the system (i.e., generating species lists) following the application of amendments over time will not provide the information necessary to evaluate the success of the effort. Studies of reclamation efforts will have to focus on the structure and functioning of the microbial component of the system as modified by the activity of the soil microfauna. The importance of microfloral-microfaunal interactions in a reclamation effort has not previously been addressed. However, evidence from undisturbed systems would suggest that these linkages are critical to system functioning and development.

Microbial Community Structure

The choice of a microbial community descriptor to examine the short and long term effects of stripmine reclamation efforts is not a trivial matter. At the community level various characteristics and parameters can be obtained and calculated that will provide the investigator the ability to examine changes in the microbial assemblages with time and with reclamation effort (Table 2). Which parameters are most useful in describing changes in community organization and structure is open for discussion. As stated previously the majority of the research concerning microbial community structure in reclaimed sites has concentrated on the fungal component. Analysis of bacterial assemblage data at the community level has been descriptive due to the difficulties and time constraints of current bacterial taxonomy. Most studies discuss the densities of bacterial colony forming units of various bacterial groups or key genera as affected by the reclamation protocol (e.g., Fresquez and Sabey 1989).

Table 2. — Parameters used to access redevelopment of microbial activity in reclaimed surface mine spoils

Level	Parameter
Population	Number of propagules Species frequencies
Community	Species composition Spatial heterogeneity Species diversity Species abundances Annual variability
Ecosystem and landscape	Decomposition rates Respiration Enzyme activity Microbial biomass carbon/ total organic carbon Metabolic coefficient (qCO_2)

Once a surface mined site is recontoured, previously stockpiled topsoil or another source of organic matter is usually applied to the site (see Visser 1985 for a review). Even under the best of conditions, the microbial richness and inherent system heterogeneity has been destroyed.

Allen and MacMahon (1985) suggested that the loss of spatial heterogeneity between the soil fungi, organic matter, and nutrient pools was a key factor affecting reclamation success on a strip-mined site in Wyoming. Fungal genera on the reclaimed site were broadly dispersed across the landscape, an indication of site homogeneity, while on the undisturbed area fungal taxa were organized around shrubs (Table 3). The loss of spatial heterogeneity within the soil and plant community affected both the species composition and structure of these fungal assemblages. Allen and MacMahon (1985) suggest that the loss of fungal spatial heterogeneity and community structure could be used to measure the success of a reclamation effort.

Table 3. — Spatial heterogeneity of fungal species in topsoiled spoil

Pot size	**Sorensen's Index of Similarity**		
	4 cm^2	**40cm^2**	**4m**
		Undisturbed	
4cm^2	1	0.25	0.30
40cm^2		1	0.22
		Topsoil Amended	
4cm^2	1	0.43	0.50
40cm^2		1	0.50

Data from Allen and MacMahon (1985); values represent the percent of similarity in fungal species composition between the various size sampling areas.

Stanton et al. (1981) proposed that changes in microbial species composition during reclamation may be as important as changes in microbial densities. This statement implies that shifts in microbial species composition will affect the structure and function of the observed microbial assemblages. Compositional changes alone may not affect either structure or function if the changes are compensatory. Fluctuations in the composition of microbial assemblages and their respective frequencies of occurrence have been used by numerous investigators to examine the outcome of a reclamation effort (e.g., Visser et al. 1983; Fresquez and Sabey 1989). For fungi the number of fungal propagules and the number of fungal taxa are reduced following mining. The microbial community may increase after the addition of organic matter and as the vegetation cover increases (e.g., Visser 1984). While some tentative conclusions concerning reclamation success can be obtained by comparing species lists from reclaimed vs. undisturbed sites, this approach does not allow the investigator to address the central issue of microbial community structure. Elucidating the structure of microbial assemblages provides a means for determining the relationships among component species as they may affect key processes such as decomposition and nutrient cycling. By examining the structure of microbial assemblages, one hopes to be able to relate change in structure with change in function, which is a more difficult parameter to measure and understand.

Measures of diversity, which include species abundance measures and diversity indices, have in general been used to examine community structure and to indicate the "well being" of ecological systems (Magurran 1988). These indices utilize both the number of species within a community and the distribution of individuals among species (evenness). By calculating a diversity index, large data sets of species frequencies can be condensed into one value for comparison between sampling times or treatments. Care should be exercised, however, when inferring changes in community stability from a diversity index since the parameters combine a number of variables that actually characterize community structure (Ludwig and Reynolds 1989). By first understanding the limitations of these indices, one can utilize a diversity index to examine large shifts in community structure. The investigator should be aware that two communities can have the same diversity index, but contain either different numbers of species or different evenness values. Although the communities are structurally different, the diversity indices are similar.

Dennis and Fresquez (1989) and Fresquez et al. (1986) have utilized the Shannon index of diversity and associated evenness parameter to examine the effects of either topsoil or sewage sludge on diversity indices of soil fungal assemblages. For a surface mine site in New Mexico, diversity of the soil fungal assemblage was higher in the undisturbed areas than in a site which did not have topsoil applied but which did receive fertilizer and 4.5 metric tons of hay/ha (Table 4). Even after 8 years, diversity and the evenness parameter were low in the nontopsoiled area as compared with the undisturbed site. A second and younger series of areas that received the same hay and fertilizer treatment plus a 20-30 cm layer of topsoil was examined over a 4-year period (Table 4). For the topsoil plots, fungal diversity and evenness increased compared to those sites not receiving topsoil. Based on the diversity indices, evenness, and species composition of the fungal assemblages, Fresquez et al. (1986) indicated that the microbial component may have begun to stabilize in the topsoil treatment by four years following application. With increasing application rates of sewage sludge applied to a degraded range land, Fresquez and Dennis (1990) found that fungal diversity declined by 2 months after application. (Table 5). Evenness declined in a similar manner. In the succeeding three years of the study, diversity was lowest in the plots receiving the highest sewage application with diversity similar in the other treated plots and the control. The results of Fresquez and Dennis (1990) indicate that initial application rates of sewage will have a long term impact on the resultant structure of the soil fungal community.

Table 4. — Diversity indices for fungal assemblages from topsoil amended surface mine soil

Treatment	Reclamation age (yrs)	Shannon (H) Diversity Index
Undisturbed		1.09
Nonamended	6	0.48
	7	0.65
	8	0.45
Topsoil amended	1	0.86
	2	0.88
	4	0.75

Data from Fresquez et al. (1986).

Table 5. — Shannon diversity indices for fungal assemblages in sewage amended soil

Sludge application (Mg/ha)	Time from reclamation			
	2 mon	1 yr	2 yr	3 yr
0	0.94	0.89	1.03	0.72
23	0.69	0.87	1.01	0.89
45	0.51	0.97	0.84	0.75
90	0.24	0.66	0.50	0.61

Data from Fresquez et al. (1986).

Species abundance distributions provide an alternative to diversity indices in describing community structure. Microbial data sets which contain information on the number of species and their relative abundances are used to generate these distributions. Magurran (1988) has emphasized that a species abundance distribution utilizes all the community information and is the most complete mathematical description of the data. Four models have been developed which describe species abundance distributions (Fig. 1). These distributions are characterized by specific biological parameters which account for the form of the distributions (May 1975, 1981) (Table 6). With species abundance distributions, changes in the structure of the microgbial community as a consequence of reclamation can be followed over time. Tilman (1982) observed marked changes in plant species abundance patterns in response to fertilizer addition of a permanent pasture.

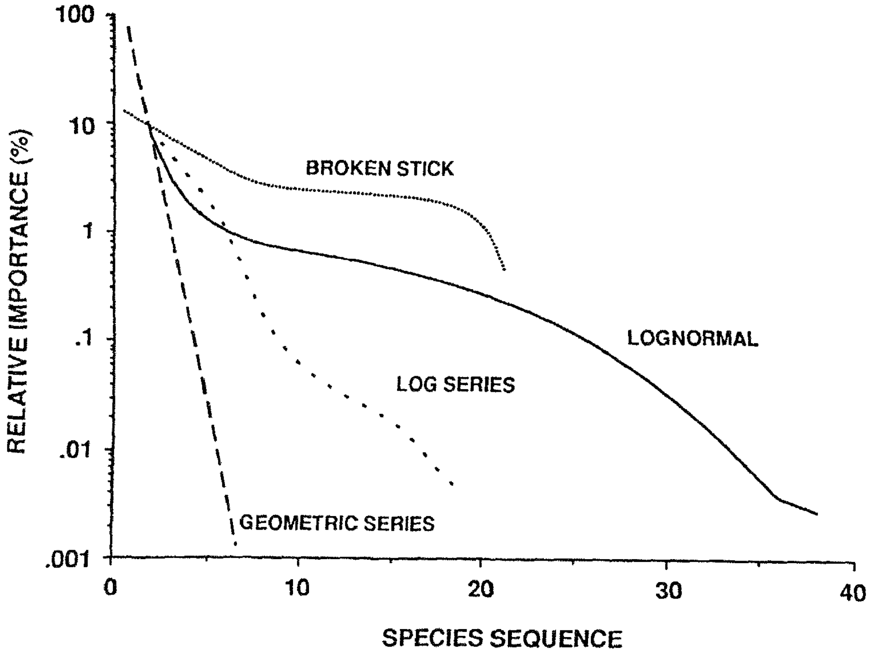

Figure 1. — Species abundance distribution models.

Table 6. — Parameters which characterize the four major species abundance models

Geometric	Log series	Lognormal	Broken stick
Dominant species preempt a portion of limiting resource	Small assemblage of species	Large and varied community	Subdivision of niche space is random and simultanous
Abundance is proportional to amount of resource utilized	Propagules arrive at random intervals	Abundances governed by many independent factors	
Associated with species poor habitats	Small number of abundant species, large number of rare species	Resource utilization is multidimensional	Abundances governed by even division of major resource
Propagules arrive at regular intervals of time	Abundances governed by one or a few factors	Equilibrium community	Characterized solely by the number of species
Nonequilibrium community	Nonequilibrium community		

Information from Magurran (1988) and May (1975, 1981).

Zak and Parkinson (1984) and Zak (1988) have used this approach to examine the effects of nutrient and organic matter application to an extracted oil sands spoil and a subalpine coal mine spoil, in Alberta, Canada, on the structure of root surface fungal assemblages of *Agropyron trachycaulum* (Link, Malte.), slender wheatgrass. Zak and Parkinson (1984) originally showed that the fungal species abundance distributions were not described by the truncated lognormal model. Reexamining the initial 2-year oil sands data and including a fourth year, Zak (1988) found that the forms of the distributions in the control, peat, and fertilizer amended plots were best described by a logarithmic function over the 4-year period. Assemblages from the sewage treated plots differed in that they were best described by an exponential function, and may approximate the geometric model (Zak 1990). The high nutrient concentrations in the sewage shifted the distributions from logseries to geometric (Zak 1990). Although fungal species composition and species numbers associated with the grass root surfaces were altered by the reclamation program (Zak and Parkinson 1984), except in the sewage plots, the structure of these assemblages was not modified by either organic matter addition (peat) or by fertilizer application. The fungi associated with the grass roots in the amended oil sands spoil, except in the sewage treated plots, represent stable assemblages that differ only in composition. The observed structure of these assemblages reflect the paucity of fungal species in the spoil and may be a consequence of the initial homogeneity of the system.

Utilizing the fungal data set from decomposing timothy litter on a subalpine mine spoil of differing reclamation ages (1 to 7 years), Zak (1990) observed that species abundance distributions from the original material were best described by the geometric model. Following field placement of 1, 2, and 4 months, the distributions were found not to change with decomposition age or time from reclamation. At all sampling times, the species abundance distributions were best described by either the geometric of logseries models. The forms of the timothy litter fungal species abundance distributions would indicate that disturbance effects were still evident 7 years after reclamation (Zak 1990).

Calculating species abundance distributions from the data set from Fresquez et al. (1986), we observed that the structure of these fungal assemblages did not change after the first year following topsoil application (Fig. 2A) (Table 7). While both the geometric and logseries models could describe the assemblage from the undisturbed area, the geometric was a better fit. The geometric series is characteristic of species - poor habitats where the abundance pattern is governed by one major factor. The logseries is similar to the geometric, but for which abundances may be governed by more than one factor. The change in the species abundance distribution after topsoil application may reflect the increased heterogeneity of the system. Although fungal diversity was found to change with reclamation age (Fresquez et al. 1986), the structure of these assemblages as determined by species abundance distributions did not differ with time. For those sites which did not receive topsoil, the structure of the fungal assemblages was also found not to change after 6, 7, and 8 years (Fig. 2B). The structure of the assemblages was best described by the logseries model at all sampling times (Table 7). The assemblage from the topsoil amended and control sites represent depauperate systems that are very stable. The addition of sewage sludge at 23, 45, and 90 mg/ha to a degraded range land in west-central New Mexico (Fresquez and Dennis 1990) also shifted the species abundance distribution for the fungal assemblage from this site from geometric to logseries (Fig. 3, Table 8). Although diversity indices and numbers of species declined with increasing sewage application, at the 2-month sampling period the structure of the fungal community was not altered. Except for the 2-month sample in the undistributed area, which was best described by the geometric model, all other distributions over the 3-year period were best described by the logseries (Table 8).

These results may imply that other factors besides nutrients and organic matter may be responsible for the structure of these assemblages. As with most arid systems, moisture availability may be the key regulator affecting structure and function (Whitford 1989).

Recent work by Woolhouse and Harmsen (1987) has suggested that the following community descriptor:

Annual variability (AV) = Variance ($\log N_i - \log N_j$) where N_i = the density of species 1 at time i and N_j = the density of species 1 at time j, may prove to be useful in examining the stability of the microbial component following reclamation. The parameter focuses on the stability in the densities of component species over time. While the descriptor cannot shed light on the underlying mechanisms for or against stability, AV can be used to examine the relative stability of microbial assemblages. The validity for employing this parameter as a measure of stability is based on the assumptions that (1) the densities of component species will not fluctuate widely over time; and (2) although individual species densities will change over a year, the pattern is repeated from year to year. Calculating the AV parameter from the fungal decomposition study of Durall (1984) on a subalpine-coal mine spoil, Zak (1990) found that during decomposition of timothy litter over a 4-month period, the AV parameter for the fungal assemblages on the litter was lowest in the oldest reclaimed site (7 years). In the more recently reclaimed sites, the parameter increased with decomposition age.

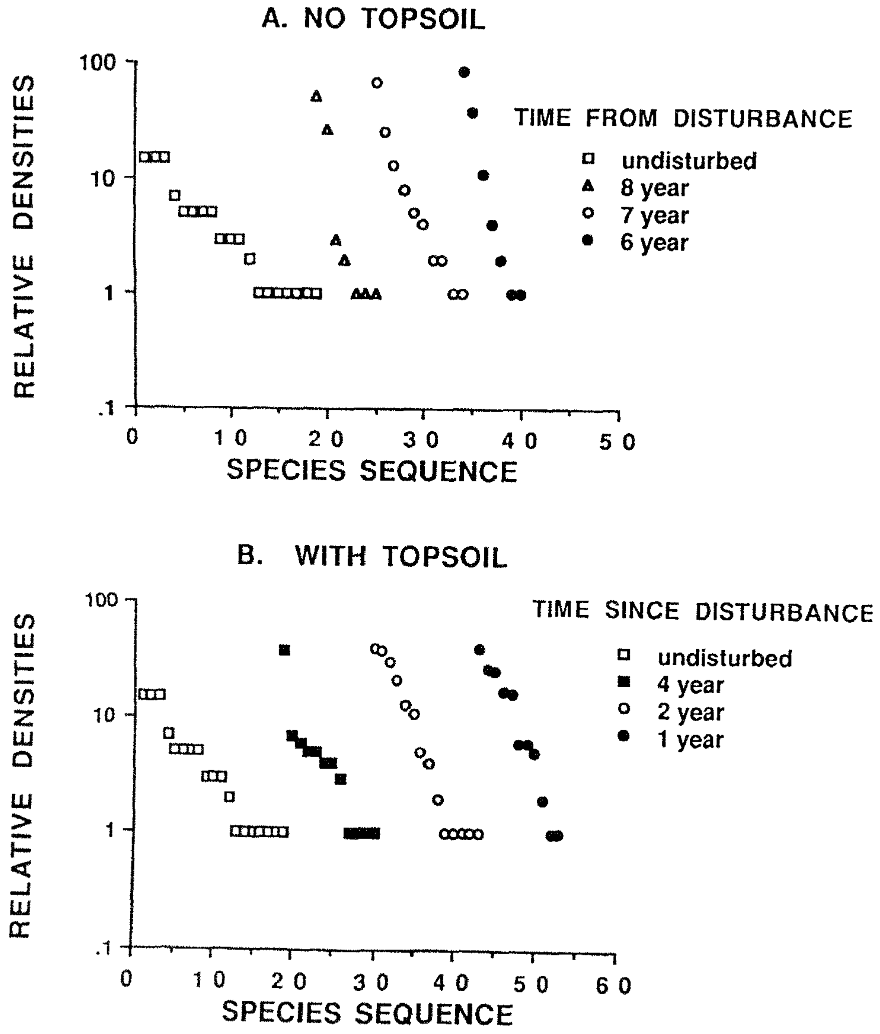

Figure 2.—Effects of topsoil application on fungal species abundance distributions from a surface mine spoil (original data from Fresquez et al. 1986).

Table 7. — Effects of topsoil addition on fungal species abundance distributions

Treatment (time from reclamation (yrs))	Model	P Values
Undisturbed	Geometric	>0.99
	Logseries	>0.30
Topsoil		
1	Geometric	>0.30
	Logseries	>0.50
2	Geometric	>0.80
	Logseries	>0.90
3	Geometric	>0.01
	Logseries	>0.20
Nonamended		
6	Geometric	>0.20
	Logseries	>0.50
7	Geometric	>0.05
	Logseries	>0.99
8	Geometric	>0.02
	Logseries	>0.30

P values indicate probability that Geometric or Logseries species abundance models fit observed data.

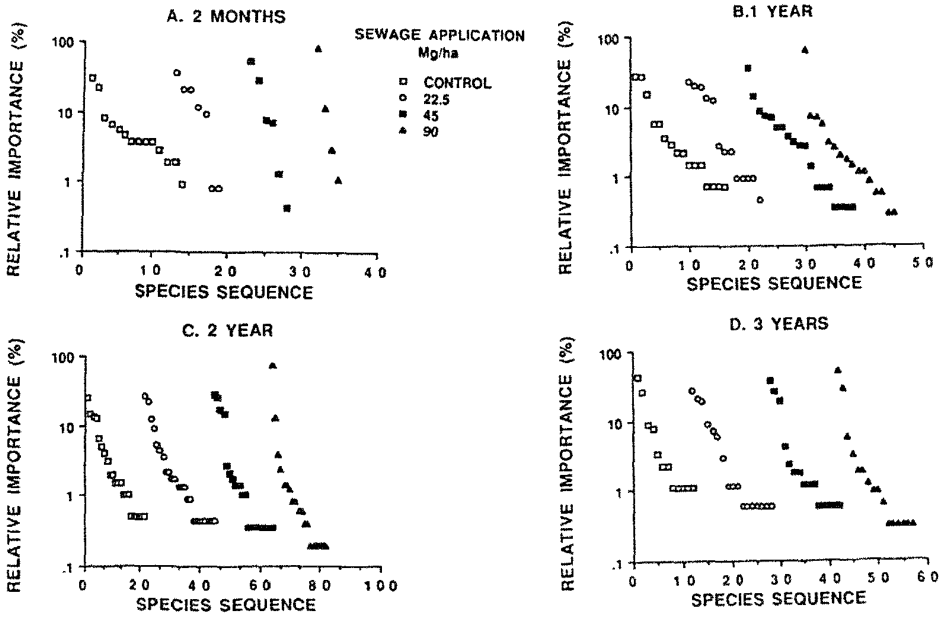

Figure 3. — Fungal species abundance distributions from a dergraded rangeland soil amended with varying application rates of sewage sludge (original data from Fresquez and Dennis 1990).

Table 8. — Effects of sewage sludge on fungal species abundance distributions

Treatment (Mg/ha)	Model	Time from reclamation			
		2 mon	1 yr	2 yr	3 yr
Control	Geometric	>0.50	>0.10	>0.20	>0.02
	Logseries	>0.20	>0.70	>0.80	>0.50
23	Geometric	>0.001	>0.02	>0.0001	>0.10
	Logseries	>0.01	>0.70	>0.90	>0.30
45	Geometric	>0.01	>0.01	>0.001	>0.001
	Logseries	>0.80	>0.80	>0.30	>0.30
90	Geometric	>0.02	>0.001	>0.001	>0.001
	Logseries	>0.70	>0.90	>0.50	>0.50

P values indicate probability that Geometric or Logseries species abundance models fit observed data.

The AV parameter for the mine site in New Mexico examined by Fresquez et al. (1986) did not change for 3 years after topsoil application (Fig. 4). By the end of the fourth year, the AV was lower compared with the previous years. Whether the decrease indicated the beginning of a stable system cannot be evaluated. For the area which did not receive topsoil, the AV value was lower 8 years after the initial reclamation effort than for the topsoil amended site (Fig. 4). These results indicate that while mine spoils may have a reduced number of fungal species, those that do function within the disturbed system represent a stable assemblage of species that is adapted to the abiotic constraints of the habitat. Moreover, unless nutrients and organic matter are added to most mine spoils, the soil fungal community that establishes after the disturbance will remain unchanged for long periods of time.

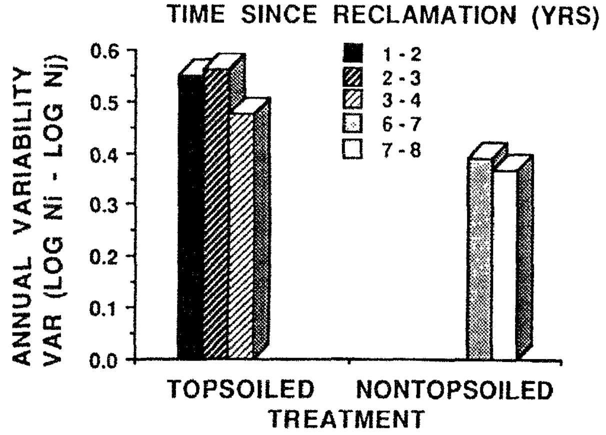

Figure 4. — Stability of the soil fungal community in topsoiled vs. nontopsoil amended surface mine spoil as measured by the annual variability parameter (oirlginal data from Fresquez et al. 1986).

Sewage sludge has been examined in several studies as a source of nutrients and organic matter for ameliorating the adverse effects of surface mine spoils and as a primer for initiating the activity of the soil microflora in these systems (Visser 1985). The rates at which sludge is applied are usually based on plant response without considering the effects on the soil microflora. The development of the soil microbial component of terrestrial systems is dependent on soil organic matter levels and inputs via plant litter in order to maintain the microbial biomass necessary for efficient decomposition and nutrient mineralization. The initial rates of sewage sludge addition become critical in maintaining the microbial component of the system prior to establishing sufficient inputs of carbon and nutrients via plant litter. The choice of sewage application rate can be based on the assessment of plant growth response coupled with the determination of the impact of the sewage application on the AV value for the soil fungal assemblage. Ideally, the amount of sewage applied to a spoil should significantly increase plant growth without causing large fluctuations in AV values. Excessive nutrient inputs to a system can lead to destablization (e.g., Tilman 1982).

Based on plant growth response (Fresquez et al. 1990 a, b) coupled to diversity indices of the soil fungal community, Fresquez and Dennis (1990) found that an application rate of 45 mg/ha as compared to 23 or 90 mg/ha maximized growth without significantly affecting microbial diversity over a 3-year period. All sewage application rates in the study of Fresquez and Dennis (1990) increased the AV value for the fungal assemblages between the first and second growing seasons, with the highest variance found in the 90 mg/ha treatment (Fig. 5). Comparing years one and two, the variance parameter declined in all treatments with the highest application rate being most similar to the value observed for the control area. Even in the control there was a considerable change in the AV value over the first two years of the study. Contrary to the original results of the study by Fresquez and Dennis (1990), the fungal assemblage in the highest sewage application treatment was as stable as that observed in the control plots by the fourth growing season. For the specific system studied by Fresquez and Dennis (1990), the lowest application rate decreased fungal stability while the highest rates resulted in increased plant growth and the maintenance of fungal community stability. The species composition in the 90 mg/ha application (highest) was least similar to the control site over the 4-year period (Table 9). The high nutrient and subsequent litter inputs changed the interactions between fungal species and possible microfaunal regulators, allowing a new assemblage of fungal species to become dominant. Although fungal species composition in the high sewage plot differed from the control area, the structure of the two assemblages was similar.

Table 9.—Sorensen's similarity coefficients for the fungal assemblages from nonamended and sewage amended soil

Treatment (Mg/ha)	2 months				4 years			
	0	23	45	90	0	23	45	90
0		38	30	11		76	67	57
23			77	73			75	67
45				80				71

Data from Fresquez and Dennis (1990).

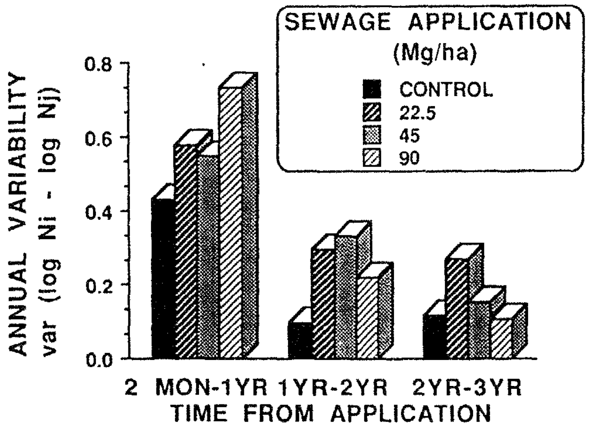

Figure 5. — Changes in fungal community stability following application of sewage sludge application to a degraded rangeland soil as measured by the annual variability parameter (original data from Fresquez and Dennis 1990).

Based on the analyses of the previous fungal data sets, the choice of a community descriptor will significantly affect the conclusion reached concerning the effects of reclamation on the soil microbial community. Comparisons of species lists, while important, do not convey any information concerning how these species are organized into a community or assemblage, and the potential relationship to function. Ultimately we are interested in determining the stability in the composition and structure of the microbial community in space and time as these two components affect function. The previous examples illustrate that at the level of the community, reclamation efforts may impose a stability on the microbial component that may be long term and that may lead to an undesirable recovery sequence. Based on the observed structure of the microbial community, we can infer the probability of reclamation success.

Microbial Processes and Activity

Through its role in decomposition and nutrient cycling, microbial activity is a key factor affecting the functioning of all terrestrial systems. Measurement of process rates governed by the soil microflora and the general metabolic activities of these organisms can therefore be used to address the question of system "health and stability." As with examining reclamation success via microbial community dynamics, several approaches have been utilized to evaluate reclamation efforts, each with varying degrees of success (Table 2).

Decomposition potential is the simplest parameter with which to estimate microbial activity in surface mine spoils. Visser (1985) found that decomposition rates in two mine spoils (prairie grassland and subalpine coal mine) and associated undisturbed areas were dependent upon the material examined, filter paper vs. wood. Over the short term no differences in decomposition rates were detected between disturbed or undisturbed areas. After 33 months in the field, however, decay rates were greater in the disturbed areas than on the control sites (Table 10). Durall et al. (1985) also reported higher decomposition rates of timothy litter on a 1-year-old reclaimed subalpine coal mine site as compared to the 3- and 7-year-old reclaimed areas (Table 11). The low level of available nutrients in the 7-year-old site, resulting from a large amount of undecomposed litter, retarded decomposition rates on this reclaimed site with time. Durall et al. (1985) emphasized that management practices will need to stimulate decomposition of those reclaimed sites where grass litter is increasing so as not to retard long term decomposition potential. The decline in decomposition rates would suggest that the linkages between decomposition and primary production in this system were breaking down.

Table 10. — Percent mass loss of filter paper in a prairie and subalpine surface mine spoil

	Grassland		Subalpine	
Months (no.)	Undisturbed	Disturbed	Undisturbed	Disturbed
6	1	0	0	15
12	1	3	1	10
24	1	49	16	23
44	18	99	39	72

Table 11.—Percent mass loss of timothy litter on different aged, reclaimed subalpine coal mine spoil

Months (no.)	Reclamation age		
	1 yr	3 yr	7 yr
4	52	49	50
13	75	62	60
16	82	63	62

Soil or litter respiration rates (e.g., Durall 1984) and soil enzyme activity (e.g., Fresquez et al. 1987) have also been followed in mine spoils as an alternative approach to estimating microbial activity. Cundell (1977) originally advocated the use of respiration rates and enzyme activity to monitor reclamation efforts. Visser (1985) reported significantly lower CO_2 efflux from disturbed vs. undisturbed grassland and subalpine, surface mine spoils. She attributed the lower activity in the spoil to the loss of microbial biomass and soil organic matter. For surface mine spoils varying in age and vegetation in West Virginia, Stroo and Jencks (1982) observed that respiration rates were lowest for barren sites and highest on the youngest amended sites (3 to 5 yrs after reclamation) which were vegetated with grasses and legumes (Table 12). While microbial activity was high on the younger sites as a consequence of the mulch and fertilizer addition, respiration rates and organic matter content generally declined over time. Only in one 17-year-old locust site were respiration rates not significantly different from values detected in the native soils. The decline in respiration rates suggests that the necessary linkages between microbial acitvity, plant growth, and decomposition were not present in the majority of these sites. Stroo and Jencks (1982) emphasized that reclamation practices at the time would not generate stable systems over the long term.

Table 12.—Spoil respiration rates ($\mu O_2/g/h$) for vegetated and nonvegetated surface mine spoils of varying ages

Site characteristic	Age (years)	Respiration Rate
Barren	17	0.58
Black locust	11	0.84a
	20	1.96b
Grass and legume	3	5.17a
	17	2.07b
Native site		2.77

For each site, letters differ at P=0.05; data from Stroo and Jencks (1982).

Enzymatic activity of the associated soil microflora was followed by Stroo and Jencks (1982) in the same mine spoils as described for the respiration studies to examine the potential of the microbial component to carry out decomposition and mineralization. Amalyase activity (carbon utilization) was lowest in the barren sites and highest on the youngest amended sites (Table 13). With increasing time since reclamation, activity declined in both the grass-legume and locust plots. Phosphatase activity was significantly higher in the the native soils compared to any reclaimed site. Stroo and Jencks (1982) indicate that the cycling of phosphorous may be repressed in these systems. The activity level of microbial enzymes in the mine spoils was found to be positively correlated with available carbon and nitrogen. Fresquez et al. (1987) measured microbial enzyme activity in an unamended and topsoil amended mine spoil in the arid Southwest and found that in the nontopsoiled area enzyme activity was low, particularly in the 8-year-old site (Table 14). Three months after topsoil application the activity of most enzymes was not significantly different from the undisturbed area. Microbial activity increased subsequently and peaked 1 to 2 years following the initial reclamation effort. By 4 years after reclamation, enzyme activity was similar to that observed in the undisturbed site.

Table 13.—Enzyme characteristics of vegetated and nonvegetated surface mine spoils of varying age

Site characteristic	Age (yrs)	Amalyase	Phosphatase
Barren	17	0.6	1.2
Black locust	11	1.1a	0.6
	20	3.5b	4.8
Grass and legumes	3	6.5a	2.9
	17	3.6b	2.8
Native soils		4.6	8.6

Amalyase = μm of reducing sugar/100g soil; phosphatase = μm of PNP/g soil/hr. For each site, letters indicate significant differences at P=0.05; data from Stroo and Jencks (1982).

Table 14. — Soil enzyme activities in surface mined spoils with and without topsoil

Treatment (Time from reclamation (yrs))	Nitrogenase (nm C2H4)	Phosphatase (μg PNP)	Cellulase (μg glucose)
Undisturbed	204a	368a	52bc
No topsoil			
6	809bd	1561c	32cd
7	21c	885b	4e
8	9c	698b	11de
Topsoil			
1	367a	379a	161a
2	1774d	1018bc	91ab
4	208a	325a	35bcd

Data from Fresquez et al. (1987). Within a column, values followed by the same letter do not differ at P=0.05.

Some organic matter and nitrogen amendment should be added to most mine spoils to prime the microbial component. Lindemann et al. (1989) examined the effects of various topsoil and spoil mixtures with or without sewage sludge or sorgum straw on nitrogen mineralization rates (Table 15). For unamended soil and spoil mixtures, nitrogen mineralization declined as the proportion of spoil increased. These results emphasize the importance of the microbial biomass in the immobilization of nutrients. For the sewage amended mixtures, mineralization was highest at the intermediate levels of soil and spoil. The increased nutrients may have compensated for the lower microbial biomass in these soil:spoil mixtures.

Table 15.—Nitrogen mineralization in mixtures of topsoil and spoil material amended with sewage sludge

Treatment	Net organic N mineralization
	Percent
Topsoil only	8.2
Soil:spoil (80:20)	7.1
Soil:spoil (60:40)	6.1
Soil:spoil (20:80)	4.2
Topsoil + sludge	9.8
Soil:spoil + sludge (80:20)	9.4
Soil:spoil + sludge (60:40)	10.1
Soil:spoil + sludge (20:80)	8.4

In some cases microbial biomass, respiration rates and enzyme activity on vegetated, reclaimed sites may recover to near predisturbance levels within a short period of time following reclamation. Insam and Domsch (1988) have argued that the problem with these single microbial parameters as descriptors of reclamation success or system development is that while the level at which they are measured may achieve some stability, they cannot assess the stability and development of the linkages between levels that are necessary for total system stability. This argument supports the minimal structure hypothesis of Pickett et al. (1989) which emphasizes the importance of linkages between levels in examining the impact of disturbance. For measuring system stability, descriptors should link carbon and nutrient flow through the microbial component and organic matter accumulation. Insam and Domsch (1988) proposed that for a steady state to be achieved, there should be a equilibrium level for the ratio of:

$$\frac{\text{microbial biomass carbon } (C_{micro})}{\text{total soil organic carbon } (C_{organic})}$$

If the equilibrium constant is known, then deviations from this value should provide a measure of system stabiity and whether the soil is gaining or losing carbon. Examining a chronosequence of reclaimed agriculture and forest soils in the German Rhineland mining district, Insam and Domsch (1988) reported that annual carbon accumulation was 0.03% during the first 5 years after reclamation and 0.01% for the subsequent 40 years. They calculated an equilibration time of 90 years to achieve a C organic level of 1% in the agricultural site. For the forest site, annual carbon accumulation was 0.59% during the first five years, and 9.05% over the next 40 years. Equilibration was not reached within 46 years. The ratio of C_{micro}/C_{org} declined with time for both sites. While soil organic carbon and microbial biomass taken separately had indicated a steady state by 15 years following reclamation, the ratio of C_{micro}/C_{org} had not reached an equilibrium value even after 50 years (Insam and Domsch 1988). Recently Insam and Haselwandter (1989) have proposed that the metabolic quotient, which is much easier to measure than C_{micro}/C_{org}, is an appropriate integrating parameter with which to measure reclamation success:

qCO_2=Microbial respiration (R)/Microbial biomass (B)

The metabolic quotient is based on the work of Odum (1971), who showed that as long as production is greater than respiration, organic matter and biomass will accumulate in the system and R/B will decrease with time. Two reclaimed sites revegetated with a mixture of grasses and legumes were sampled over a 5-year period to estimated the qCO_2 parameter (Table 16). The qCO2 values for both sites reached a peak 2 years after reclamation and decreased subsequently (Table 16).

Table 16.—Metabolic quotients ($qCO_2.10^3$) of two reclaimed sites

		Time from reclamation			
Site	Initial	1 yr	2 yr	3 yr	4yr
I	3.0	3.8	4.2	4.0	1.8
II	4.2	6.1	6.5	6.2	2.5

Data from Insam and Haselwandter (1989).

Summary

Evaluation of reclamation success must focus not only on plant growth response but, as importantly, on determining the dynamics and functional status of the soil microbial community. While at the outset this may seem to be a formidable task, the approaches recently presented by Picket and White (1985 a, b) and Pickett et al. (1989) concerning patch dynamics and disturbances provide a framework within which to address reclamation success from a microbial perspective. However, we should not lose sight of the fact that the microbial component of the soil is intimately linked to primary production through decomposition and nutrient cycling. Ultimately, it is the efficiency of these linkages that we are trying to access. The problem we are confronted with is the difficulty in measuring the linkage strengths between decomposition and primary production and the stability or instability of the interactions. Since we are not able to directly access these system characteristics, methods have been developed at the community or ecosystem level. Surface mining can impose a functional homogeneity on the system which results in severely altered structure and organization. Thus, this paper addressed two questions: (1) at which ecological level should we be concerned in trying to reestablish a functional unit, and (2) what system parameters should we utilize to measure our success in reestablishing the critical microbial-plant linkages? We also propose parameters that would allow the investigator to determine changes in microbial dynamics and function as they affect system stability.

Management Implications

Based on the minimal structure concept of Pickett et al. (1989), microbial species lists and similarity indices among treatment and times by themselves will not provide the means to address changes in community structure, implied function, and reclamation success. At the community level, parameters which estimate the structure (i.e., numbers of species and their relative abundances) of the microbial assemblages are most appropriate. Of the several approaches that can be used, species abundance distributions (see Magurran 1988 for the procedures to generate these distributions) and the annual variability (AV) parameter (this text) will provide the most complete assessment of community structure and stability. The difficulty with utilizing community descriptors is that they require a considerable investment in time to obtain the species frequency and density data (specifically fungal) required for the calculations. While they provide a theoretical basis for studying reclamation success, the time involved in data collection limits their usefulness as quick methods for evaluation. However, if facilities and expertise are available these parameters should not be overlooked. The paper by Zak and Parkinson (1984) describes the methods to determine fungal frequencies from roots and can be used as a guide for collecting fungal community data. These same approaches can be utilized to examine the plant components of the system.

At the functional level, single parameter measurements (e.g., respiration rates, enzyme activity, decomposition) cannot estimate the long term stability aspects of the system. Variables such as C_{micro}/C_{org} (Insam and Domsch 1987) and qCO_2 (Insam and Haselwandter 1989) will be the most useful since they integrate microbial function, primary production, and decomposition at the ecosystem level. The ratio C_{micro}/C_{org} is predicted to increase over time to some equilibrium level for the system. The direction and rate of change will provide an indication as to the stability of the system. The metabolic quotient qCO_2 may be the easiest functional parameter with which to measure system stability and potential reclamation success. Like the previously described carbon ratio, the qCO_2 value will decrease with time as systems accumulate carbon. Microbial respiration measurements can be assessed by either placing soil in Mason jars with beakers of 1.0 N NaOH solution to trap the CO_2, and titration of the alkali solution to obtain amounts of CO_2 or utilizing automatic CO_2 measuring devises (e.g., infrared gas analyzer). Details of the various procedures can be found in Anderson (1982) and the papers cited in this chapter. Microbial biomass is best determined by the substrate-induced respiration procedure of Anderson and Domasch (1878). Respiration is expressed as CO_2 /h/g dry soil, and converted into microbial biomass (μg C micr/g dry soil) by applying the formula y= 40.04x+0.37 (Insam and Haselwandter 1989), where x is the glucose induced respiration. Once the equipment is obtained and protocols established, this method should allow for effective short and long term monitoring of reclamation success.

Acknowledgment

The efforts of Ms. A. Paige Cooper and Mr. Randy Fuentes in compiling some of the data used in this chapter are appreciated.

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Mycorrhizae and Reclamation Success: Importance and Measurement

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Introduction

A mycorrhiza is a mutualistic association between plants and fungi that affects all terrestrial and some aquatic communities. By affecting the success of individual plants, the association may play a role in the success of reclamation efforts by their presence (improving the growth and fitness of desirable species) or in failure by their absence (enhancing the competitive ability of species with little positive response to the association such as many weedy annuals). Thus, any effort to gauge reclamation success must measure the structure and functioning of the mycorrhizal symbiosis.

The study of mycorrhizae began at the ecosystem level when it was recognized that mycorrhizae aid in the acquisition of critical nutrients for individual plants. Efforts to establish forests in the Caribbean, Australia, and Rhodesia, and shelterbelt plantings in midwestern North America and the Ukrainian region of the Soviet Union demonstrated the importance of ectomycorrhizae to tree survival (e.g., Hacskaylo 1967). More recent work demonstrating that mycorrhizal fungi are sensitive to human perturbations initiated several studies into the demise of mycorrhizae with such anthropogenic disturbances as surface mining, tillage agriculture, and road building (M. Allen 1989). Studies demonstrating that inoculation by mycorrhizae could improve growth and survival of outplanted individuals onto recontoured mined soils (e.g., Aldon 1975; Marx 1975; Carpenter and Allen 1988) clearly indicated a need to understand better when and how to measure adequately mycorrhizal associations in reclaimed sites.

Types of Mycorrhizae and Reclamation Goals

One difficulty in measuring mycorrhizae in natural or human-disturbed communities is differentiating the types of mycorrhizae and developing techniques to characterize them. There are 5 major types of concern to most reclamation efforts. Vesicular-arbuscular mycorrhizae (VA, often referred to as endo-, phycomycetoid, or coenocytic mycorrhizae) are the most prevalent, associating with fungi in the Glomeales (Zygomycete) and forming associations with all grasses, most herbs, and many trees. In this association, the fungus penetrates the cell walls of the root cortex and forms a hyphal matrix radiating out into the surrounding soil. The second most common mycorrhizae type is the ectomycorrhizae. This type is common to many trees important to reclamation, such as pines and oaks. The hyphae of the mycorrhizal fungi (in this case, members of the Ascomycets and Basidiomycetes in addition to members of the genus *Endogone*, a Zygomycete) are found only externally to the root cortical cells but completely encase the short roots in a hyphal mantle. There are three other predominant mycorrhizal types that form a gradient in structure between these two. These include the ericoid mycorrhizae in which the fungus penetrates the cortical cells of ericaceous plants. They form only minimal external hyphae but have a vast array of enzymes enabling them to utilize organic nutrients. They are found in highly organic (peaty) soils, or in soils with high concentrations of heavy metals. The arbutoid mycorrhizae also occasionally penetrate the cells of ericaceous plants but primarily form a hyphal mantle as in the ectomycorrhizae, but with a less extensive covering. Ectendomycorrhizae are formed when the mycorrhizae of pines and other ectomycorrhizal plants form associations with a specialized group fo mycorrhizal fungi that penetrate cell walls and form only a partial mantle.

Other mycorrhizal types of a less general nature include orchid mycorrhizae and monotropoid mycorrhizae. More detail can be found on all types in other books (e.g., Harley and Smith 1983; Allen 1990). These less dominant mycorrhizal types should be considered only if rare and endangered plants forming these types of associations are of concern to the success of reclamation on a given site.

The above descriptions refer to the types of mycorrhizae. Recalling that a mycorrhiza is composed of both a plant and a fungus, that means that both groups of organisms are of concern. As an aid, most plants only form one specific type of mycorrhiza. Thus, knowing the plants of interest can tell the land manager which mycorrhizal types to be concerned within any given community. Relatively comprehensive lists are continually being updated but most plants and their mycorrhizal associates can be found in Trappe (1987) or Harley and Harley (1987). Unfortunately,

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the paper by Harley and Harley (1987) is still the only list to the generic level and it is compounded for Britain. A similar listing of North American plants is needed.

As a generalization, we can suggest that virtually all North American terrestrial plants form mycorrhizae except for annuals in the Chenopodiaceae, Brassicaceae, and Amaranthaceae. Reports of nonmycorrhizal perennials are probably inaccurate because of the difficulties in adequately sampling for mycorrhizae, especially in highly variable habitats (e.g., Allen 1983; E. Allen and M. Allen 1990).

Reclamation Success: A Function of Mycorrhizae?

Do mycorrhizae hasten the rates of reestablishment of desirable plant communities? To test this hypothesis, several research groups have inoculated small plots or individual plants to assess the role of mycorrhizae in reclaiming surface mines following the replacement of the plant growth medium, be that medium spoil (unconsolidated subsoils) or respread topsoil.

Since ectomycorrhizal fungi can be mass produced in culture, they have been used to enhance reclamation of ecosystems where conifers and oaks predominate for several decades. Importantly, these plants approach obligate dependence on mycorrhizae. Without natural or artificial inoculation, these plants simply will not reestablish. Artificial inoculation with selected species found to tolerate the harsh conditions of most reclamation sites enhances the regrowth of many forest systems dramatically. Greater detail on these systems can be found in Marx (1977) and Molina and Trappe (1982). These studies have demonstrated the importance of artificial inoculation of individuals in a reclamation site. In many large scale efforts, such procedures can become very expensive. However, these inoculated forests have value as sources of wood and this may recover much of the reclamation costs. In such systems, considerable efforts have been made to develop mechanical means to plant inoculated trees in easily harvestable rows.

In many habitats, the predominant mycorrhizae are the VA type which cannot be cultured without an actively growing host. Thus, artificial inoculation can only proceed in small patches created by planting inoculated individuals. Also, many of these lands have minimal agricultural value and are primarily reclaimed for wildlife and grazing. In the first study of reclamation of a western arid system that utilized artificial inoculation, Aldon (1975) demonstrated that inoculation with VA mycorrhizae could enhance the survival of *Atriplex canescens.*

In other studies, the potential role of mycorrhizae on the reclamation of VA mycorrhizal dominated disturbed sites was expanded in an important direction. Nicolson (1960) noted that the early successional species of dunes in Scotland, *Salsola kali* (Russian Thistle), was nonmycotrophic. That is, this species does not form mycorrhizae while the later successional species were mycorrhizal. Reeves et al. (1979) and Miller (1979) found similar patterns in disturbed versus undisturbed arid lands in surface mined areas of Colorado and Wyoming. E. Allen and M. Allen (1980) noted that when topsoil containing inoculum was replaced immediately following stripping, mycorrhizal associations were rapidly reestablished with the planted grasses. Alternatively, where topsoil was not replaced, no mycorrhizae were found and the nonmycotrophic weeds persisted for up to 10 years following replanting with no reestablishment of the seeded grasses and shrubs.

E. Allen and M. Allen (1984) hypothesized that mycorrhizae should enhance the succession rates on distrubed areas such as mined lands by enhancing the competitive ability of mycotrophic grasses and shrubs in the presence of nonmycotrophic weeds that naturally colonize these sites. Details of these studies can be found elsewhere and will not be repeated here (see E. Allen 1989; E. Allen and M. Allen 1990; Carpenter and Allen 1988). To summarize, the responses of the dominant plants representing the successional stages can be arranged in continuum (Fig. 1). The species response to mycorrhizae ranged from a negative response of the early successional weeds *Salsola kali* and *Atriplex rosea* to the highly positive responses of *Artiplex canescens* and *Artemisia tridentata.* The important N-fixers, *Purshia tridentata* and *Hedysarum borealis,* depended on both the bacterial and the mycorrhizal fungal symbionts to survive and reproduce. While competitive and facilitative interactions can complicate the successional patterns created by mycorrhizae (E. Allen and M. Allen 1988; E. Allen 1989), in general, the fungal associations are essential to the recovery of these sites.

Measuring Mycorrhizae in Disturbed and Undisturbed Lands

Identifying most mycorrhizae is relatively easy with a minimum of experience. Determining the presence or absence of any given mycorrhizal type is within the capabilities of land managers. The only requirement is a microscope and a lab bench to process samples. Sterile technique is not necessary. Mycorrhizae invade cortical cells and form specific structures that can be readily identified following staining of root segments. Arbuscules, peletons and vesicles are the predominant structures. External mycorrhizae (ecto-, ectendo-, or arbutoid mycorrhizae) are distinguished by the hyphal mantle and hartig net that forms between the cortical cells within a root. These features are well illustrated in other books (e.g., Schenck 1982; Harley and Smith 1983; Allen 1990). In addition, most land grant universities currently have faculty capable of teaching land managers how to identify the symbiosis.

Perhaps the greatest challenge to the study of mycorrhizae is to quantify the association to determine whether the mycorrhiza is present in adequate quantities to be effective in improving the growth and survival of the desired plants. Estimating the activity of mycorrhizae or even the functional

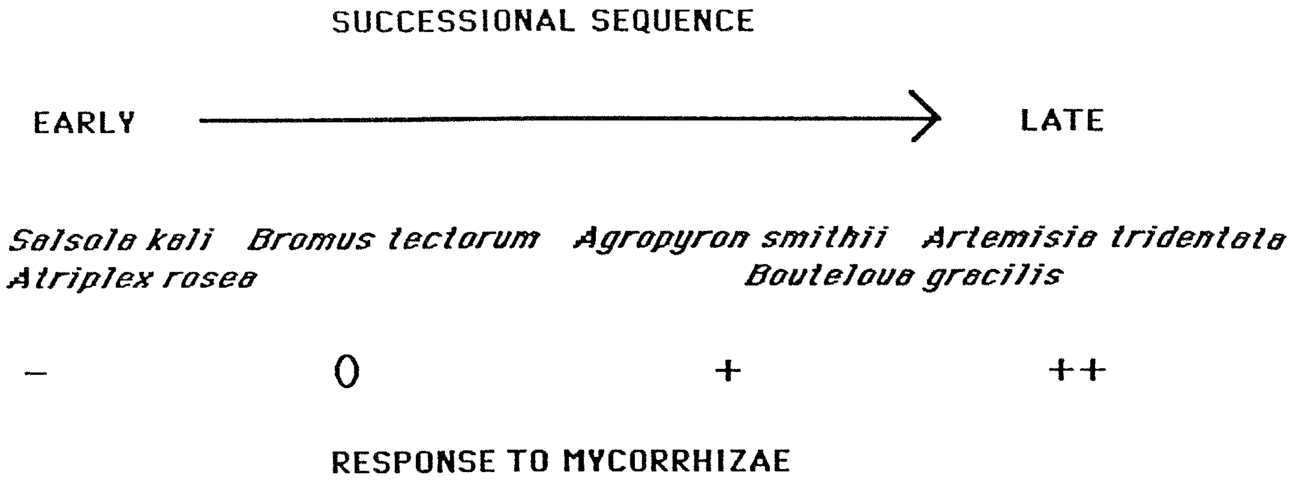

Figure 1.—The relationship between successional sequence of some plant species from mineland shrub-steppe ecosystems and their response to mycorrhizae.

significance of the association within a community is difficult and controversial. Nevertheless, the presence of mycorrhizal associations and of a diversity of fungal species is important and some indications of the presence and the fungal species may be critical to predicting reclamation success.

Techniques for Assessing Mycorrhizal Activity

Several methods currently are used to assess mycorrhizal activity. These include both direct and indirect methods, some simple and some requiring considerable training and time to complete. Also of consideration are the sampling scheme and the types of samples that should be analyzed. Specifically, what sampling regimes are important to the reclamation officer as opposed to researchers currently involved in attempting to understand the specific roles of mycorrhizae in ecosystems? At what scales should samples be taken to assess reclamation success as opposed to determining the fine structure of mycorrhizal dynamics within a single plant or across the entire landscape?

Bioassays of soils for mycorrhizal fungi have been commonly used for a long time. For example, White (1941) determined that a lack of natural ectomycorrhizal inoculum in prairie soils inhibited the establishment of ectomycorrhizal trees. More recently, Reeves et al. (1979) and Schoenberger and Perry (1982) surveyed differing disturbances for their effects on mycorrhizal inoculum. In these techniques, soils to be tested are simply seeded with plants that readily form mycorrhizae and the plants are grown for a given time period in the greenhouse. They are then harvested and the roots stained and assessed for the presence of mycorrhizae by looking for the structures listed above.

There are two indirect techniques for quantifying mycorrhizal activity based on bioassays for mycorrhizal fungi. These have primarily been used to test soils prior to planting to estimate the potential for recovery of mycorrhizae. Mycorrhizal inoculum potential (MIP) was proposed by Moorman and Reeves (1979) as a means to determine the potential for mycorrhizae to reestablish following a disturbance. A similar procedure is called the most probable numbers estimate (MPN) of mycorrhizal fungal densities. This procedure is outlined by Wilson and Trinick (1982) in greater detail. In both of these procedures, a known amount of test soil is mixed into a standard, sterile soil and seeded with a given mycorrhizal plant. After a known period, the plant is harvested and the number of propagules (MPN) or mycorrhizal inoculum potential (MIP) estimated by the percentage of root length infected by the mycorrhizal fungi (for VA mycorrhizae) or by the proportion of root tips infected (ectomycorrhizae).

Several methods are available for direct estimation of mycorrhizal activity. Primarily, this involves determining the percent of the root length containing VA mycorrhizae or the percent of root tips that are ectomycorrhizal using plants collected from the field at differing times following the replacement of the growth medium. Generally, soil cores are taken and the results extrapolated to the surface 10-20 cm of the soil using a known soil volume. To gain a greater understanding of the activity, root length can also be estimated using the same cores. Deeper cores can be used to estimate total mycorrhizal root length per plant or per community. These data give the actual mycorrhizal activity and can best be used to understand the recovery of mycorrhizae of given plants at specific times following reclamation. Examples of these studies giving greater detail of the methods can be found in E. Allen and M. Allen (1980), Allen et al. (1989) and Schenck (1982).

To determine the reproductive potential of the mycorrhizal fungi, direct counts of the reproductive propagules also can be made for many mycorrhizae. VA mycorrhizal fungal reproduction can be assessed using asexual spores isolated directly from field soils. Although several procedures are available in the literature, we have concluded that differential water/sucrose centrifugation obtains a more reliable estimate of the total spore numbers (Ianson and Allen 1986). However, if the reclamation officer does not have access to a centrifuge, simple sieving of soils can separate many spore types in several soils (Ianson and Allen 1986). The fungi forming ectomycorrhizae often form mushrooms fruiting above the soil surface or just under the litter layer (hypogeous fungi). Upon fruiting, these fungi can be surveyed using sample techniques also used for plants or by simple density estimates. Both of these assessments also indicate the species composition of those fungi actively reproducing.

Existing Sampling Strategies for Assessing Mycorrhizal Activity

Several strategies for sampling and statistical analyses have been used to assess mycorrhizal activity. This is especailly important as most research efforts at understanding the distribution and ecology of mycorrhizae have concentrated on small-scale studies or theoretical assessments of their distributions. Few studies have assessed the large-scale practical effects of land management practices. A complicating factor is that mycorrhizal fungi appear to have differing dispersion patterns in different regions and ecosystems. Moreover, since the association is of plant and fungus, the distribution is highly dependent upon the dispersion patterns of the host.

Many studies have stated clearly that one or two strategies must be followed when sampling and assessing the mycorrhizae of a site or of a research effort. Most efforts have concentrated on using a regular (e.g., grid or line intercept) or a random sampling procedure. St. John and Hunt (1983) stated that mycorrhizal infection is clumped and that a negative binomial distribution for assessing percent mycorrhizal infection should be used. St. John and Koske (1988) also stipulated that spores are distributed according to a negative binomial model. What this model states is simply that the infection units or spores are aggregated into discrete patches in a sample area, i.e., the data are skewed with most values being low and a few high.

Alternatively, the negative binomial distribution does not always describe either infection or spore count data. In evenly planted grasslands, mycorrhizae can often be distributed evenly (if a later-successional site) or randomly (for a newly planted site). At the other extreme, in arid lands where plants are widely spaced, both infection and spores can be extremely skewed if both interspace and plant are considered or a value approaching a normal distribution can be obtained if only under-plant samples are considered. More importantly, all of these models only considered the distribution of data from within community sampling as opposed to assessing the influence of a management technique across a hillside or group of communities.

When comparing several studies that looked at the distribution of spore counts and percent VA mycorrhizal infection across geographical regions, the statistical distribution of points and sizes of patches varies considerably across biomes. For example, Anderson et al. (1983) reported that a patch was approximately 10cm^2 whereas Allen and MacMahon (1985) found that even a 2 cm^2 sample grid may have not been small enough to identify the patch size. Even within a site, the statistical distribution of values varies considerably depending on the size of area of interest. Moreover, this variation is not linear with area size or plant diversity (Fig. 2). As can be seen in Fig. 2, our data indicate that the degree of skewness can change across sample scales even on the same site.

Finally, mycorrhizal fungi are sensitive to both seasonal (Allen 1983) and annual (Allen et al. 1987) variations. Simply assessing the percent mycorrhizal infection, sporocarps, or spores at one time will not always give an investigator an accurate assessment of activity. Importantly, simple assessments of surrounding undisturbed (reference) areas may not give an accurate assessment of the density or diversity that one should expect of a reclaimed site. We have found the curvilinear relationship between propagule size and number (Chaung and Ko 1981) to provide a good estimate of the maximum propagule density by which to gauge both the reference and the reclaimed area (Fig. 3).

Responses of Mycorrhizal Fungi to Reclamation Efforts: A Case Study

We conducted a study to determine how differing reclamation strategies affect the outcome of both plant establishment and mycorrhizal redevelopment. We assessed two major factors that could affect reclamation success: soil management and initial plant density. Our indicator of success are spore counts of VA mycorrhizal fungi. We believe that this parameter gives the best indicator of recovery for the following reason: mycorrhizal root infection is exttremely variable in time and the spores represent an indication of the reproductive potential of the fungus and, thus, its fitness on the site. If the fungus is actively reproducing, then it is actively associating with a plant (M. Allen and E. Allen 1990).

Three soil treatments were tested: (1) zero topsoil (using the spoil material only), (2) stored topsoil (wherein the topsoil was removed, stored for 8 years, then replaced back over the plots at a depth of 10 cm), and (3) nonstored topsoil (where fresh topsoil was scraped and directly respread over test plots at a depth of 10 cm). These treatments were subjected to further treatments. The stored topsoil was subdivided onto three subtreatments: unplanted, planted with a moderate density of shrubs, grasses and forbs, and the normal management treatment. The nonstored topsoil treatment was subdivided into two subtreatments, unplanted and planted, also using a moderate density of shrubs, forbs and grasses. A moderate

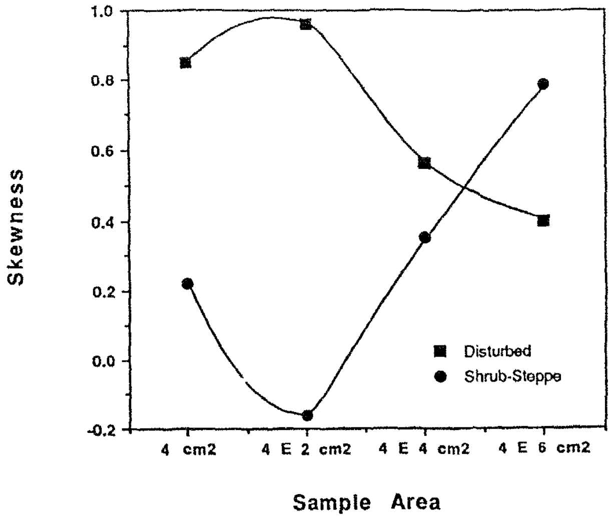

Figure 2.—Skewness using a constant sample size as the area samples increases. Data are from the 1981 Kemmerer site (Allen and MacMahon 1985). All samples were taken using a grid over the sample area indicated (E=10^n).

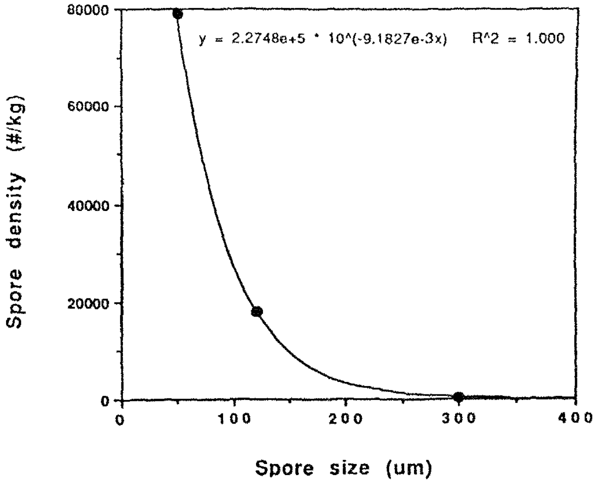

Figure 3.—The relationship between VA mycorrhizal fungal spore density and size using the spore data from Ianson and Allen (1985).

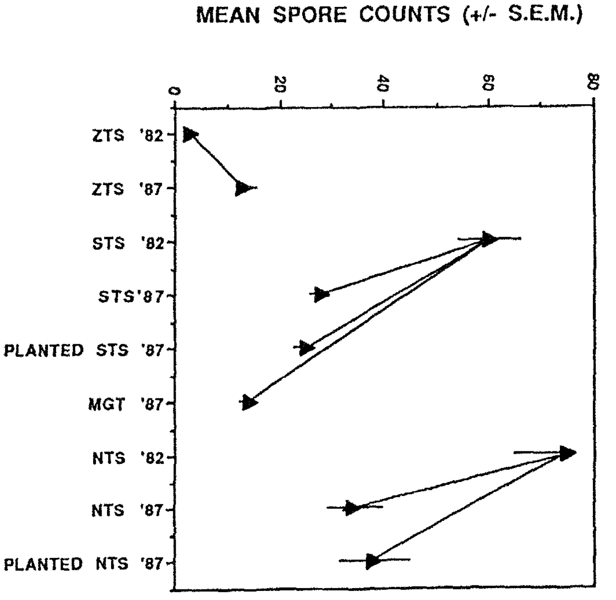

Figure 4.—Influence of initial soil treatment and subsequent management on VA mycorrhizal fungal spore counts in 1982 (the first growing season following the initialization of the site) and 1987. ZTS means no topsoil added, STS means stored topsoil added, NTS means non-stored topsoil added. Planted means that the plots were replanted with a moderate density of shrubs, grasses and legumes (see text for details). MGT means that the plots were managed using the normal company procedures (see text).

density planting meant that individual shrubs were planted at 0.6 x the density of shrubs in the adjacent undisturbed area. An identical number of patches of grasses and legumes were also planted. The normal management treatment consisted of planting barley for the first year after respreading of the stored topsoil, followed the second year by seeding a mixture of grasses, forbs and shrubs. All soil treatments were initiated during the autumn of 1981 and all plantings occurred in the spring of 1982. Each treatment was replicated using three 20 x 50 m plots. We took four soil samples from each plot evenly distributed downslope through the plot using 4 cm diameter by 10 cm deep cores. The data from 1982 (initial conditions) and 1987 (the final sample year) are shown here. Detailed spore, plant production and climate data associated with the El Nino years of 1983-84 and the drought of 1985 can be found in Allen et al. (1987).

Clear differences in the reproductive potential of the fungi were apparent at the whole-plot scale due to the initial conditions to which the plots were exposed. The initial soil treatments and the subsequent management practices had significant impacts on the reproductive activity of the VA mycorrhizal fungi. In both the stored and nonstored topsoil, the total spore density declined to levels that were not significantly different from each other. However, total fungal reproduction was significantly reduced in the management treatments to a level not significantly different from that of the plots that had no topsoil added (and no spores initially present, Fig. 4). Also, we found that the variance to mean ratio (an indicator of the development of patchiness, important to the redevelopment of "islands of fertility" in these semi-arid conditions) was also affected by initial soil treatment (Fig. 5). This ratio increased in the zero topsoil

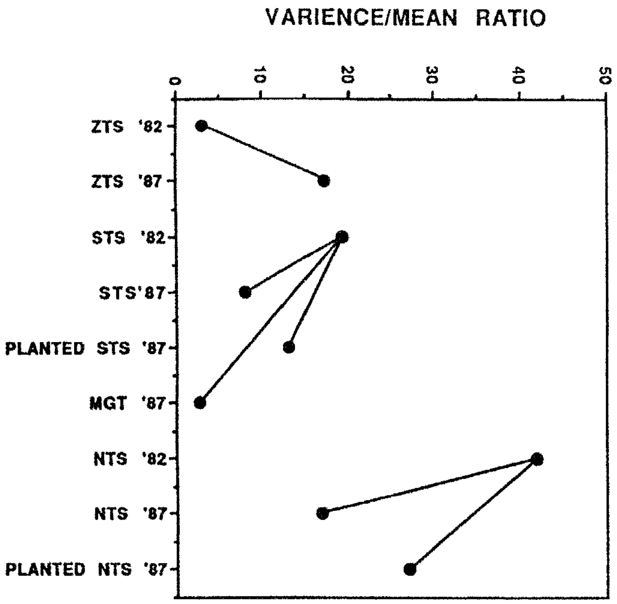

Figure 5.—Variance to mean ratio of the VA mycorrhizal fungal spore counts of different soil and management treatments in 1982 (the first growing season following the initialization of the site) and 1987. ZTS means no topsoil added, STS means stored topsoil added, NTS means non-stored topsoil added. Planted means that the plots were replanted with a moderate density of shrubs, grasses and legumes (see text for details). MGT means that the plots were managed using the normal company procedures (see text).

and declined in all others. Importantly, it remained relatively high in the plots where the plants were arranged in patchiness but declined to the lowest levels in the management treatment. These data clearly indicate that, in arid lands where the reformation of spatial patches that serve as cues for animals, as windbreaks for snow and propagule deposition, the traditional covercrop/seeding management treatment resulted in a less desirable site for mycorrhizae.

We were also interested in how changing the initial planting density might affect the reestablishment of mycorrhizal associations. Using the same site described above, we altered the planting pattern as follows. We created low density (0.2 x normal), moderate density (0.6x), normal density (1.0x), and high density (1.5 x normal) plantings. A normal density planting was equal to the shrub density of the surrounding native areas (16,000 plants per ha). In addition, shrubs were planted in clumped or regular patterns to test for the development of patchiness. The higher density plantings resulted in greater total fungal reproduction than the lower density plantings (0.6x). However, when we tested for the differences in spore counts as a function of individual plant groups, no differences were detected between adjacent plots that contained differing numbers of plant shrubs (Fig. 7).

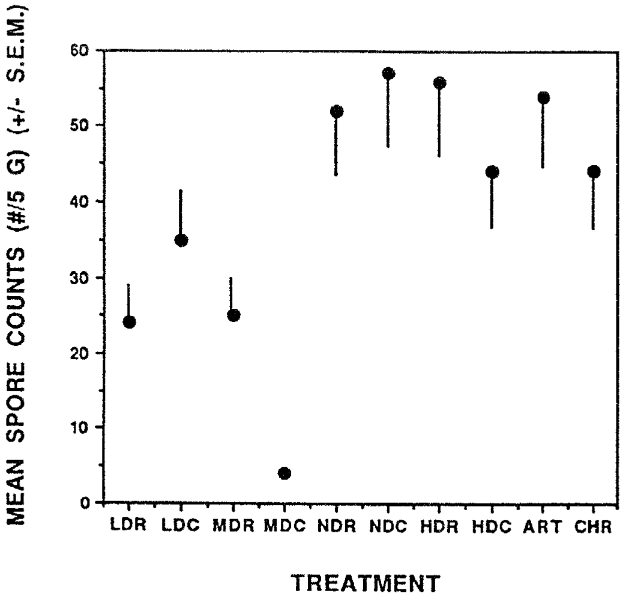

Figure 6.—Mean spore counts from samples taken at regular intervals (regardless of the plant location) in plots with the differing planting treatments (see text for treatment details). The data are from 1987 counts (S.E.M.=standard error of the mean).

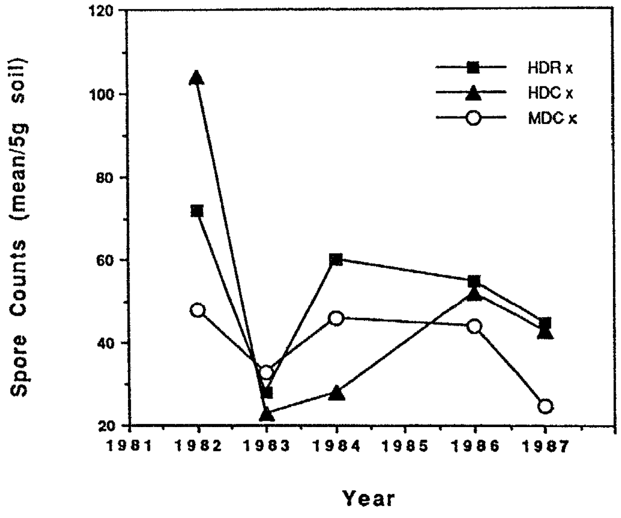

Figure 7.—Mean density of spores from adjacent plots with differing plant densities using samples taken only in association with individual plants. Shown are the mean spore counts from a plot with planting of a moderate density (clumped), a high density regular, and a high density clumped planting. These represent data from treatments showing the greatest extremes in mean spore counts (Fig. 6.).

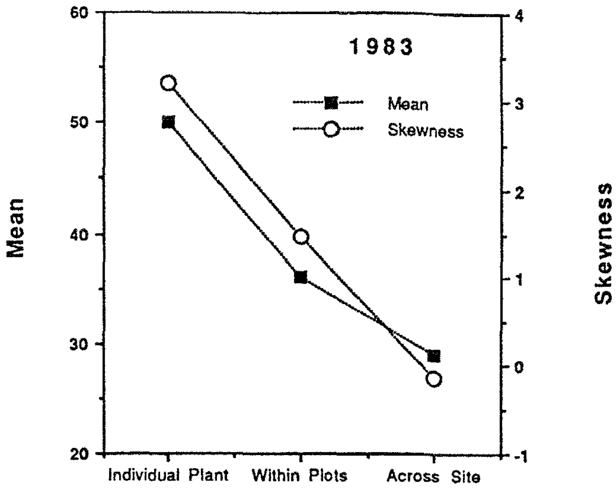

Figure 8.—Mean and skewness of VA mycorrhizal fungal spore counts in 1983 comparing samples taken among individual plants, within a single plot at regular intervals, and at regular intervals among plots with the same treatment across the site. All samples were taken using the medium density treatments (see text for details.

These data suggest that the major factor affecting the fungal reproduction on a plot basis is the total density of plant patches placed on the plot.

No clear pattern of spatial variation versus planting treatment emerged either at the scale of the whole plot or at the scale of the individual plants. However, when we contrasted the spore count mean and distribution of individual plants with the larger units, an interesting pattern emerged. In 1983, the year following planting, both the means and the skewness (degree of clumping of values) were highest at the scale of the individual plant and lowest at the whole site scale (Fig. 8). By 1987, the mean values tended to be greatest at the individual plant scale but the skewness increased (Fig. 9). These data suggest that both the mean reproduction of the fungus and the distribution in space change with successional time. We also suggest that habitat heterogeneity may be important as the patch size appears to be localized around individual plants and different plants represent different habitats.

Finally, we were interested in the potential for changes in the fungal species composition with time. Is there any predictability to changes in species composition? Is the species composition simply a product of initial treatment or are the species richness and composition simply random? We hypothesized that island biogeography theory may provide a model for understanding aspects of the reestablishment of mycorrhizal fungi. To this end, we looked at changes in spore density and the VA mycorrhizal fungal species richness in 1987. Interestingly, both VA mycorrhizal fungal species richness and total spore numbers were related to the size of the shrub "islands" (Fig. 10).

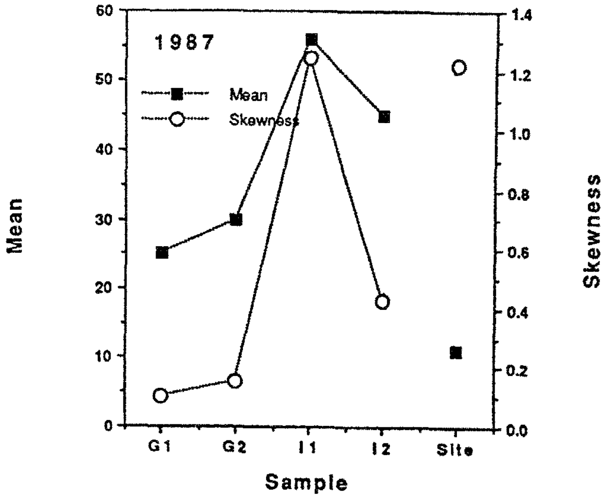

Figure 9.—Mean and skewness of VA mycorrhizal fungal spore counts from 1987 comparing samples taken among individual plants (G1, G2, I1, I2) and at regular intervals among the plots with the same treatment across the site. All sample were taken using the medium density treatments (see text for details).

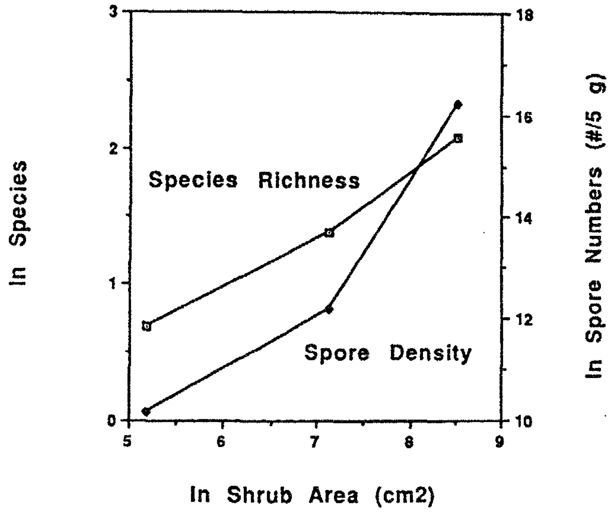

Figure 10.—Relationship between the size of shrub island and the density and richness of VA mycorrhizal fungal species using a natural log scale.

Recommendations for Assessment of Mycorrhizae by Land Managers

In conclusion, after wading through the research jargon, how should a land manager determine mycorrhizal activity in the context of assessing reclamation success? We suggest that a few simple observations can not only help the reclamation officer but can also contribute in a major way to our scientific data base. This will enable us to make better recommendations to individual reclamation officers and to regulatory agencies. We suggest that the reclamation officers conduct simple surveys of mycorrhizal activity in conjunction with their assessments of plant parameters. This would include two parts. First, a survey of percent infected root segments of the dominant plants should be made. This would indicate whether or not mycorrhizal activity was present. This would entail no more than 5 plants of each of the dominant species from each major community on the reclamation site. Second, some measure of reproduction should be undertaken. This would require a survey of fruiting structures of the fungi themselves. Density estimates and diversity of sporocarps of ectomycorrhizal fungi (mushrooms) should be made as plants are assessed. Spores of VA mycorrhizal fungi should be extracted, identified (if the expertise is available) and counted. These surveys should be conducted at two scales, within a group of individual plants within each community type, and at regular intervals across the site. This would not necessarily need to be a large sample size, just enough to establish that the fungi are reproducing and that desired associations with the plants of interest are occurring. These data should be shared among reclamation researchers, officers, and regulators to understand better the dynamics of reestablishment of an association critical to the success of reclamation efforts. Continued cooperation among all concerned will better enable us to attain the common goal of reclaiming the structure and functioning of desired communities following the extraction of important resources.

Acknowledgments

We acknowledge the helpful comments of Edith Allen on the manuscript. And technical assistance by Dave Ianson, Dan Duce, Wendy Jones, and Karen Leunk. This research was supported by grants from the National Science Foundation (BSR 8807342 and BSR 8818076) and USDA Competitive Grants Program (Environmental Stress Program 88-37264-4026).

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Nutrient Cycling: The Key to Reclamation Success?

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Abstract

The results of research into nutrient cycling on reclaimed land has enabled the control of nutrient input and species content during reclamation schemes to provide for faster nutrient cycling leading to improvements in ground cover, erosion control and appearance. However, the ecological value of abandoned mine lands where nutrient input is low and cycling very slow is often greater than that of many reclamation schemes. Research has indicated that a "capital" of nutrients is required in reclamation schemes for an ecosystem to be self sustaining and that intermittent inputs of nitrogen often lead to moribund vegetation with nutrients immobilized. Also small but frequent pulses of nutrients as observed in legume -supported swards or others where nutrient cycling is not obstructed can support effective reclamation schemes of some ecological value. These findings are considered in the context of the manipulation of nutrient cycling in order to enhance the ecological value of reclaimed sites.

Introduction

The supply of nutrients has been recognized as an important factor governing the establishment of vegetation on despoiled land since researchers began investigating vegetation establishment on mined-out areas in the early part of this century (McDougall 1918; Croxton 1928). Even before such research work began, enlightened mine owners were using techniques such as spreading used animal bedding on spoil as a mulch to encourage the establishment of nutrient cycling in some early reclamation attempts (Richardson 1984). It is not surprising therefore that much research into mined land reclamation during this century has concentrated on ways of establishing and accelerating nutrient cycling. Much of this research has concentrated on nitrogen supply with a fair body of work also on phosphate supply. Because of the scale of coal mining, probably more research has been carried out on reclamation after the mining of coal than on all other types of mining combined.

The primary emphasis of the reclamation of mined out land this century has been to put the land to beneficial use. Other important requirements have been the removal of visual scars, the prevention of erosion and the prevention of the contamination of water courses or land. The requirement to put the land to "beneficial use" has meant that much of the research carried out has been related to nutrient input and cycling to maximize herbage and tree production levels. Reclamation success has been measured in terms of productivity comparisons with similarly cropped unmined land. However in more recent times, measures such as the Surface Mining and Control and Reclamation Act (SMCRA) in the USA have done much to raise the level of importance of the consideration of ecological parameters in reclamation (Brenner 1984). In the United Kingdom such considerations have also become an important factor in land reclamation and rehabilitation.

This paper aims to summarize what has been learned from research into nutrient cycling in reclaimed land and to relate this to the concept of ecological success.

Types of Despoiled Land

There are a wide variety of ways in which land becomes despoiled and the characteristics of despoiled land vary considerably (Fig. 1). Many of the constraints on the development of vegetation on such land interact and on some types of despoiled land one factor, toxicity, for example, may dominate all other considerations. Nutrient deficiency is however a characteristic of most types of despoiled land and results from either a lack of one or more of the major nutrients or an impediment to the cycling and supply of one or more of those nutrients. In this paper three types of derelict land will be used as examples. These are colliery spoil, china clay waste, and spoil from metalliferous mining. The characteristics of these spoils are presented in Table 1.

Nutrients Cycling

A major limiting factor on despoiled land is lack of nitrogen and discussion here will be primarily concerned with nitrogen supply. A nitrogen cycle which could be applied to most types of reclaimed land is presented in Figure 2. In nitrogen deficient systems, cycling is at a low level because inputs of nitrogen are low and are also often impeded because nitrogen is bound up in compounds of high-C:N ratio or immobilized in microbial biomass. In such situations the C:N ratio in organic matter is too high for microbes to decompose the organic matter and any available nitrogen arising is quickly further taken up and immobilized by microbes. This interruption of cycling is also often assisted by factors affecting microbial action such as low pH. Mineralization and nitrification are the most common places of interruption of the nitrogen cycle on reclaimed land (Fig. 2). The effect of this interruption has been shown in the United Kingdom by Williams (1975) and in the United States by Reeder and Berg (1977). Some results of William's experiments are summarized in Figure 3 and illustrate well how substrate availability in terms of ammonium and fluctuations in pH as affected by lime additions influence nitrification, a microbially controlled process. The message is clear: for nitrogen to be supplied to vegetation there has to be a steady supply of ammonium to be nitrified and the pH has to be high enough for that nitrification to take place.

	Stability	Combustion	Slope Angle	Flooding. Stress	Toxicity	Compaction	Temperature	Wind Erosion	Nutrients	Stoniness	Uneven Surface	Erosion	Soil Fauna and Microbes
Colliery Spoil	●	●	●	●	●	●	●	●	●	●	●	●	●
Smelter Slag	●	●	●	●	●	—	●	●	●	●	●	●	●
Slate/Shale	●	—	●	●	—	—	●	●	●	●	●	●	●
Metal Wastes	●	—	●	—	●	●	●	●	●	●	●	●	●
Quarry Pits	—	—	●	●	—	●	●	●	●	●	●	●	●
Brick Pits	—	—	●	●	—	●	●	●	●	●	●	●	●
Peatland	—	●	—	●	●	—	●	●	●	—	—	●	●
China Clay	●	—	●	●	—	—	●	●	●	—	—	●	●
Ironstone	●	—	●	●	—	●	●	●	●	—	●	●	●
Chemical Waste	●	—	—	—	●	●	●	●	●	—	—	●	●
PFA	—	—	—	●	●	●	●	●	●	—	—	●	●
Sand and Gravel	—	—	●	●	—	●	●	●	●	—	●	●	●
Domestic Refuse	—	●	—	●	●	●	—	●	—	—	●	—	—

Figure 1.—Constraints on vegetation development on different wastes (after Kent 1982)

Table 1.—Characteristics of colliery spoil, china clay waste and metalliferous mine spoil

Item	Colliery spoil	China clay waste	Metalliferous mine waste
pH	<3 to >8	<4 to 6	<3 to 8
Physical properties	Weathering silt and mudstones, high clay fraction	Mica (silt) or sand waste, the latter free draining	Clay to gravel size, thixotropic to free draining
N	Nitrogen deficient	Nitrogen deficient	Nitrogen deficient
P	Phosphate deficient, fixes phosphate	Phosphate deficient, easily leached out	Phosphate deficient
K	Adequate supply	Deficient, easily leached out	Deficient
Potential toxicities	Al and Zn at low pH, salinity on some sites	None	Heavy metals
Moisture	Waterlogged in winter, potentially droughty in summer	Sand waste: low moisture-holding capacity; mica waste: less of a problem	Sands and gravels have low moisture-holding capacity; silts retain moisture
Erodibility	Easily eroded	Easily eroded	Easily eroded

The ammonium has to come from either organic matter or external sources such as fertilizer and there have been many studies to show that intermittent applications of nitrogen on nutrient deficient colliery spoil sites can encourage the accumulation of material of a wide C:N ratio which then acts as a nitrogen sink. In such situations nitrogen supply to vegetation is only adequate shortly after fertilizer application (Wieder et al. 1983; Palmer et al. 1986).

A way of boosting a supply of organic matter rich in nitrogen is by the encouragement of a legume with an active nitrogen symbiont. It has been suggested by work in the UK on the colonization and reclamation of china clay waste that a legume component is an important factor in successional stages leading to a "self sustaining ecosystem" (Dancer et al. 1977). Studies in natural succession after the retreat of glaciers at Glacier Bay Alaska have also reinforced the importance of a nitrogen-fixing component in succession (Crocker and Major 1955). On china clay waste the amount of nitrogen accumulated under vegetation of various types and ages was compared to that of natural ecosystems (Table 2) and it was concluded that a level of nitrogen in the soil of 700 kg/ha or in the soil and vegetation

Table 2.—Comparison of nitrogen content and compartmentation between major ecosystem pools in china clay wastes and semi-natural temperate ecosystems (from Marrs and Bradshaw 1982)

Ecosystem	Total N	Shoots (%)	Roots (%)	Litter (%)	Soil (%)	Sampling depth (cm)	Reference
	kg/ha					cm	
Naturally colonized china clay wastes							
Pioneer (Lupinus arboreus)	291	37	1	7	56	21	
Pioneer (Calluna vulgaris/Ulex europaeus)	823	13	5	3	79	21	
Intermediate (Salix atrocinerea)	981	8	18	6	68	21	Marrs et al. (1981)
Woodland (Betula pendula/ Rhododendron ponticum/Quercus robur)	1770	30	3	0	67	21	
Reclaimed china clay wastes							
Sand tips	211	11	59	nd	30	21	
Mica dam walls	441	8	61	nd	31	21	Roberts et al. (1981)
Semi-natural temperate ecosystems							
Heathland	5644	combined shoot root=3		1	96	20	Robertson and Davies (1965)
Grassland	10460	0.7	0.5	0.3	98.5	30	Perkins (1978)
Oak woodland	7940	combined shoot root=4.9		0.9	94.2	30	Duvigneaud and Denaeyer-de Smet (1973)

of 1,000 kg ha^{-1} was necessary for the ecosystem to be self sustaining (Roberts et al. 1981).

Work on colliery spoil has not been able to quantify the accumulation of nitrogen in relation to ecosystem development in the same way as on china clay waste. However, the importance of legumes, not only in accumulating nitrogen but also in providing a steady supply of nitrogen throughout the growing season, has been demonstrated on colliery spoil (Palmer and Chadwick 1985; Palmer et al. 1986). Despite these findings and the many records of naturally occurring legumes on colliery spoil, there is little evidence in the literature to suggest that legumes are at a competitive advantage or are contributing significantly to the nitrogen input to the soil in situations where legumes are naturally colonizing a despoiled site (Palmer 1984). It is on sown, reclaimed land that the real advantages of legumes have been found. This has undoubtedly been due to the fact that these sites have often received phosphate applications and have been limed. Adequate levels of phosphate are necessary of maximize nitrogen fixation and for the accumulation of nitrogen in legume-supported swards (Palmer and Iverson 1983).

Metalliferous mine sites are nitrogen deficient also but the overriding limitation on plant growth is due to toxic metals. Nevertheless, the principles governing nutrient cycling still apply. The metals affect not only higher plants but also the microbes responsible for N-cycling and vegetation establishment can be improved by the addition of fertilizer and organic matter (Winterhalder 1988). Two mechanisms seem to operate. One is that at higher nutrient levels there is an increased ability of the plant to cope with stress due to toxicity and the other is that the chelation of metals by organic matter reduces the availability of these metals for uptake by plants.

Ecological Considerations

It is not the purpose of this paper to discuss at length how ecological success in reclamation might be evaluated but it is worth pointing out how ecological considerations might be treated in land reclamation. There are perhaps two

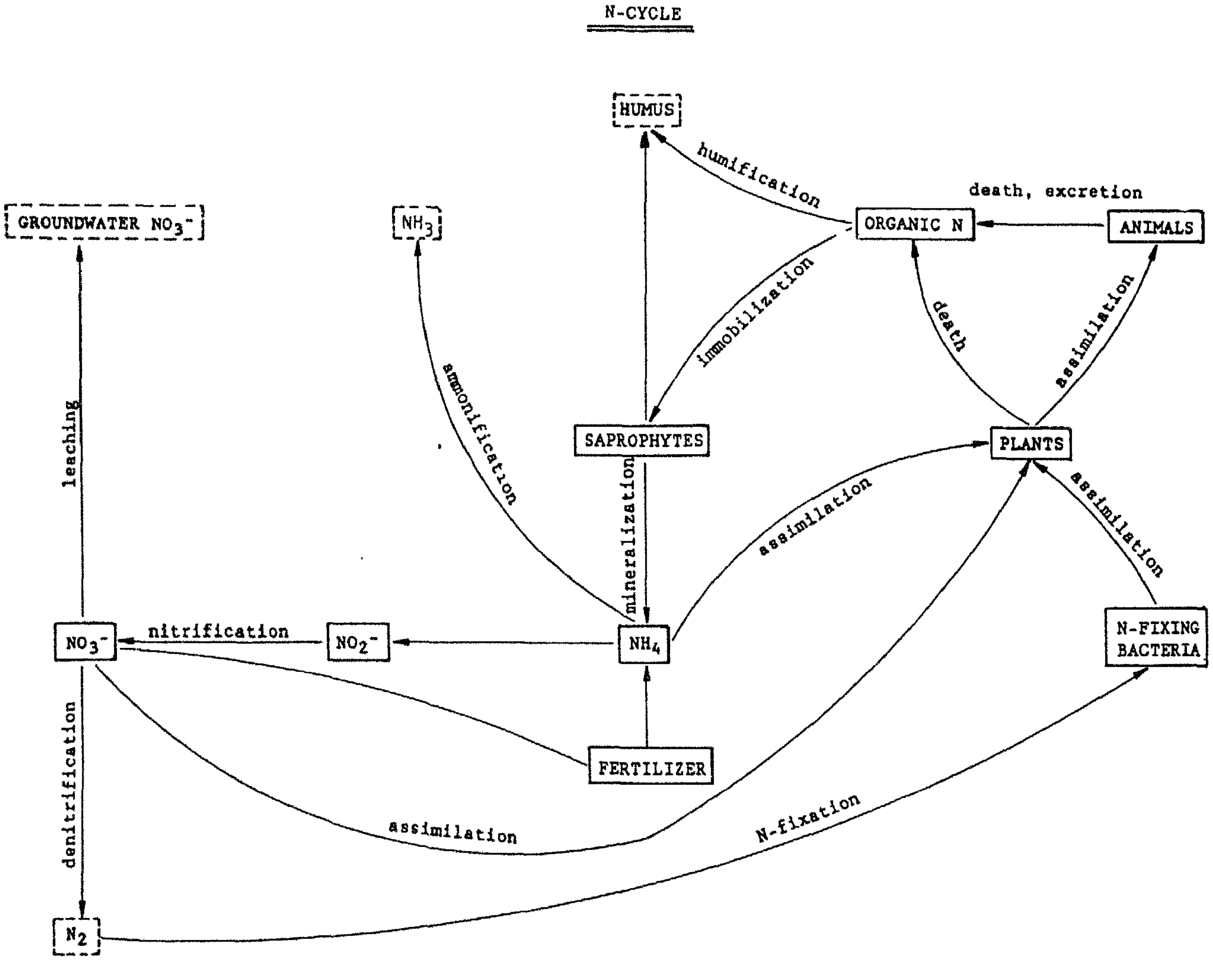

Figure 2.—A nitrogen cycle applicable to reclaimed land. Decomposition, mineralization and nitrification are processes likely to be impeded on reclaimed land.

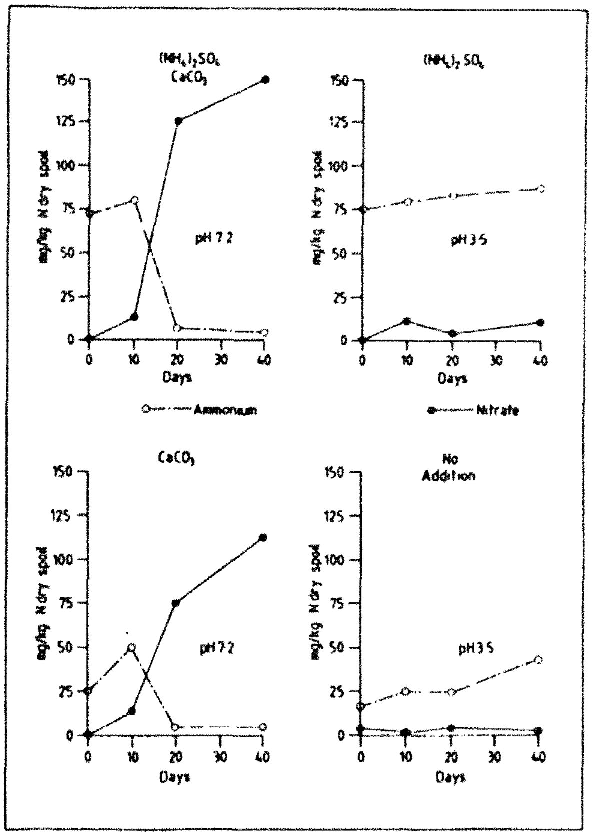

Figure 3.—Nitrogen transformations on reclaimed colliery spoil incubated at 25°C and 10% moisture for 40 days with additions of ammonium sulphate, calcium carbonate or no additions. (from Williams 1975).

distinct ways in which this may be done. The first is where land of some ecological merit is being destroyed as part of, say, a mineral extraction operation and land reclamation has to restore the land to the same ecological status and type. Here the requirements of land reclamation in terms of vegetation composition and type and even nutrient cycling can be quite well defined and success can be measured by assessing similarity with undisturbed land of a similar type to that disturbed. In the United Kingdom where land of some ecological value is to be destroyed as part of a mining or other activity, a technique becoming more common to ensure restoration of ecological value is to move the vegetation and soil in blocks to either a holding area to be replanted as part of the reclamation scheme later or to a different area of less ecological value. This has been successful for areas of limestone grassland (Park 1985) and is even being attempted for the ground flora of ancient woodland. In this type of reclamation, if effective, nutrient cycling processes are transferred with the soil, vegetation, and management.

The second way in which ecological considerations may be taken into account is where the opportunity is taken during reclamation to increase the "ecological value" of the site. For the purposes of assessing ecological value the factors listed in Table 3 have been considered to be important. The concept of diversity has of course in been embodied in United States reclamation legislation since SMCRA and is important in the United Kingdom too. However in the United Kingdom the concept of "naturalness" is currently being given more weight than previously. Although most of the vegetation of the United Kingdom is semi-natural and has been disturbed by man, the extent to which the vegetation conforms to recognized natural communities and the lack of recent gross disturbance are considered important. Even with grossly disturbed sites, such as mined-out land, lack of recent intervention by man allows colonization and ecosystem development to take place naturally. This development, in response to the disturbed environment, can lead both to communities which may fit into recognized natural community types and those which are modifications of natural types. Such communities are nevertheless the result of the natural processes of colonization and be considered important because of this.

In order to consider the influence of nutrient cycling where ecological value is a goal of reclamation it is instructive to consider what has happened on naturally vegetated mined out sites. It is a paradox facing the reclamation practitioner that in order to "reclaim" a site one is often faced with the

Table 3.—Criteria used in wildlife conservation evaluation in nine studies reviewed by Margules and Usher (1981) as adapted from Usher (1986)

Criterion	Frequency of use in nine published studies
Diversity (both of species and habitats)[a]	8
Naturalness[a]	7
Rarity (both of species and habitat)[a]	
Area (extent of habitat)[a]	6
Threat of human interference	
Amenity value	
Education value	3
Recorded history[a]	
Representatives	
Scientific value	2
Typicalness[a]	
Uniqueness	
Availability	
Ecological fragility[a]	
Management considerations	
Position in ecological/geographic unit[a]	1
Potential value[a]	
Replaceability	
Wildlife reservoir potential	

[a]Criteria used by the Nature Conservancy Council for assessing sites in the United Kingdom.

destruction of spoil heaps of some considerable wildlife value. Many sites abandoned to be colonized naturally are of some ecological interest. This often the case if the waste is alkaline or has a high heavy metal content. In the United Kingdom many of these sites are protected because of their ecological interest (Table 4). It is also the case with some sites which are low in nutrients such as colliery spoil sites where the naturalness of the sites is considered of value. Whether these sites have special characteristics (e.g., heavy metals and alkalinity) or not, they have all developed their vegetation and ecological value under very low inputs of nutrients.

In the United Kingdom the annual input of nitrogen from rainfall would be of the order of 9 to 10 kg/ha in industrialized areas. Phosphate levels would also be low, thus reducing the chance of large inputs of nitrogen fixed by legumes. Under all of these systems cycling of nutrients would have been at a low level and slow, limited by factors such as substrate nitrogen required to allow the sward to develop but at which the growth of productive species is not too great. Another important trigger is pH. Chadwick and Hardiman (1976) in the United Kingdom and Glenn-Lewin (1979) in North America have both shown that species number on colliery spoil heap is correlated with pH rather than age of the spoil. Lime applications to raise pH would also act as a trigger to facilitate colonization. Facilitating the natural development of vegetation in this way while at the same time meeting reclamation goals from other stand-

Table 4.—Examples of disused mineral extraction sites in Britain which have acquired conservation value status on botanical grounds (from Johnson 1978)

Location	Origin	Distinguished botanical features
Brickfields, Buckinghamshire[a]	Brick-clay extraction	Blackstonia perfoliata, Lotus tenuis, Ophrys apifera
Sanford, Berkshire[a]	Sand extraction	Dactylorhiza incarnata, Equisetum variegatum, Epipactis palustris, Potamogeton coloratus
Aston Clinton, Buckinghamshire[a]	Chalk ragstone quarrying	Hippocrepis comosa, Ophioglossum vulgatum, Polygala calcurea, various Orchidaceae
Hurley, Berkshire[a]	Chalk quarrying	Anacamptis pyramidalis, Gymnadenia conopsea, Ophrys apifera
Millers Dale, Derbyshire[b,c]	Limestone quarrying/ calcining	Botrychium lunaria, Daphne mezerem, Parnassia palustris, mezerum Saxifraga tridactylites, various Orchidaceae
Wingate, Durham[b]	Magnesian limestone quarrying	Botrychium lunaria, Crepis mollis, Gentinella amarella, Salix nigricans, various Orchidaceae
Honister Crag, Cumbria[d]	Slate quarrying	Alchemilla alpina, Saxifraga aizoides, Oxyria digyna, Sedum rosea

[a]Kelcey (1975).
[b]Davis (1976).
[c]Bradshaw (1977).
[d]Ratcliffe (1974).

points such as erosion control and visual appeal would seem to be a valid way of achieving ecological success. In the reestablishment of heathland, a vegetation type which normally receives only atmospheric nutrient inputs, the use of low levels of fertilizer to accelerate establishment has been found to be beneficial (Putwain et al. 1982). Alternatives to allowing development of ecologically valuable swards, such as wholesale transplantation of ecosystems ahead of mining, are valid but costly. The reclamation practitioner should not overlook the possibility of leaving some of the reclaimed area unsown and unfertilized in order to facilitate natural colonization.

This review has concentrated primarily on nutrient cycling in spoil materials. Where a very fertile topsoil is used in reclamation, other kg N/ha in the soil suggested by china clay reclamation research may be an achievable goal. At the same time, the rate of mineralization of nitrogen must be low enough to allow a diverse sward to develop, but not too low to prevent ingress of new species. Where this level should be is not clear from research so far but previously unpublished data on the ingress of unsown species into trial plots on colliery spoil provides some clues. These plots were established by the Derelict Land Reclamation Research Unit, University of York , at Thorne Colliery in the United Kingdom. The plots had received phosphate fertilizer but no nitrogen and had been sown with a legume. Nitrogen fixation was initially at a very low level and the level of mineral nitrogen in the soil remained extremely low for three years. The sward was largely moribund during this time with bare ground between plants. In the third year after sowing, nitrogen fixation was much higher, legumes grew vigorously, soil mineralizable nitrogen levels increased and the sward closed up. This was accompanied by an increase in the number of unsown species establishing in the plots (Fig. 4). The level of mineralizable nitrogen was between 10 and 40 mg/kg when the number of unsown species entering the plot increased (Fig. 5).

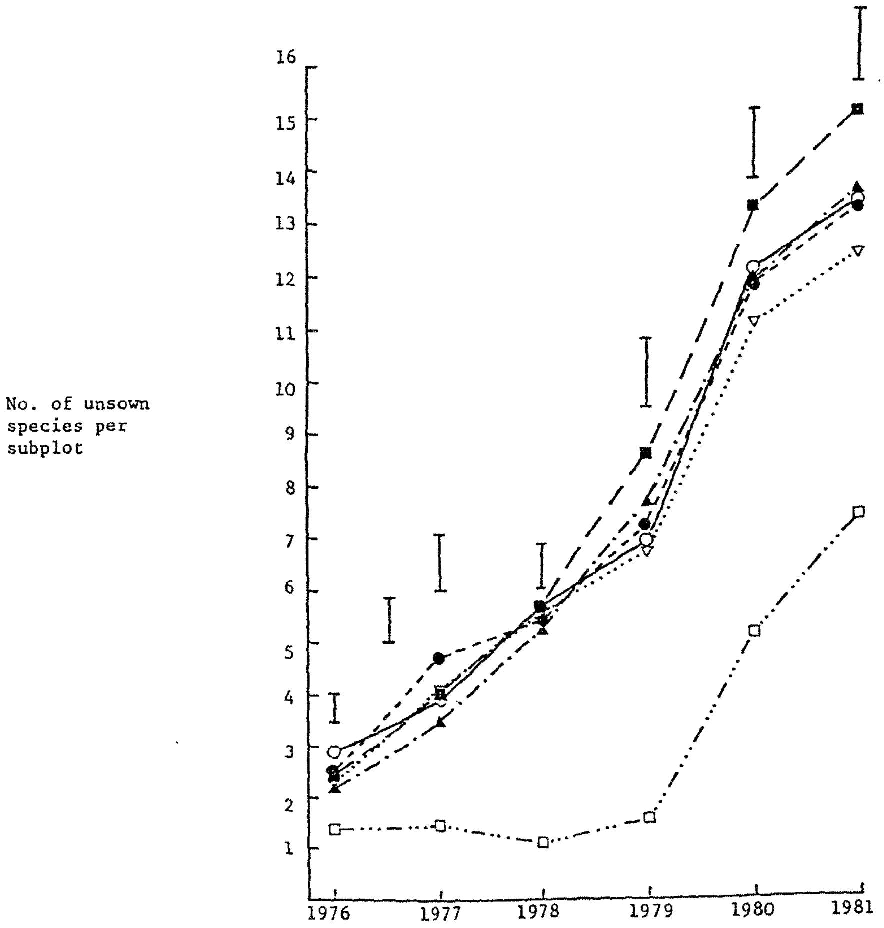

Figure 4.—The ingress of unsown species into legume and fertilizer plots at Thorne Colliery, South Yorkshire, UK between 1975 and 1981. Symbols: □ legume plots, ▲, ▼, ■, ○, ●, various forms of applied nitrogen fertilizer. Least significant differences (p 0.05) for each year are shown.

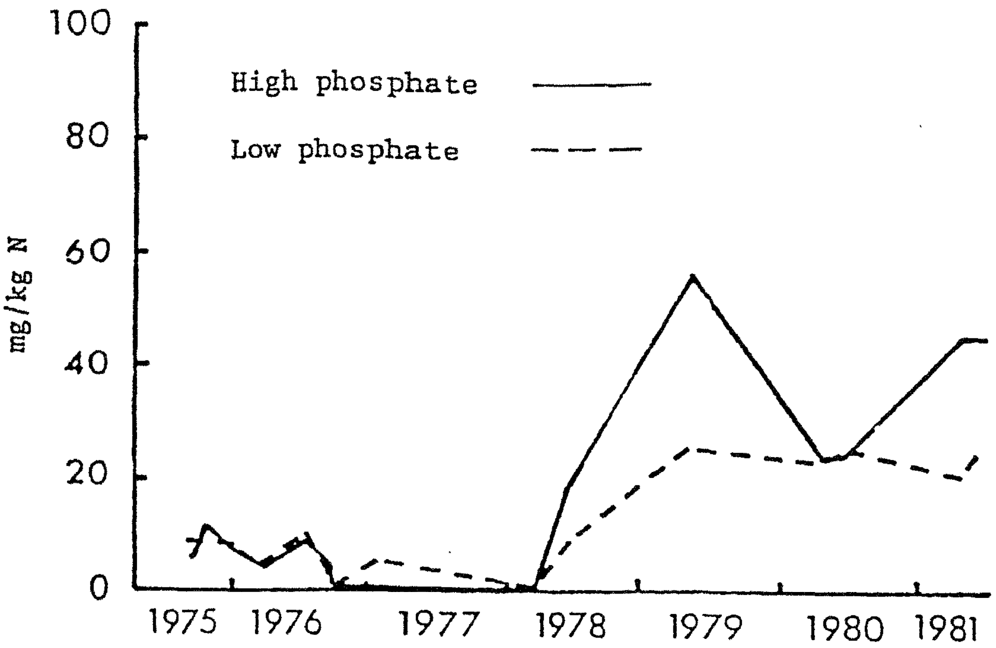

Figure 5.—Mineralizable nitrogen levels in legume plots at two different phosphate levels at Thorne Colliery, South Yorkshire, UK between 1975 and 1981.

Such data indicate that there is a "trigger level" of mineralizable availability and pH already discussed. Nutrients would accumulate slowly often bound up in organic matter with thin humic layers developed at the surface. At one site in Wales, for example, after 60 years a closed sheep-grazed sward had developed on colliery spoil. An organic layer of about 50 mm had developed above the spoil. In harsher environments, due to factors such as toxicity or climatic extremes, the development of such a layer would take much longer. Metalliferous mines in the same area of Wales had virtually no vegetation or organic layer development after similar periods of abandonment. They were still of ecological value however because of their lower plant flora. In this context the ecological value of bare slopes and open communities must also not be overlooked in reclamation scheme design.

Although ecologically valuable sites have developed with very small inputs of nutrients on abandoned mine lands, the reclamation practitioner cannot wait the length of time required for new sites to be colonized in this way.

On many sites a balance has to be struck between obtaining a vegetation cover which will be visually acceptable and performing an erosion control function but which will develop in an ecologically valuable way. For example, difficulties may be encountered in restoring land of high ecological value because the high nutrient status of replaced topsoil will encourage productive species in the sward at the expense of species diversity. In such situations a cropping regime may have to be practiced which will over a period of years remove nutrients from the system and allow a more diverse sward to develop. Experiments using this technique on grasslands which have begun to lose diversity due to changes in management have met with some success in the United Kingdom (Marrs 1985). The management problems associated with over-fertile soils and productive species can, however, be prevented by avoiding the use of these soils at the time of land reclamation. The successful management of nutrient cycling, whatever the nature of the substrate, must be based on a proper assessment of the subtrate and the tailoring of the land reclamation techniques to substrate characteristics and reclamation goals.

Conclusions

1. Mine spoils are often low in the major nutrients. Nitrogen is a major limiter of plant growth and the nitrogen cycle may be interrupted by microbial substrate availability, low pH, or toxicity.
2. Additions of high levels of fertilizer lead to the promotion of productive species in the sward. If these additions are intermittent, then moribund swards of high C:N ratio often result.
3. Legume-based swards can provide a steady but low level of nitrogen input which will give a continuous input of mineral nitrogen throughout the growing season and thus avoid accumulation of moribund productive grasses.
4. Work on china clay waste reclamation has suggested that a capital of 700 kg N/ha is needed in the soil before the ecosystem becomes self-sustaining.
5. Naturally colonized spoil can even with very low nutrient inputs result in ecologically valuable vegetation after many years.

6. On colliery spoil a trigger of a low level of mineralizable nitrogen is needed to increase species diversity by ingression of unsown species. Between 20 and 40 kg mineralizable N/ha appears to be the level at which this occurs.
7. Ecologically valuable swards are most likely to be achieved and sustained by providing a low level of mineralizable nitrogen throughout the season. This is best obtained by the use of a legume which in itself necessitates an input of phosphate.
8. Where the soil material is rich in nutrients care should be taken with species choice and management in order to achieve a diverse sward.
9. Opportunities should not be missed during mining and reclamation schemes to consider transplanting habitats and leaving areas unsown to colonize naturally.

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Population-Level Processes: Seed and Seedbed Ecology

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Abstract

Seeds play an important part in the vegetation of disturbed areas through germination from natural or artificial seedbanks, natural seed dispersal, or artificial seeding. The density and species composition of the plant communities thatt colonize disturbed areas depend on the inherent physiological systems of the seeds and potential of the seedbed to support germination. Inherent physiology can be modified by pretreatment such as seed scarification of species with indurate seed coats. The potential of the seedbed can be modified by seedbed preparation and seeding techniques.

Introduction

The most economical method of establishing vegetation cover on an area that has been drastically disturbed is usually through planting seeds, seed germination, and subsequent seedling establishment. This process can be the result of natural succession, germination from conserved topsoil that was reapplied to a disturbed site, or through artificial planting. The success of the revegetation process depends on the inherent potential of the physiological systems of the seeds that are naturally dispersed across the site or artificially placed in the seedbed, and equally important, the potential of the seedbed to support germination.

The concept of the seedbed having varying potentials to support germination may seem odd, but it is the basis for most farming practices designed to assure adequate stands. Tillage practices such as plowing may be initially done for seed control, but subsequent operations such as harrowing and cultipacking are primarily designed to make certain seeds are planted or drilled at the proper depth, covered with soil, and adequate contact between soil particles and the seed coat is established. The interesting aspect of seedbed potential is how we can manipulate the beds to make them a selective filter to govern the species composition of post-disturbance communities.

It is perhaps easier to contemplate that seeds vary in their potential to germinate in a specific seedbed. We can manipulate such differences by selecting specific cultivars to be included in revegetation mixtures. It is also possible to change the germination characteristics of seeds by physical, chemical, or physiological manipulations done before the seeds are planted.

The purpose of this synthesis paper is to draw together various aspects of seed and seedbed ecology that have practical applications to the revegetation of disturbed environments.

Seedbanks

Seedbanks refer to the number of viable seeds per volume of surface soils. In agronomic soils, the seeds in the seedbank may be distributed through the tillage depth. In wildland situations where tillage is not practiced the bulk of viable seeds may be located in the litter and duff, on or very close to the soil surface. Fine textured soils that crack will have the cracks filled with seeds. There is always some seed burial by small mammals and insects. In temperate desert environments seed collection and caching in scatter hoards by Heteromyid rodents is a very significant component of seedbank enrichment (Longland 1987). At the same time, predation of seeds in seedbanks may be a major source of seed loss (Louda 1989). Microscopic seeds may find their way to considerable depths in the soil through soil pores or by ingestion by soil arthropods (Major and Pyott 1966).

It is very tempting to generate flow diagrams to illustrate the processes involved in seedbanks. An often cited example is that of Harper (1977) or the more recent effort by Simpson and others (1989). I admit to attempting such diagrams (Young and others 1981; McAdoo and others 1983; Young and Evans 1985). These flow diagrams probably serve a purpose as conceptual models, but unfortunately, they also seem to become absolute. Numerous young scientists are in the process of quantifying and assigning probability levels to the various aspects of specific models for seedbanks (e.g., William Longland and *Oryzopsis hymenoides,* unpublished research ARS-USDA, Reno, NV). Diagrams for seedbank functions are available in the literature for reference by readers, but remember they are probably incomplete generalizations that should not curb experimentation.

If you are going to manage and manipulate soil seedbanks as a source of seeds for revegetation of disturbed areas, you are concerned with a rather thin, organically rich layer near the soil surface. Much thicker layers of the soil profile may be stripped and stockpiled for topsoiling disturbed areas, but the primary purpose of such operations is to provide a medium for plant growth.

Seedbank transfer may be incidental to such manipulations. Seedbanks reflect the heterogeneity of the site where they were produced. It may be profitable to selectively collect topsoil from specific sites that have seedbanks rich in seeds of desired species. In degraded big sagebrush *(Artemisia tridentata)* communities, virtually all viable seeds are located beneath the shrub canopies, and the interspace between shrubs is devoid of seeds (Eckert et al. 1989)

Nature of Seeds in Seedbanks

Depending on the season of the year, the seedbank may contain seeds of the current year's seed rain or may be composed entirely of seeds carried over from previous years' production. When dealing with seedbanks as a means of revegetating disturbed areas, one must remember that you can get weeds as well as desirable species. In fact, many weed species have seeds that are adapted to prolonged burial in soil seedbanks and they are triggered to germinate by disturbance. For seeds to remain in the seedbank for long periods, they must either be physiologically dormant or induced to dormancy by being in a position where they cannot germinate.

In a synthesis chapter in the most recent review of seedbanks, Parker et al. (1989) suggested that soil seedbanks are more likely to resemble early successional than late successional vegetation. They cite many examples, but as with any generalizations there are exceptions. In the annual ranges of California where the vegetation must be renewed annually from the seedbank, the seeds of the dominant grasses dominate the seedbank (Young et al. 1981).

Seeds may be physiologically dormant in the seedbank because of hard seedcoats that inhibit moisture or gas diffusion to or from the embryo because of light requirements for germination, or because of insufficient concentrations of chemicals that promote germination (see Baskin and Baskin 1989). In very wet muck soils, lack of oxygen may inhibit germination. When you store soils for later use in topsoil applications it is important to realize that the conditions that promote dormancy may be changed and the dormant seeds may germinate and die before the soil is spread on the disturbed area to be vegetated. Remember that the biologically most active zone of the soil profile, as far as seedbanks are concerned, is the surface, with associated litter and duff. If this organic matter is placed in a position where active decomposition is enhanced, the environment for the seedbank will be dramatically changed. I am not experienced with storing topsoil for future plant establishment of plant communities from seedbanks, but piled or piled and covered topsoil may offer better storage of seeds than spread layers of soil.

Johnson and West (1989) obtained samples from topsoil stockpiles of three different ages on a coal strip mine in order to determine the infuence of time and depth of burial on germination and species composition. The number of viable seeds increased with increased storage time. Differences in predisturbance vegetation and soil depth are likely factors determining the seeds in stockpiled topsoil.

Storage Life of Seedbanks

Some seeds have very short half-lives in the soil and therefore are largely absent from seedbanks. Big sagebrush is an excellent example of this type of seed. In western Nevada, seed of big sagebrush is mature in late November or early December, and by the following June, seeds are largely lost from the seedbank (Table 1) (Young and Evans 1989). Establishing big sagebrush has been a major problem on restored strip mine areas in the American West.

Table 1.—Emergence of seedlings of big sagebrush (*Artemisia tridentata*) from soil samples collected periodically throughout the year at 5 locations; based on means of seedling emergence data collected from 1983 through 1986 (Young and Evans 1989)

		Big sagebrush seedling emergence, sample collection dates							
Location	**Subspecies**	**Dec. 1**	**Jan. 15**	**March 1**	**April 15**	**June 1**	**July 15**	**Sept. 1**	**Oct. 15**
		(no./0.01m^2)							
Granite Peak	Mountain	0	560 ab	300 b	30 b	0	0	0	0
Churchill Canyon #5	Mountain	0	680 a	560 a	180 a	0	0	0	0
Churchill Canyon #1	Basin	0	10 c	0 c	0 b	0	0	0	0
Churchill Canyon #4	Basin	0	120 c	30 c	0 b	0	0	0	0
Medell Flat	Basin	0	480 b	510 a	190 a	0	0	0	0

[a] Means within rows followed by the same letter are not significantly different at the 0.01 level of probability as determined by Duncan's multiple range test. No letters indicate no significant differences.

The concept of donor seedbanks has been proposed for restoring wetlands (van de Valk and Pederson 1989). Many wetland species are very difficult to establish from direct seeding (e.g., George and Young 1977; Cluff and Roundy 1988). Muck from an established wetland is excavated and dumped in the new area. Careful water management is required for this system to be successful.

Ecophysiology of Seeds

A major problem in environmental restoration is that plant communities reproduce in the minds of ecologists and plants reproduce from seeds. Unfortunately seeds do not always cooperate with the efforts of man. For many agronomic species, germination problems have been overcome through selection unless absent some basic physiological block. The seeds of many wildland species have germination systems that have been naturally selected to fit specific environmental situations where they are adapted. If these species are adapted to colonizing highly disturbed areas, there may be no problems, but if the desired species are adapted to some advance successional stage in a woodland, there may be considerable difficulties in getting the species established.

There are two basic self-evident truths in seed ecophysiology: 1) the seed of most species germinate without pretreatment, and 2) the physiological requirements conditioning dormant seeds can often be overcome by more than one method. At maturity, most seeds of plants of the temperate zone dry to about 8 to 10% moisture content and are kept in a dormant state by desiccation until they are allowed to imbibe moisture. This is a very simple system. There are recalcitrant seeds that cannot be dried to low levels of seed moisture content. Members of the genus *Araucaria* usually have recalcitrant seeds. The seeds of numerous species of tropical forests have recalcitrant seeds that have very short half-lives in storage.

The fact that dormant seeds can often be made to germinate by more than one and often dramatically contrasting methods is attributed to the cyclic nature of the biochemical processes that control the physiology of seeds (Roberts 1973). One of the many examples of this is the seeds of the common mullein *(Verbascum thapsus)*. The seeds of common mullein are initially dormant, but the dormancy is easily overcome by exposing imbibed seeds to red light. The seeds can be made to germinate in the dark if incubated at widely fluctuating incubation temperatures (Semenza et al. 1978). There are hundreds of other examples of similar phenomena in seed germination that may lead the casual observer, who is only interested in establishing a vegetation cover on disturbed lands, to suggest that seed physiology is a perfect example of chaos. There are repeated patterns in the entire germination process that bring relative understanding to chaos.

Hard Seed Coats

Indurate seed coats that interfere with moisture imbibition and sometimes the movement of gasses to embryos are a major source of seed dormancy. Members of the legume family often produce seeds with hard seed coats. The high-protein seeds of legumes are the target of numerous seed predators.

In many parts of the world, seed beetles (Bruchidae) infest seeds of the *Acacia* species (Halevy 1974). The insect larvae will render the seeds nonviable if the insect is allowed to complete its life cycle. Large herbivores are attracted to maturing legumes of *Acacia* species in East Africa and they consume large quantities of the fruit. Digestive fluids in the rumen selectively kill the larvae of the bruchid beetle without reducing the viability of the seeds (Lamprey et al. 1974). Digestive juices may also influence the seed coats of the seeds. Domestic livestock consume significant quantities of fallen *Acacia tortilis* seed pods in arid pastoral ecosystems of northwest Kenya (Coughenour and Detling 1986). These seeds are subsequently defecated in corrals where dense populations of this species of *Acacia* arise. Periodic wildfires that burn through *Acacia* stands also selectively influence the bruchid beetle. The larvae of the beetle is killed by lower temperatures than the minimal lethal temperature for seed viability. Again the fire is a double action factor in that it not only kills the insect larvae but also may influence subsequent germination by altering the seed coat. The net effect is that legumes often develop large seedbanks, even in subtropical environments with rapid decomposition.

Using leguminous seedbanks for revegetation of disturbed areas is dependent on triggering the hard seed coats to become permeable. There is no good working hypothesis of how seeds with hard seed coats become permeable in seedbanks. Microbial decomposition, cracking from freezing or diurnal temperature fluctuations, and heat from wildfires have been suggested as natural means of breaking down hard seed coats. The greatest experimental evidence exists for the influence of fires on germination of hard seeded species (e.g., Gratkowski 1962).

Planting seeds with hard seed coats requires several considerations. If a portion of the seeds in a given seed lot has hard seed coats, prolonged germination over more than one season is assured, which may be an advantage in some situations. If the majority of the seeds in a given lot are dormant because of hard seed coats, pretreatment in the form of scarification is required.

Scarification can be accomplished with mechanical, acid, or thermal shock treatments. Virtually all scarification treatments, even when applied correctly, reduce total seed viability and potentially can drastically reduce seed viability. The basic problem with scarification treatments is that the amount of scarification that is required varies among lots of the same species, making preliminary testing of each seedlot advisable.

In developing countries with large seeds of valuable tree species, scarification is often done by hand with a knife or file. Usually it is not necessary to remove the entire seed coat, only to break a small area. Many seeds with hard seed coats have very specific, often small structures in the seed scar area that control moisture imbibition. Mechanical scarification involves rubbing the seeds against an abrasive surface so the seed coat is broken. In some species mechanical scarification results in damage to exposed embryos and loss of viability. The most common treatment is acid scarification with concentrated sulfuric acid. Before you undertake acid seed scarification consider: 1) sulfuric acid is dangerous to handle, 2) prolonged treatment will kill the seeds, 3) acid scarified seeds are often susceptible to soil borne pathogens, and 4) you are creating a toxic waste, used acid, that requires disposal. The heat and chemical reactions associated with acid scarification have ramifications far beyond mere removal of seed coats (e.g., McDonald and Khan 1983). If you have no choice but to proceed with acid scarification, review the classic paper on the subject by Heit (1967).

Hot water treatments are much simpler than mechanical or acid scarification, but unfortunately they are not effective on all species. Bring a volume of water four times the volume of the seeds being treated to a boil and drop in the seeds. Remove the container from the heat and allow the seeds to stand in the cooling water overnight.

When dealing with seeds of species that potentially have hard seed coats, an old nurseryman's trick is to harvest the seeds early, before they are fully dry, and plant the seeds at once, being careful to never allow them to dry. This is effective for a surprising number of species (Young and Young 1990).

Afterripening Versus Prechilling Requirements

This is an often misunderstood portion of seed ecophysiology. I define afterripening requirements as dormancy that is overcome by passage of time after harvest, independent of storage conditions. In contrast, prechilling requirements are a form of dormancy that is overcome by a period of incubation, in an imbibed condition, at temperatures too low to permit germination.

Seeds of Russian thistle *(Salsola australis* R. Br.) provide a good example of afterripening requirements (Fig. 1). In this case, the afterripening requirement is related to incubation temperature, but not to storage temperature (Young and Evans 1972). Environmentally sensitive dormant seeds include seeds with warm-moist pretreatment requirements, or temperature specific incubation requirements that decrease over time. I must warn you that no one else follows these definitions, so you are faced with a bewildering array of dormancy conditions collectively termed afterripening requirements.

Afterripening that is immune to external stimuli requires only patience and it goes away. If you have a revegetation contract that requires plants to be emerged by a certain date and only fresh seed is available, you are in a difficult position. The seeds of some grass species can have the duration of afterripening requirements reduced by heating in forced draft ovens in accelerated afterripening treatments (Taylorson and Brown 1977). Note that by my definition this would not be termed afterripening because the dormancy can be broken by an external stimuli. It has been suggested that my definition is invalid because all cases of apparent afterripening requirements can be manipulated by external stimuli. We just have not discovered the correct treatments.

For species native to temperate regions the most frequent form of dormancy can be overcome by moist prechilling (Fig. 2). Some seed physiologists have even suggested that seeds of certain tropical species have enhanced germination following prechilling, even though the species are native to environments where it never gets cold. If you are not familiar with the term prechilling it is because it recently has come into general use to replace stratificiation. Despite its rich historical roots in northern European forestry, stratification is being discarded for the self descriptive prechilling.

The general rationale for prechilling requirements is that seeds mature in late summer or fall and are dispersed to the seedbed where they ideally lie under snow cover until the spring. The interface between snow and soil usually provides an ideal prechilling environment of near 0°C. Unless you are going to artifically pretreat seeds that have prechilling requirements you are forced to use fall planting. For species such as bitterbrush *(Purshia tridentata)*, where the seeds are subject to predation, fall seeding may lead to large losses of seeds to rodents. The problem with prechilling seeds artificially is that the seeds cannot be dried after the prechilling requirement is satisfied so you are faced with mechanically handling a wet seed. This can be done by gel seeding (Booth 1985).

Warm-Moist Pretreatment

A considerable number of species, especially those native to subhumid and humid regions, produce seeds that have dormancy-breaking requirements which include a period of incubation under warm-moist conditions. For want of better terminology these are still referred to as warm stratification requirements. Without pretreatment of seeds of these species, they have to be planted very early in the fall or emergence will be delayed until a second year after planting. Several species of woody plants native to eastern North America have prechilling requirements that must be preceded by warm-moist incubation. Termed double dormant species, these species require 2 years after natural seed fall to emerge.

Practical revegetation attempts, including these double dormant seeds in mixtures, can lead to problems if

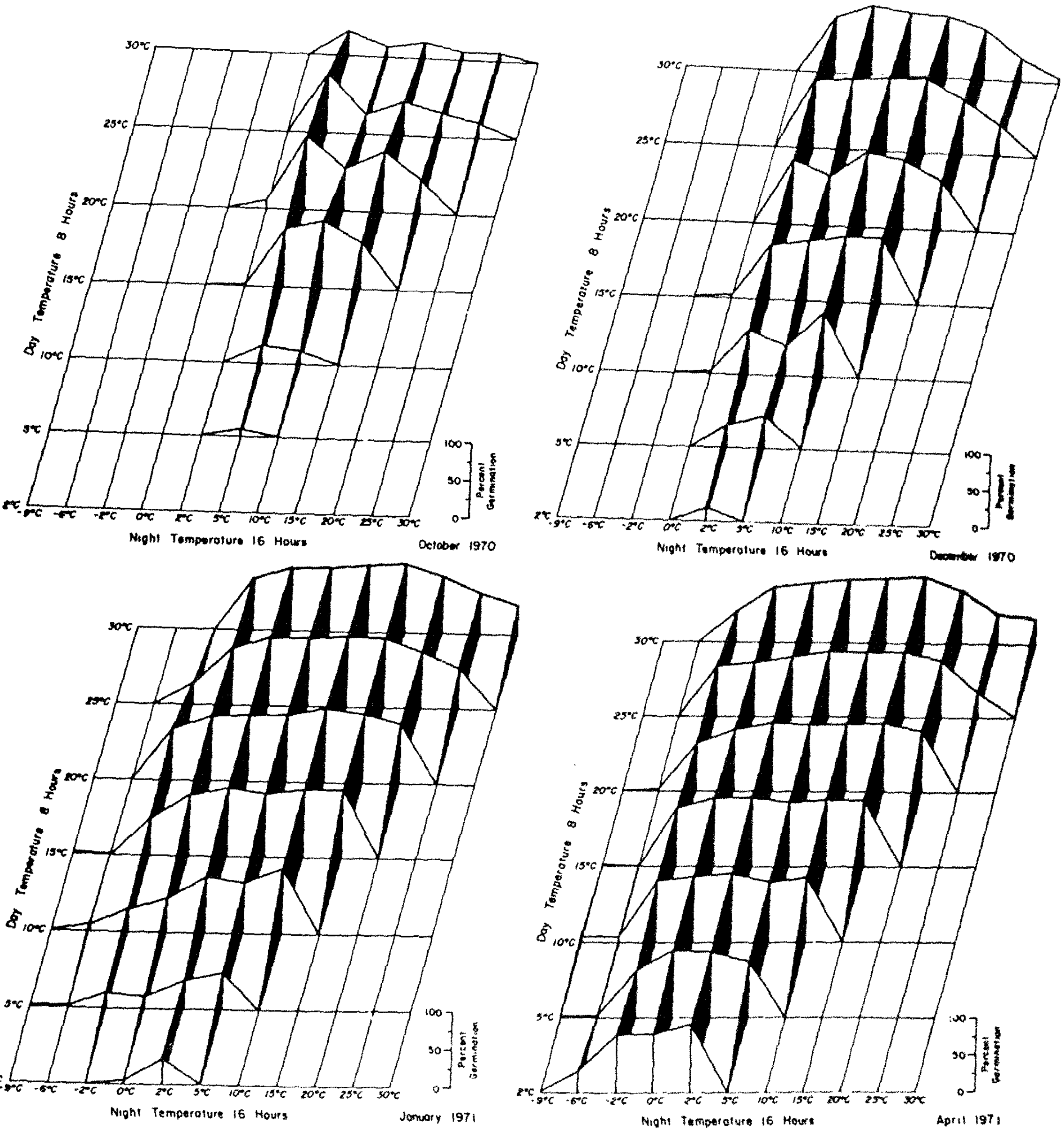

Figure 1.—Germination of *Salsola australis* (*S. iberica*) seeds in relation to temperature related afterripening requirements (Young and Evans 1972).

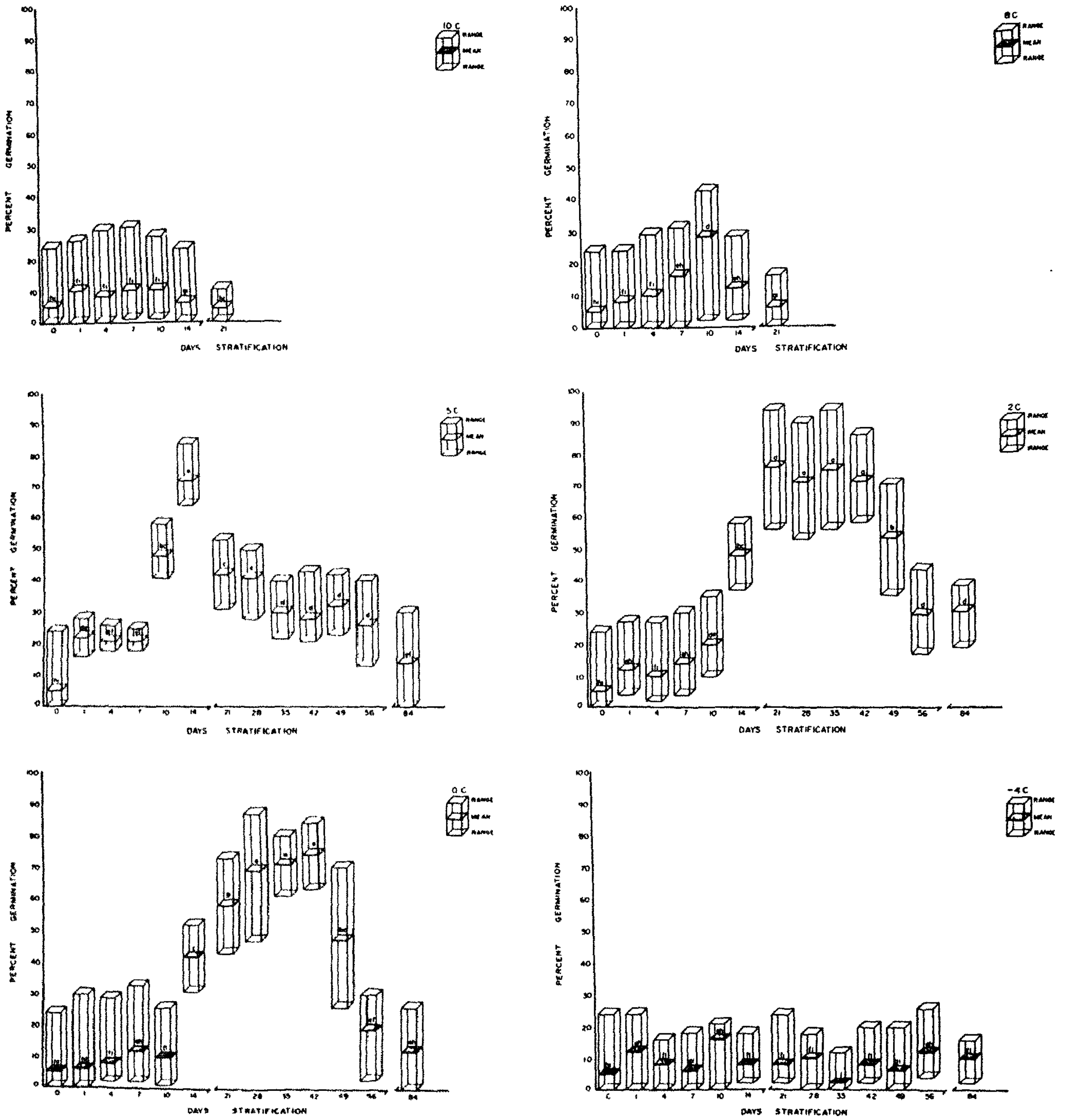

Figure 2.—Germination (mean and range) of six lots of bitterbrush seed (*Purshia tridentata*) incubated at 15°C after stratification at temperature of -4, 0, 2, 5, 8, and 10°C, from 0 through 84 days. No germination was observed in seed stratified longer than 21 days at 8 and 10°C. Means (bars) with the same letter are not significantly different at the 0.01 level of probability as determined by Duncan's multiple range test (Young and Evans 1976).

herbaceous species are successfully established the first season after seeding. By the time the double dormant species germinate, there may be no environmental potential available for them for establishment. It may be more biologically and cost effective to transplant seedlings of nursery grown stock of double dormant species than to attempt to grow such species from seed in a mixture.

Light

Seed germination can be affected by the quality of light, duration of light (photo period), and, rarely, by the presence of light. One of the most spectacular means of inducing dormant seeds to germinate is by exposing imbibed seeds to red light if they are sensitive to phytochrome (e.g., Borthwick et al. 1952). The reversible phytochrome reaction is a very important part of seed ecophysiology. Not all seeds are sensitive to light. Many seeds are neutral to light in their germination response. In 30 years of research in this field I have experienced one species of *Galium* where seed germination was inhibited by light.

It seems to be a general consensus that little light is transmitted through even thin layers of soil. Studies by Kasperbauer and Hunt (1988) indicated that less than 0.005% of the incident light penetrated 4 mm in dark colored soils or 10 mm of gray-white soil. Light requiring seeds of tobacco (*Nicotiana tobacum* L.) used to bioassay for light had 99% germination on the soil surface and no germination when buried 4 mm in the soil. This means that light-sensitive seeds must be exposed on the surface of the seedbed if they are going to have germination stimulated by light. Exposure on the surface of seedbeds can place seeds in very unfavorable moisture conditions. The seed must take up moisture from partial contact with the particles that compose the seedbed while at the same time lose moisture to the atmosphere. If the moisture gradient to the atmosphere is severe, the seeds may never obtain sufficient moisture to germinate. Remember that most seeds germinate without pretreatment to break dormancy, but all seeds are initially held in a dormant state by desiccation. Moisture relations can impose dormancy upon any seed.

In the humid East and in other areas with growing season precipitation, seeds that require light have a good chance of germination during transitory periods of satisfactory moisture conditions. Even in the arid Southwest, several species of perennial grasses require light for germination, but these species germinate and establish during the summer monsoonal rains. In the portion of the American West where precipitation is restricted to winter months and is largely out of phase with temperatures for plant growth, surface germination becomes extremely hazardous. On disturbed sites with soils with low organic matter content, the risk is increased.

When a seed that requires light for germination is placed in a situation where moisture conditions override the phytochrome response and induce dormancy, the seed remains dormant until a secondary physiological system is triggered that takes over control of germination. Suppose our light-requiring seed that lacked sufficient moisture for germination is gradually buried by litter fall and rain drop splash on its harsh seedbed. Burial blocks the operation of the physiologically active phytochrome system, but the seed is still close enough to the soil surface so that it is exposed to extreme diurnal fluctuations in temperature. As previously noted for seeds of common mullein, the temperature shift may trigger germination in the absence of light and the slight coverage of soil and litter would be sufficient to bring moisture relations into balance and permit germination.

Nitrate

When you work in the field dealing with revegetation of disturbed lands it is common to note a flush of spring germination. Part of this is accounted for by seeds responding to warmer temperatures after prechilling requirements were satisifed. A contributing factor to spring flushes of germination is available nitrate through nitrification. The germination of many dormant seeds is enhanced by available nitrate ion enrichment. Raw, low organic matter content soils on disturbed sites may be so starved for nitrogen that this form of germination enhancement is not available. We know that this is true where you have accumulations of litter under woody plants where the carbon:nitrogen ratio is high (Young and Evans 1975) and have demonstrated in bioassay procedures that germination can be enhanced through nitrate enhancement (Evans and Young 1975). Can germination characteristics be manipulated in seedbeds on disturbed sites through nitrate fertilization? No one, to my knowledge, has conclusively demonstrated that this is possible.

Artificial Seedbeds

It is a huge step to go from the laboratory to actually planting seeds in a seedbed in the field to establish vegetative cover on a disturbed site. Planting seeds is among the most basic instincts of human culture, and millions of subsistence agriculturalists in the Third World are dependent upon this simple task. The planter that has not had a stand failure has never attempted to grow plants from seeds in the field.

The more ecological factors pertaining to seed germination and juvenile seedling growth you control, the less the chance of failure. Ideally you have a weed free, firm but friable seedbed covered with mulch, and control over irrigation, predation, and grazing.

In my experience, the most common sources of seedling stand failure on wildlands are in order of occurrence: 1) seeding too deep; 2) improper seed placement and coverage; 3) competition from weeds; 4) seeding nonadapted species; 5) predation of seeds or seedlings; and 6) natural disasters, floods, drought, or wind erosion.

Technology of Seeding

Before we explore the various hazards to seedling establishment, let us review the necessary steps in seeding technology (Young and Evans 1987). There has to be some method of transporting the seeds as they are planted. Your hand, a coin envelope, the hopper on a hand seeder, or a drill seed box with the capacity of several hundred kilograms of seed serve the same purpose. If seeds of more than one species are being seeded at the same time in a mixture, separate containers for each species or agitation to keep the seeds mixed in the same box are required. If the seeds have fluffy appendages or hairs that can not be removed, special agitators such as arms rotating in the opposite direction are required.

The second technological requirement for seeding is some means of metering the seeds. Obviously an evenly spaced stand is usually most desirable to fully occupy the site and to minimize intraspecific competition. The most common mechanical metering system for planting is a fluted roller or shaft. Occasionally in technologically advanced machines, slotted plates with the slot machined to the size of specific seeds are used. Metering systems are usually driven from traction mechanisms.

If the seeds are to be broadcast on the surface of the seedbed, this is all that is required for mechanical seeding except for some means of distributing the seeds, such as a spinning impeller. If the seeds are actually to be planted in the seedbed, an opener is required. The opener is a fixed shovel, a rolling or a cupped disk, or a fluted colter that forces the soil apart for a small opening or drill for the placement of the seed. For the mechanical drill to function properly there are two other requirements: tubes to transport seeds between the metering device and the opener, and some means of closing the drill opening and covering the seed.

Seeding Depth

The general rule for depth of seeding is two and one half times the diameter of the seed. If you stop and consider this rule, it is obvious that most seeds need to be planted at very shallow depths. In order to accomplish this, the seedbed has to be rather uniform. Lateral variations in topography ruin depth control on a microscale. If you have nearly flat microtopography, the wider the drill the more precise the depth control. The reciprocal is true with uneven microtopography. The most narrow drill, an independently suspended opener, has the potential of providing the most precise depth control with irregular microtopography. Unfortunately, the only depth control for an independently suspended opener is a depth band on the opener.

Remember that some seeds have dormancy systems that require extreme diurnal temperature fluctuations for germination. Soil is an insulator and the deeper the seed burial the less the temperature fluctuation. For the majority of seeds, extreme temperature fluctuations depress rather than enhance germination. Small seeds of this nature can be planted at shallow depths in the bottom of furrows. The furrows serve as microtopography to dampen extremes in diurnal temperatures fluctuations (Evans et al. 1970).

What happens when you are seeding mixtures with different sized seeds? If you only have one box to hold seeds on the drill, the depth setting has to be a compromise. Generally the optimum depth for the dominant species in the mixture is chosen. Drills are now being manufactured with multiple boxes that feed alternate openers that can be set at different depths. The choice of openers and the durability of the mechanism, because of the obvious complexity, may not be suitable for all seeding situations, especially on disturbed lands. Even with the relatively crude rangeland drill, it is possible to vary seeding depth if the drill is equipped with more than one seed box. Larger grass seeds can be seeded through the openers and small seeded legumes or very small seeds of big sagebrush can be dribbled ahead of the opener, perhaps in alternate rows to the grass seeding.

Why are seedings consistently made at too great a depth? Part of the problem is human nature to sock the seeds in the ground so they have a good chance to grow. While conducting research on hydroseeding, I had a revegetation contractor tell me that he could obtain any depth I wanted by just increasing the pressure. We never had the opportunity to check this hypothesis because the pumps used to obtain the pressure largely destroyed seed viability before the seedbed was reached (Kay et al. 1977).

Optimum seed placement implies a balance among contact between soil particles and the seed coat for hydraulic conductivity, adequate soil aeration, and minimal resistance to seedling emergence. Obtaining proper seed placement is often a function of soil moisture content. If the seedbed is too wet, there is the danger of crusting and problems with seed tubes clogging and openers not self cleaning. In wildland seeding on disturbed sites, soil moisture content is much more often a problem. Without supplemental irrigation, contractors are dependent on nature for moisture events. At high elevations the first fall moisture event may be snow that closes the revegetation site. Interesting experiments have been conducted by Hauser (1987) in injecting water through the drill opener to enhance soil-seed contact.

Plant Material for Environmental Restoration

In the past, most of the emphasis on restoration of disturbed lands was to enhance grazing or forestry use of the area. By using well-designed methodology, notable success has been obtained in interseeding shrubs and forbs in grass stands on disturbed sites (e.g., Richardson et al. 1986). With the current interest in biological diversity, longer and more varied species lists are being proposed for restoration of complete communities. We need to give a lot of thought and research effort to the timing and spacing of these diverse plantings to reduce intraspecific competition among the seeded species.

Competition From Weeds

Remember that in using top soil as a seedbank source you get the bad and the good species. Many weed species have seeds highly adapted to remaining viable for long periods in seedbanks. I am not experienced in seeding mine spoils, but on a variety of wildland situations I have found a season of fallow certainly assists in weed control and ultimately in seedling establishment.

Predation

The key to manipulation of seed predators is habitat changes to reduce predator populations and to encourage predation of the granivorous. The use of repellents on seeds has almost ended because of lack of environmentally acceptable materials. A great deal more research is needed on the influence of granivors on regeneration of native plant species. The influence is not always negative. The regeneration of several species is often tied to scatter hoard caching. Scatter hoard caching consists of multiple caches of small numbers of seeds buried at a shallow depth in the surface soil. Often this type of burial is ideal for cool-moist chilling of seeds and subsequent germination and emergence of seedling. Rodents also cache seeds in larder hoards located in underground burrows. These hoards usually remove seeds from the possibility of germination.

Green Thumbs

Experience is a great teacher in the science and art of revegetation and environmental restoration. Providing a regulatory environment that allows individuals to be innovative is essential for environmental restoration to be successful.

Evaluating Success

The final m`easure of success in terms of seed ecology in environmental restoration is if the reestablished community is successful in renewing itself. Many of the crested wheatgrass *(Agropyron cristatum)* stands in the Great Basin have persisted for four decades with little evidence of aggressive spread or stand renewal. For fully stocked stands there has been no need or environmental potential available for seedling recruitment. However, it is possible to find isolated plants in seedings that were termed unsuccessful, with no progeny evident around the plant. In more favorable environments crested wheatgrass has spread from the sites of original seeding (Hull and Klomp 1967).

At this time there is no estimate of how long these stands of exotic perennial grasses will persist, but it appears that eventually they will be replaced with other species or mixtures of species. How many other artifically revegetated stands of exotic, naturalized, or native species are merely transitory occupiers of the site and eventually through seedling establishment will change in botanical composition?

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Evaluating Community-Level Processes To Determine Reclamation Success

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Abstract

The 1977 Surface Mining Control and Reclamation Act (SMCRA) refers to the reclamation of community level structure and processes in Sec. 515 (19) when it requires the operator to: "establish . . . a diverse, effective, and permanent vegetative cover of the same seasonal variety . . . and capable of self regeneration and plant succession at least equal in extent of cover to the natural vegetation of the native area . . . ." Naturally high levels of diversity can seldom be economically replicated on mined land, but a functional diversity that includes the dominant life forms and plants of varying phenology can be achieved. Within- and between-habitat diversity also need to be considered, as natural landscapes contain heterogeneous patches of vegetation. An effort to simulate natural patchiness may result in more successful reclamation if plant mixtures are chosen for adaptations to microenvironmental conditions. To maximize species diversity and productivity, plants should be chosen to minimize competitive interactions in mixtures. This can be done by choosing species that are different in their temporal and spatial use of resources. The goals of high diversity and high productivity are sometimes at odds, as there are numerous examples of high yielding species that respond to fertilization, and that exclude more slowly growing species. The goal of reclamation is to create communities that are as stable as are undisturbed communities, but there are as yet few examples of stable, artificially reclaimed communities that have also met the criteria of diversity. With time, researchers will be able to test hypotheses concerning stability on older stands. Practicing reclamationists can add to this body of knowledge by creating and measuring communities that are planned using ecological concepts.

Introduction

Many of the measurements that are typically taken to assess reclamation success reflect community-level structure or functioning, including diversity, phenology, stability, production, and others. In fact, the 1977 Surface Mine Control and Reclamation Act (SMCRA, Sec. 515, para. 19) refers to the community level when it requires the mine operator to "establish . . . a diverse, effective, and permanent vegetative cover of the same seasonal variety . . . and capable of self regeneration and plant succession at least equal in extent of cover to the natural vegetation of the native area . . . ." Each of the terms of this paragraph, while written in legal language, refers to various ecological concepts (Table 1). A reanalysis of this language, the implied concepts, and allied concepts that are not directly implied are of value in defining the kinds of measurements that need to be taken. Such a reanalysis is the subject of this paper.

Table 1.—Legal requirements for assessing reclamation success as set forth by SMCRA of 1977, Sec. 515 (19) and ecological concepts that pertain to them

Legal requirement	Ecological concept
Diverse	Species diversity
	Structural diversity
	Species composition
	Dominance
	Rare and uncommon species
Seasonal variety	Phenology
	Life form
Effective, permanent, self-regeneration	Establishment
	Production
	Competition
	Reproduction
	Stability
	Resilience

Some of these measurements (Table 1) do not fall exclusively into the area of community ecology, such as stability, which is also used in an ecosystem sense, or establishment, which is used when discussing individuals or populations. However, ecologists recognize that it is often difficult to draw a line between the processes that occur in populations, communities, and ecosystems, and the intent of the law surely was to promote ecosystem and population as well as community processes. Thus, some of the community processes discussed here will interface with population and ecosystem ecology. Succession will not be covered in this chapter, as it is the subject of the chapter by Chambers, MacMahon, and Wade (this volume).

Both the terms reclamation and restoration are used here, and they are as defined in the introductory chapter. Restoration is the attempt to create identical conditions before and after disturbance, while reclamation is the attempt to create somewhat similar conditions, and may include introduced species.

Diversity

Both structural and species diversity are important considerations for reclamation. Species diversity refers to the number of species (richness) and their relative importance, while structural diversity is the patchiness and vertical distribution of organisms. Species diversity can be measured by alpha, beta, and gamma diversity. Alpha diversity is typically measured by enumerating species richness and eveness with a habitat, patch, or homogeneous land unit (Whittaker 1975). Alpha diversity can be measured using various indices such as H' (Shannon-Wiener diversity index) or S (Simpson's index), and should be similar in the reclaimed and reference areas. In fact, there are questions about what these indices actually measure (Peet 1975; Chambers 1983), and the choice of index needs to be carefully considered to reflect the needs of any comparison, e.g., whether the goal is to have many of the original species or only similarity of the dominant species as in most reclamation projects. A species list with dominance values of individual species should be included when an index of diversity is used for comparison of reclaimed and reference areas, and may often be more effective and more interpretable than an index in determining success.

Beta and gamma diversity incorporate concepts of species and structural diversity (Whittaker 1975; Routledge 1984). Beta diversity describes between-habitat or between-patch diversity, and gamma diversity is known as landscape or total diversity, the sum of the diversity of all the patches. Thus the terms can be used to help describe patchiness or horizontal structure, although not vertical structure. Beta diversity is high if there is low similarity between patches, and is calculated as $\beta=\gamma/\alpha$. Gamma diversity is high if the landscape contains many dissimilar patches. In recent years reclamation plans have begun to call for more than one seed mixture to conform to the different edaphic and topographic characteristics of a site. This is an attempt to increase gamma diversity. DePuit (1984) described topsoiling strategies to increase gamma diversity, such as using different topsoil depths at different locations. Gamma diversity is not only important in assuring a maximum plant species richness on the site, it is extremely important for recolonization of animal species that require different habitat types (Parmenter and MacMahon, this volume). High gamma diversity increases the likelihood that colonizing species will find habitats to which they are adapted. Pre-disturbance and reference area assessments need to incorporate gamma diversity, which can be measured by Simpson's or the Shannon-Wiener indices (Routledge 1984).

Structural diversity includes both vertical and horizontal structure. Large-scale horizontal structure is discussed above as gamma diversity, but small scale structure also needs to be considered. Plants in natural communities form small patches because of initial colonization events, patchiness of resources, clonal growth, and competitive interactions. The dispersion of the patches may be regular, random, or clumped, depending on a number of factors such as age of the stand and interactions between individual species (Pickett and White 1985). One pattern that is important to natural and reclaimed stands is the "islands of fertility" formed by shrubs in desert and semidesert biomes. These islands have an important role in providing nutrients and water to the shrubs because the plants can "mine" the interspaces for available resources (West 1989; Garner and Steinberger 1989). They can also capture dispersing propagules of plants, animals, and microorganisms and thus increase the rate of succession on natural or reclaimed lands (M. Allen 1988a). Standard reclamation practices often obliterate plant structural patterns by drilling seed in rows. Following these practices, the soil resources remain random rather than dispersed, the self-fertilization process does not occur (M. Allen 1988b), and the stand may decline in productivity (Whitford 1988). Thus simulation of natural plant patchiness may be critical to reclamation of productive communities.

Vertical structure can be recreated on disturbed lands by reestablishing the same or similar plant life forms that existed prior to disturbance. Even the simplest communities have vertical structure, and these vary in complexity from deserts and grasslands that have only one or a few layers of vegetation to multi-layered tropical forests. Over- and understorey vegetation may be interdependent, as in the case of shade-dependent seedlings that mature to become sun-dependent trees or of obligately shade loving plants (e.g., Young and Smith 1979). The omission of important plant life forms not only will change community functioning but also is the most obvious error in recreating an estheticially pleasing stand that a reclamationist can make. Creating a grassland where there was once a shrubland or forest brings immediate attention to the inappropriateness of some reclamation practices, and creates an incongruous patch in the landscape. It is not necessary to restore all species to achieve similarity in vertical structure of reclaimed and natural areas, but it is necessary to simulate this structure. This goal can be achieved by choosing representatives of all of the major life forms, e.g., trees, shrubs, grasses, and forbs if these were originally present.

While the goal of restoration is to include as many of the original species as possible, a reclamation plan might include only the dominant plant species. Ideally these would be planted so that they are similar in order of dominance to the reference area, but in reclamation practice this occurs less often than is desirable from an ecological standpoint. A 1988 symposium on reclamation success indicated that most reclamation efforts are still aimed at planting pasture or grassland, even if these were not the original vegetation types (E. Allen 1988).

Rare and uncommon species are seldom considered for reclamation, even though they are important components of plant communities. There are many rare species, but only a few common ones (Rabinowitz et al. 1986). The uncommon and rare species may, taken as a whole, form a greater proportion of the plant community than many of the abundant species (e.g., Allen and Forman 1976). The abundance of uncommon species is illustrated by dominance-diversity curves from reclaimed and undisturbed Wyoming sagebrush-grassland (Fig. 1). The undisturbed area has many species, but the reclaimed mined site has few. While the reclaimed site has many of the same dominant species as the undisturbed, there are also many missing, such as some of the dominant shrubs.

The dominants that are excluded may in some cases be "keystone species." These are species that control the abundance of important neighboring species, both plants and animals, and thus the functioning of the entire community (Terborgh 1986). Shrubs may be such keystone species in that they create microsites for establishment of other species (e.g., Franco and Nobel 1988) and form the islands of fertility described above. Plants with nitrogen fixing symbionts have a similar keystone role (see paper by Palmer, this volume). The functioning of species that are purposefully excluded from reclamation mixtures must be carefully examined to assure that their absence does not lead to the creation of a community that is entirely different from the reference area or from what is intended.

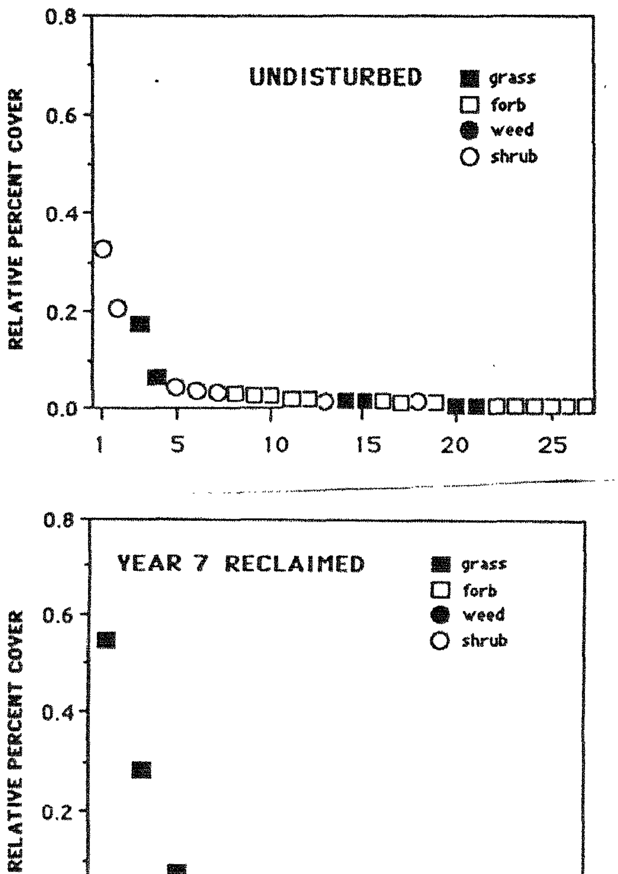

Figure 1.—Dominance diversity curves of 7-year-old reclaimed and native undisturbed areas in Wyoming sagebrush-grassland. The y axis is plotted on a numeric rather than a logarithmic scale show the large number of species that are low in abundance on the undisturbed site. Data from Parmenter et al. (1985).

Uncommon and rare species perform some important functions and should not be overlooked. For instance, all of the leguminous species in the undisturbed areas of Figure 1 are naturally inabundant, yet none has been included in the reclamation mix. Inabundant species are also important to increase diversity. A high diversity of species may help maintain year-to-year and site-to-site productivity, as each species has different responses to environmental conditions and will have the greatest productivity under its peak conditions. In other words, diversity leads to stability. There are many arguments for the maintenance of diversity (e.g., Soule 1986), and restoration has even been suggested as a method to maintain diversity of severely disturbed habitats (Jordan et al. 1988).

However, there are economical and ecological limitations to restoring a high diversity on most sites. One method that has partially overcome the problem is to use the native seedbank in freshly applied topsoil (Bell 1988; Thompson and Wade 1990). Where the seedbank is insufficient the hope is that with natural dispersal and succession, the lesser abundant native species will colonize the site and increase the diversity without the difficulty and expense of planting inabundant species. Colonization has occurred on some sites, as between planted, relatively widely spaced shrubs in a mine restoration project in southwestern Wyoming (West, personal communication), but has not occurred on other sites that used the more typical reclamation practice of densely seeding fast-growing plants that establish rapidly to stabilize the soil (e.g., Bock et al. 1986). The latter practice has the advantage of reducing soil erosion, an understandable goal when the expense of preserving and recontouring the topsoil is considered. However, there is a trade-off in that this aggressive reclamation mixture may preclude establishment of later colonizing species (Bock et al. 1986; Palmer, this volume). A solution to the problem might be to use other methods to stabilize the soil, such as mulching or furrowing, and seeding a more open stand of plants that allows establishment of colonizers. Even this may not always be a viable solution, where the colonizers are undesirable weeds. In general, a site that is rich in soil resources will be low in diversity (Tilman 1987). Reclamationists need to experiment with rates of fertilization and irrigation to achieve the desired diversity. Most often a low or moderate level of fertility will achieve the highest diversity (Powell et al. 1990).

Phenology

The legal requirement for seasonal variety of plants can be interpreted as including plants of varying phenology. One of the most obvious physiological determinants of phenology is made by photosynthetic pathway. Warm season plants have the C_4 photosynthetic pathway, they have relatively high temperature optima for photosynthesis, they are seldom shade tolerant and they photosaturate at light intensities that are somewhat greater than full sunlight, and they have high water use efficiency. Conversely, cool season plants have C_3 photosynthesis, relatively low temperatures for photosynthesis, are often shade tolerant and photosaturate at about one-third of full sunlight, and have low water use efficiency. The Gramineae have more C_4 species than any other family. There are C_4 representatives among forbs and shrubs, especially from arid and semiarid climates, but there are few C_4 trees. This is likely due to the self shading of leaves caused by trees with multilayered canopies, coupled with the lack of shade tolerance.

There appears to be no pattern of drought stress tolerance among C_3 and C_4 plants. In the Nebraska Sandhills the C_3 grasses developed lower xylem water potentials than the C_4 grasses (Barnes and Harrison 1982). There is a larger proportion of C_4 grasses in southern North America. Regression analysis showed this proportion to be best correlated with July temperatures and not at all with precipitation (Teeri and Stowe 1976). However, C_4 grasses only occur in areas that receive summer precipitation or have access to stored soil moisture during the period when temperatures are optimum for photosynthesis.

Within the photosynthetic pathways there are further phenological variations. For instance, there are many shade intolerant, C_3 spring ephemerals that complete their life cycles before the co-occurring tree canopies leaf out. The C_4 shrub *Atriplex confertifolia* has a leaf temperature optimum of only about 25°C, explaining its ability to grow actively both earlier and later in the season than would be expected of a C_4 plant (Caldwell et al. 1977). Thus it is important to know the phenological characteristics of individual species, as well as their photosynthetic pathway, to include a diversity of phenologies in a plant community.

Rooting depth also plays a role in phenology. Shrubs tend to be more deeply rooted than herbaceous plants, and therefore may have greater access to water. This may in part explain the long growing season of *Atriplex confertifolia* in a climate with an arid summer (Caldwell et al. 1977). Phreatophytes, which have their roots in the water table, are capable of season-long growth in arid climates (Nilson et al. 1984).

Plant communities are typically composed of species with varying phenology and life form that are adapted to the local climatic conditions. For instance, in the Chihuahuan Desert, where both summer and winter rains occur, C_4 plants grow in summer while C_3 plants are most active in late winter to early spring (Kemp 1983). Plants that have co-existed for long time periods are thought to have evolved mechanisms to reduce interspecific competition (Pianka 1981). The temporal division of resources that comes with phenological diversity is one likely strategy for achieving reduced competition. The inclusion of species with diverse phenologies will not only assure reduced competition, it will assure maximum utilization of soil resources throughout the growing season. This will then maximize productivity throughout the season, another important goal for reclamation.

Competition

One of the goals of reclamation is to hasten the rate of succession by introducing late seral species. However, colonizing annuals, often weeds, still are part of this process, and they are known to compete with later seral species (Connell and Slatyer 1977; Allen and Knight 1984). Reclaimed sites consist of the juxtaposition of pioneer species with climax species, and are represented by few intermediate stages (E. Allen 1988). In communities with many complex stages of succession, such as eastern deciduous forest, this juxtaposition is not so prevalent naturally, as the pioneer species have often disappeared by the time the late seral species are abundant. However, where the "initial floristics," model applies, or colonization of late seral species in early seral stages occurs (Egler 1954), late and early seral species do interact. This occurs especially in communities with few stages, such as deserts or semideserts, where the late seral species are likely to encounter pioneer species as neighbors even under conditions of natural succession (E. Allen 1988).

I studied the interactions between planted reclamation grasses and colonizing annuals during 1982-1988 in Lincoln County, SW Wyoming. The native vegetation is shrub-steppe and receives 230 mm precipitation annually, about 60 percent as snow. May and June receive the greatest precipitation, with drought typically ending the growing season in July or August. The topography is hilly, and soils tend to be clay loams. More detail on the site description can be obtained in Parmenter et al. (1985). The research area was prepared after coal extraction by regrading the overburden to approximate original contour of 1:5 and 1:6 slopes. Part of the area received 25 cm stored topsoil, and part was left with a subsoil. Native grasses *(Agropyron dasystachyum, A. smithii, A. trachycaulum, A. spicatum)* were seeded in the fall of 1981. Annual weeds, primarily *Salsola kali* and *Bromus tectorum*, colonized naturally in the spring of 1982. Other less abundant weeds include *Amaranthus albus, Chenopodium album, Descurainia sophia, Lepidium perfoliatum,* and *Polygonum arvense.* These all constituted 2 percent or less of the weed cover in any year, while *S. kali* had up to 44 percent cover and *B. tectorum* up to 25 percent. To test the effects of the annuals on grass growth, they were removed by hand from one-half of 144, 0.5 m^2 permanent plots at each of the five sites. See Allen and Allen (1986, 1988) for details on the experimental design.

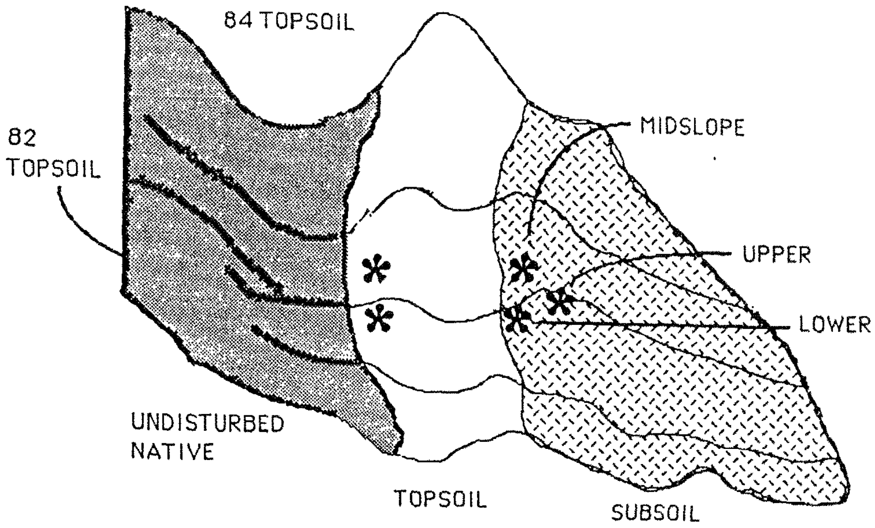

Figure 2.—Locations of the five research sites at the Kemmerer Mine. Three were on subsoil (upper, 1984 midslope, and lower), and two were on topsoil (1982 topsoil, 1984 topsoil). Two of the sites were initiated in 1984, three in 1982.

Five sites were chosen for study, two on topsoil and three on subsoil. The test the effect of year of initiation of succession (and therefore climatic variability), two of the sites, one each on topsoil and subsoil, were initiated in 1984. This was done by discing the site to destroy existing vegetation and reseeding with the same grass mixture in early spring of 1984. A schematic of the area (Fig. 2) shows the relative location of the 1982 and 1984 topsoil sites, the upper and lower subsoil sites, and the midslope 1984 subsoil site.

The patterns of interactions were different among the five sites in that the annuals either competed with, facilitated, or had no effect on the percent cover and density of the planted *Agropyron* grasses (Figs. 3-6). An increase in grass density caused by the presence of annuals was the most frequent occurrence. For instance, the weeds resulted in an increase in both grass density and cover in the upper subsoil site during one or more years (Fig. 3). This indicated that the annuals were nurse plants for the establishing grasses, or in other words facilitated grass growth. This occurred because the dominant annual was the tumbleweed *Salsola kali,* which remains on the site as a litter cover during the winter. Where it was not removed as part of the experimental treatment, it caused increased snow capture and provided an improved microenvironment for grass establishment during the spring (Allen and Allen 1988). Grass density was higher where weeds were present by the second growing season, but it took several years before grass percent cover was also improved (Fig. 3).

Grass density was also improved by the presence of *S. kali* in the lower subsoil and 1982 topsoil sites (Figs. 4 and 5). However, there was no change in percent cover of the grasses at either of these sites. Apparently, the less dense grass individuals were able to compensate for lower density by greater growth on these two sites.

An entirely different pattern was observed at the 1984 topsoil site, where the percent cover of the grasses was decreased by the presence of annuals by the final year of measurement, or in other words they experienced competition from the annuals (Fig. 6). The grass density was not significantly affected at this site, although in 1984 and 1987 grass density was higher in the weed removal plots at $p = 0.08$. The reason for this different pattern may be that *Bromus tectorum* became the most abundant annual weed at this site after a few years, although *S. kali* colonized in the early years. (The unusually high colonization of *B. tectorum* at this site is likely due to its proximity to the native unmined area, where previous grazing disturbances had resulted in colonization. At all the other research sites, it had less than 2 percent cover).

Unlike the later growing C_4 *S. kali, B. tectorum* is C_3 as are the *Agropyron* species During some years *B. tectorum* germinated in fall and survived the winter even in this cold climate, and apparently competed with *Agropyron* species and reduced the growth of the perennial grasses the next spring. This pattern of competition between *B. tectorum and A. spicatum* was also noted by Harris (1967).

The final pattern of interaction was observed at the 1984 midslope subsoil site (data not shown), where there were no significant effects of the annuals on *Agropyron* species density or percent cover. A summary of the effects of the annuals on the five sites is shown in Table 2. Facilitation

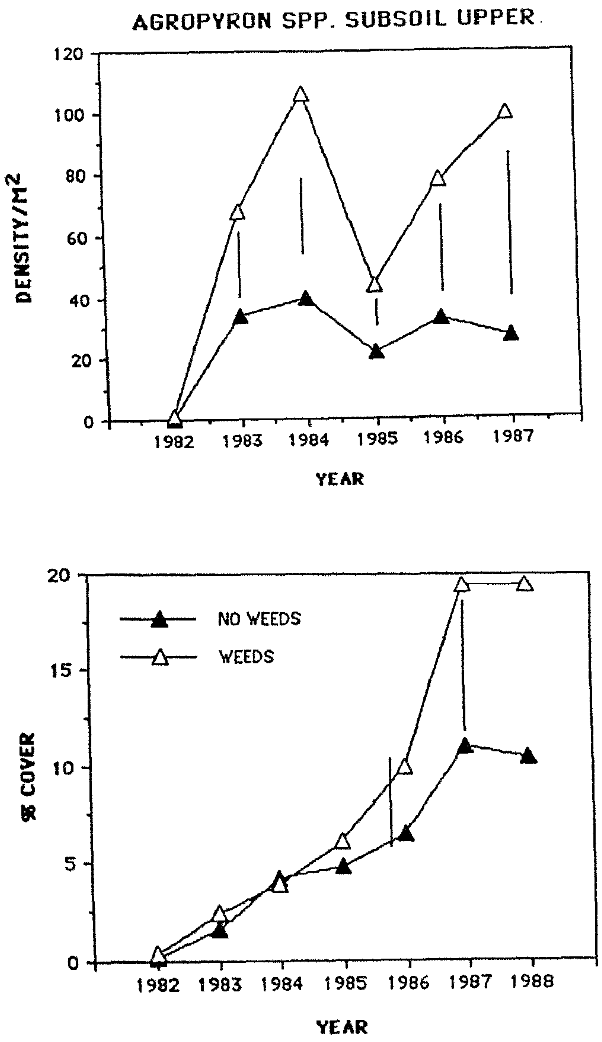

Figure 3.—Density and percent cover of the four planted species of *Agropyron* on the upper site (subsoil) with and without annual weeds from 1982-88. Individual grasses were difficult to distinguish after 1987 and were not counted. Weeds were removed by hand to create the "no weed" treatment.

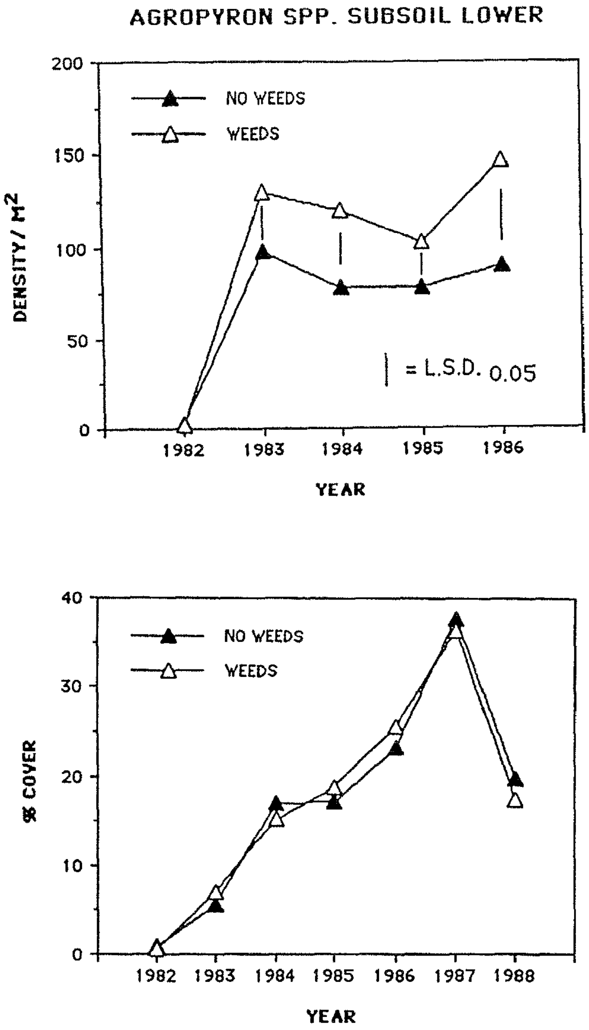

Figure 4.—Density and percent cover of the four planted species of *Agropyron* on the lower site (subsoil) with and without annual weeds from 1982-88. Individual grasses were difficult to distinguish after 1986 and were not counted.

Table 2.—Effects of annual weeds on density and percent cover of *Agropyron* species

Item	*Agropyron* density	*Agropyron* cover
Increase (Facilitation)	1982 Topsoil Upper subsoil Lower subsoil	Upper subsoil
No charge (Tolerance)	Midslope subsoil	1982 Topsoil Lower subsoil Midslope subsoil
Decrease (Competition)	1984 Topsoil	1984 Topsoil

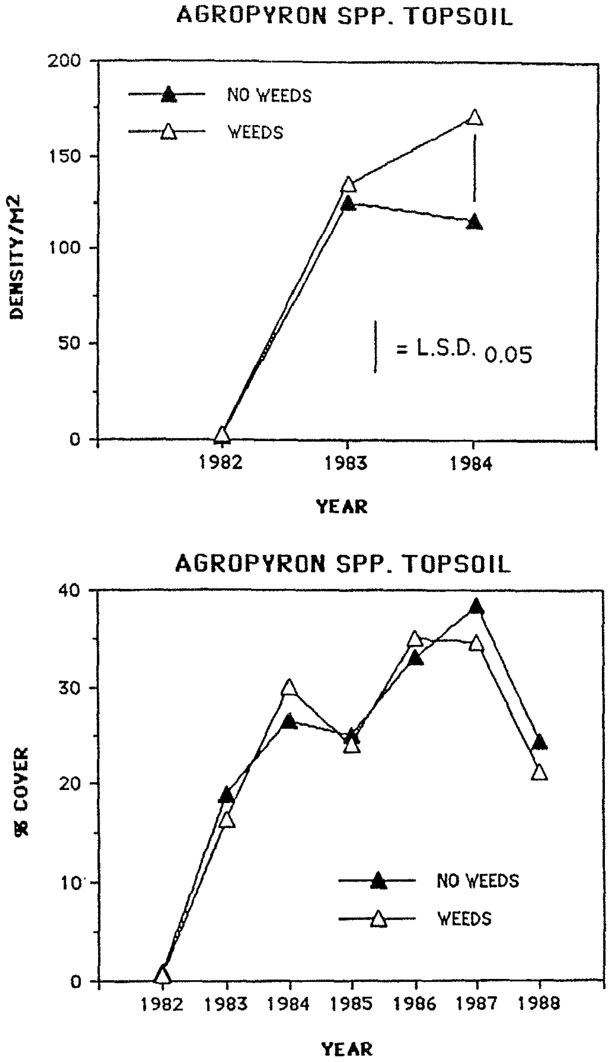

Figure 5.—Density and percent cover of the four planted species of *Agropyron* on the 1982 topsoil site with and without annual weeds from 1982-88. Individual grasses were difficult to distinguish after 1984 and were not counted.

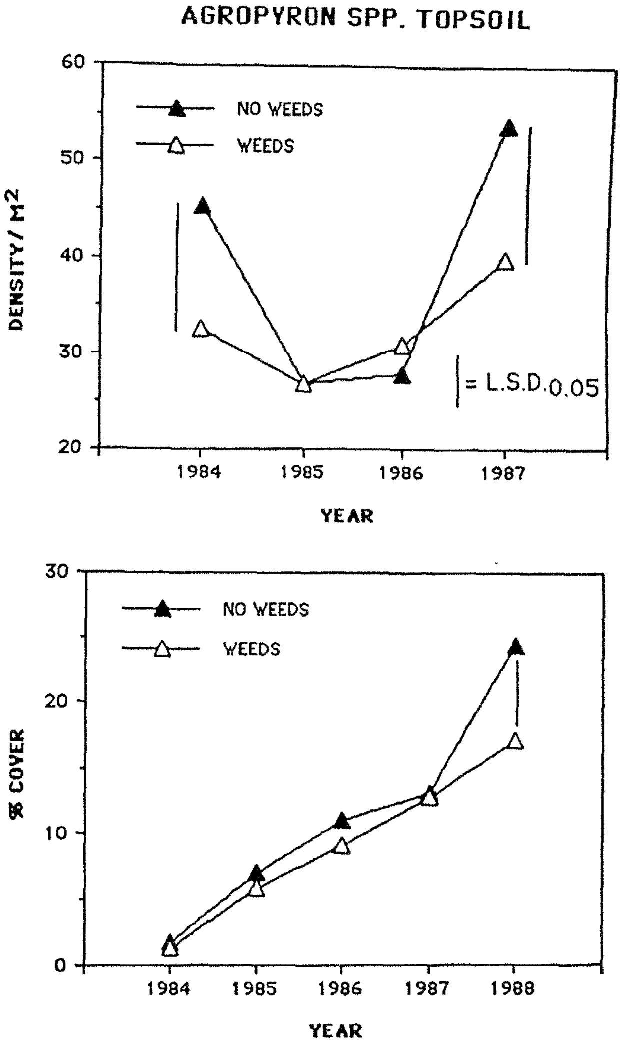

Figure 6.—Density and percent cover of the four planted species of *Agropyron* on the 1984 topsoil site with and without annual weeds from 1984-88. Individual grasses were difficult to distinguish after 1987 and were not counted.

was the most frequent interaction when grass density was measured, but no change was observed most frequently when percent cover was measured. Again, this occurred because the grasses that had decreased density without annuals (facilitation) were able to compensate by producing increased cover per individual.

The general patterns of annual weed colonization are shown in Figures 7 and 8. The upper and lower subsoil sites had an increase primarily in *S. kali* in the early years and then a decline, although the annuals were much more abundant in the upper plot. *S. kali* was virtually the only annual to increase again after 1985 in the upper site, but was only one of 13 species in the lower site, each with about 1 percent cover, after 1984, and the 1984 topsoil site *S. kali* dominated until 1985, and then *B. tectorum* became the dominant annual (see Allen and Allen 1986, 1988 for more details).

The main point for reclamationists to be aware of is that weedy colonizing annuals do not necessarily indicate failure, and may in some situations contribute to the success of reclamation where they facilitate establishment of desirable plants.

These patterns of facilitation, competition and tolerance between planted grasses and colonizing annuals indicate that the interactions are complex, and make the effects of weeds quite difficult to predict. The major pattern that emerges in that if *S. kali* was the dominant weed, facilitation or tolerance were operative. If *B. tectorum* were dominant, competition was observed. These results differ from those observed in eastern Wyoming in disced sagebrush grassland, where *S. kali* competed with seral species that included grasses and native annuals (Allen and Knight 1984). The main differences between the two studies are that the densities and percent cover of *S. kali* and the seral

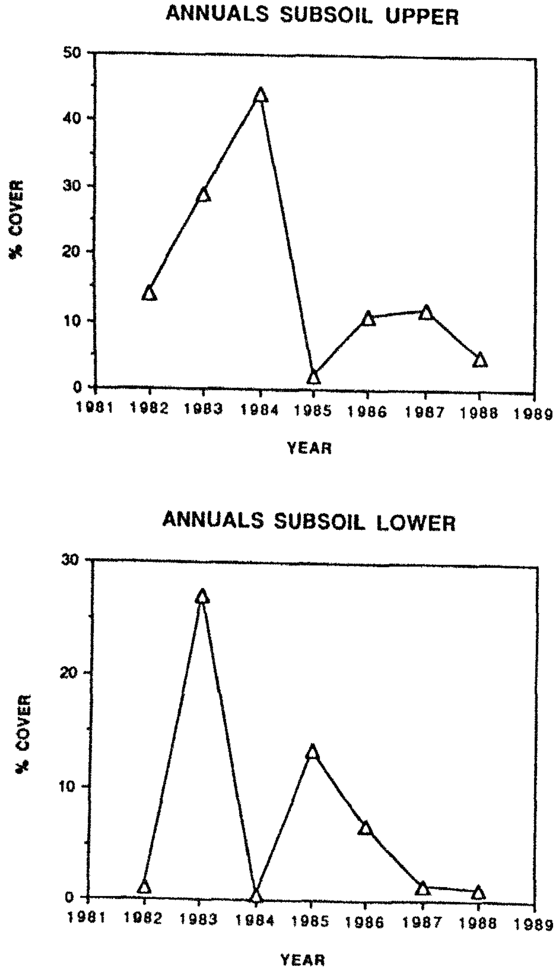

Figure 7.—Percent cover of annual weeds on the upper and lower subsoil sites between 1982 and 1988.

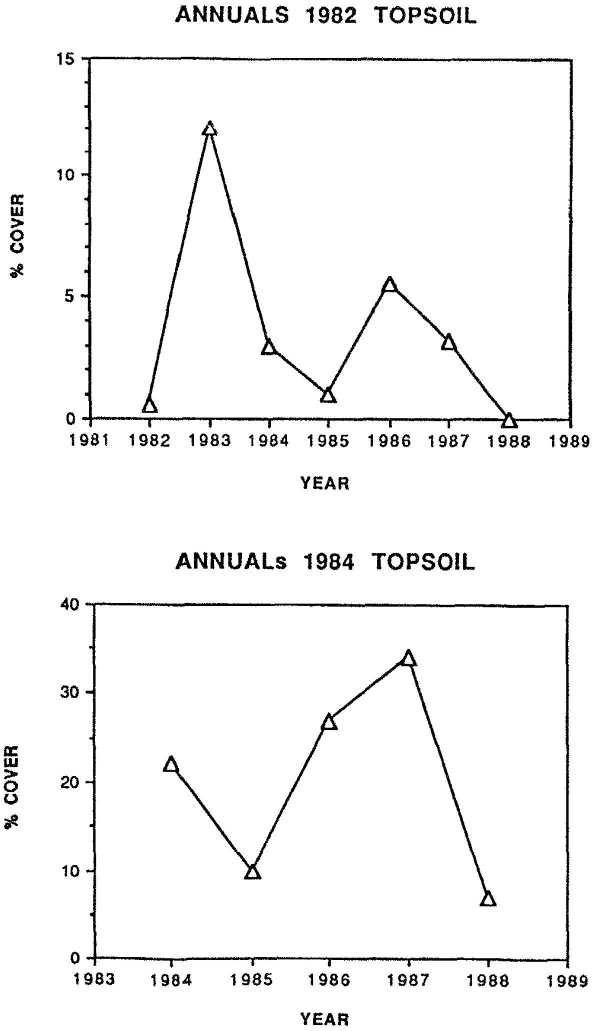

Figure 8.—Percent cover of annual weeds on the 1982 and 1984 topsoil sites until 1988.

species were higher in the moister climate, longer growing season, and richer soils of eastern Wyoming. *Salsola kali* may act to facilitate grass growth if it is relatively low in abundance, as in the study reported here, but it may have the net effect of competing with neighboring plants where its abundance is high.

Stability

Stability has two definitions, and has been divided into resilience stability, the ability of a disturbed site to return to a predisturbance state, and resistance stability or inertia, the ability of a site to resist change after a disturbance (Westman 1978; Fig. 9). Reclaimed sites are often not designed to return precisely to a predisturbance state, as a low diversity of native species, or introduced species are planted. Thus the resilience of the reclaimed sites is low. Whether they have functional resilience, e.g., whether nutrient cycling and other aspects of ecosystem functioning return to a former state, is an open question. A restored site, on the other hand, by definition has high resilience if the restoration is successful.

The question of resistance or inertia of reclaimed lands to minor disturbances is also open. The SMCRA requires 5 or 10 years to determine whether the vegetation exhibits resistance, but a decline after this time period is possible. An interesting example can be taken from the competition data presented above. Both the lower subsoil and the 1982 topsoil sites had a decline in percent cover of *Agropyron* species between 1987 and 1988 (Figs. 4 and 5). This decline cannot be ascribed to climatic variation, as the younger 1984 topsoil site had a continued increase in percent cover in 1988. Whitford (1988) and Palmer (this volume) have suggested the reasons for a decline in productivity after several years may be related to nutrient cycling. If plant litter accumulates over time, nitrogen and other nutrients are immobilized both in the undecomposed vegetation and in saprophytic microorganisms. Densities of microorganisms and available nutrients declined over time in the studies of Whitford (1988) and Palmer. A visual inspection of the two sites described in Figures 4 and 5 showed a dense litter layer on the ground, although data on nutrient immobilization were not obtained. In any case, both sites showed a decline in productivity by the seventh growing season, well before the 10 years required for legal reclamation success, and after seven years standing crop was not as high as in the adjacent undisturbed native area (Waaland and Allen 1987). Further observations are necessary to determine whether productivity would increase again after seven years.

To date, reclamation success has been legally declared on few mined areas (Allen 1988). Thus, I will draw examples of long term stability from other sources of literature. These are older planted stands that have shown stability for up to 40 years (e.g., Bock et al. 1986; Johnson 1986), but these typically have few species or consist only of introduced species and would not meet present day legal requirements. In most cases the lifespan of the planted perennials has not been exceeded, so the longevity of the stand beyond the longevity of the planted individuals is not known. For instance, if a plant survives 50 years, the individuals may be considered established, but the stand is not established unless mortality is replaced by reproduction. In a young stand on an abandoned oil drill pad in arctic tundra (about 5 years old), mortality was high, but was compensated by reproduction. However, plant density was not as high as in the adjacent undisturbed areas (Gartner et al. 1983; Bishop and Chapin 1989). The most hopeful predictions for long-term stability come from prairie restoration projects. The Curtis Priarie at the University of Wisconsin was established more than 50 years ago, and has been a site of intensive study. It shows similarity in structure and functioning to unplowed prairie

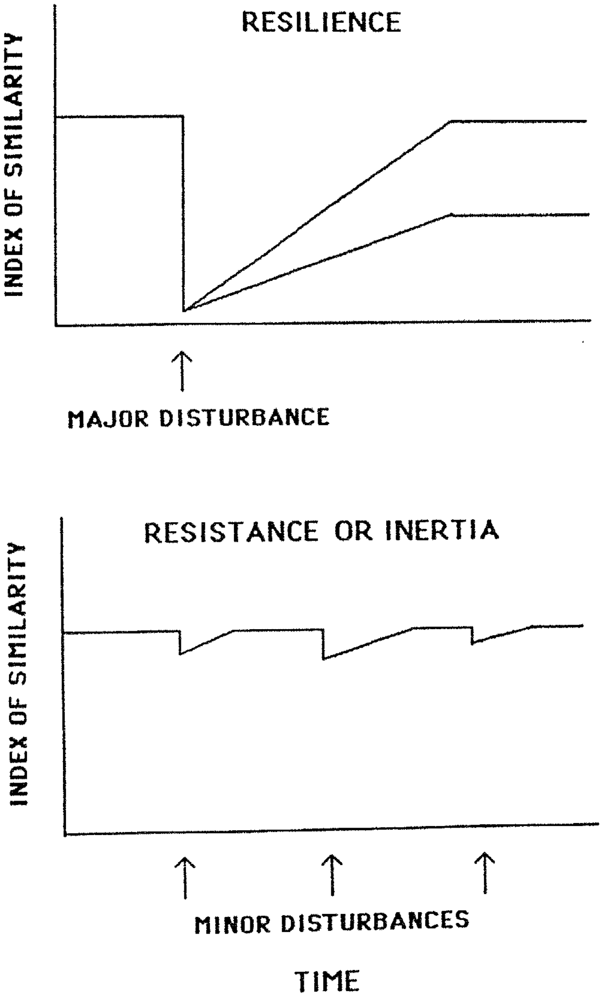

Figure 9.—Diagram of resistance and resilience stability. Resilience means that the site can be returned to a pre-disturbance structure and functioning as measured by a similarity index. Resistance means that the site can recover from a perturbation, often associatled with smaller disturbances than surface mining.

Table 3.—Measurements commonly used to determine the success of reclaimed community structure, and the functional measurements that are needed

What is measured (structure)	What is needed to determine functioning
Production (g/m^2)	Productivity (g/m^2/yr)
Density	Turnover (mortality, reproduction)
Species richness	Potential functional loss caused by missing species.
"Seasonal variety" of species	Changes in function caused by absence of some phenologies
Number of life forms	Changes in function caused by absence of some life forms
Indices of diversity, similarity	More specific information on species composition, and relationship to functioning
Production, density during 2-3 years	Long-term stability

remnants, and has exhibited resilience for many years (Cottam 1987). Whether reclamation projects with less intensive input and management can meet this impressive goal remains to be seen.

Conclusions

Ecologists have spent much effort in relating community and ecosystem structure to functioning. Structural measurements are preferable because they are usually simpler and quicker to take than measurements of functioning, and they are the types of measurements that are legally required (Table 3). The assessment of reclamation success will provide practical opportunities to determine if and when this relationship exists. For instance, plant density is a required measurement for shrubs and trees, but does not indicate rate of turnover. Reclaimed stands that have adequate density of long-lived plants will persist only for the lifespan of those plants, if they cannot reproduce on the site. Reduced species diversity is a characteristic of reclaimed stands, but normal functioning to maintain long-term stability is expected. If the proper vertical and horizontal structure has been recreated from a limited species pool, and species with important functions such as N fixation are included, then a functioning community may reestablish.

There are many functional attributes of reclaimed communities for which no structural measurements are required (Table 4). Again, the hope is that the structural measurements can be further extrapolated to relate to all important aspects of funtioning. Whether the communities function in the manner designed by reclamationists can only be determined with experimentation and observation over time. However, many basic ecological principles are accepted among ecologists and can be applied in the design of reclaimed sites. Where a self-perpetuating community that resembles the native community is to be created, reclamationists can help assure long-term stability by applying the following principles:

- Simulate gamma or landscape diversity of the reference area in the reclaimed site.
- Simulate smaller scale patchiness of species.
- Re-create a similar horizontal structure by introducing he same or similar lifeforms.
- Include species with varying phenologies to decrease temporal overlap in resource use and to maximize resource utilization throughout the growing season.
- Include known keystone plant species in mixtures, e.g., shrubs that provide microsites for later plant establishment.
- Include some uncommon species in mixtures that have important functions, e.g., nitrogen fixation, that will increase diversity.
- Use species mixtures and cultural techniques that will allow natural plant colonization to increase diversity.
- Take advantage of non-aggressive weedy colonizers that may facilitate establishment of desirable species—control only those weeds that are known competitors.

Table 4.—Structure and functioning for which measurements are not legally required to determine reclamation success

Diversity of horizontal structure (patches and landscapes)

Diversity of vertical structure

Root architecture

Trophic structure

Species turnover

Species interactions (competition, facilitation, symbioses)

Recolonization by animals and resultant composition

Recolonization by microorganisms and resultant composition

Ecosystem functioning: nutrient cycling, mineralization, immobilization, etc.

By following some of these simple steps toward recreating reclaimed communities, reclamationists can likely increase the success of reclamation. Such steps should be coupled with increased monitoring to provide better information to plan for the success of future projects.

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Differences in Successional Processes Among Biomes: Importance in Obtaining and Evaluating Reclamation Success

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Introduction

Following disturbance land undergoes a "repair" process that is termed succession. Because of the universality of this process, the concepts of succession are central to any discussion of land reclamation—a form of managed succession. The rate and direction is affected by a variety of abiotic and biotic processes. In contrast to natural succession, the reclamation of disturbed areas is under some degree of control of humans. The key is in knowing, for any given disturbance, what effects inputs of the reclamationist have on these processes and, consequently, on trajectories and outcomes of succession. This information gives reclamationists the ability to select the most appropriate end land use and to "direct" successional processes towards development of ecosystems that support that land use. It also provides an objective basis for evaluating whether or not reclamation success has been achieved and, in some instances, indicates the best measures of success.

We define succession simply as the change in physiognomy, species composition, or proportion of species on a plot of ground over a moderate time interval (decades to a few centuries) following a disturbance to that site (MacMahon 1980). Relationships among the factors and processes that affect succession and the manner in which they influence succession are summarized in the flow diagram developed by H.S. Horn and J.F. Franklin (1981) (Fig. 1). Many of these factors or processes have been discussed in detail in the previous chapters in terms of how they relate to the reclamation of disturbed lands and reclamation success.

Successional processes span all levels of organization (molecular to biome). Herein, we have chosen to address successional processes at a very general level—that of the biome. Successional processes common to six major biomes are reviewed and differences among those biomes are examined. The manner in which these differences influence inputs of reclamationists and consequent successional outcomes is then explored. Finally, we examine whether current concepts of reclamation success are appropriate given the differences in successional processes among biomes.

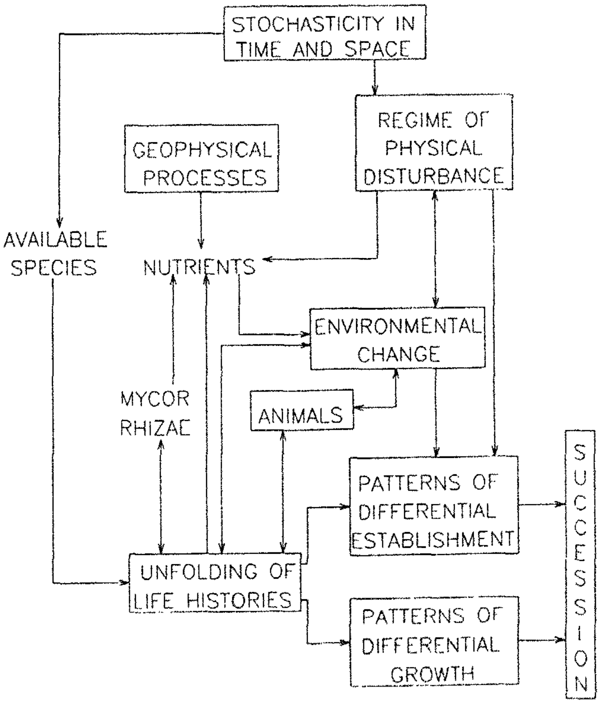

Figure 1.—Flow diagram of major elements with current concepts of succession (compiled and arranged by H.S. Horn and J.F. Franklin).

Successional Processes Common to all Biomes

Clements (1916) accurately portrayed the three major phases and six essential processes of succession early in the twentieth century (MacMahon 1980, 1981; Pickett et al. 1987). In his original description of succession, Clements (1916) stated that secondary succession begins with the "initiation" phase through some type of disturbance and ends with the development of a "climax formation." The processes involved depend upon the type of disturbance (nudation), the propagules remaining in the soil following disturbance and the colonizers that reach the site (migration), the success of the propagules and colonizers in establishment and growth (ecesis), and the alteration of the abiotic environment by these individuals (reaction). During the "continuation" phase species compete with one another (competition) and this results in an additional alteration of the environment. The end result of these processes, the "climax formation," occurs when the mix of species on the site tends to persist for long periods of time. A schematic diagram of succession applicable to many natural systems is presented to illustrate relationships among the stages of succession and the various processes that are involved in succession (Fig. 2).

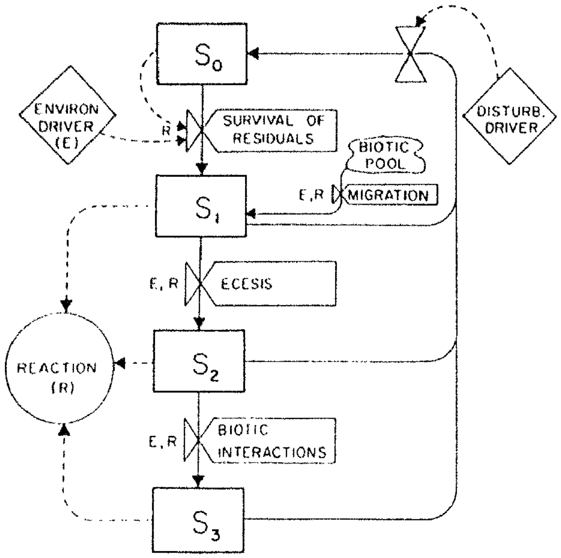

Figure 2.—Model of status of components (organisms and the physical and chemical and physical conditions) of a plot of ground over time. Boxes are states of the plot at any instant. Diamonds are system drivers. The circle is an intermediate variable. Dashed arrows represent information flows. Letters next to the control gates replace dotted lines from that point to the control for graphic simplicity (from MacMahon 1980).

Successional Processes and Reclamati in Different Biomes

A generalized comparison of the differences in successional processes across the various biomes is presented in Table 1. A detailed description of how the information for this figure was compiled is given in MacMahon 1981. It has been updated for inclusion here.

We recognize that exceptions to the comparisons presented exist and emphasize that our purpose is an examination of general patterns. Since we assert that the basic processes of succession parallel the processes required for land reclamation (MacMahon 1987; Redente and DePuit 1988), we discuss the differences in reclamation methodology among biomes as they are dictated by differences in successional processes.

Desert

Desert communities are highly responsive to the climate regime under which they grow (Turner 1990). Periods of mortality are often correlated with prolonged or extreme drought and episodes of establishment are related to unusually heavy precipitation during certain seasons (Turner 1990; MacMahon and Schimpf 1981). Fires are infrequent and usually occur only after a period of above-average rainfall and a buildup of plant fuels (Humphrey 1974). Wind is an everpresent factor and, consequently, plants are adapted to it. Localized mortality may result from outbreaks of pathogens or herbivores.

Because deserts are characterized not only by low but also by highly variable precipitation (MacMahon 1981), plant species adapted to the harsh environment are limited in number. Consequently, the species pool for colonization is often small. Although annual species may be abundant, their density and production is dependent not only upon soil moisture but also upon the relationship between soil moisture and soil nitrogen (Gutierrez and Whitford 1987). Seed production is highly variable among years. Seeds may be dispersed or consumed by rodents, birds, or ants (Reichman 1979; Parmenter et al. 1984; Morton 1985). The final distribution of dispersed seeds both on the soil surface and in the soil seedbank can be highly dependent upon vegetation structure and topography (Price and Reichman 1987). A high correlation between the seed bank and vegetation may exist because of extreme and unpredictable variations in growing conditions and the effects of seed predation (Henderson et al. 1988). Many succulents establish underneath existing "nurse" plants, probably due to more favorable temperature and nutrient relations, even though seedling competition for water with "nurse" plants and shading may result in lower growth rates (Franco and Nobel 1988).

In desert ecosystems species composition and overall physiognomy may be slow to change (Goldberg and Turner 1986). The initial colonizers may persist for long periods of time and the attainment of "climax" following disturbance may appear to be fairly rapid—40 to 100 years (Vasek et al.

Table 1.—Comparison of differences in successional processes involving plants among various biomes. Comparisons include nudation, most important factors creating disturbances; migration, the major types of plant propagules colonizing the site; competition, relative importance of different resources; reaction, degree to which seral biota (plants) alters chemical and physical environment of a site; stabilization, rate at which physiognomy (but not necessarily the composition) of biota (plants) on the site stabilizes; miscellany, degree of physiognomic and species turnover during succession (Revised from MacMahon 1981)

Item	Desert	Tundra	Grassland	Coniferous forest	Deciduous forest	Rain forest
Nudation	Drought	Cryo planation/ burrowing	Fire	Fire/ wind	Wind/ senescence	Senescence/ wind
Migration	Seeds	Nonseeds/ seeds	Seeds and nonseeds	Seeds and nonseeds	Seeds and nonseeds	Seeds and nonseeds
Ecesis	Periodic	Slow-periodic	Moderately fast	Mod fast (variable)	Fast	Very fast
Competition	Water/ nutrients	Nutrients/ water	Water/ nutrients/ light	Light/ water/ nutrients	Light/ nutrients	Light/ nutrients
Reaction	Moderately low	Moderately low	Moderate	Moderately high	High	Very high
Stabilization	Fast	Fast	Moderately Fast	Slow	Slow	Moderately Slow
Miscellany	No physiog. No species	No physiog. No species	Mod physiog. Mod species	High physiog. High species	High physiog. High species	High physiog. High species

1975; Webb et al. 1986; Webb et al. 1987). Depending upon the vegetation assemblage and the severity of disturbance, primarily the degree to which the soils are altered, the dominant species may or may not be the same (Prose et al. 1987; Webb et al. 1987). The overall severity of the site may also determine the degree of physiognomic change. Due to low plant cover and production, reaction or change in the physical or chemical environment is dominated by the effects of individual plants. Scattered vegetation results in "islands of fertility" that are associated with nitrogen accumulation and nitrogen availability (West and Klemmedson 1978). Plant competition varies depending upon water availability. On extreme desert sites plants may never attain densities sufficient to compete.

In the reclamation of desert areas, the highly variable seed supply and low seedling establishment rates necessitate addition of viable propagules of the desired species. Since adapted plant materials are often unavailable commercially, it is usually necessary to collect seeds of the desired species on or near the disturbed area. Several reclamation techniques can be used to improve water relations. Topographic contours can be manipulated to concentrate precipitation, and topdressing materials can be used that allow water infiltration, reduce evaporation, and have high water holding capacities (Aldon and Oaks 1982). Planting schemes can be designed to take advantage of seasonality of precipitation. In areas that receive less than 250 mm of annual precipitation, irrigation can be used to enhance establishment of shrubs and perennial grasses and forbs (DePuit et al. 1982). Irrigation can increase the number and productivity of individual plants that establish (Hunter et al. 1980) and the number of species that establish during a given time interval (Redente and DePuit 1988). In some cases it may be advantageous to accelerate the critical establishment phase through the use of tublings.

Tundra

Small-scale soil movements are the most common forms of disturbance in tundra ecosystems. These occur primarily via cryoturbation processes (Johnson and Billings 1962) and small mammal burrowing (Thorn 1982). Fire and wind are seldom important and the effects of herbivores are often highly localized.

Plant establishment occurs from both seeds and vegetative propagules (Bliss 1985; Chambers et al. 1990). The low heat budgets that characterize tundra ecosystems result in short growing seasons during which temperatures often drop below freezing. Highly variable and generally low seed production (Chambers 1989) and a limited number of seed "safe sites" (Chambers et al. 1991) contribute to low, possibly episodic, seedling establishment. Seed banks of both alpine and arctic tundra appear to be comparable in size to those of temperate forests (McGraw And Vavrek 1989). Seed production and longevity varies among species (Chambers 1989) and the resemblance of the seed bank and standing vegetation is also variable (McGraw and Vavrek 1989). Seedling mortality is generally high (Bliss 1985, but see Chambers et al. 1990) and may result from needle-ice activity (Roach and Marchand 1984), soil drought, or slow seedling development (Bell and Bliss 1980).

As in deserts, the pool of species adapted to the extreme environments is small. In tundra ecosystems there are few annual species (Bliss 1985) and long-lived perennials are the most stable component. Although definite shifts in species composition occur during succession, once establishment occurs many of the "climax" species are in place (Bliss et al. 1973), a case of "autosuccession" *sensu* Whittaker 1974. Tundra species are low in stature and community physiognomy changes little during succession.

Tundra plants can alter their environment by moderating temperatures within their own canopies and insulating the soil surface from temperature extremes. Because of the generally cold temperatures, most processes, such as decomposition, occur at slower rates than in temperate systems. Competition may involve both nutrient and water availability (Gartner et al. 1983; Johnson and Caldwell 1975).

Several factors must be considered in the reclamation of disturbed tundra ecosystems. Depending upon the size of the disturbed area, residuals and migrants from surrounding native areas seldom result in plant densities sufficient for immediate soil stabilization, even with direct replacement of topsoil. In both deserts and tundra seeds may be redistributed by the wind following initial contact with the soil (Chambers et al. 1991). In wind dominated tundra ecosystems it is often necessary to hold both soils and seeds in place with stabilizing mulches (Chambers et al. 1990). Organic mulches can ameliorate the microenvironment by increasing soil temperatures and soil water and decreasing the deleterious effects of wind (Cochran 1969). Fertilization can increase seedling growth in nutrient poor environments, but may result in a short-duration nutrient pulse in poorly developed mineral soils (Chambers et al. 1990). Also, high nutrient levels may result in high seedling competition in well-developed soils (Gartner et al. 1983). To obtain plant materials adapted to the harsh environment, it is often necessary to collect them on or near the disturbed area. Because plant growth is slow, it may be necessary to transplant established seedlings or plants in areas where immediate erosion control is a reclamation goal.

Grassland

Major and frequent disturbances in grasslands include both fire and drought. Herbivores can effect long-term changes in species composition and create small-scale disturbances (Huntley and Inouye 1988). Seed production is abundant except during droughts (Rice 1989). Seed banks often exhibit strong seasonality and spatial variability (Hassan and West 1986) and there are often significant differences between the seed bank flora and above ground plant community. Depending upon the length of time since disturbance, an abundance of both early seral natives and exotics may persist in the seedbank (Rice 1989). Because of the nature of the seed bank, it may be difficult to assess the composition of the flora that will regenerate following disturbance (Douglas 1965). Establishment occurs rapidly, depending on the scale of disturbance, and may be from residuals or sexual propagules dispersed by wind or animals to the site. During the reaction phase significant changes in soils and microclimate may occur (Hinds and Van Dyne 1980). Rapid growth of above ground plant parts results in shading and competition for light. Nutrient availability is a system attribute that is often closely linked to competition (Tilman 1988). Competition for water, although less than in deserts, may be intense during droughts. Relationships between water and nutrient availability can have significant impacts on competition and, consequently, on successional trajectories (Walker 1981). During succession moderate species turnover and physiognomic changes occur (Weaver 1954) and attainment of "climax" may require several decades.

In the reclamation of disturbed grassland areas it is seldom possible to depend solely upon seed reserves or propagules in topsoil. Since soil seed banks are frequently comprised primarily of early seral or exotic species (Douglas 1965), it is often necessary to seed more desirable or later seral species. On reclaimed areas late seral species may have to compete for resources with naturally occurring early seral species, exotics, or cultivated species that are seeded to hasten the stabilization of the site. Early seral species frequently exhibit higher growth rates, lower root-to-plant ratios, and higher nutrient uptake efficiencies than late seral species (Grime 1979; Chapin 1980). Many exotic species naturally exhibit these characteristics and cultivated species are often bred for these traits. Under the proper growing conditions, these species have highly positive growth responses to nitrogen and phosphorus fertilization (e.g., Aldon et al. 1976; DePuit et al. 1978). At high rates of fertilization or high nutrient levels in general, species that exhibit the greatest growth response tend to outcompete slower growing species (DePuit and Coenenberg 1979; Berg 1980).

Reclamation strategies can attempt to mediate resource competition by careful selection of seeded or planted species, modifying nutrient availability, or direct manipulation of species that are planted or become established on the site (Redente and DePuit 1988). The numbers of highly competitive species that are seeded can be reduced and those that are included can be seeded at lower rates. Less competitive species and species that are difficult to establish from seed can be transplanted onto the site. Fertilizer can be omitted or applied at moderate levels and, if necessary, herbicide or other specific plant removal methods can be used to eliminate or decrease the densities of extremely competitive species.

Coniferous Forest

The primary natural agents of change in coniferous forest are fire (Heinselman 1981), wind (Sprugel 1976), insects, or pathogens, which are often carried by an insect vector (Barbosa and Schultz 1987). Regeneration of disturbed areas occurs from residual seeds, vegetative propagules, and animal dispersed dissemules (Archibold 1979; Van Cleve and Viereck 1981). Seed production often exhibits high periodicity. Coniferous forests tend to be characterized by trees that are either dispersed by wind or cache-boarding mammals and birds (Fenner 1985). Many late seral conifer species are wind dispersed. There is frequently low correspondence between the composition of the seedbank and that of the vegetation (Archibold 1989). Late seral forest species often have low seed longevity and early seral species are often as or more abundant in the seed bank than late seral species in all stages of succession (Archibold 1989). Initial seedbed conditions can have a significant effect on seedling establishment. For certain forest species, establishment may be higher on fallen logs than on soil, apparently due to decreased competition from herbs and mosses (Harmon and Franklin 1989). In cool temperate areas, soil moisture and temperature relations are critical determinants of seedling establishment. Frost-heave and soil drought are primary causes of seedling mortality, especially of late seral forest species. Initial colonizers of severe disturbances are usually herbs and shrubs that are often rapidly topped by fast-growing deciduous tree species (Schimpf et al. 1980; Heinselman 1981; Van Cleve and Viereck 1981; Walker and Chapin 1986). Some conifer species exhibit slow growth rates and higher shade tolerance. Consequently, they may be able to establish in later seral stages (Walker and Chapin 1986) or persist as understory species established early in the successional sere (Bergeron and Dubuc 1989).

Depending upon the severity of the disturbance, significant changes in microclimate and in soil properties occur during succession (Vitousek and White 1981; Vitousek 1985). Plants in coniferous forests may compete for light, water, or nutrients. Shade tolerance has been equated primarily with late seral species and intolerance with early seral species in all forested biomes (Oliver 1981). The degree to which a species responds to shade determines its successional role can be mediated by the availability of other resources. It has been suggested that within a given ecosystem temporal changes in species composition occur primarily in response to autogenic or internal changes in light availability (Smith and Huston 1989). Species zonation or differences among areas has been interpreted as spatial changes in species composition that occur primarily in response to the effect of allogenic or external changes in resource availability on the dynamics of competition for light. In coniferous forests of the Pacific Northwest, shade intolerant species such as *Pseudotsuga menziesii* can form a stable climax if environmental conditions exclude the more shade tolerant species (Franklin and Hemstrom 1981). If environmental conditions favor the more tolerant species, they may dominate in both early and late succession. Several pine species in the southeastern United States exhibit a similar pattern (Smith and Huston 1989). In the absence of fire, they are replaced by hardwoods on more favorable sites but persist and dominate on exposed sites with xeric moisture regimes.

If the conifers are slow to assume dominance over broadleaf early seral stages, there is strong physiognomic contrast between early and late succession. Because of the length of time required for the conifers to assume dominance in boreal, temperate, and subalpine forests, it may take centuries or even more than a thousand years to reach "climax" (Van Cleve and Vierick 1981; Franklin and Hemstrom 1981; Schimpf et al. 1980). However, warm temperate forests may require much shorter times to obtain "climax" (Christensen and Peet 1981).

Since late seral species are often difficult to establish, reclamation of severely disturbed coniferous ecosystems requires creating an adequate seedbed and providing a seed or propagule source. Although it may be desirable to reduce competition, nurse species that ameliorate the environment or maintain required mycorrhizae or other symbiotic soil microflora should be present. For example, the early successional hardwood shrubs madrone and manzanita form ectomycorrhizal associations with some of the same fungal species required for establishment of Douglas fir seedlings in Oregon. If the hardwoods are absent, Douglas fir is often unable to reestablish after severe disturbance (Perry et al. 1989).

In the natural regeneration of these systems, soil conditions are often moderated by the initial colonizers or early seral species (Connell and Slatyer 1977). During reclamation it is frequently necessary to ameliorate the soil environment with natural or artificial mulches (Cochran 1969). Late seral forest species often exhibit significant lag times between seedling establishment and reproductive maturity. Also, seed production of late seral species often exhibits periodicity or occurs only during favorable years (Oliver 1981). If a disturbance does not coincide with a high seed production year, late seral species may be slow to establish or may not establish on the site at all (Connell and Slatyer 1977). To insure the presence of late seral forest species in all of the forested biomes, it is usually necessary to seed, or more frequently, to transplant the desired species.

Deciduous Forest

In the deciduous forest, fire is a less significant cause of disturbance than in the coniferous forest, but minor fires may be relatively common under natural conditions. Trees often become senescent, die, and are blown down by wind (tip-ups). If trees fall individually, isolated gaps in the forest are created and a regeneration mosaic may develop. Vegetation regeneration can result from suppressed seedlings, root-sprouts, rhizomes, and other surviving vegetative structures in addition to seeds (Oliver 1981). In mature and old northern hardwood forests of the United States, many of the tree species are shade tolerant and can germinate and establish under a closed canopy (Canham 1989). The number of canopy trees growing in gaps as saplings can be high (37 to 67%) and may result in significant regeneration of these sites (Lorimer et al. 1988). Seed production of late seral species may be highly variable among years and seeds of late seral species are often short lived. The composition of the seed bank in mature forests may be poorly correlated with that of the standing vegetation and seeds of early species may have the highest abundance (Pickett and McDonnell 1989). Microsite requirements for seedling establishment of late seral species can be highly specific (e.g., Borchett et al. 1989).

Plants with many different life history and physiological traits establish rapidly following disturbance. In many cases late seral dominants establish immediately following disturbance (Drury and Nisbet 1973; Christensen and Peet 1981). As succession proceeds, changes in both the physical and chemical environment occur that substantially influence competitive interactions and species replacement (Bormann and Likens 1979). Categories of shade tolerance have been developed for many deciduous tree species (Hicks and Chabot 1985). As in coniferous forest, species responses to shading may be determined in part by availability of other resources (Smith and Huston 1989).

Secondary succession in deciduous forest appears to have a deterministic component that is based on the life history traits of the available species and resource availability. A stochastic component reflects the site history and the characteristics of the disturbance (Halpern 1989; Houston and Smith 1987; Smith and Huston 1989). Initial colonizers may influence successional outcomes (Drury and Nisbet 1973; Connell and Slatyer 1977), but species composition often becomes more predictable as succession proceeds (Christensen and Peet 1984). Late seral stage dominants may be present from the beginning of the successional sere and apparent successional stages may be expressions of changing dominance due to differences in lifespans and environmental tolerances (Egler 1954). Regeneration series are often sufficiently predictable that they can be modeled using an individual-based approach that accounts for resource availability (e.g., JABOWA/FORET forest simulation models) (Botkins et al. 1972; Shugart 1984; Huston and Smith 1987). Development of the climax phase is often more rapid (200 to 500 years) than in all but the southern coniferous forests.

Reclamation of severely disturbed deciduous forests involves many of the same methods used in coniferous forests. A protective mulch may or may not be required depending upon the nature of disturbance and climate regime. It is usually necessary to provide seed or propagules of the mature forest species and to control competition. Post disturbance forest regeneration can result from suppressed seedlings, root-sprouts, rhizomes, and other surviving vegetative structures in addition to seeds (Oliver 1981). In the reclamation of severe disturbances in deciduous forest, topsoil can be used as a propagule source to significantly increase the diversity and productivity of early seral ecosystems (Wade 1989). The effectiveness of topsoil as a propagule source will be dependent upon the harshness of the site, the degree to which the topsoil has been degraded through past disturbance, and the current vegetation on the site from which the topsoil originates.

On reclaimed areas in deciduous and other forest biomes, vegetation structure can influence seed dispersal by both birds and mamnmals. Seeds of many species pass through the intestinal tracts of birds and are deposited in feces. The presence of vegetation of suitable height and size for perches, or even artifical perches, can serve as centers of invasion for many plant species (Hoppes 1988).

Boundary shape of disturbed lands and species composition of nearby undisturbed land can influence natural rates of species invasion and succession. Hardt and Forman (1989) found that stems of woody species extended farther into surface mines reclaimed with grass cover opposite concave than convex boundaries. A "concave-convex reversal" occurred a decade later when filled-in coves extended more rapidly toward uninvaded areas. Overall species composition within adjacent forest was a poor predictor of composition of the colonizing community. Abundance of animal-dispersed species was positively correlated with their abundance in adjacent undisturbed areas. No such relationship existed for wind-dispersed species.

Rain Forest

Mature rain forests are rarely subject to fire (Whitmore 1975) and drought and ice storms do not occur. Consequently, wind is the major agent of disturbance, acting either on individual trees or moderate-size groups of trees (tens to hundreds of hectares). In areas with steep topography, mass wasting may be important. Individual trees may persist for relatively long periods—100 to 1,000 years (Budowski 1970), with the accumulation of epiphytes increasing the probability of blow down (Strong 1977). Openings in the forest that result from tree blown down often form a mosaic of small patches that are characterized by varying moisture, temperature, and light regimes (Rickleffs 1977).

Rain forests exhibit a high degree of complexity and birds and mammals are usually more important dispersal agents than wind (Gentry 1982). Mammal and bird dispersal to a gap may increase over time as both shelter and food resources increase (Schupp et al. 1989). Ants may serve as important secondary dispersers (Roberts and Heithaus 1986). Seed longevity in tropical forests may be shorter than in more temperate areas due to the abundance of fungi, pathogens, and predators (Garwood 1989). Seedling mortality may also be high as a result of fungi, pathogens, and predators (Schupp et al. 1989). Germination and establishment occur rapidly following disturbance, but the majority of early colonizers that originate from seed are not those that are present in the mature forest (Farnworth and Golley 1974; Gomez-Pompa and Vazquez-Yanes 1981; Saulei and Swaine 1988). Regeneration also occurs from a variety of vegetative propagules and, depending upon the severity of disturbance, advance regeneration can be a significant factor in establishment (Uhl et al. 1988).

The vegetation not only alters the soil environment but also creates distinct temperature and light gradients that may extend dozens of meters into the canopy (Ricklefs 1977). Competition for water may occur in humid rain forests that have several consecutive months in which potential evaporation exceeds precipitation (Deshmukh 1986). More often competition is related to light or nutrients. Although the importance of light gradients and degree of shade tolerance in tropical forest regeneration is generally recognized, species responses to these gradients and consequent effects on successional patterns have been variously interpreted (Whitmore 1989a; Lieberman et al. 1989; Martinez-Ramos et al. 1989).

It has been estimated that 1 to 2% of the canopy areas is disturbed yearly through natural processes in both temperate and tropical forests. In tropical forests this appears to result in a high degree of spatial and temporal variability in the extent of canopy disturbance and, consequently, in light gradients (Lieberman et al. 1989), species composition and turnover, and population sizes (Whitmore 1989b). On severely disturbed sites (e.g., abandoned farms), succession may be retarded by the slow rate at which primary forest species become established (Uhl 1987). The time required to reach "climax" is relatively long (250 years or more) because the mature canopy species grow slowly (Richards 1952) and many environmental and floristic changes occur during succession.

Due to the high precipitation, severely disturbed sites in rain forests have a high potential for soil erosion. Thus, reclamation in rain forests requires extreme care to insure slope stability and to prevent the loss of soil and its nutrients. Reclamation of nutrient-poor ancient soils (primarily Oxisols) and of soils in which nutrients have been depleted through long-term cropping may require the addition of organic matter, nutrients, or both to reestablish forest vegetation (Deshmukh 1986). Artificial establishment of vegetation (seeding or transplanting) is seldom necessary because of the rapid natural recolonization of vegetation following disturbance (Hamilton 1990). Since late seral species may be slow to establish following severe disturbance, enrichment planting may be useful in some cases. However, the full complement of species can seldom be restored and, except on nutrient-poor soils, natural regrowth outcompetes planted trees.

Relationship of Reclamation Goals to Succession

There are three common goals of reclamation. All have a strong successional basis. Perhaps the most frequent goal of reclamation is to construct or reestablish an ecosystem that will structurally and functionally resemble the predisturbance native ecosystem (Allen 1988). When this is actually accomplished, it is most accurately termed restoration. A necessary assumption for restoration is that the predisturbance ecosystem is at or near climax and exhibits a high degree of stability.

A second possible goal of reclamation is to arrest succession at a low or intermediate seral stage. This may be desirable when the end land use is dependent upon species (plants or animals) that are abundant in the arrested seral stages. For example, in coniferous forests ungulate species such as elk and deer are often dependent upon early to mid-seral communities to provide the necessary habitat for their survival. If a reclamation goal were to increase or maintain deer and elk habitat, it would be desirable to promote establishment of shrubs or browse species but to delay establishment of late seral tree species for as long as possible.

A final goal may involve a totally different successional endpoint and, consequently, trajectory than that of the native undisturbed ecosystem. There are several instances in which this may be appropriate. The nature of the disturbance may place constraints on both the ability to reclaim a site and the subsequent successional processes that occur on that site. Reclamation of abandoned mine sites often involves substrates that differ significantly from those of the surrounding ecosystem and that may also contain toxic materials. It is often impossible and sometimes undesirable to establish the original native vegetation (due to plant uptake of heavy metals, etc.) on these sites without extensively ameliorating the substrates and topsoiling them. The reclamationist may have the opportunity to reclaim a site so that it will have a more desirable successional trajectory than the predisturbance ecosystem. This is often the case when surrounding areas have been previously seeded to monocultures or degraded due to overgrazing or other land uses. The reclamationist can improve the nutrient and water holding capacity of soils through proper soil handling and amelioration and can increase species diversity by direct replacement of topsoil and careful selection of seeding mixes.

Reclamation success is often evaluated by comparing the reclaimed area to a reference area or other standard chosen to represent the desired successional stage (Chambers and Brown 1983). The reference area is often part of an adjacent undisturbed native ecosystem. There are several potential problems with this approach. Comparisons of the reclaimed and reference area or standard are based on a relatively short time interval (usually 3 to 10 years) (see Surface Mining Control and Reclamation Act 1977). Consequently the stage of succession of the reclaimed and reference areas or standards may or may not be similar at the time of measurement or the successional trajectories may differ. Also, the attributes measured may not provide a good indication of the real successional trajectories.

The system attributes measured to evaluate reclamation success usually only include those related to vegetation structure such as number of trees and shrubs, percent cover or biomass of herbaceous species or species diversity (Chambers 1983; Chambers and Brown 1983). A necessary assumption is that if the vegetation structure of the reclaimed and reference area or standard are similar, the functional system attributes will also be similar. The practical reasons for this are that the time frame allotted for determining reclamation success in short and that detailed measurements of many functional system attributes are much more expensive and difficult to obtain than measurements of structural system attributes. A serious omission in many current evaluations of reclamation success is the assessment of soil properties and of the soil biota (see M. Allen and Zak et al., this volume). In many cases, soil properties, including the soil biota, provide an excellent indication of the potential long-term vegetation productivity (Doll and Wallenhaupt 1985) and of successional trajectories (Biondini et al. 1985). Also, disturbance is an integral part of most natural ecosystems and a successfully reclaimed area should exhibit recovery rates and processes similar to the reference area. How a restored or reclaimed ecosystem reacts to normal disturbance (fire, grazing, tree blowdown, etc.) compared to the reference area ecosystem can be one criterion used to evaluate reclamation success.

Some potential successional trajectories of reclaimed and reference areas are illustrated in Figure 3. The y-axis reflects any attribute, abiotic or biotic, measured to evaluate reclamation success. The x-axis is the time interval required to obtain climax. The ideal situation is that over time the reclaimed area converges with the reference area (Fig. 3a). A less desirable outcome is that the reclaimed area diverges from the reference area (Fig. 3b). This may occur even if certain attributes of the reclaimed and reference areas are highly similar at the time or under the conditions that the attribute are measured. Another possibility is that the reclaimed area parallels the reference area without divergence or convergence (Fig. 3c). This may

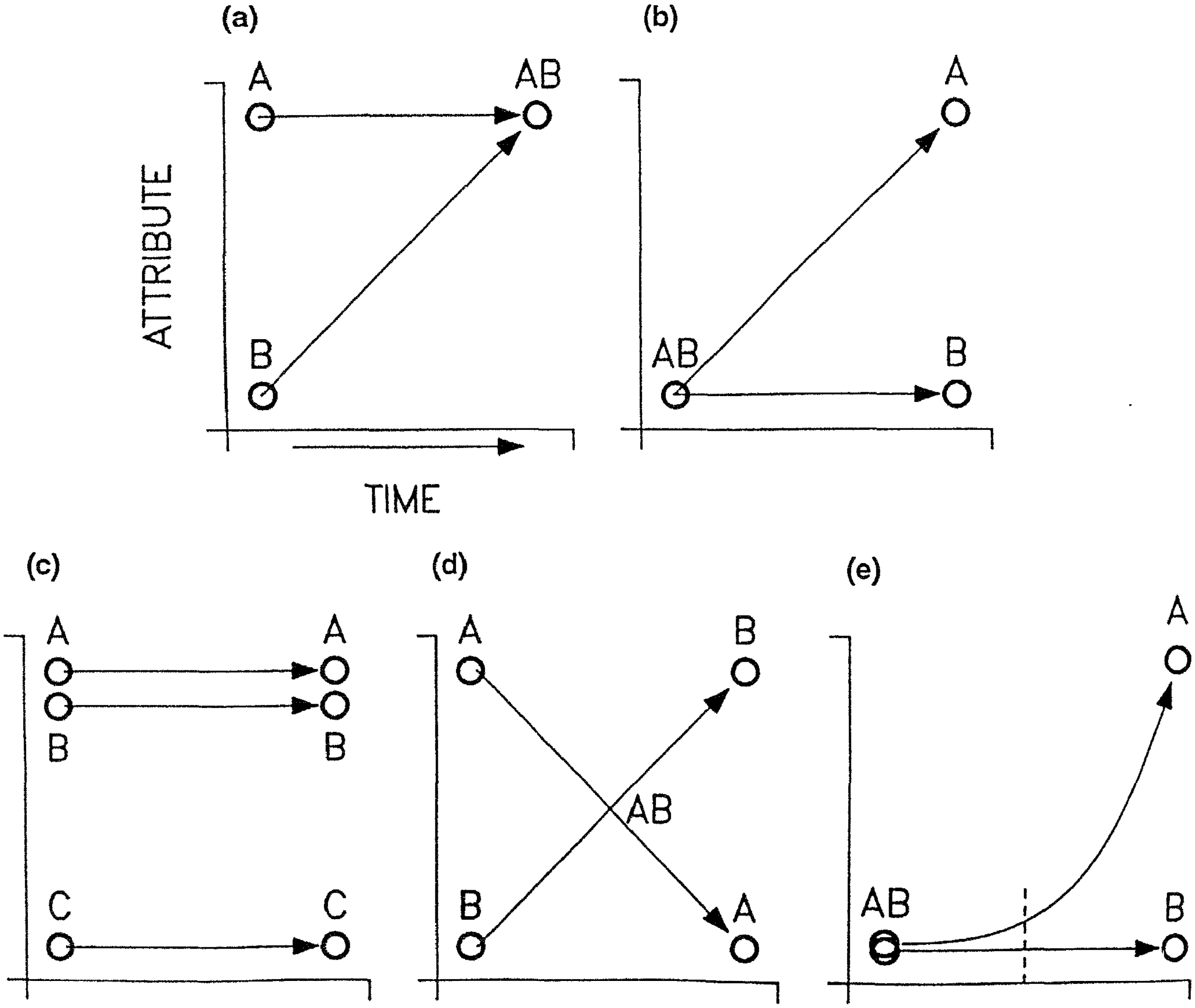

Figure 3.—Potential successional trajectories of reclaimed and reference areas. The y-axis represents any attribute measured to indicate the progress of succession or the success of reclamation. The x-axis is the time interval required to obtain climax. Different scenarios include: (a) the reclaimed area converges with the reference area; (b) the reclaimed area diverges from the reference area; (c) the reclaimed area parallels the reference area without divergence or convergence; (d) the reclaimed and reference areas may change in different directions (e.g., due to new or different types of management); and, (e) changes may exhibit various functions. The point in time when the attribute is measured affects perceived reclamation success.

be more or less desirable depending upon how similar the two areas were initially. Figures 3d and 3e illustrate the potential effects of land uses and management on comparisons of reference and reclaimed areas. The reference area may deteriorate during the reclamation period through disturbance associated with mining activities (Fig. 3d). In contrast, the successional trajectory of the reference area may change if it is released from land uses such as grazing through exclosure or some other form of protection (Fig. 3e). The point in time during which the measurements are made often determines the degree of perceived similarity and, consequently, reclamation success.

The same figure can be used to visualize the manner in which different ecosystem attributes change during succession. Attributes that exhibit little or no change or that vary randomly during succession are of little value in detecting successional trajectories. In some cases, it may be necessary to examine more than one attribute to ascertain real trajectories.

The effects of different successional starting points and trajectories on successional outcomes are related to a hypothetical reclamation effort in Figure 4. Arrows in the diagram represent successional trajectories. The original condition reflects the "normal" successional sere which may also be the reclamation goal. The reclaimed community may resemble either a successional or a climax-type community initially. If restored environmental conditions are similar to the original community and the proper species establish, a reclaimed early successional community may resemble the original successional community, or eventually, the climax community. If the reclaimed community is similar to a "normal" climax for the region in terms of both environmental conditions and species. It will undergo little change and the reclamation effort can be termed restoration. If environmental conditions or species are significantly different from the original conditions, reclaimed successional or climax-type communities may follow new trajectories toward new, possibly unique, climax-type communities. Some reclaimed successional communities may also go to arrested seral stages for significant periods of time.

Reclamation Goals and Success in Different Biomes

The successional processes that occur in the different biomes often dictate reclamation goals. Because of the differences among biomes in the length of time required to obtain climax, the degree to which reclamation success can be measured varies considerably.

Due to their extreme climatic regimes, desert and tundra ecosystems have often been cited as being among the most difficult biome types to reclaim following disturbance (Brown et al. 1978). They often exhibit variable temperature and precipitation regimes that result in a high degree of unpredictability (MacMahon 1981). Seed production is highly variable among years and plant establishment is frequently episodic. Despite these limitations, specific reclamation methodologies have been developed for both deserts and tundra that are highly effective (e.g., Aldon 1982; Brown et al. 1978; Brown and Chambers 1989). If the original soil conditions are restored, a carefully selected mixture of native species is seeded or transplanted, and the proper conditions for establishment are provided, the reclaimed ecosystem will rather quickly resemble the surrounding area. It may not be possible to establish the complete compliment of species immediately. In many desert and tundra ecosystems the predisturbance physiognomy can often be approximated through natural regeneration within 10 to 50 years. This timeframe can often be accelerated with proper reclamation methods.

For both deserts and tundra, often the only reclamation goal that makes ecological sense is the reestablishment of a native, late seral ecosystem. Native perennial species are the most stable component of these biomes and they should be used to evaluate reclamation success. Since dense stands of species with similar life forms and growth responses often retard colonization of other species (e.g., Brown et al. 1984), a variety of species with diverse life forms and growth responses should be present. Plant cover and production may be highly variable among years in deserts (West 1983). Also, since plant establishment is often episodic and plant growth can be slow in both deserts and tundra, it is necessary to monitor reclaimed areas for a reasonably long period to determine if reclamation success has been achieved.

In grassland and forest biomes, seed and other propagule availability is often high and, given the proper edaphic conditions, plant establishment can occur rapidly. For grasslands, the physiognomy of the site is often readily duplicated. Obtaining the desired species compliment and rare species is more difficult, especially if high densities of early seral, exotic, or introduced species occur on the site (Douglas 1965; Rice 1989). Reclamation goals may include restoration of a predisturbance or of a late seral type ecosystem if the predisturbance ecosystem is in an earlier seral stage. Native prairie restoration in the United States has been quite successful, but the cost is often high and other ecosystem components such as fire and herbivores must be carefully managed (Kline and Howell 1987). The ability to completely restore and manage these ecosystems will improve with increased research (Cottam 1987). Other reclamation goals for grasslands include successional trajectories that differ from the late seral condition. Arrested successions are seldom appropriate since they are often dominated by "undesirable" early seral species. Many former grasslands have been converted to pastures or croplands which are restored to the same end land use during reclamation.

Disturbances in forested biomes, especially those on mesic sites, are often considered the easiest to reclaim because of an abundance of propagules and rapid plant establishment. However, because of the nature of successional processes and the length of time required for

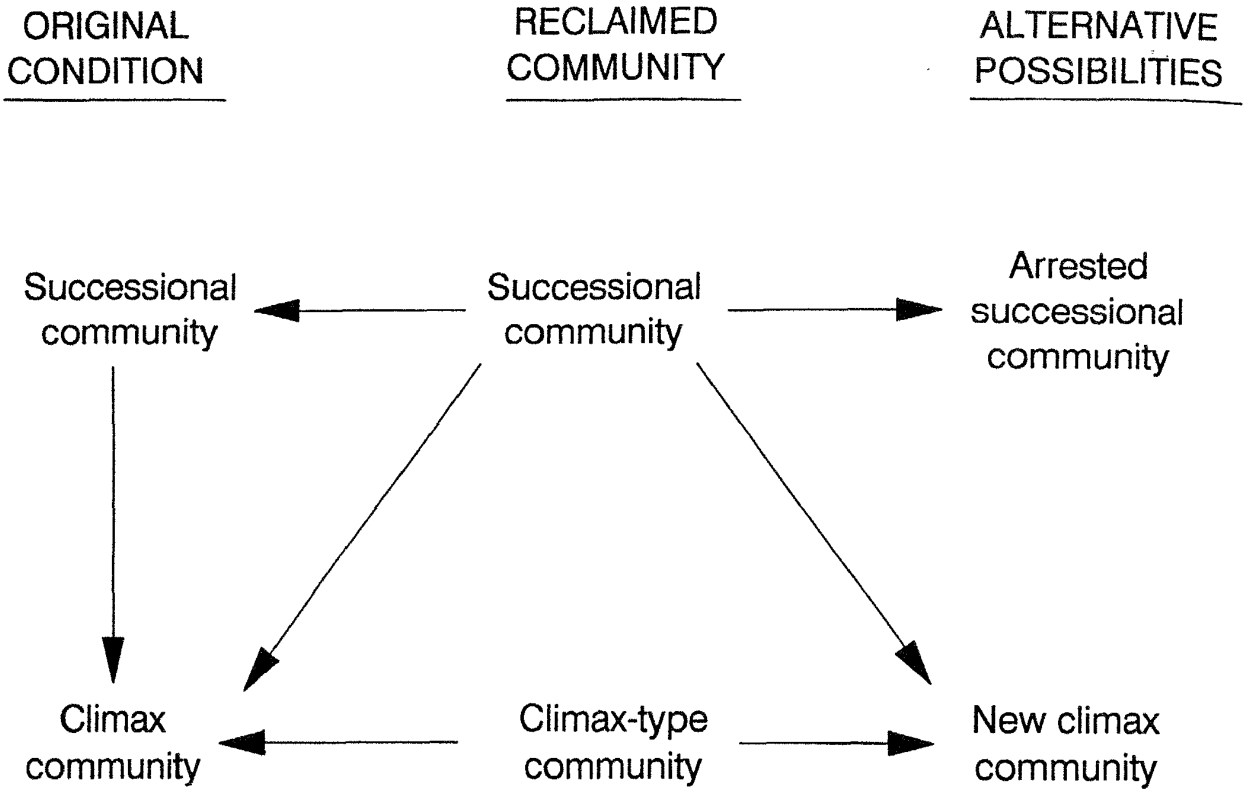

Figure 4.—Possible successional outcomes for a hypothetical reclamation effort. Initial reclamation may result in communities that resemble successional or climax-type communities. The reclaimed communities may eventually resemble the original communities or they may develop into new climax-type communities. Reclaimed early seral communities may remain unchanged as arrested successional communities.

succession in forests, it will almost never be possible to reclaim a forest so that it resembles a late successional stage within the bonding period. Late seral forest species are often difficult to establish, especially if propagules of early seral, exotic, or introduced species are already abundant on the site (Oliver 1981). Seed production of late seral forest species is often periodic and the seeds are often short lived. Since late seral forest species frequently exhibit slow growth rates and are slow to reach reproductive maturity, the physiognomic complexity of forests is slow to develop.

In forests, reclamation goals can include the establishment of early, mid- or late seral ecosystems representative of the predisturbance area or of ecosystems with totally different successional trajectories. Due to the length of time required for forests to reach late seral stages, reclamation success must necessarily be evaluated from early to mid-seral type ecosystems, regardless of the reclamation goal. Successional outcomes in forests may be highly dependent on the initial colonizers and are not always predictable (Drury and Nisbet 1973; Connell and Slatyer 1977). It has been suggested that species composition should become more predictable as succession proceeds and the influence of the pioneer community decreases (Christensen and Peet 1981, 1984). If the reclamation goal is to reestablish a particular mid- or late seral ecosystem, it will be necessary to insure that the dominant species for that particular seral stage have been established within the reclaimed area before bond release.

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Faunal Community Development on Disturbed Lands: An Indicator of Reclamation Success

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Abstract

The contemporary approach to reclamation of disturbed lands usually follows an "agricultural" strategy which often results in the development of spatially homogeneous flora dominated by a small number of species. In addition, the planting of alien or "native-but-not-resident" species creates unique floral communities that historically have not existed on the site or in the region. As a result, initial colonization by animals is accomplished by only a small subset of the potential species pool, and may result in dramatic population increases of certain opportunistic species that have been released from resource limitation or competitive and predatory pressures. In many cases, the faunal community structure of reclaimed sites bears little resemblance to those of pre-disturbance or nearby undisturbed faunas. In view of the importance of invertebrate and vertebrate wildlife to ecosystem function (e.g., pollination, herbivory, soil aeration, litter decomposition, seed predation, and dispersal), we suggest that an analysis of the faunal community development (incorporating trophic organization, species dominance hierarchies, guild apportionments, population demographics, and species turnover rates) can be used as an evaluation method for successful reclamation. We further suggest that deliberate inclusion of both horizontal and vertical heterogeneity in vegetation architecture, at various spatial scales (micro-, meso-, and macro-scale), will result in a landscape mosaic more conducive to the development of a highly diverse, self-perpetuating faunal community.

Introduction

To a large extent, reclamation "success" is more a human perception than an ecological reality. "Successful" reconstruction of a disturbed ecosystem depends on the attainment of certain anthropogenic, predetermined goals, such as the development of a prescribed flora and fauna, or the replacement of the original, pre-disturbance biotic community. In reality, of course, some form of biological assemblage will develop on all but the most severely disturbed sites, regardless of the reclamation effort. The question then becomes, "Is the resulting community a reasonable facsimile of the desired community (both in structure and function), and will it persist through time to become self-perpetuating?"

Clearly, one of the prerequisites for determining the degree of reclamation success is a model community with which to compare the reclaimed community. If the reclamation goal is to restore the original biotic system, then the model community can be either the pre-disturbance community of the site, or a nearby undisturbed community of similar type. If the reclamation goal is to create a different type of ecosystem from the pre-disturbance one, then one has two options: (1) a comparison can be made to similarly reclaimed sites that have been declared "successful;" or (2) if no such comparison sites exist, then biological surveys and long-term monitoring efforts can be implemented to evaluate the structure, functioning and persistence of the reclaimed site's community. Such studies would provide the data from which a judgement of "success" would eventually be made.

The purpose of this symposium is to address a number of ecological considerations in evaluating reclamation success. These considerations can provide the framework for assessing the degree to which a reclaimed site has developed a suitably complex and stable biotic community. The subject of this paper is the use of data on faunal assemblages as a means of measuring reclamation success. Animals (both vertebrates and invertebrates) can be useful indicators of ecosystem development, because, as with plants, their life cycles integrate a wide variety of abiotic and biotic variables. However, animals contribute additional layers of ecological complexity to ecosystem structure and functioning, and may themselves influence the development of a reclaimed site. Hence, inclusion in site evaluations of at least some animal species has long been recognized as an important component of reclamation success (Majer 1989).

In this paper, we first briefly discuss the properties and processes of biotic systems in which animals are important participants, and identify those components that ecologists might use to measure, compare, and evaluate different undisturbed and reclaimed communities. We will then provide an example data set from studies performed on the faunal communities of a shrub-steppe, surface coal-mine site in western Wyoming. Next, we will report on preliminary results of a long-term experiment designed to accelerate successional processes following reclamation of the Wyoming site, and discuss the implications for faunal community development. Finally, we will delineate future research needs and techniques for evaluating reclamation success with an integrated approach using both faunal and floral community data.

Properties of Biotic Systems

In the context of evaluating reclaimed ecosystems, the faunal component of the ecosystem both influences and responds to a number of biotic system properties. First, animals require a suite of habitat resources, that provide food, shelter and reproduction sites. A major component of an animal's habitat is the architectural structure of the vegetation. This includes both vertical architecture (grass vs. shrub vs. tree) and horizontal architecture (patch size and spatial distribution of vegetation types). Because animal species vary tremendously in their vagility and movement patterns, horizontal vegetation architecture is an important resource factor at a number of scales, ranging from landscapes (km^2) through stands (m^2) to individual plants (cm^2).

Animals also impart a trophic structure to the ecosystem. Through their feeding activities, animals influence the plant community in a number of ways (e.g., herbivory, granivory, pollination, and seed dispersal). In addition, animals can transport spores of beneficial mycorrhizal fungi (Rothwell and Holt 1978; Ponder 1980; Warner et al. 1987). Animal trophic interactions also influence nutrient cycling and energy flows; numerous invertebrate species (e.g., earthworms) are detritivores and play important roles in decomposition and soil development (Abbott 1989; Hutson 1989)

Faunal diversity (species richness and evenness) and biomass are additional properties of ecosystems that can be measured and used to compare various sites. Species richness and diversity are ecologically significant attributes of an ecosystem, as they can be a measure of the amount of redundancy in functional groups and trophic guilds. High levels of species redundancy may promote a greater stability of ecosystem functioning (e.g., numerous species of detritivores may increase decomposition rates and efficiencies, enhancing nutrient availability to vegetation). Animal biomass will, to some degree, determine the amount of herbivore pressure on the floral assemblage, and may ultimately influence the dispersion and species composition of the plant community.

Successional Processes

Succession is perhaps the most important ecological process influencing the outcome of a reclamation effort. While numerous models of successional processes have been developed (see MacMahon 1981), in this paper we will use Clements' (1916) classic succession model as a conceptual framework in which to discuss ecosytem development on reclaimed lands. In Clements' scheme (which applies to both flora and fauna), the ecosystem sustains a disturbance ("nudation") that reduces or eliminates resident populations. Surviving species ("residuals") undergo the process of establishment ("ecesis"), during which some species that are unable to cope with the new environment are eliminated. Through time, newly colonizing species ("migrants") join the residuals. Species that successfully establish alter the abiotic environment ("reaction"), thereby influencing the potential establishment of future migrants and the survivorship of the offspring of both residuals and past migrants. Biotic interactions ("coactions," e.g., competition, predation, parasitism) also influence the species compositions of the community. These successional processes continue until an equilibrium ("stabilization") is attained among the extant species and the environment. This state is often termed the "climax."

In the following section, we present a summary of faunal successional studies conducted on reclaimed surface coal mines in southwestern Wyoming. Many of our data can be interpreted using Clements' successional model outlined above. In addition, we will emphasize the relationships among the developing faunal and floral communities, and provide interpretations and implications for different management strategies.

Faunal Community Development In Shrub-Steppe Ecosystem

In 1979, a multi-disciplinary group of scientists (of which we were a part) at Utah State University began an NSF-funded long-term study of successional patterns on reclaimed surface coal mines in southwestern Wyoming. During the initial period of the project, we examined a number of reclaimed mine sites of varying ages in an attempt to describe the early successional patterns of the floral and faunal communities, given the conventional regime of reclamation efforts.

We conducted our studies on seven reclaimed surface coal mines, and an adjacent, undisturbed site, on the Pittsburg and Midway Coal Mining Company's Kemmerer Mine near Kemmerer, Wyoming. The region is characterized by rolling hills covered with a mosaic pattern of shrub-steppe vegetation (Fig. 1). Precipitation, mostly snow, averages 226 mm/yr, and is highly variable. Mean monthly temperatures in this high elevation site (2,200 m) range from -8°C in January to 17°C in July.

Vegetation in the region was shrub-steppe (West 1983), and was composed of two physiognomically distinct plant associations (Fig. 1). The first, a "tall shrub" association (with shrubs greater than 30 cm high) was dominated by big sagebrush (*Artemisia tridentata*), antelope bitterbrush *(Purshia tridentata)*, and Utah serviceberry *(Amelanchier* *utahensis)*. Shrub density was approximately 16,000 shrubs/ha (Parmenter et al. 1987). Understory vegetation was composed or forbs and grasses. This association of plants was patchily distributed throughout the region, but occurred most frequently in moister areas (e.g., along ephemeral stream courses and the leeward side of hills and ridges where deep winter snowpacks accumulate). These "islands" of tall shrubs were surrounded by areas of much shorter vegetation (less than 30 cm high), dominated by the diminutive low sagebrush *(Artemisia arbuscula)*, rabbitbrush *(Chrysothamnus viscidiflorus)*, and a variety of cushion plants, forbs and grasses.

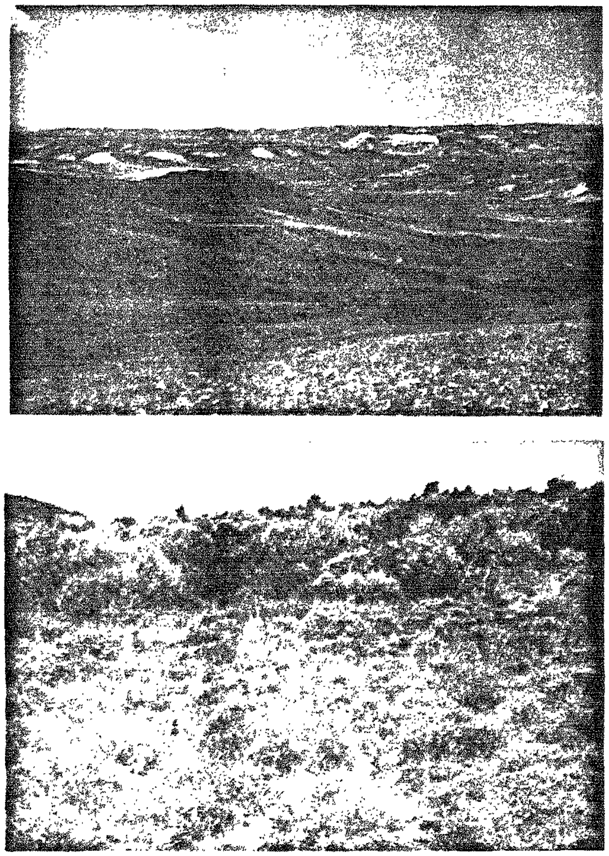

Figure 1.—Shrub-steppe vegetation near Kemmerer, Wyoming. Top: View of undisturbed area west of surface coal mines. Note landscape mosaic patterns of tall and short shrub associations. Bottom: Tall shrub association in background, short shrub association in foreground.

The primary goal of the reclamation effort at the Kemmerer Mine was to establish a shrub-steppe ecosystem that would be used both for wildlife habitat and livestock grazing (sheep and cattle). The seven reclaimed sites on the Kemmerer Mine had been sequentially revegetated at one year intervals beginning in 1977. The mines had been excavated by drag-lines, and the coal removed by truck-and-shovel procedures. Pit sizes ranged from 8 to 67 ha. Following coal extraction, the pits were recontoured with overburden materials (shales and sandstone). The mine pits revegetated from 1977 to 1979 received no topsoil, as they were not included in the Federal Government's 1977 Surface Mining Control and Reclamation Act (P.L. 95-87). The sites reclaimed from 1980 to 1983 received approximately 10 cm of stored topsoil. All sites were drill-seeded with a mixture of four-wing saltbrush *(Atriplex canescens)* and several wheatgrasses (western wheatgrass *[Pascopyrum smithii]*, beardless wheatgrass *[Pseudoroegneria spicata]*, thickspike wheatgrass *[Elymus lanceolatus]*, and slender wheatgrass *[Elymus trachycaulus])* (Fig. 2). Fertilizer and irrigation were applied during the first or second years to facilitate plant establishment. At the time of revegetation, each site was contiguous with undisturbed vegetation; however, as new, adjacent mines were developed, older mine sites became surrounded by newer ones. Details of each mine site's reclamation history can be found in Parmenter et al. (1985).

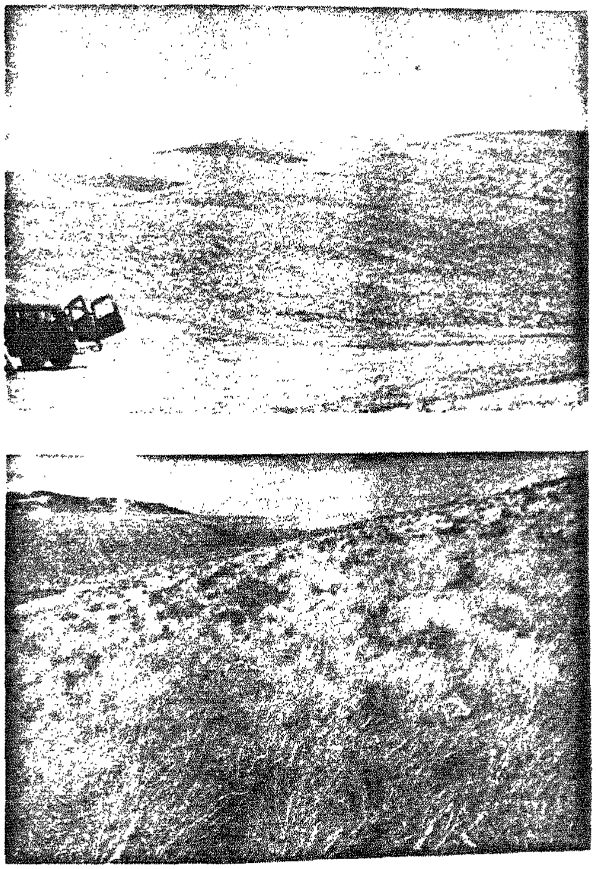

Figure 2.—Top: Recontoured surface coal mine pit on the Kemmerer Mine, Wyoming, following initial seeding and shrub planting. Bottom: Reclaimed mine site 6 years after revegetation with drill-seeded saltbush and wheatgrass seed-mix.

At the time of our sampling, the sites represented a series of plots containing various successional stages of plant communities, ranging from a newly seeded site (0 yr old) dominated by annual forbs, through young sites (1-2 yr old) with mixed forb/grass vegetation, to older sites (3-6 yr old) with established stands of bunchgrasses and saltbush. Unfortunately, no replicate (i.e., similarly aged) mine sites were available in the region; hence, we could not estimate the degree of faunal or floral variation that may have existed within each successional stage. In addition, these sites may not have represented a strict, temporally-linear successional sequence, due to variation in weather conditions (e.g., precipiation amounts) that existed on the sites at the time of their revegetation. However, the successional sequences and time-frames that we observed on these sites were comparable to those reported for other grassland mine sites in western states (e.g., Iverson and Wali 1982).

We sampled the flora and fauna of the seven mine sites and the undisturbed area in 1983. Sampling methods for vegetation and vertebrate wildlife are described in Parmenter et al. (1985), and methods for invertebrate sampling are given in Parmenter and MacMahon (1987) and Parmenter et al. (1990). Briefly, (1) vegetation was sampled on a series of 0.5m² quadrats, (2) rodent population densities were estimated from mark-recapture data, (3) large mammal, reptile and amphibian presence/absence data were collected from general surveys, (4) bird presence/absence data were collected from line transects, and (5) anthropod data were collected from pitfall traps (ground-dwelling species) and barrier traps (flying species).

The vegetation data from the mine sites showed a pattern of increasing dominance by grasses (Fig. 3, Table 1). Floral diversity was greatest on the one-year-old site, due to the large number of annual forbs that had colonized this site. As grasses became dominant, however, diversity decreased. Some shrubs (virtually all four-wing saltbush) eventually became established (see Figure 2, bottom). Although conspicuous in appearance, shrubs did not achieve very high densities or percentage cover (Table 1). The floral communities on the site bore little similarity to the undisturbed site's vegetation (Fig. 4). This was due to the absence of the seeded mineland plants in the undisturbed area; while four-wing saltbush and various wheatgrasses occur patchily in western Wyoming, they are generally restricted to locally favorable sites (e.g., alkali soils for the saltbush, and disturbed sites for the wheatgrasses). As a result, the shrub-steppe ecosystem initiated by the mine company was very different from the surrounding, undisturbed shrub-steppe.

Surveys of vertebrate wildlife on the sites produced 24 species of mammals, 35 birds, 3 reptiles and 3 amphibians (Parmenter et al. 1985). The undisturbed area contained more species of mammals, birds, and reptiles than any of the mine sites; on mine sites with extant ponds, the amphibian species were generally common.

The early colonizing species of mammals on the reclaimed mines included shrews *(Sorex* spp.), deermice *(Peromyscus maniculatus)*, sage voles *(Lagurus curtatus)*, montane voles *(microtus montanus)*, and jackrabbits *(Lepus townsendi)*. Larger carnivores (coyotes *[Canis latrans]* and foxes *[Vulpes fulva]*) and ungulates (mule deer *[Odocoileus hemionus]* and pronghorn antelope *[Antiolocapra americana]*) were observed traversing and foraging on the mine sites, but did not appear to be permanent residents; these animals had home ranges that were much larger than any of our mine sites. Dominant birds on the mine sites were vesper sparrows *(Pooecetes gramineus)* and horned larks *(Eremophila alpestris)*, both of which prefer grassland habitats. No reptiles were ever observed on any reclaimed mine site.

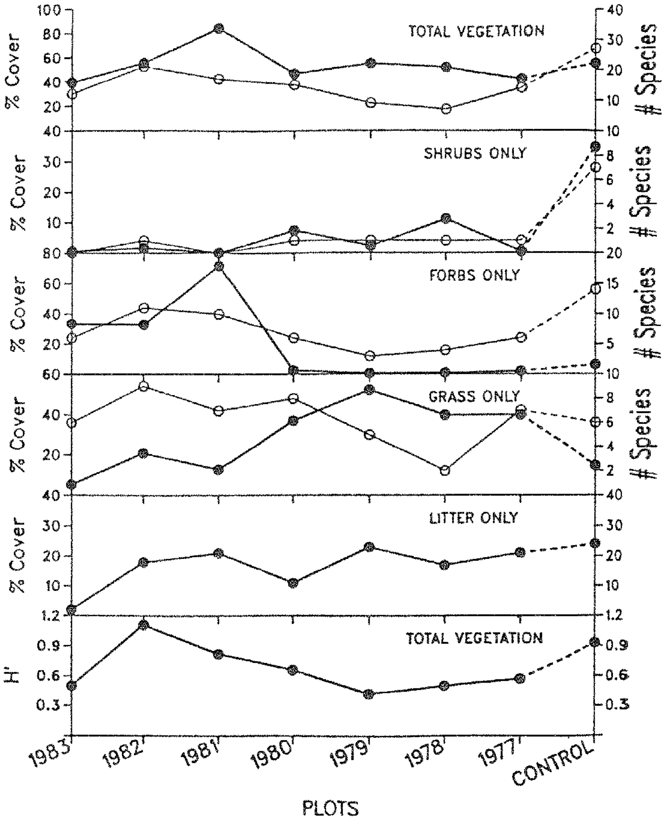

Figure 3.—Early succession trends in vegetation of reclaimed surface mine sites on the Kemmerer Mine, Wyoming. Plots are listed by year of vegetation seeding. Solid circles with heavy lines are percentage cover or diversity. Open circles with light lines are number of species. H'=Shannon-Weiner Index (Margalef 1957) (from Parmenter et al. 1985, with permission).

In analyzing the floral and faunal data from these sites, we first tested the hypothesis that faunal recolonization was independent of time since revegetation. This analysis (linear and non-linear regressions) failed to reject the hypothesis, indicating that time alone was not a significant factor in faunal re-establishment. We then tested a series of 63 potential relationships among floral and faunal community variables. The floral variables included species

		PERCENT SIMILARITY							
		CONTROL	1977	1978	1979	1980	1981	1982	1983
COMMON SPECIES	CONTROL		9	7	7	7	7	10	6
	1977	4		50	62	62	18	15	10
	1978	2	5		81	75	14	9	6
	1979	3	4	5		76	14	9	7
	1980	3	6	6	8		16	16	8
	1981	4	11	6	5	8		31	28
	1982	4	11	5	6	9	13		32
	1983	3	7	4	5	7	9	10	

Figure 4.—Schoener's (1968) percent similarity indices (upper right triangle) and number of "mutually common" species (lower left triangle) from inter-plot comparisons of vegetation on reclaimed mine sites on the Kemmerer Mine, Wyoming. Plots are listed by year of vegetation seeding (from Parmenter et al. 1985, with permission).

Table 1.—Vegetation characteristics of undisturbed and reclaimed mine plots near Kemmerer, Wyoming, 1983, reclaimed mine plots are labeled by year of revegetation (values compiled from Parmenter et al. (1985).

		No topsoil plots			Topsoil plots			
Parameter	Undisturbed	1977	1978	1979	1980	1981	1982	1983
No. Species:								
Shrubs	7	1	1	1	1	0	1	0
Forbs	14	6	4	3	6	10	11	6
Grasses	6	7	2	5	8	7	9	6
Total	27	14	7	9	15	17	21	12
Percent Cover:								
Shrubs	35	<1	11	2	7	0	1	0
Forbs	6	2	1	1	2	72	33	34
Grasses	15	40	40	52	38	12	21	5
Total plants	56	42	52	55	47	84	55	39
Litter	24	2	19	21	11	23	19	22
Diversity (H')	0.93	0.57	0.51	0.42	0.66	0.82	1.10	0.50

richness and percentage cover for the total plant community, for grasses, forbs, and shrubs (separately), and for percentage cover of litter. Faunal variables included species richness for both resident and non-resident mammals, birds, total vertebrates, and deermouse densities (the only rodent species with sufficiently large sample sizes). Of these regression analyses, only one significant relationship consistently emerged: as the shrub component increased in the floral community, vertebrate species richness increased (Fig. 5). Similarly, deermouse densities were positively correlated to shrub cover (Fig. 6).

Few of the vertebrate species present on the reclaimed mine sites fed extensively on four-wing saltbush; hence, the importance of this shrub to the recolonizing vertebrate fauna was probably not food-related. Rather, we suggest that the scattered shrubs on the older mine sites contributed to architectural heterogeneity of the vegetation, providing small patches of shelter (for small mammals and birds in particular), as well as sources of alternative foods (e.g., concentrated seed clumps and insect populations). However, the density of shrubs was still below the threshold needed to support many species of vertebrates. For examples, shrub-dwelling species of rodents (such as long-tailed voles *[Microtus longicaudus]*, least chipmunks *[Eutamias minimus]*, and Uinta ground-squirrels *[Spermophilus armatus]*) and birds (such as Brewer's sparrows *[Spizella breweri]*, sage thrashers *[Oreoscoptes montanus]*, and green-tailed towhees *[Pipilo chlorurus]*) failed to successfully colonize the mine sites. Addition of these species into the mineland fauna will likely occur if and when shrub densities increase; however, this may take many decades to accomplish without additional reclamation efforts.

From the perspective of Clements' successional model, we can not only interpret our Wyoming results but also contrast succession in the shrub-steppe ecosystem with succession in more mesic, forested ecosystem. In terms of vertebrates, mining operations ("nudation") effectively leave no surviving "residual" species; hence, the initial vertebrate fauna is composed of "migrants." As forb and grassland vegetation develop, providing food and shelter resources, vertebrates (generally omnivores) become established ("ecesis"). As shrub vegetation becomes more dominant and begins to modify the architecture of the site ("reaction"), more vertebrate species may be added to the community. As the animal diversity and biomass increases, predators (weasels, badgers, hawks, snakes) can move in and exploit the prey populations ("coactions").

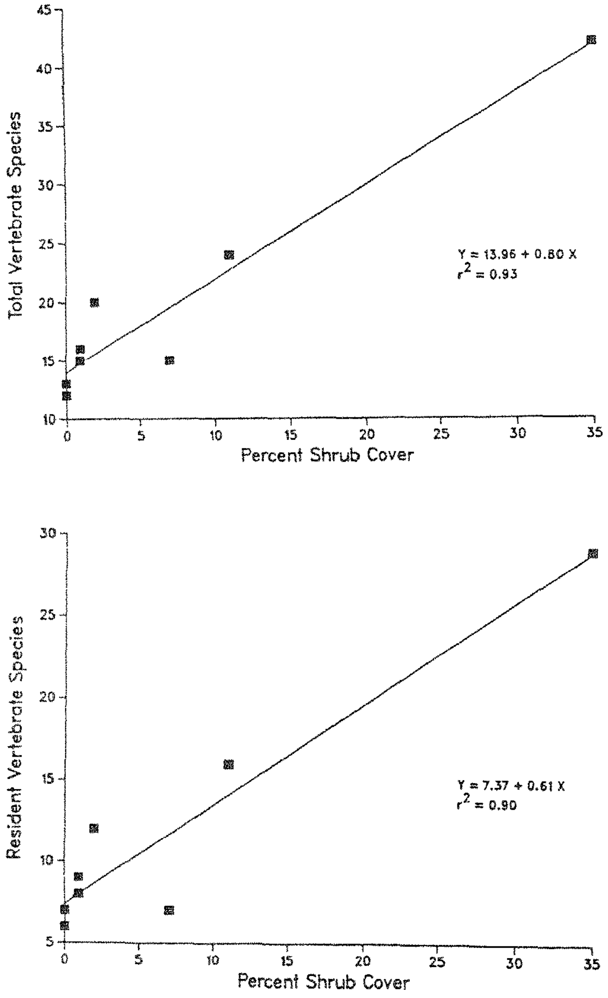

Figure 5.—The relationship between percentage cover of shrubs and vertebrate species richness on reclaimed mine sites on the Kemmerer Mine, Wyoming. Top: Total vertebrate species (includes residents, non-residents, and amphibians). Bottom: Resident species only (does not include amphibians) (from Parmenter et al. 1985, with permission).

In the shrub-steppe, as in other arid and semi-arid ecosystems, the successional process can be viewed as a simple accumulation of species, in which plants and animals are sequentially added to the community without extensive losses or replacement (species turnover). This is because arid lands generally do not progress beyond shrub-dominated vegetation patterns. This type of succession, based predominantly on the initial species list, can be termed auto-succession. In contrast, mine land succession in more mesic, forested regions follows a pattern of distinct species turnover (or relay succession) as a site goes from a forb/grassland system through shrubland into forest. Such patterns have been well documented for small mammals (Yeager 1942; Verts 1957; Kirkland 1976; Sly 1976; Hansen and Warnock 1978) and birds (Karr 1968; Chapman et al. 1978; Crawford et al. 1978; Bejcek and Tyrner 1980; Krementz and Sauer 1982; Schaid et al. 1983).

While general analyses of vertebrate community structure on reclaimed mine lands can yield insights into the complexities of successional processes, such studies often suffer from the logistical difficulties of obtaining large, statistically suitable sample sizes. Field population sizes of many rodents or shrews, for example, may be too small to sample adequately, and in many cases fluctuate dramatically from year to year. Home ranges of larger vertebrates may extend across multiple mine sites and unmined areas, making data interpretation difficult. In addition, detailed statistical comparisons using various indices of vertebrate assemblages (e.g., species diversity, richness, evenness, similarity), when based on small numbers of species and/or individuals per plot, can be virtually meaningless. In contrast, studies of invertebrates (principally arthropods) can yield more extensive data sets from which to evaluate successional trajectories and reclamation success.

Unfortunately, only a few studies have examined arthropod community development on reclaimed mine sites (e.g., Neumann 1971; Usher 1979; Hawkins and Cross 1982; Majer et al. 1982; Urbanek 1982; Schrock 1983; Majer 1985; Nichols and Burrows 1985; Parmenter and MacMahon 1987; Sieg et al. 1987; Parmenter et al. 1991; see also references in Majer 1989). Given the importance of insect pollinators, herbivores, predators, and detritivores to ecosystem funtioning, and the potential for economic impact on reclamation efforts, knowledge of insect recolonization and successional patterns would be useful to ecologists and land managers in their attempts to develop successful strategies of reconstructing disturbed ecosystems.

Studies of the successional development of insect communities on disturbed sites have shown that initial colonization and dominance is generally accomplished by scavenging and omnivorous species, and that the herbivore assemblage changes as a function of vegetation diversity and abundance (Bulan and Barrett 1971; Teraguchi et al. 1977; Southwood et al. 1979; Butt et al. 1980; Force 1981; Hawkins and Cross 1982; Majer et al. 1982; Brown and

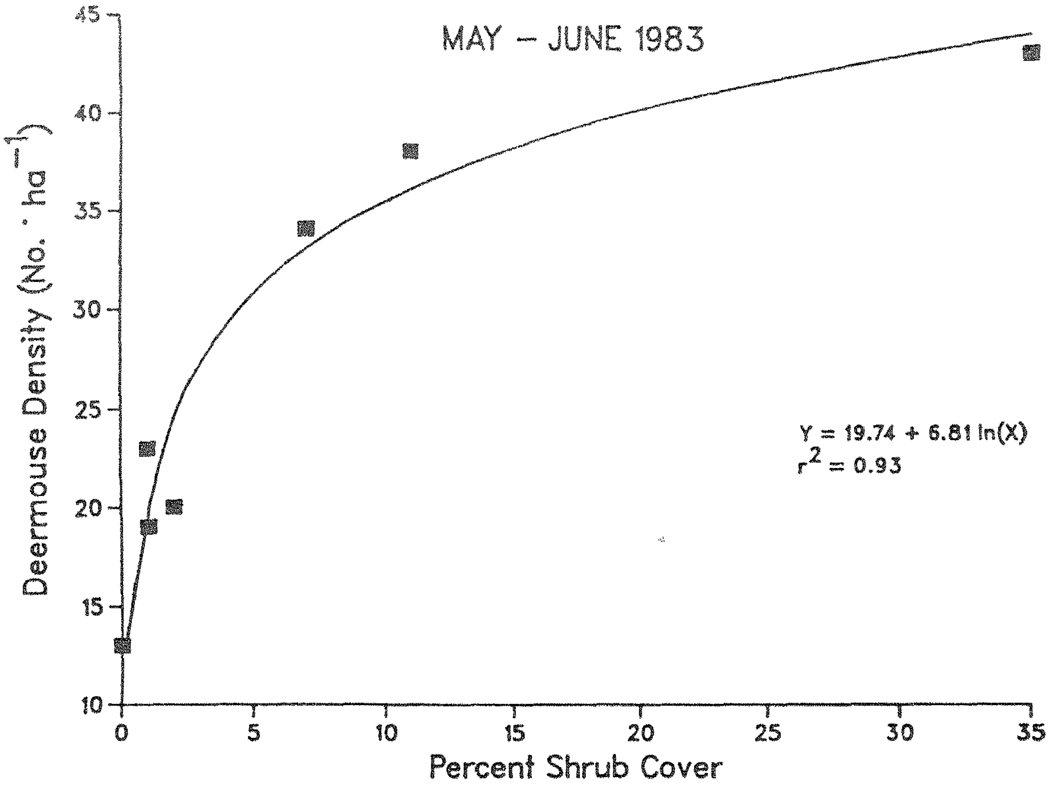

Figure 6.—The relationship between percentage cover of shrubs and deermouse densities on reclaimed mine sites on the Kemmerer Mine, Wyoming (from Parmenter et al. 1985, with permission).

Southwood 1983; Majer 1985; Parmenter and MacMahon 1987; Parmenter et al. 1991). In addition, numerically dominant species in post-disturbance environments are oftentimes exceedingly rare in the undisturbed community; such opportunistic "pioneer" species typically exhibit large and rapid population increases following a disturbance. These successful colonizers benefit not only from a suite of newly available food resources, but also from a combination of changes in the abiotic (e.g., temperature and moisture regimes) and biotic environments (e.g., predation and competition pressures).

On the Kemmerer Mine, we selected two groups of insects as representative taxa for examining trends in faunal community development: beetles (Coleoptera) and grasshoppers/crickets (Orthoptera). Grasshoppers represented an important and abundant group of herbivores/omnivores that should respond to floral community development. Beetles constituted a highly diverse order, with representatives of a wide range of trophic groups (e.g., predators, herbivores, omnivores, fungivores, granivores, palenivores, parasitoids, detritivores, and carrion feeders). As such, beetle assemblages had the potential to illustrate overall trends in the development of the faunal community.

During our field study, we trapped 12,196 beetles, belonging to 98 species in 19 families (Parmenter and MacMahon 1987), and 7,125 orthopterans, belonging to 26 species in 3 families (Parmenter et al. 1991). Summary statistics for the beetle and grasshopper assemblages on the reclaimed mines and undisturbed areas are presented in Tables 2-5. Analysis of beetles species-dominance curves (Fig. 7) showed approximately similar structural patterns, although the reclaimed sites tended to have more rarer species. However, cluster analysis revealed that the beetle assemblages on the reclaimed mines were considerably different from that of the undisturbed site (Fig. 8). A similar pattern was recorded for the orthopteran fauna (Fig. 9). In addition, the beetle faunas on older, grass-dominated sites appeared similar to one another, indicating that successional changes had slowed. We also noted lower beetle species richness and diversity indices on older mine sites (Fig. 10).

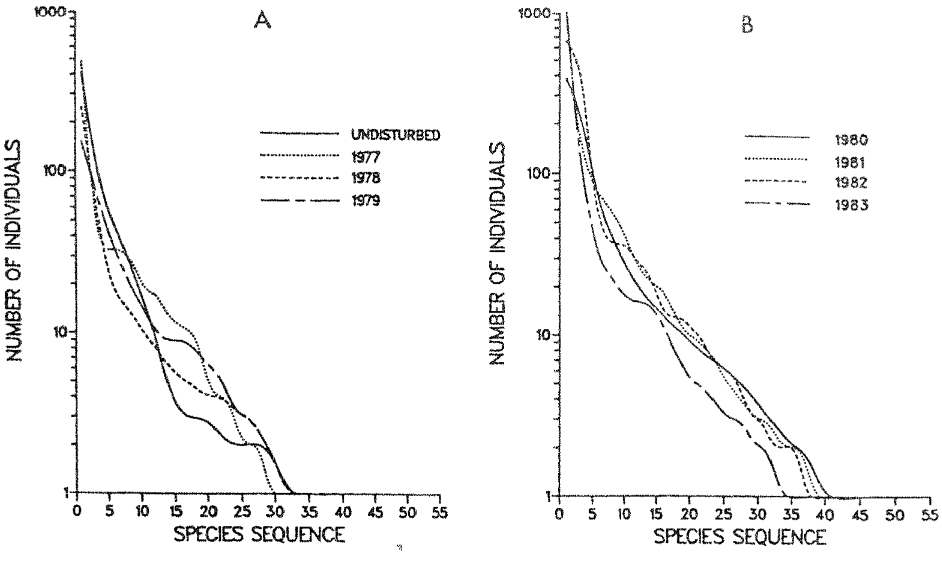

Figure 7.—Species-dominance curves of beetle assemblages on (A) undisturbed and non-topsoiled reclaimed mine sites, and (B) topsoiled reclaimed mine sites (Kemmerer Mine, Wyoming) (from Parmenter and MacMahon 1987, with permission).

Table 2.—Taxonomic and community parameters of beetle assemblages on undisturbed and reclaimed surface coal mines near Kemmerer, Wyoming, 1983; reclaimed mine plots are designated by the year in which they were revegetated (diversity computed using Shannon-Wiener (H') index)

	Plot							
Parameter	**Undisturbed**	**1977**	**1978**	**1979**	**1980**	**1981**	**1982**	**1983**
No. Families	15	13	13	14	14	13	13	13
No. Species	45	35	37	39	52	48	46	44
No. Individuals	1189	1072	684	686	1651	2503	2461	1950
Diversity (H')	0.91	0.94	1.00	1.18	1.12	0.89	0.99	0.77
Evenness (J')	0.55	0.60	0.64	0.74	0.65	0.53	0.60	0.47

Table 3.—Numbers of individuals of beetles in trophic grôups found on undisturbed and reclaimed surface mine plots near Kemmerer, Wyoming, 1983

	Plot							
Trophic group	**Undisturbed**	**1977**	**1978**	**1979**	**1980**	**1981**	**1982**	**1983**
Herbivores	241	52	50	74	103	89	90	154
Predators	277	141	90	130	195	264	242	139
Omnivores	643	340	348	252	1123	722	1551	200
Fungivores	3	12	18	21	38	137	66	419
Dung feeders	5	1	0	0	0	0	1	0
Carrion feeders:								
on vertebrates	17	36	25	104	13	2	2	5
on insects	1	490	153	105	179	1289	508	1033

Table 4.—Numbers of species of beetles in trophic groups found on undisturbed and reclaimed surface mine plots near Kemmerer, Wyoming, 1983

Trophic group	Plots							
	Undisturbed	1977	1978	1979	1980	1981	1982	1983
Herbivores	11	6	7	7	9	10	9	8
Predators	14	11	10	14	16	18	17	18
Omnivores	10	12	11	11	17	14	12	11
Fungivores	3	2	5	4	6	4	4	5
Dung Feeders	2	1	0	0	0	0	1	0
Carrion feeders:								
on vertebrates	4	2	3	2	4	1	2	1
on insects	1	1	1	1	1	1	1	1

Table 5.—Species and numbers of Orthopetera collected from undisturbed sagebrush-steppe site and seven reclaimed mine sites near Kemmerer, Wyoming, summary computations of diversity and evenness based on Shannon-Wiener function, H' (Margalef 1957)

Taxon	Undisturbed sagebrush-steppe	Reclaimed mine sites (No. years since revegetation)						
		0	1	2	3	4	5	6
ACRIDIDAE								
Gomphocerinae								
Aeropedellus clavatus	0	0	4	3	28	2	3	3
Amphitornus coloradus	0	0	0	0	1	1	0	20
Aulocara elliotti	4	2	76	5	11	2	3	20
Psoloessa delicatula	139	0	2	1	0	0	0	1
Melanoplinae								
Melanoplus bivitattus	1	0	0	0	1	0	0	0
Melanoplus complanatipes	5	1	0	0	0	1	0	0
Melanoplus infantilis	25	1	10	1	7	0	0	5
Melanoplus sanguinipes	3	1	362	7	25	64	32	48
Oedipodinae								
Arphia conspersa	12	0	0	0	0	1	0	0
Arphia pseudonietana	1	0	4	0	0	0	0	0
Camnula pellucida	4	0	160	22	6	96	69	229
Circotettix carlinianus	1	1	9	9	0	0	0	1
Circotettix rabula	0	0	0	0	0	0	1	0
Cratypedes neglectus	3	0	10	17	3	0	3	16
Trimerotropis gracilis	2	0	0	0	1	0	0	0
Trimerotropis pallidipennis	11	0	11	8	4	5	0	9
Trimerotropis sparsa	1	0	0	0	1	0	0	0
Trimerotropis strenua	1	0	0	0	0	0	0	0
Xanthippus corallipes	3	0	1	10	0	3	2	6
GRYLLACRIDIDAE								
Rhaphidophorinae								
Ceuthophilus caudelli	48	219	270	133	1031	1023	1948	699
Stenopelmatinae								
Stenopelmatus fuscus	23	0	0	1	1	0	0	1
TETTIGONIIDAE								
Decticinae								
Anabrus simplex	0	0	7	5	1	1	0	3
Steiroxys pallidipalpus	3	0	9	0	0	0	1	10
Total individuals	290	225	935	222	1121	1199	2062	1071
Species richness, (R)	19	6	14	13	14	11	9	15
Diversity, (H')	0.79	0.07	0.68	0.67	0.19	0.25	0.12	0.50
Evenness, (J')	0.62	0.09	0.60	0.60	0.16	0.24	0.13	0.42

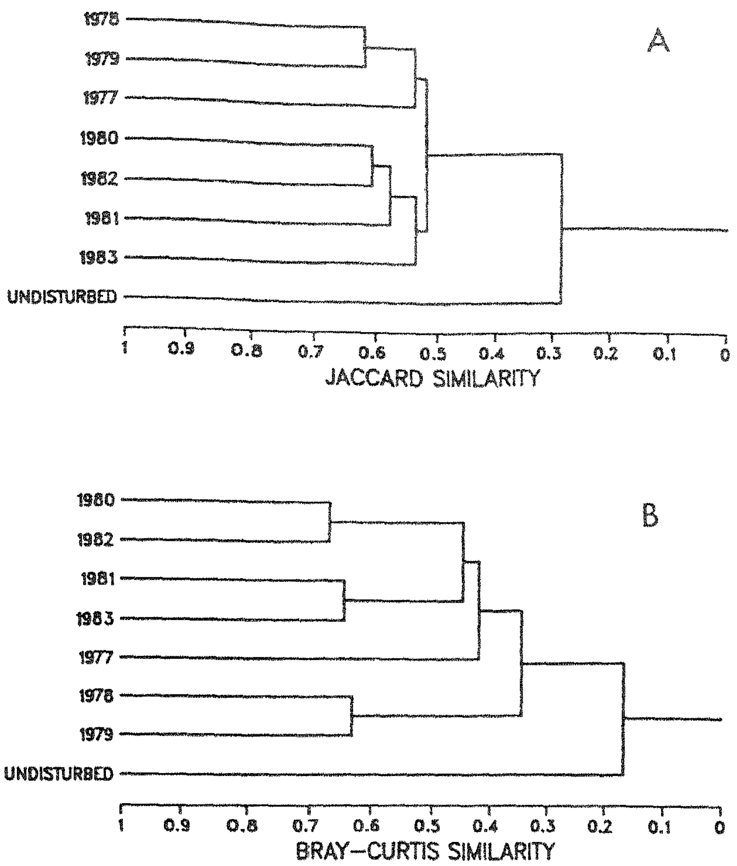

Figure 8.—Dendrograms of similarity clusters from hierarchical cluster analysis of beetle assemblages on undisturbed and reclaimed mine sites of the Kemmerer Mine, Wyoming. (A) Community similarities based on species presence/absence data (Jaccard's [1908] Similarity Coefficient). (B) Community similarities based on both species and numbers of individuals per species (Bray-Curtis [1957] Similarity Coefficient) (from Parmenter and MacMahon 1987, with permission).

The beetle and grasshopper assemblages on the sites proved to be reasonable indicators of vegetation development. Both herbivorous beetles and grasshoppers showed significant correlations between insect and plant species richness (Figs. 11 and 12). Grasshopper diversity also was correlated to plant diversity, although herbivorous beetle diversity was not. In addition, we observed no significant correlation between the total beetle fauna and any floral variable; this was not unexpected, as many of the beetles were predators, omnivores, fungivores, and carrion feeders, and should not be highly sensitive to differences in vegetation. However, some species require shrubs as an architectural resource that ameliorates temperature and moisture extremes (Parmenter et al. 1989a,b). For example, in the undisturbed area, darkling beetles (Tenebrionidae) partition their microhabitats based on the presence/absence of tall shrub cover (Fig. 13). As with vertebrates, shrub architecture can have a significant influence on invertebrate recolonization of reclaimed sites; we found that the two shrub-dwelling species illustrated in Figure 13 (*Eleodes pimelioides and E. constrictus)* had failed to establish populations on the mine sites.

One interesting aspect of the beetle faunal succession was the general decline in the number of fungivores on the reclaimed mines (Fig. 14). As stored topsoil is respread on the mine sites, organic debris (leaves, branches, and small animal carcasses that had been swept up with the topsoil) is suddenly made available to saprophytic fungi. Large numbers of fungivorous organisms can exploit this resource, as evidenced by the abundance of fungivore beetles. However, as this resource (actually, the detrital substrate) is depleted through time, fungivore numbers decline toward the normal background level.

In summary, our studies of the Kemmerer Mine faunal and floral communities, taken together, have shown clearly that

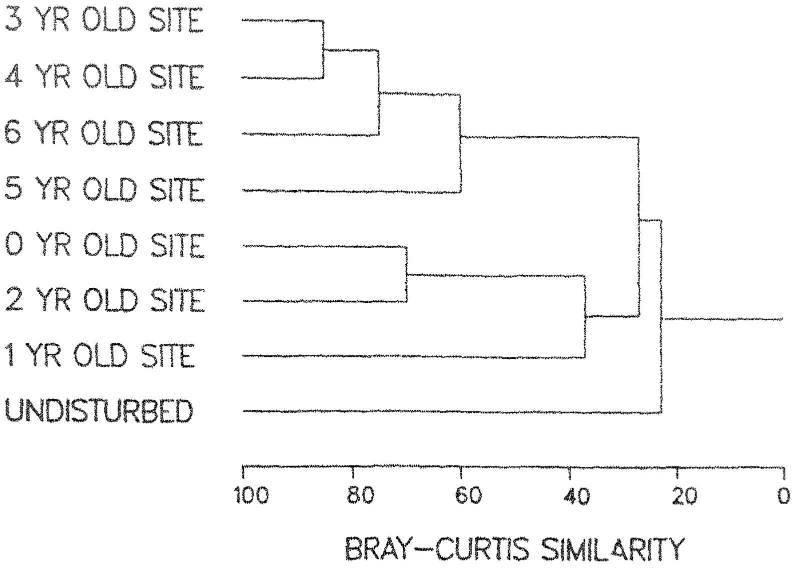

Figure 9.—Dendrogram of orthopteran faunal similarity clusters from hierarchical cluster analysis. Faunas of the disturbed mine sites are listed by the number of years since revegetation. Analyses based on Bray-Curtis (1957) similarity coefficients (from Parmenter et al. 1991, with permission).

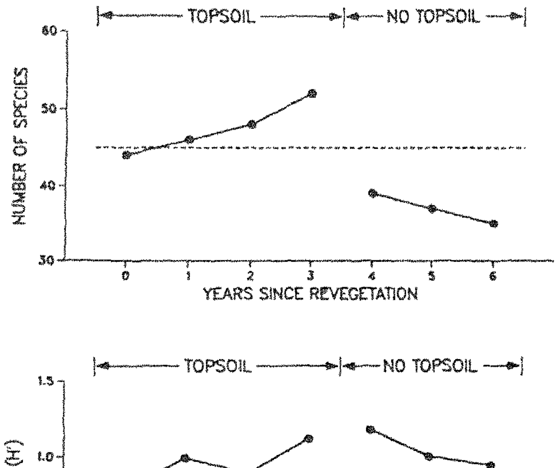

Figure 10.—Early successional trends in the beetle fauna (number of species and diversity) on reclaimed mine sites at the Kemmerer Mine, Wyoming. Dashed horizontal lines indicate undisturbed site values. Diversity was computed using the Shannon-Wiener function (Margalef 1957) (from Parmenter and MacMahon 1987, with permission).

the shrub-steppe ecosystem on the reclaimed sites bears little resemblance to the native, undisturbed shrub-steppe ecosystem. This outcome can be attributed to the use of "native-but-not-resident" shrubs and grasses during the reclamation effort. As a result, the use of an undisturbed area of shrub-steppe as a comparative "model community" is invalid for our study sites. Rather, in the absence of comparable saltbush/wheatgrass communities in the region, we should continue to monitor the successional progress of these sites to determine if they are indeed self-perpetuating and sustainable.

Accelerated Reclamation: An Experimental Approach

In arid and semi-arid environments, one of the most difficu successional stages for shrub vegetation is the "ecesis," o establishment, phase. Shrubs often require several consecutive "good" precipitation years to germinate and establish root systems of sufficient depth to survive drougi periods. As a result, shrub populations in arid lands often consist of a few discreet cohorts, rather than a wide variet of differently aged individuals. This constraint on shrub establishment suggests that the formulation of a successf reclamation plan for a disturbed shrub-steppe ecosystem should include a management strategy that circumvents t "bottleneck" of shrub establishment.

With this in mind, and an appreciation for the value of landscape heterogeneity involving shrub vegetation, we began a large-scale experiment on the Kemmerer Mine tc

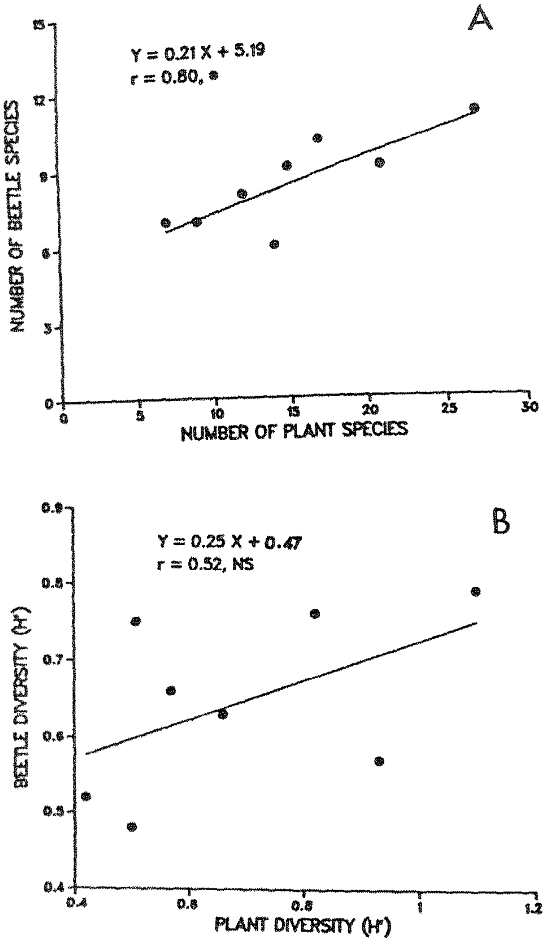

Figure 11.—Linear regression analyses of the relationship between (A) herbivorous beetle species richness and plant species richness, and (B) beetle and plant diversity indices (Shannon-Wiener function) on undisturbed and reclaimed mine sites on the Kemmerer Mine, Wyoming. (*=Significant correlation, $P<0.05$; NS=Not Significant) (from Parmenter and MacMahon 1987, with permission).

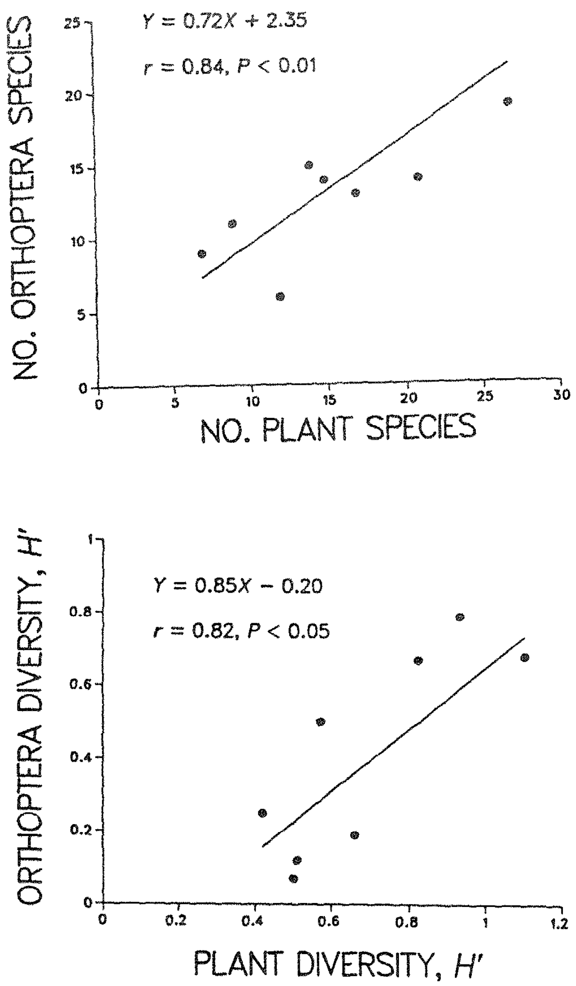

Figure 12.—Linear regressions showing the relationships between orthopteran community attributes and vegetation attributes on reclaimed mine sites near Kemmerer, Wyoming. Top: Species richness; Bottom; Diversity (from Parmenter et al. 1990, with permission).

evaluate successional processes under a variety of management conditions, paying particular attention to both floral and faunal components. The major portion of the research focused on a comparison of two reclamation "strategies." The first was the conventional "agricultural" approach, wherein the mining company utilized farm equipment to drill-seed prescribed plant species in a spatially homogeneous pattern across re-topsoiled mine sites. The second, alternative approach was based on our ecological knowledge of succession in semi-arid, shrub-steppe ecosystems. We hypothesized that young native shrubs, grown in greenhouses for a year and transplanted onto the mine sites in varying densities and spatial patterns, would provide a large number of "islands of fertility." These "islands" of shrubs would act as foci for wind-blown seeds, organic matter, fungi spores, and animal activity, and would eventually expand and coalesce, thereby greatly accelerating the successional process. The use of greenhouse-grown plants would allow the shrubs to avoid the critical germination and establishment period experienced by seeds. While the initial planting of the shrubs would be more expensive than simply drill-seeding the site, we hypotheized that the accelerated development of the biota would shorten the time span required to achieve certification as "successfully revegetated," thereby facilitating the early release of the bond posted by the mine company.

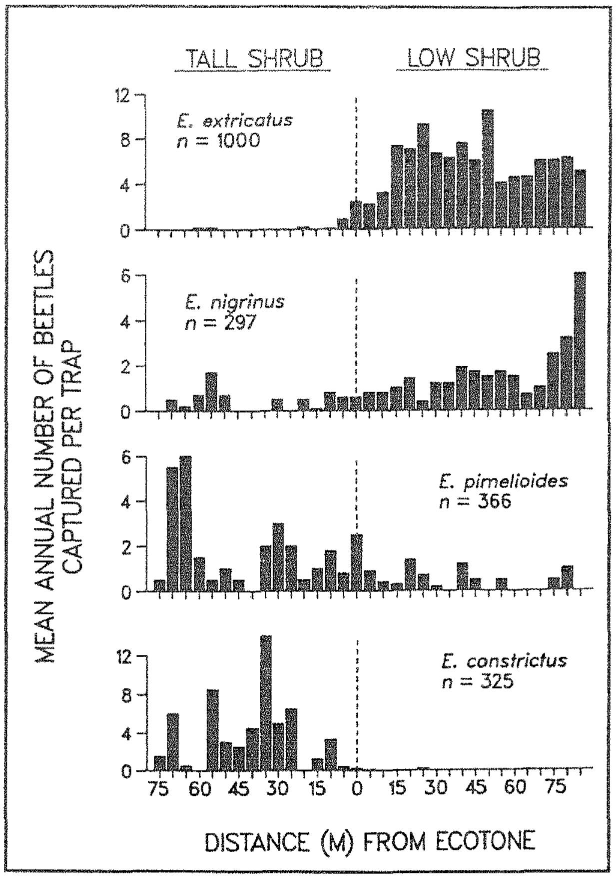

Figure 13.—Spatial distributions of captures of darkling beetles, *Eleodes* spp., in two adjacent habitat types in the undisturbed shrub-steppe site on the Kemmerer Mine, Wyoming. Total sample sizes *(n)* represent pooled data from 1983 to 1986. The ecotone was the interface between stands of tall and low shrubs (from Parmenter et al. 1989a, with permission).

The overall design of the experiment was an incomplete, replicated factorial in which the treatments were (1) soil type (no topsoil, stored topsoil and fresh topsoil, (2) shrub density (normal [16,000/ha], high [2X normal], medium [0.6X normal], and low [0.2X normal]), (3) shrub dispersion pattern (regular or clumped), and (4) shrub species (big sagebrush, Gardner's saltbush, and rabbitbrush). Additional treatments included controls and "conventional" reclamation plots (plots that were revegetated by the mine company using standard practices and seed mixes). Each treatment was replicated 3 times on plots ranging in size from 0.06 to 0.22 ha. Vegetation and animals were sampled annually on all plots from 1982 through 1987. Periodic sampling (at 5 to 10 year time intervals) is planned for the future.

Shrub establishment on the sites was very successful (Fig. 15), with high rates of survival and establishment of big sagebrush and rabbitbrush. Mortality of Gardner's saltbush was about 70% in the first year, but because it was a relatively small shrub, it's loss did not have a significant influence on vegetation architecture of the plots. For forbs and grasses, results to date indicate that the herbaceous communities on the plots are converging to a common (cover-weighted) diversity (Hatton and West 1987), regardless of topographic position, planting density, or freshness of topsoil. The shrub-planted plots also have produced consistently greater numbers of species and higher percentage cover of vegetation than the unplanted controls or the plots seeded by the mine company.

Results of the animal studies have thus far been consistent with our previous successional studies in that during the early stages of succession, the colonizing species and community structure are similar among all treatments. However, as the shrub vegetation has developed, we have observed colonization by shrub-dwelling species, such as least chipmunks, Brewer's sparrows, and the darkling beetles (*Eleodes* spp.) discussed above. Hence, the deliberate inclusion of patches of shrubs for the purpose of diversifying the vertical and horizontal vegetation architecture on reclaimed mines has led to predictable increases in recolonizing animal species.

Summary and Future Needs

From our research experiences on the Kenmerer Mine, we have concluded that faunal analyses can successfully distinguish communities in disturbed and undisturbed ecosystems and among various seral stages of ecosystem succession. Comparative model communities, when appropriate, can be used to test whether a community on a reclaimed site is statistically similar. These test results, in turn, can be used to evaluate the success of the reclamation effort. Faunal assemblages worldwide have been shown to be sensitive to changes in floristics and abiotic variables, and can therefore be expected to reliably perform as indicator variables of successional status in disturbed ecosystems. We would emphasize that an ecologically based, holistic approach to the evaluation of reclamation success, incorporating both abiotic and biotic variables, will have the greatest probability of accurately determining the degree to which a disturbed site has been reclaimed. We also suggest that in future monitoring and research activities, more attention be paid to landscape-scale heterogeneity of vegetation. We predict that landscape heterogeneity, if incorporated into reclamation plans, will increase both floral and faunal richness and diversity. Finally, we recommend that more resources be devoted to the examination of invertebrate assemblages, as these faunas are likely to have extremely important influences on, and responses to, the various successional trajectories of disturbed ecosystems.

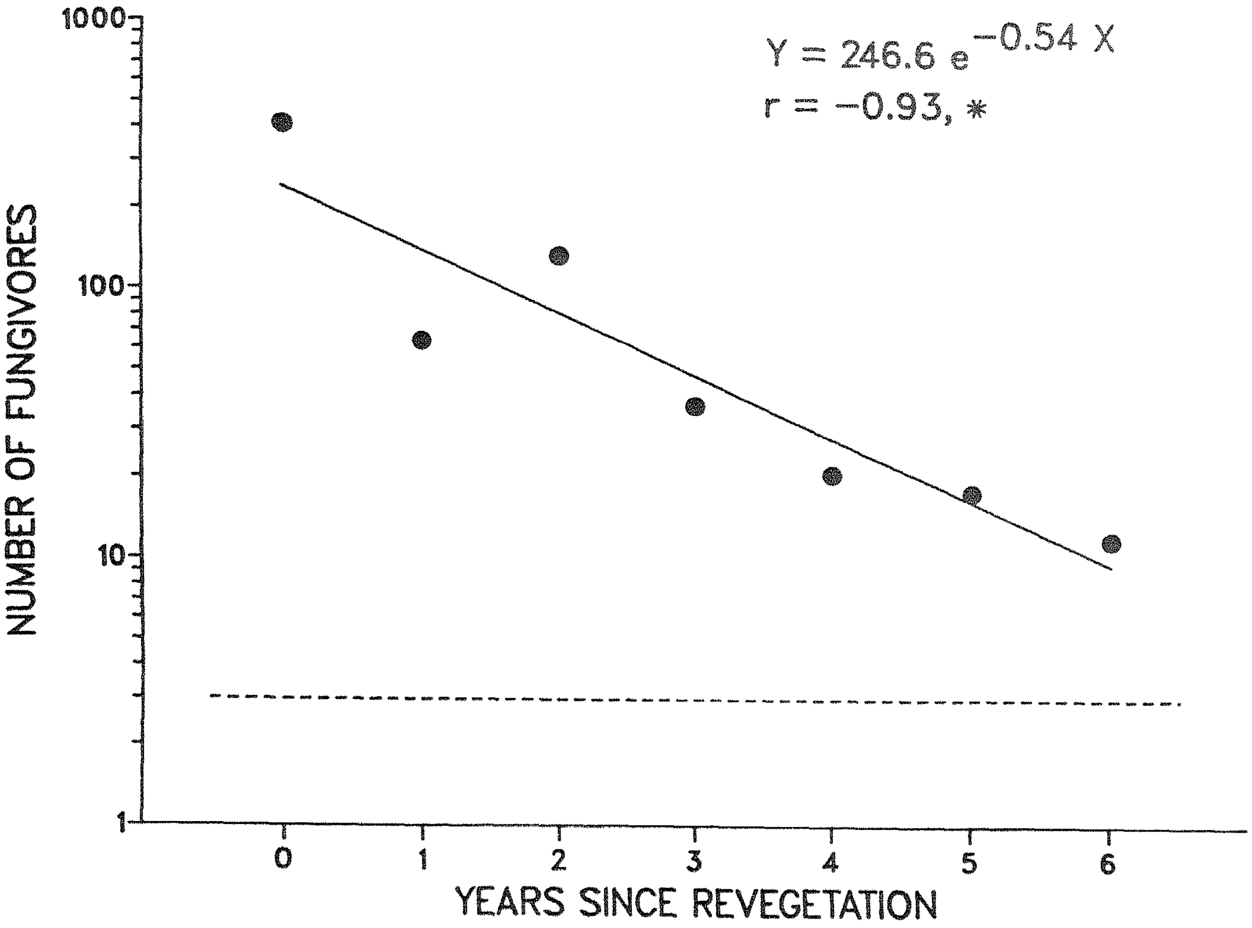

Figure 14.—Temporal changes in fungivorous beetle abundances on reclaimed mine sites on the Kemmerer Mine, Wyoming. The dashed, horizontal line represents the undisturbed site's value (from Parmenter and MacMahon 1987, with permission).

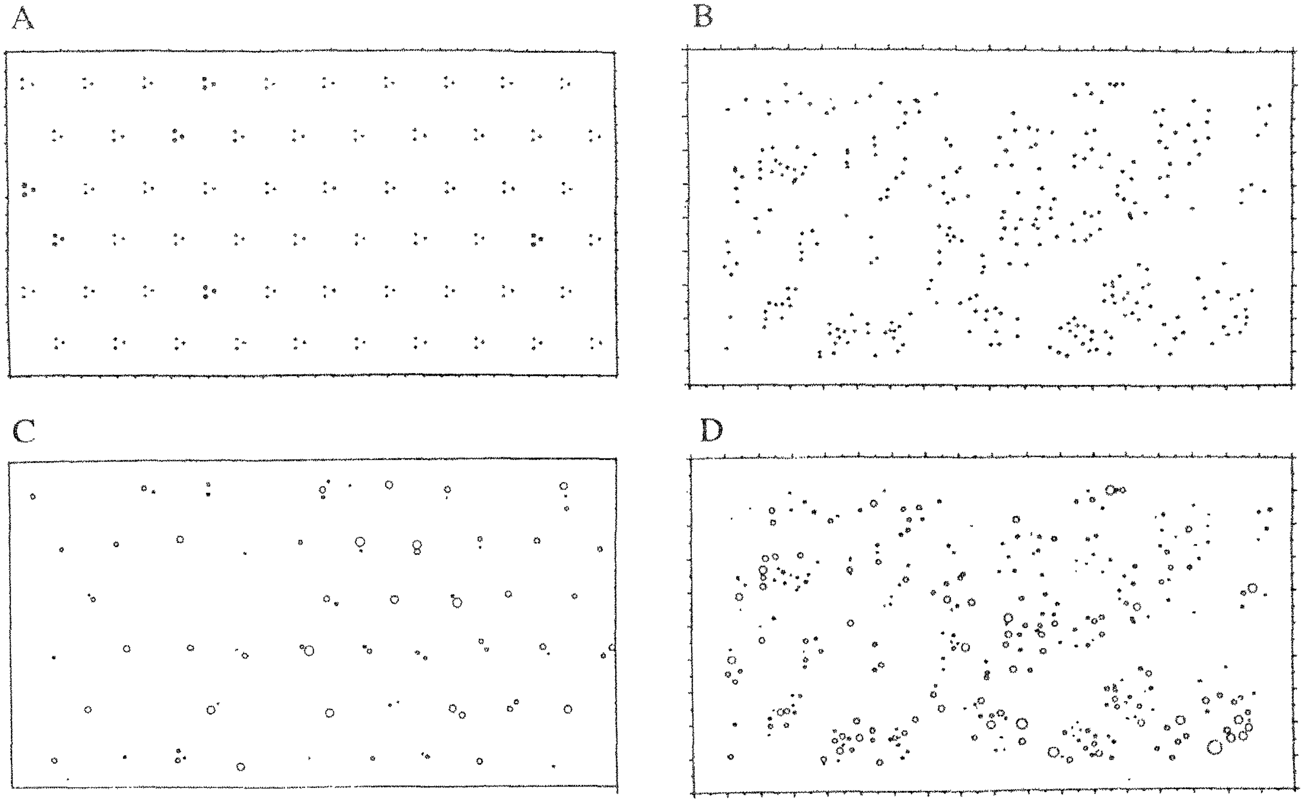

Figure 15.—Examples of distribution maps of shrub dispersion patterns from the Kemmerer Mine reclamation experiment. A,B: Original shrub planting distributions at the start of the experiment in 1982. C,D: Surviving shrub distributions 5 years later (1987); sizes of circles corresponds to diameters of individual shrubs. A,C: The low-density, regular treatment pattern. B,D: The high-density, clumped treatment pattern. Plot sizes are 20x36m.

Acknowledgment

We wish to thank the Pittsburg and Midway Coal Mining Company (CHEVRON, USA) for their cooperation and the use of their lands for the field research. J. Chambers and C. Sieg contributed thoughtful critiques of the manuscript. Funding for this research was provided by the National Science Foundation (Grants DEB81-01827 and BSR 83-17358). Support for manuscript preparation was provided by the University of New Mexico, Sevilleta Long-Term Ecological Research Program (NSF Grant BSR-8811906).

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Landscape Ecology and Reclamation Success

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Abstract

To all landscapes, including reclaimed ones, there is a spatial diversity with both horizontal and vertical boundaries. Understanding of this spatial diversity and the reasons for it can be enhanced by the application of synecological principles, remote sensing techniques, geographic information systems (GIS), and multivariate analysis procedures resulting in improved long-term management of reclaimed or rehabilitated landscapes.

Introduction

Mines have been developed on almost every conceivable landscape. These include agroecosystems and wetlands as well as forest and range ecosystems. The spatial distribution of these landscapes can be considered from a landscape ecology perspective. Landscape ecology is the study of landscapes, their spatial patterns, and how such patterns develop. Landscape pattern is organized in special ways: the component events and patches occur at characteristic scales that are positively correlated in time and space (Urban et al. 1987).

Landscape ecology considers ecosystem functions at a landscape scale. The various definitions include a particular interest in space (Golley 1987). Forman and Godron (1981) defined a landscape as a distinct, measurable unit with several interesting ecological characteristics. Within the landscape is a recognizable and repeated cluster of ecosystems and disturbance regimes. Elements of energy distribution, flow of water, mineral nutrient cycling, species distribution, landform, soil development, ecosystem change, and human impact come into play. Landscape ecology is an integrative science. Its value as a discipline is the viewpoint it affords. The interrelationships between human society and its living space and its perception is a holistic one. If knowledge about a landscape can be organized and put to use, we can think in terms of accomplishing reclamation or rehabilitation of a landscape.

Landscape ecology provides a "new" framework upon which to base land management research and ultimately decisionmaking. Each ecosystem has developed under a different set of external variables and, therefore, each ecosystem has different capacities to resist or recover from stresses (Risser 1985). Landscape structure is a complex arrangement of patches, corridors, and matrices. The levels of homogeneity and heterogeneity and the complexity of such structure are studied to provide information for land managers, including restoration ecologists.

From a biological standpoint, landscape ecology requires understanding of both autecological and synecological principles. Synecology tends to be emphasized within landscape ecology, although ecophysiological and genetic principles must not be ignored. This chapter reviews some aspects of synecology, landscape ecology, remote sensing, and multivariate analysis of ecosystem parameters. The landscape perspective has been used for many years by foresters, range managers, and wildlife managers. The focus has been on forest and range ecosystems and their productivity, and on the edge effect and its importance for wildlife habitat.

Spatial Diversity

Each landscape that we work on has both vertical and horizontal dimensions. The horizontal dimensions are the polygons that we describe when we map the various ecosystems. The vertical dimensions are the geology, soil, soil chemistry, ground water systems, lake and river depths, and other variables encountered in the earthward dimension. The structure of the vegetation and its various layers constitutes another vertical dimension as does the atmosphere. These vertical dimensions must be examined along with the polygons.

Landscape patterns are the consequence of numerous disturbances or perturbations creating patches. Patch implies a relatively discrete spatial pattern, but does not establish any constraint on patch size, internal homogeneity, or discreteness (Pickett and White 1985). Patch implies a relationship of one patch to another in space and to the surrounding, unaffected or less affected matrix. Patch dynamics refers to patch change. Perturbation activities including logging, grazing, croping, and other land uses. In addition, other perturbations can influence these landscapes either pre- or postmining such as fire, flooding, insect infestation, and plant disease.

Clearcutting in forests is one example. Franklin and Forman (1987) used a checkerboard model and five other models to evaluate the probability of disturbance with regard to wildfire, windthrow, and species diversity. Fortunately, ecological restoration (based on restoration ecology principles) can allow for strategies to restore and conserve biological diversity (Jordan et al. 1988).

Patch dynamics as exemplified by changes in patch bodies must be understood as we examine landscapes. Johnston and Naiman (1987) described how patch bodies are either created or modified by beaver impoundments. They further discussed impoundment geographic constraints associated with energy and material transfer in aquatic ecosystems impacted by the beaver. Beaver create patch bodies through browsing pressure, an ability which they share with many other large animals. Like other animals, beaver affect the spatial distribution of landscape resources by transporting materials across patch body boundaries, thus creating patches with new characteristics.

Patches can influence the distribution and abundance of animal species. Influences include the size of the woodland or habitat unit, interpatch distance, and density of similar habitat patches in the surrounding landscape. For example, bird abundance is strongly regulated by such patch dynamics in woodlands. This is the result of a smaller number of individuals that can be maintained in a smaller patch. If the patch becomes too small, no birds can be maintained (VanDorp and Opdam 1987). Another term used to describe both landscape patterns and patch dynamics relative to wildlife is habitat fragmentation. Perry (1988) discussed how various forest pests influence landscape pattern.

Plant synecology deals with the classification and/or ordination of vegetation and the relationships of plant communities to environments (West 1988). West suggests that "it is the ecosystem that responds to management action or inaction, not just vegetation, soils and micro-climate. The land manager would thus like to recognize where one ecosystems stops and another begins and how similar or different an ecosystem is from other areas where he has observed success or failure of a particular management approach."

An important part of the landscape ecology perspective is the question of vegetation/soil relationships as considered from a synecological perspective. First there is no 1:1 relationship between soil and vegetation classification units. However, since many soil boundaries are based on landform boundaries, a better correlation often exists between soil and vegetation map units than with the classification units themselves (Leonard et al. 1988). Certain correlations have been developed between vegetation, soils, and landform. The success or failure of past management can be determined. New management trials and new research results can be extrapolated to new areas with greater confidence based on these relationships.

Reclamation Landscapes Premining: Planning for Reclamation

Mining protocol suggests that a mine will not be developed unless a prudent man can make a profit from the enterprise. Ecologists can suggest an analogy that would state that a mine should not be developed unless prudent men can agree that the environment can retain its ecological integrity and be ecologically restored over the long term as landscapes are mined and managed. However, it is often very difficult to determine if a mine can be restored and maintained after product extraction. This requires a knowledge of landscape ecology, the identification of ecological equivalency among the various ecosystems, and a review of empirical results. Then evaluations can be made of the success of land restoration procedures. (Fig. 1).

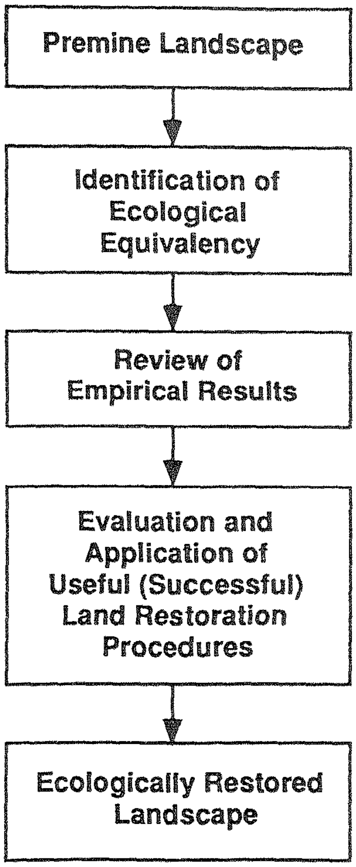

Figure 1.—An often used approach for evaluating information that can be applied to assure ecological restoration of landscapes.

How would we go about doing this? First, the prudent man would recommend that an inventory of landscape resources be developed. Ideally this would include all aspects of the ecosystem. The inventory should include geology, soils, vegetation, wildlife (including fishes, reptiles, and invertebrates), water (including water quality and ground water), air quality, visual resources, and all other important resources both within the landscape to be mined and in the surrounding region since cumulative impacts are important and must be considered. My purpose is to explore how landscape ecology can provide perspective and insight into how best to manage mined landscapes both pre and post mining based on a knowledge of landscape ecology.

The inventory must be followed by a reclamation plan. Within the reclamation plan, consideration must be given to numerous aspects of ecological land restoration. Among these would be such things as surface soil stockpiling, waste rock disposal, tailing disposal, erosion control practices, species selection, recommended species mixtures, supplemental irrigation for vegetation establishment, fertilization, the possible use of container-grown plant materials, seedbed preparation, recommended mulches where thought necessary, maintenance of roads and fences, and a host of other questions and recommendations.

Multivariate Analysis Procedures

In this paper I am emphasizing natural vegetation. There are new alternative procedures for looking at vegetation distribution and vegetation patterns. These involve numerical classification of vegetation and ordination of these vegetation attributes (Gauch 1982). Several ordination approaches are available including reciprocal averaging (RA) and detrended correspondence analysis (DCA). Both of these methods locate samples along ordination axes representing one or more environmental gradients by eigenvector ordination techniques. (Gauch 1982; Whittaker 1967, 1987). Non-metric multidimensional scaling (NMDS) is another ordination technique that is thought to be useful in successfully discovering or documenting gradients of high beta diversity (species changes) (Minchin 1987). These techniques integrate the vegetation data into a form that portrays distribution of species and stands along environmental gradients.

A rotational correlation analysis can be used to determine if various relationships fit along an ordination axis or at some significant rotation at an angle away form the axis. In several desert studies we have developed rotational correlation matrices to examine the relationship of the following factors to vegetation distribution along the axes: (1) a radiation index (based on elevation, slope, and aspect), (2) elevation, (3) total species, (4) shrub cover, (5), grass cover, (6), forb cover, (7) slope, (8) number of shrub species, (9) total cover, (10) total biomass, (11) shrub biomass, (12) grass biomass, (13) forb biomass (Tueller et al. 1991). Numerous other factors can be added to such a matrix, quickly evaluated via the computer, and interpretations made relative to changes on various landscapes. This, of course, depends upon the availability of data and the reasonableness of gathering certain kinds of data. For example, even though biomass information and or ecosystem productivity information is often of great interest, the cost of accuisition tends to be prohibitive.

These multivariate approaches have been designed to evaluate the individual distribution of species in space and time, to identify unique successional trajectories driven by a variety of life history features, and to uncover or discover many possible unique biotic interactions only intermittently operative. There is often a large element of chance especially in the initial floristic composition (West 1988). Also, the multivariate approach can lead to the determination of subtle relationships that may or may not be related to the dominant species on a site (Tueller and Eckert 1987).

The multivariate approach can lead to a better understanding of landscape processes. The predictability of vegetation assessment can be improved. Sites with similar ecological characteristics can be defined and empirical results of ecological restoration procedures can be better extrapolated to those sites (polygons) where high success can be assured.

Landscape Awareness Principles: A Remote Sensing Viewpoint

Remote sensing information useful for landscape management is derived from measurements of electromagnetic radiation by air- or satellite-borne cameras and scanners, video cameras, infrared detection appartus, and by radar and radio frequency receivers. Spatial, temporal, and spectral characteristics of these various systems provide data in various formats. Applications tend heavy toward inventory, evaluation, and monitoring of land resources and the incorporation of remote sensing data to support and improve the decision processes related to wise land use.

With remote sensing we have a view of resources and input to the management of resources that is unique. This view is a synoptic one giving a single-glance intergrative, interpretive impression of large land areas. Such a view has not been previously available prior to the development of remote sensing technology. Remote sensing data, data analysis, and interpretation, along with the synoptic view, given greater understanding of land-based resources such as forests, rangelands, terrestrial and aquatic wildlife habitats, riparian zones, wetlands, watershed and water resources, outdoor recreation, and uses of landscapes for mining.

If we accept the thesis that remote sensing can play an important part in the process of restoration ecology, then it is necessary to describe how this can come about. Remote sensing includes many forms, many kinds of data, numerous scales, numerous spectral bands, and a variety of data gathering and interpretation paradigms. We should further consider how this new knowledge of the ecosystem, alluded to above, and derived from remote sensing might come about.

Numerous authors have described procedures for mapping landscape polygons. For rangelands the procedures have been reviewed by McGraw and Tueller (1983) and Tueller (1989) utilizing remote sensing techniques. These studies need not be reviewed here. One can use remote sensing procedures, using either photo interpretation techniques or digital image processing techniques, to map real landscape differences.

What is not always known without considerable reliable field experience is the exact nature and status of the landscape resource represented by each polygon. Remote sensing technique provides the mapping and can provide some of the actual resource data if the spectral characteristics of the landscape are known (Tueller 1987, 1989). Pixel modeling and vegetation indices and various kinds of resource spectral signatures can provide data sets helpful in allowing understanding of the landscape resources. The coupling of the various data sets, both ground and remotely sensed, can lead to an understanding beneficial to the ecologist charged with the requirement of restoring a given landscape.

Several examples can be given which relate to the distribution of resources in a spatial sense over a landscape. The relationship of drainage, landform, and waterpoints to forage utilization on rangelands constitutes one example. Another example is to examine the levels of productivity of range or forest vegetation as related to landform, soil, insolation, and other factors. A third example would be to consider how landscape diversity provides edge effect and habitats suitable for a variety of wildlife species. Also, numerous algorithms have been developed to determine resource changes and can be used as part of the monitoring process (Jensen 1981).

Recently, my student Norbert Schweyen and I have been able to predict vegetation cover in desert shrublands with Landsat Thematic Mapper data. A high correlation ($r = 0.923$) has been found between desert shrub plant cover and the Perpendicular Vegetation Index (PVI). The PVI is a vegetation index that has the capability of providing information relative to vegetation quantity and quality on arid landscapes. The ability to measure and predict vegetation attributes using satellite digital data can be very valuable for studying pre- and postmined landscapes.

Presently much consideration is being given to the development and application of Geographic Information Systems (GIS). A simple and workable definition of a GIS has been given by Parker (1987) — information technology systems which store, analyze, and display both spatial and non-spatial data. Points, lines, and polygons or areas can all be depicted, overlayed, stored, and retrieved and then changed as land management problems are resolved.

A review of recent papers on GIS suggests to me that this sparkling new technology is doing some very simple things, but doing them rather quickly and efficiently (Johnston 1987; Nellis and Lulla 1990; Moran 1990; Astroth et al 1990). GIS interest, as currently viewed, is thought of as a "final solution" to many resource management problems. This quick fix is unfortunately not provided by GIS and is dependent upon a well-trained cadre of professional applied ecologists and resource managers. A GIS system does nothing new but it does old things fast. It places landscape features and data concerning landscape features together rapidly and in ways that we have not been able to do before. A colleague of mine refers to GIS as simply a different kind of filing cabinet, one with a screen and printer/plotter.

The real value of GIS is in the process of interpreting adjacent/competing land uses and showing quickly where these various uses occur in relation to one another. Probably the greatest potential benefit from GIS techniques is the ability to couple data base information directly to various polygons.

Armed with information accumulated concerning plant synecology, vegetation/soil relationships, and other landscape variables, we are then prepared to evaluate these landscapes using principles of landscape ecology, multivariate analysis techniques, and remote sensing/GIS. This allows the mapping and organization of landscape knowlege resulting in recommendations for ecological restoration (Fig. 2).

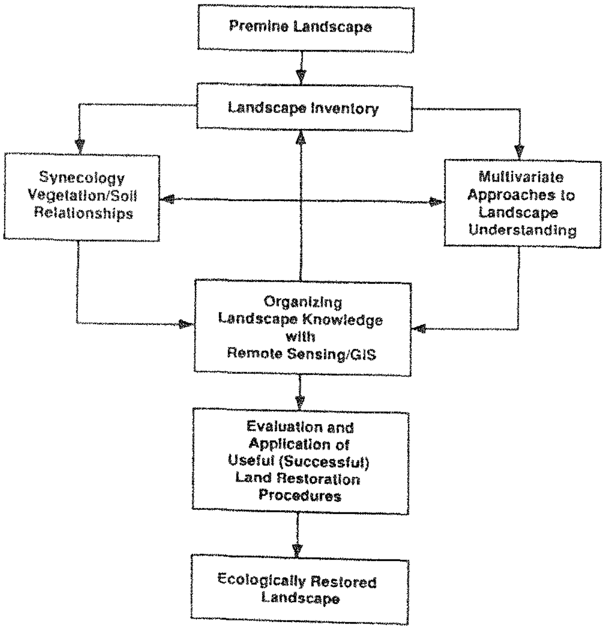

Figure 2.—Four-part diagrammatic model directing information to the ecological restoration of mined landscapes. Product extraction is the objective of a prudent man mine. Land restoration procedures can result in an ecologically restored and productive postmined landscape.

Reclamation Landscapes Postmining: Consequences of Mining

Environmental Effects

Every mine, because of its very nature, results in a variety of land perturbations. These include exploration roads, assessment work, access roads, haul roads, open pits, adits, overburden dumps, waste dumps, tailings ponds, and administrative areas. Each of these land use requirements during the mining process impacts the land. The land is fragmented. Ecosystems are often altered completely. Secondary successions are set into motion and management strategies based on a reclamation plan are invoked to ameliorate the impacts. Landscape ecology provides a holistic view of this process. This leads to understanding which in turn leads to good management and the recreation of landscape usefulness.

Long-Term Management of Reclaimed Landscapes

According to Bradshaw (1988), the endpoint in all reclamation must be a robust self-sustaining ecosystem which, if not achieved for any reason, negates the entire reclamation process. If the best known methods are used and if all underlying problems associated with the site are solved along with repeat and concurrent reclamation, we can reasonably expect success. In some cases the cost may tend to become prohibitive and perhaps the cost will have to be borne by the public interested in environmental integrity rather than the land owner.

The barriers to effective ecosystem exploitation and rehabilitation are often both ecological and sociopolitical (Maguire 1988). I am not sure why Maguire used the term exploitation since that term has a negative connotation. We must think positively and agree that to extract precious products from our landscapes, as demanded by society, is a positive management oriented endeavor and should be viewed in a positive light. This becomes a social, political, legal, and ethical problem and has little to do with a prudent mining human. Either the mining company or one or more public agencies acting in behalf of the environment will have to evaluate, manage, and monitor these lands into the future. The landscape ecology perspective can assist various interested persons and institutions to understand the reclamation process, the necessity for such action, and the probabilities for success.

Economic, sociological, and ecological concerns must be considered in any analysis of a land use system (Franklin and Forman 1987). Great importance must be attached to the evaluation of spatial and long-term consequences of human-imposed landscape patterns. These include such questions as patch configuration and size, edge lengths, and their configuration and disturbance susceptibility, as influenced by the mining process.

Long-term management can be prescribed with greater confidence as suitable landscape ecology data are obtained and insight from remote sensing and multivariate data analysis procedures is used. The long-term management, in most cases, will require only the typical management that would normally be used to manage the various resources found on the site postmining. However, the prudent land owner, whether the mine owner or a subsequent land owner, should be concerned enough about the landscape to see that the land and its vegetation and other resources are monitored. There are many good monitoring methods and scenarios to assist in the determination of ecological resource integrity. Monitoring should include periodic evaluations of several resources: vegetation (timber, wood, pasture, and range), soil (erosion), water (ground and surface), wildlife, and other important resources.

Landscape ecology principles coupled with remote sensing/GIS can assist with the monitoring process.

Cumulative Effects

Whereas cumulative impacts have always been specified in the National Environmental Protection Act (NEPA) process, only recently have impact studies begun to emphasize cumulative impacts. The Council on Environmental Quality (CEQ) defines cumulative impacts as "the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (federal or non-federal) or person undertakes such other actions. Cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time." Cumulative impacts are judged as to type—piecemeal, slow degradation, off-site, homotypic, heterotypic—and as to the way they occur—additive, synergistic, multiplicative, or threshold. Cumulative impact assessment takes a broader view than conventional impact assessment: the boundaries, the number of disturbances, the geographic area, and the time frame considered all tend to be larger (Preston and Bedford 1988). The restoration ecology/landscape ecology approach suggested here, particularly referencing remote sensing/GIS, can enhance the cumulative assessment process on mined land and relate the mining activity to the totality of other landscape impacts in a predefined region.

Conclusions

The outcome of the approach that I have outlined here is a holistic assessment of the reclamation process. This assessment includes premining resource inventory, the development of a reclamation plan, the accumulation of appropriate multivariate data sets, and interpretation of the data via remote sensing/GIS. In particular GIS provides the opportunity to combine data base information with polygons quickly and efficiently. These are analyzed in order to understand the polygons of the landscape and their relationship to one another. The principles of landscape ecology, reclamation planning, multivariate data anlaysis, and remote sensing/GIS combined in appropriate ways to improve the probability of successful restoration of impacted ecosystems.

I was intrigued by Cairns' (1988) fear that some scientists feel that even partial effectiveness in restoration will be viewed as a license for further ecological destruction in the name of progress and growth. They further fear that some of the wilderness areas, national parks, and other ecosystems that now have exceptional protection will have this protection reduced as a consequence of the feasibility of repairing damage following exploitation of various resources in these systems. Cairns further asserts that some attempts will almost certainly be made to intrude on previously protected areas on the grounds that the restoration process is now sufficiently advanced to make this possible. Being naturally optimistic, I have less concern. However, I will concede that we have much yet to learn to maintain the integrity of our landscapes. Landscape ecology and each of the subjects of this symposium offers hope in the process of landscape restoration.

We are looking toward new levels of landscape understanding. Landscape ecology, remote sensing, synecology, vegetation/soil relationships, GIS, and multivariate statistical approaches to vegetation analysis can lead to improved management models based on new and innovative methods. An improved approach to landscape understanding is the result of these integrated activities.

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Cumulative Effects of Mining on Hydrology, Water Quality, and Vegetation

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Abstract

Evaluating the cumulative effects of mining and other land disturbances at the watershed-scale poses a challenge to researchers and practitioners. Offsite cumulative effects may arise as a result of multiple mining disturbances occurring simultaneously within a watershed or from the interaction of mining activities and of adjacent land uses (grazing, residential development, recreation, timber harvest). Progressive surface and underground mining may generate onsite cumulative effects by modifying soil and vegetation composition, slope stability, and surface erosion potential. In addition, this temporal sequence, typical of many coal mining operations in the East and gold mining operations in the West, can affect offsite resources such as water chemistry, channel sedimentation, riparian vegetation, aquatic habitat, and peak flows. To address these important issues of cumulative effects, we need to consider the physical, chemical, and ecological processes within the time and space in which these activities occur.

Introduction

The concept of cumulative effects involves the interaction of natural ecosystem processes with the effects of land use distributed through time or space, or both. At the watershed scale, mining activities often occur in combination with other land use such as road construction, residential development, agriculture, timber harvesting, grazing, and recreation. Interaction among these activities can result in effects on offsite resources (water quality, aquatic habitat) that are cumulative through time or that are not accurately predictable when evaluating each land use separately. Onsite cumulative affects can result from sequential mining activites and may affect soil development, vegetation composition, and erosion potential.

Concerns regarding cumulative effects related to mining vary by region. For instance, in Appalachia, coal mining may occur in the same watershed along with grazing, agriculture, or woodlot management. All of these activities can accelerate soil loss and the delivery of sediment to streams as influenced by their spatial and temporal distribution. In the Northeastern and Lake States there are concerns over effects of various types of mining along with concurrent impacts from atmospheric deposition, insect epidemics, and the increasing level of forest harvest. These activities may lead to depletion of nutrients from forest soils and changes in chemistry of streams, lakes, and groundwater. In the West the recent increase in gold mining has encroached upon wilderness areas and land uses such as grazing and timber harvest. The major concerns about effects relate to erosion, fish habitat, and riparian ecosystems.

Progressive Underground Mining

Underground mining operations have the potential to generate both on- and offsite cumulative effects. An important example of an onsite effect is in the progressive underground mining of coal seams. When only minimal structural pillars are left in place, land subsidence may occur. This is exacerbated in seismically active areas. Subsidence is common around underground coal mines in the following regions: agricultural lands in the Midwest (Darmody and others 1988); rangelands in Colorado, Utah, and Wyoming (Dunrud 1976; Degraff and Romesburg 1981); and anthracite mine communities in northeastern Pennsylvania.

Subsidence not only destroys property and alters topography and surface drainage, but also influences outlets for acid mine drainage, and may create areas of future instability by oversteepening of slopes and redirecting of subsurface flow. The generating of acid drainage by opening new pyritic deposits to oxidation and allowing redirected subsurface flow to contact these deposits is a major offsite cumulative effect. Acid mine drainage can be a problem in areas where no subsidence has occurred. However, the additional flow pathways created by subsidence would complicate the problem as well as any ameliorative solutions.

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Surface Mining

Surface mining creates the greatest environmental disturbances of any mining technique because of the vast land areas disturbed, the volume of spoil material generated, and the changes in the properties of the disturbed matrix. Typically, strip mining of coal proceeds in a progressive manner whether the operation involves overburden removal or relatively flat terrain, contour mining along coal outcrops in relatively steep terrain, or mountaintop removal. Temporal cumulative effects from these mining practices can arise because of the lag time in reclamation and changes in the hydrologic function, erodibility, and nutrient status of the reclaimed spoils.

Traditionally, hardrock mining employed underground techniques and the laws governing reclamation of hardrock mining were tailored to these early extraction methods. Recently, surface mining methods have been adapted for removal of lower grade ores of precious metals. Most notably, the cyanide-heap leach process has made profitable the extraction of gold from very low-grade ore bodies. The process has opened up large land areas in the Intermountain West for goal exploration and mining. Cumulative effects related to these activities may occur because of both the vast spoil deposits that must be reclaimed and the open pit mine sites that are not required to be filled under current laws.

The exploration phase of precious-metal mining operations can cause considerable disturbance. Core drillings normally are taken in a grid matrix to determine the location of the high-grade ore body. When a high level of precious metal is assayed, a more detailed grid system is overlayed in the immediate vicinity to map the ore body more accurately. This exploration procedure requires a vast network of low-grade roads to facilitate movement of drilling rigs from point to point. Many times steep and unstable slopes must be accessed. In relatively steep watersheds, exploration roads crisscross mountainsides and potentially concentrate surface and subsurface runoff, increase the hazard of slope failure, accelerate surface erosion, and detract from the esthetic quality of the natural landscape. Since the exploration phase of precious-metal mining requires more intensive road networks than during the extraction phase, the temporary roads must be considered in a cumulative effects analysis. Similar problems exist in the exploration of oil and natural gas, especially in mountainous terrain.

Long-Term Weathering of Spoil Material

Soils are a product of climate, vegetation, topography, parent material, and time (Jenny 1941). Soil development of mined areas is unusual because vegetation, topography, and even parent material to a limited extent can be controlled. In the eastern coal region the climate is humid, precipitation is well distributed throughout the year, and vegetation growth is good. These conditions probably would encourage soil development on disturbed areas such as mines. However, some geologic strata in this region have experienced accumulation of solutes or formation of sulfides in reducing environments during past depositional periods (Evangelou and Karathanasis 1984). The effects of these depositional minerals on soil and water acidity can be predicted by physiochemical relationships. Soil development does appear to be quite rapid in the eastern coal areas. Wade and Halverson (1988) found recognizable O, A, and B soil horizons had developed after 22 years under both hardwood and conifer plantations. The majo difference among vegetation types was in the 0 horizon. The organic layer was thicker under a mixed pine than under hardwood plantations.

Comparisons also were made with the undisturbed soil adjacent to the mine. Soils were less developed when the A horizon was approximately 3 cm on the mine and 10 cm in the surrounding forest. However, when the soil chemistry on and off the mine was compared there were few significant differences between mined and unmined sites. Thirty-one soil variables, including both nutrients and toxic materials, were measured and only K, Mn, Cu, Cr, and A horizon thickness were found to differ significantly from adjacent unmined soils (Wade and Halverson 1989). This site appears to be returning rapidly to original conditions.

Davidson and others (1988) studied the progression of soil pH in several eastern coal areas over many years and found a significant pH shift toward premining conditions. These results also suggest soil development over short periods.

In a study of 70- to 130-year-old shaly iron ore spoils in northern West Virginia, Smith and others (1971) found that rooting depth was greater in mine spoils than in nearby native forest soil in spite of the higher bulk density and better structural development in forest soil. Forest site quality, pH, and mineralogy were not greatly different between natural soils and mine soils.

Mine spoil development in southeastern Kansas was evaluated one year after reclamation of a coal mine (Welch and Humbleton 1982). The only visible pedogenic development was the physical weathering of shale fragments, due largely to freeze-thaw activity. The chemistry of the mine soil was similar to that of native soil in the area, except the pH was approximately one unit lower in mine spoil samples.

Multiple Land Uses

Road construction is a common companion of mining because there may be no existing roads or only an inadequate network. The impact of roads has been mixed. Earlier mined, recently mined, and unmined watersheds in Tennessee were instrumented in the early 1980's to evaluate the effect of roads (Woods and others 1986). The combined effect of road construction and mining was not uniform. Total dissolved solids (TDS) were greatest on newly mined watersheds where levels approached 1,000 ppm. On earlier minded watersheds, TDS levels were in the hundreds and unmined watersheds had values ranging from 10 to the low hundreds.

In general, magnesium and sulfate levels were lowest in undisturbed watersheds and watersheds with old mines. Conversely, sodium levels were high on undisturbed watersheds and newly mined watersheds. These results were further confounded by physiographic regions with Cumberland Mountain watersheds responding differently from Cumberland Plateau streams.

The cumulative effects of road constructed and mining are not simply additive. Rather, effects appear to be a complex function of road and mine location in the watershed, road construction and mining technique, geologic substrate, and reclamation efforts on both old roads and mine surfaces.

In the West, grazing often occurs within the same watershed as mining. The effects of grazing have been cumulative through this century and have resulted in catastrophic channel incision and drastic changes in riparian vegetation in many areas. Recent mining activities in these watersheds may exacerbate fine sediment inputs to streams, but the long-term effects of mining activities on channel morphology are uncertain (Sidle 1989). In Appalachia, conventional or limited-till agricultural practices often occur near active coal mines. The effects of these agricultural practices on sediment and nutrient transport must be evaluated along with those from mining.

Large strip mining operations in forested settings require extensive clearcutting before exploration and earthmoving. Depending on lag time between timber harvest and actual mining disturbance, harvesting may affect sediment production. Additionally, harvest activities may occur in the same watershed independent of, and concurrent with, mining operations. This poses additional concerns regarding cumulative effects, depending on harvesting techniques and silvicultural methods.

In certain areas of the East, residential and industrial development occurs in the same drainage as mining. In addition, reclaimed mine sites sometimes become sites for housing tracts or businesses. This is especially true in Appalachia where flat, accessible land is limited. Variations from approximate original contour reclamation of coal strip mines permit such development to occur (Zipper and others 1989). Thus, the evaluation of potential cumulative effects on such resources as water quality must address not only the effects of reclamation, including revegetation and changes in spoil chemistry, but also the influence of the new land uses.

Strip mining and associated road construction in steep terrain increase the potential for landslides (Sidle and others 1985). These mass failures, together with erosion caused by other types of land use and sources, contribute to the overall cumulative effects on basin sedimentation. Mining activities in steep terrain generate massive quantities of spoil material that require placement of spoil material at sites that already may be unstable. Most reclamation efforts primarily are concerned with revegetation of spoil material with little emphasis on long-term stability. Factors affecting the amount of regrading needed on hillsides include depth of overburden removed, thickness of the extractable resource, and slope gradient. Recommendations have been made to regrade and compact spoil material to approximately 34° on outslopes (Allen 1973); however, this may be difficult or undesirable in steep terrain.

The Surface Mining Control and Reclamation Act of 1977 requires mine operators to regrade mine spoils to approximate original contour. This concept may have shortcomings when applied in steep terrain because regraded spoil material may have significantly different hydrologic and geotechnical properties from native soil (Sidle and others 1985). The natural slope-regraded spoil interface may provide an area for positive-pore water pressure to develop, and this may initiate shallow landslides. Research on coal mines of Appalachia reclaimed by approximate original contour methods indicates potential instability in many of the highwall backfills (Bell and Daniels 1985; Bell and others 1989). Alternatives to original contour reclamation at steep sites may need to be considered to afford better protection for slope stability (Zipper and others 1989).

Only a few detailed inventories are available that assess the degree of landsliding associated with mining. A study by Lessing and Erwin (1977) indicated that 8 percent of the landslides in seven greater urban areas of West Virginia (3,367 km² total area) were attributable to mine spoil banks. The major causes of these slides in coal spoils were overloading slopes with poorly consolidated spoils, inhibiting subsurface drainage, and oversteepening slopes. A study of abandoned tailings from silver, lead, and zinc mines in northern Idaho indicated that there was a low potential for mass failure in more than 80 percent of the sites surveyed (including both reclaimed and unreclaimed spoils) (Gross and others 1979).

Surface Erosion

Accelerated erosion frequently follows any form of land disturbance. Land disturbance due to mining is a particular concern because large areas and steep terrain often are involved. The absence of a protective vegetative cover during and immediately after mining also contributes to erosion.

Erosion studies in the eastern coal region in Tennessee and Kentucky showed greatly accelerated erosion after slopes had been restored. Some of the slopes were steep, exceeding 50 percent. The site in Tennessee lost 33.9 mm of soil material while the steep slopes in Kentucky lost 23.8 mm, and gentle slopes had no loss (Curtis and Superfesky 1977). However, the erosion was not uniform in time or space. The bulk of the erosion occurred early, before vegetation was established, and at a point about 100 feet below the top of the slope. Rills and gullies also tended to form on long, uninterrupted slopes. Davidson (1981) concluded that vegetation was the most effective erosion control measure although terraces, mulches, and chemical stabilizers can be helpful (Plass 1987).

Mines in western coal regions showed similar wide variation in soil loss, with erosion removing 2.5 to 50 mm per year from an Idaho mine (Morris and Taylor 1987). Erosion rates on an Idaho phosphate mine were measured during simulated rain storms. Again, erosion rates were variable but not related to slope position as had been found in eastern studies (Hart and Whitson 1988). The conclusions of the phosphate mine studies suggested the incorporation of coarse material into the surface layer as an effective erosion control measure, but the subsequent development of a vegetation cover was necessary for long term protection. Toy (1989) evaluated geologic and accelerated erosion rates and found natural erosion rates of 2.7 mm and erosion on reclaimed slopes averaged 4.1 mm. These values were not considered significantly different. The slopes were modest, less than 15 percent, and the results suggest erosion is not greatly accelerated by mining on gentle slopes.

In-Stream Sedimentation

The stream channel provides an important focal point for cumulative watershed effects because it integrates the offsite effects of many land uses and natural events that occur throughout the drainage. In particular, channels respond to changes in sediment load—as sediment load increases, channels become more shallow and wide. Also, the size and distribution of pools, and important habitat feature for fish, can change with increasing sediment loads. Increasing of the fines in streambed gravels that result from accelerated sediment load can restrict oxygenation of incubating fish eggs and impede fry emergence (Cordone and Kelly 1961; Crouse and others 1981).

A central Nevada study on a watershed that has experienced long-term grazing and more recent mining showed that pools were less frequent and the channel wider in the most heavily grazed reach (Sidle 1989; Coffman and others 1989). While mining had no immediate effect on channel morphology, the fine sediment in streambed gravel was higher near a large mine dump than in the rest of the channel (Sidle and Amacher 1990).

Peak Flows

The effects of mining on peak flows and floods has long been a controversial subject. Flooding has been attributed to any form of land disturbance, including mining. Disturbances such as timber harvests are generally recognized as mechanisms to increase water yield. The impacts of mining, since they extend into geologic formations beneath the surface, have been more difficult to define.

In the eastern coal region, hydrologic records from smaller watersheds have been analyzed to isolate the effects of mining (Bryan and Hewlett 1981). The results showed that storm runoff volumes were unaffected by mining but the hydrograph was altered. Small peaks were increased in the summer, but the effect on the peak flow declined as the peak increased. The largest peaks were less than anticipated on mined watersheds. The decrease was attributed to the disruption of impervious geologic strata in the watershed followed by development of a greater water storage capacity in the material (Curtis 1978). Additional evidence supporting the concept of reduced peaks from major storms over mined watersheds was derived from impoundment studies. In a study of major storms, 41 percent of the precipitation flowed from a mined area into an included pond while 76 percent of the precipitation appeared as flow from a control watershed (Curtis 1984). The mining operations in these areas had a mixed impact with some increases in peaks from minor storms but also some reductions of peak flow, especially during large storms. At least three other studies in Appalachia (Collier and others 1979; Curtis 1972; Sidle and Menser 1980) report increases in peak flows related to mining activities in watersheds. Increased peak flows were in part attributed to concentrated areas of soil compaction and disturbance near stream channels (Sidle and Menser 1980).

Some of these results must be interpreted carefully because not all include the impact of current reclamation practices. Areas that were permitted, mined, and reclaimed under the new regulations have not been instrumented for a sufficient period to draw conclusions.

Water Quality

The effects of mining on water quality are varied and depend in part on factors such as the mining technique, overburden properties, and regional climate and hydrology. Two general types of water chemistry problems can be identified. The first case involves chemistry changes directly attributable to current or recent mines. The second situation involves acid mine drainage from old or abandoned mines. Mine drainage often appears after mining, and even reclamation, is completed.

Mining operations may produce elevated levels of sediment and dissolved minerals in surface waters, especially during active mining and before reclamation efforts become effective. In general, water quality tends to improve with time since mining with water quality variables approaching premining levies (Becker and others 1986). About 15 to 20 years are required for water quality in the eastern coal areas to approach premining conditions. Effective reclamation presumably would reduce the time needed for recovery.

Acid mine drainage in the eastern coal region is a major environmental problem. In southern Kentucky, seep waters have been measured where pH is less than 3, and sulfate and iron concentrations are in thousands of ppm (Halverson and Wade 1988). Even small flows of such contaminated water can have significant impacts on surface water quality. Both chemical and biological treatment schemes are used to treat acid mine drainage, but costs are high.

Summary

The cumulative effects of multiple land use remain undefined and are further confounded when mining occurs on the watershed. Mining, due to the subsurface disturbance, can be a major contributor to changing the ecology and hydrology of an area. However, research results have shown that the impacts of mining have great variation. Mining can have minor and temporary impacts or the changes can be major and present over long periods.

The effects of mining on hydrology depend on original watershed characteristics and reclamation techniques. Hydrology may be altered to favor greater flows and peaks in some watersheds. However, proper use of wetlands, ponds, and other impoundment techniques as well as the production of deeper soils can mitigate some of the effects.

Water quality generally declines during and immediately after a mining operation. However, the effect is temporary and water quality tends to return to pre-mining conditions after the area is reclaimed. Other reclamation procedures, such as spoil placement, offer the promise of further reducing the time required for water quality to recover.

Mining can be disruptive to the vegetation, soils, and hydrology of a watershed. We should view the potential changes as an opportunity and not simply as the source of problems. While much information remains to be developed, completed research suggests that careful planning and implementation of reclamation can yield a site and ecosystem to meet future needs. For example, water yield can be increased by creating long slopes protected with coarse fragments. Conversely, water yield could be decreased through the use of impoundments or terraces and by taking advantage of increased soil moisture storage. Planning, including clear objectives for the ultimate use of the area, is key to developing the ecosystem needed to meet future objectives.

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Evaluating Reclamation Success Using Ecological Principles: A Holistic Approach

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The reclamation of entire ecosystems is one of the greatest challenges that ecologists face. Because of the worldwide demand for mineral resources, ecosystems containing these resources are often severely and progressively disturbed. Only within the past 20 years have laws been passed in the United States requiring that disturbed ecosystems be reclaimed to resemble predisturbance ecosystems. In many cases, we have been learning about the development of these ecosystems at the same time that we have been attempting to evaluate if reclamation success has been achieved. This has resulted in a synergism in which ecological principles are used to reclaim ecosystems and to evaluate reclamation success, while the data gathered are used to develop new principles and to test ecological theory. This chapter attempts to synthesize some of the more important new principles as presented in the previous chapters.

A holistic approach for both achieving and evaluating reclamation success is emphasized throughout this volume. Successful reclamation requires fundamental knowledge of both biotic and abiotic factors and of ecological processes at several different levels and scales. Because our focus is on building ecosystems from the bottom up, it is necessary to start with the properties of the reconstructed soils. Soil physical and chemical characteristics are an extremely important component of ecosystem structure. Severely disturbed soils, e.g., minesoils, are often poorly developed as there has been little time for active soil formation. Differences in reconstructed soils and native soils include higher percentage coarse fragments and lower soil water storage capacity due to higher bulk density and lower saturated hydraulic conductivity (e.g., McFee et al. 1981; Hauser and Chichester 1989). Measurement of soil properties does not give a direct assessment of ecological functioning, but often provides a more accurate indication of potential long-term vegetation productivity (Doll and Wallenhaupt 1985) and of successional trajectories (Biondini et al. 1985) than vegetation characteristics of newly restored sites.

Measures of the activity of the soil microflora and microfauna can serve as important indicators of the level of ecosystem functioning. Decomposition and nutrient cycling are closely linked with the activity of the soil microflora (bacteria and fungi) and the microfauna (protozoa, nematodes, and microarthropods). Because they require carbon and nutrients for growth, fungi and bacteria in soils constitute an important sink and source of essential plant nutrients. Assessment of the soil microbiota to evaluate reclamation success is not a legal requirement, but provides important information about the microbial aspects of nutrient cycling. At the community level, measures of the structure of the microbial assemblages (numbers of species and their relative abundances) are the most valuable (Zak et al., this volume). To evaluate ecosystem functioning, variables that integrate microbial function, primary production, and decomposition such as C_{micro}/C_{org} (Insam and Domsch 1987) and qCO_2 (Insam and Haselwandter 1989) are most appropriate.

Mycorrhizae are mutualistic associations between plants and fungi that can determine the success of reclamation efforts by their presence or absence (Allen and Friese, this volume). Almost all North American terrestrial plants form mycorrhizae except annuals in the Chenopodiaceae, Brassicaceae, and Amaranthaceae families. Certain species, especially trees, require these associations for survival. Many other desirable reclamation species benefit from these associations through increased growth and fitness and, thus, ability to compete with weedy annual species that exhibit little or no response to mycorrhizal associations. Rigorous evaluations of reclamation success must necessarily evaluate the structure and functioning of mycorrhizal associations. Surveys of mycorrhizal activity can be conducted at the same time as vegetation assessments. To determine the presence of mycorrhizae, a survey of percent infected root segments of the dominant plants should be made. To evaluate reproduction, density and species diversity of sporocarps of ectomycorrhizal fungi should be assessed.

A properly functioning nutrient cycle is an essential element of reclamation success that is highly dependent upon both the characteristics of the soils and the activity of soil microbiota. Nutrient deficiency is a common characteristic of many disturbed soils and can result either from a lack of one or more major nutrients or an impediment to the cycling and supply of nutrients (Palmer, this volume). Low nitrogen availability often limits plant growth on mined lands, especially if the topsoil is not replaced. Palmer suggests that colliery spoil requires a minimum level of mineralizable N (20 to 40 kg/ha) before the site is naturally colonized by many unsown species. For these sites to be self-sustaining and support a level of productivity comparable to natural ecosystems, a much higher "capital" of mineralizable N (700 kg/ha) is needed. Overfertilization initially may result in dominance by highly productive species and lead to declining stands with high C:N ratios. The inclusion of legumes can provide a steady but low level input of mineral nitrogen. Sites with inherently low nutrient levels may be naturally colonized and eventually exhibit a diverse compliment of desirable native species.

The establishment of vegetation is one of the more critical aspects of successful reclamation because of the lasting effects of this process on species composition. An important prerequisite is knowledge of the germination systems of seeds and of the potential of the seedbed to support germination and seedling establishment (Young, this volume). Seeds may exhibit several forms of dormancy that may be overcome through the proper planting techniques or that may require specific physical manipulation of the seeds. For example, species with hard seed coats (legumes) may require scarification of the seed coat to promote germination. Regardless of the pretreatment, the planting methods used must be appropriate for the physiological requirements of the seeds. Young (this volume) suggests that the most common causes of seeding failure are: (1) improper seed placement and coverage; (2) competition; (3) use of nonadapted species; (4) predation of seeds or seedlings; and (5) environmental factors such as drought, wind erosion, or water erosion.

Measurements of reclamation success often emphasize the vegetation component of the reconstructed community (Allen, this volume). The 1977 Surface Mining Control and Reclamation Act provides for the evaluation of community-level structure and function by requiring mining companies to "establish . . . a diverse, effective, and permanent vegetative cover of the same seasonal variety . . . capable of self regeneration and plant succession at least equal in extent of cover to the natural vegetation of the native area . . ." Although it is often difficult to exactly replicate the native community, a functional diversity that includes the dominant life forms and phenologies can often be achieved. Within- and between-habitat diversity should also be considered as natural landscapes are often heterogeneous and exhibit a high degree of patchiness. To maximize both species diversity and productivity, plants should be selected that differ in their temporal and spatial use of resources and, thus, minimize competitive interactions in mixtures. Inclusion of keystone plant species, such as shrubs that provide microsites for later plant establishment, and of uncommon species with important functions, such as nitrogen fixation, will increase species diversity and aid community functioning.

The basic processes of succession are identical to those required for the reclamation of disturbed land (MacMahon 1987; Redente and DePuit 1988). These universal processes include the arrival of propagules at the disturbed site, species establishment, alteration of the abiotic environment by tthe initial colonizers, competition and further changes in the environment, and eventual stabilization. Differences in successional processes among the major biomes (deserts, tundra, grasslands, coniferous forest, deciduous forest, and rain forest) can influence the goals and methods of reclamation used and, consequently, the successional outcomes (Chambers et al., this volume). For example, large differences exist among biomes in the length of time required to obtain climax, and the degree to which reclamation success can be measured varies considerably. In deserts and tundra only a limited number of species are adapted to the harsh climate. Once the critical plant establishment phase has been achieved in these ecosystems, the reclaimed ecosystem will rather quickly resemble the surrounding area (10 to 50 years). In contrast, because of the nature of successional processes and the length of time required for succession in forests, it will almost never be possible to reclaim a forest to resemble a late successional stage within the bonding period.

Animals integrate numerous abiotic and biotic factors and can serve as important indicators of ecosystem development and functioning (Major 1989; Parmenter and MacMahon, this volume). Reestablishing the habitat of a single species or of species groups is often a specific reclamation goal. The assessment of faunal community development on reclaimed lands requires careful selection of both species and measurements (Parmenter and MacMahon, this volume). Assessments of vertebrate populations often suffer from small sample sizes and are difficult to evaluate statistically. Invertebrate populations are often more easily assessed and can provide useful measures of both successional trajectories and reclamation success. The development of plant structural diversity on reclaimed sites significantly influences the recolonization of both vertebrates and invertebrates, and assessments of animal species must be accompanied by habitat evaluations. Parmenter and MacMahon have tracked both vertebrate and invertebrate assemblages in a reclaimed and undistrubed sagebrush-steppe ecosystem in Wyoming. During the early stages of succession on the reclaimed sites the colonizing species and the community structure were similar among treatments. Treatments that included patches of shrubs for the purpose of diversifying the vertical and horizontal vegetation architecture were colonized by higher numbers of species representing a wider array of guilds.

A holistic assessment of the reclamation process necessarily involves a landscape perspective (Tueller, this volume). All landscapes, including reclaimed ones, exhibit a spatial diversity with both horizontal and vertical boundaries. Landscape resources include all aspects of the ecosystems of interest including geology, soils, vegetation, animals, water, and air quality. A thorough understanding of landscape properties can be obtained through the application of synecological principles, remote sensing techniques, geographic information systems (GIS), and multivariate analysis procedures (Tueller, this volume).

Data obtained using these tools can be analyzed in order to understand the polygons of the landscape and their relationships to one another. These data should be collected prior to mining and can be used in the development of the reclamation plan as well as in the assessment of reclamation success. As our knowledge of landscape ecology increases, our ability to restore the natural heterogeneity of ecosystems and the diversity of the fauna and flora will increase.

A landscape-scale perspective is useful in the assessment of the cumulative effects of mining and other land disturbances on ecosystems. On-site cumulative effects resulting from progressive surface or underground mining may alter soil and vegetation composition, slope stability and surface erosion potential (Halverson and Sidle, this volume). Off-site effects may result from multiple mining disturbances within a watershed or from the interaction of mining activities with other land uses such as grazing, residential development, recreation, or timber harvest. Effects may include changes in water chemistry, channel sediments, riparian ecosystems, aquatic habitat, and peak flows. Because all of the components of ecosystems can potentially be affected, an integrated approach that uses our knowledge of physical, chemical, and ecological process is needed to address these issues. This approach can result in reclaimed ecosystems that meet the desired land use goals without adverse affects to the landscape.

Taken together, the chapters in this volume present a broad framework for evaluating reclamation success using ecological considerations. They illustrate that a holistic approach is necessary to insure that reclaimed ecosystems remain stable and productive and that they have beneficial and not detrimental effects on the landscape. Currently, most assessments of reclamation success by mine operators only include the vegetation community. As our knowledge of reclaimed ecosystems continues to expand, it may be desirable to measure other components of ecosystems that convey a high level of functioning. In addition, it will be possible to refine our methods of assessment to more accurately reflect the level of functioning.

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