The restoration of disturbed arid landscapes with special reference to mycorrhizal fungi

Edith B. Allen*

Introduction

The accelerated rate of desertification of arid and semi-arid lands worldwide is providing new incentives for their restoration. Of some 47 million km² of these lands, 20% are classified as severely disturbed (Dregne, 1983) and in need of restoration. Six million ha additional land are estimated to be desertified yearly (United Nations, 1977).

A landscape-scale approach is essential to dealing with such a large restoration problem. Different ecosystems in the landscape have different carrying capacities and require different restoration solutions (Naveh & Lieberman, 1984). Of equal importance are the interactions between landscape patches (e.g. Risseur et al., 1984; Forman & Godron, 1986), where, for instance, movement of soil or water from one patch to another may be both an indication of desertification and a deterrent to restoration if both elements are not considered simultaneously. As an example of the need for a landscape-scale approach to restoration I will present some research on mycorrhizal fungi in a disturbed semi-arid landscape.

In keeping with the focus of this workshop, I will consider restoration only of those lands which are desert or semi-desert. The most viable restoration goal on a large scale is to return these lands to stable ecosystems which require no further anthropogenic input once vegetation has been re-established (e.g. as opposed to croplands or pastures), except for management to avoid future destruction. For the most part the only human use of these lands is for domestic grazing and shrub harvesting for firewood, or irrigated agriculture on a smaller scale. Thus the predominant disturbing force is excessive removal of vegetation followed by erosion (Dregne, 1983). Since continued human use is the major impetus for restoration, the land use after restoration will be predominantly grazing and wood cutting. Restoration of natural areas for protection of plant and animal diversity is occurring on a smaller scale (Janzen, 1988; Jordan et al., 1988), but the two restoration goals need not be incompatible.

Restoration ecology is an emerging branch of ecology which incorporates applied and basic research with land management (Allen, 1988a; Gilpin et al., 1988; Jordan et al., 1988). Much of what we know of restoration is anecdotal. There are projects to restore arid lands worldwide, but little past emphasis on collecting and analyzing data so that success of restoration projects can be evaluated and repeated. In the United States, the passage of the Surface Mine Control and Reclamation Act of 1977 provided an impetus for mine reclamation in the semi-arid West, and a data base is slowly amassing. Mining is the most severe of all disturbances, so the concepts and techniques which are devised can be applied to even the most severe forms of desertification, e.g. where all of the plant cover has been removed and the topsoil has eroded. Not all lands are so severely disturbed, so it is possible

* Systems Ecology Research Group and Department of Biology, San Diego State University, San Diego, California 92182 U.S.A.

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to take advantage of natural succession, coupled with purposeful introductions of organisms when necessary, to restore these lands (MacMahon, 1983). In spite of these advances, there are limitations to restoration of arid lands that must still be overcome.

**Limitations to restoration of arid lands**

While the restoration of any disturbed land presents obstacles to be overcome, there are special limitations to restoration of arid and semi-arid lands that are driven by the harsh climate (Table 1). Each of these limiting factors is discussed in terms of its particular application to arid and semi-arid lands or, where appropriate, to restoration of any lands.

The most important of the limitations to arid land restoration is low and variable precipitation. When mined land restoration was first proposed for the semi-arid western United States, a National Academy of Sciences committee held out little hope for restoration of any lands that received less than about 250 mm precipitation (NAS, 1974). Such mined lands were termed a ‘national sacrifice area’ (Box, 1976) because of the perceived poor prospects for, and high cost of, reclamation. A decade of mandatory restoration has shown that vegetation can be re-established on these lands at cost, although the ability to re-establish stability of the restored ecosystem is still an open question (e.g. Whinford, 1988).

The annual variability of precipitation is greater in deserts than in any other biome (MacMahon, 1980), an observation which has led to the hypothesis of the pulse phenomenon of plant establishment in deserts (West, 1948; West et al., 1979; Jordan & Nobel, 1981). Once seedlings are established after one or more wet years, the adult plants can withstand the normal fluctuations in precipitation. While this pulse phenomenon is often viewed as a limitation, in that restoration will not be successful every year, it is also possible to take advantage of years of high precipitation to establish vegetation. This may be the only solution where precipitation is low and sporadic, and irrigation is not possible.

Although wind occurs in many other biomes, it is an especially important factor in deserts. With the exception of some coastal deserts, low humidity coupled with wind desiccation is stressful for desert plants (e.g. Smith, 1978), and perhaps more so for seedlings (Allen & Allen, 1986). Because the surface soils are also dry, wind is a more forceful erosive factor in deserts than other biomes. Movement of sand dunes is an outstanding example. Sand dune restoration has received more attention than other arid lands, not because of any economic benefit but to avoid immediate environmental impacts (e.g. Gati, 1984).

The loss of topsoil due to wind and water erosion is the most serious consequence of desertification (Dregne, 1983; Yair, 1987) and at the same time poses the greatest limitation to restoration. The reversibility of desertification has been questioned (Giantz & Otvolsky, 1983; Allen, 1988a), primarily because the land is no longer able to support the same level of productivity after topsoil loss. This includes lower soil moisture-holding capacity because of reduced organic matter, reduced fertility, and loss of propagules of desired plants and micro-organisms. Restoration is an attempt to reintroduce these propagules and then to allow succession to take its course. In the case of many plant species, reintroductions are possible, but there are ecological, technological, and economic limitations to reintroduction of micro-organisms (e.g. Havny, 1984; Allen, 1988a).

Since it is not possible at this time to reintroduce most of the beneficial micro-organisms on a large scale we must rely on natural dispersal. The natural rate of dispersal of micro-organisms into disturbed lands has been little studied, and is a landscape-scale problem (Allen, 1987, 1989; Wieland & Allen, 1987).

Among the soil micro-organisms one that has received a great deal of recent attention is the mycorrhizal fungus. A mycorrhiza is a mutualistic association between a root and certain species of fungi, whereby the plant has increased growth, nutrient uptake, water uptake, and drought stress-tolerance, and the fungus receives carbohydrates from the plant (e.g. Allen & Boosalis, 1983; Harley & Smith, 1983). These fungi are widespread in arid and semi-arid lands (Trappe, 1981), where the most abundant form is the vesicular-arbuscular (VA) mycorrhiza. However, a number of man-caused and natural disturbances reduce or eliminate the fungus, including drought and erosion (Powell, 1980), tillage (Allen & Boosalis, 1983), and grazing (Bothlenlavty & Dakessian, 1984; Wallace, 1987).

Unlike ectomycorrhizal fungi, which can be used to inoculate trees to re-establish forests, there are no large scale methods for inoculation of VA mycorrhizal fungi.

The loss of mycorrhizae in arid lands is a particular problem because a large proportion of the initial colonizing species are nonmycorrhizal; e.g. do not form a mycorrhizal association (Allen & Allen, 1989). Although mycorrhizae are important to the growth and survival of later colonists, species, annuals in such families as the Chenopodiaceae, Brassicaceae, Amaranthaceae, and Zygophyllaceae are non-mycorrhizal and appear to be well adapted to growth in desertified soils. These annuals persist for 0-15 years in arid and semi-arid mine soils in Wyoming which had no initial mycorrhizal inoculum (Allen & Allen, 1988; and unpublished). Even though dispersal of mycorrhizal spores can occur within a few years on these sites (Wieland & Allen, 1987), the continued presence of nonmycorrhizal annuals precludes mycorrhizal establishment. Thus even at the small scale of a micro-organism the phrase ‘desertification begets desertification’ may be true. Without the introduction of propagules of mycorrhizal plant species, and in some cases cultural manipulation to assist in their establishment, the mycorrhizal component of the ecosystem cannot re-establish.

The social, political, and economic limitations to arid and semi-arid land restoration have been dealt with in other publications (e.g. Mabbett, 1984; Tolba, 1987). They will not be further discussed here, but it is important to keep them in mind while solving ecological problems. The economic return from arid lands is low, so many governments are not willing to invest in their restoration. However, ecological solutions may be economically viable solutions, even if they are not as rapid as the more common agronomic practices (e.g. seed drilling, irrigation, fertilization). An ecological approach might include taking advantage of the pulse phenomenon of seedling establishment and natural succession. It might also involve natural dispersal of mycorrhizal fungi combined with an understanding of their effects on various planted species and in different environments. The next section of this paper is a report on an ecological, landscape scale approach to restoration which incorporates mycorrhizal fungi.

**VA mycorrhizal fungus and succession in a disturbed landscape**

Restoration research was initiated in 1982 on a recoloured coal mine in SW Wyoming. The undisturbed native area is an Artemisia steppe in hilly terrain at 2000-2200 m elevation which receives 230 mm precipitation, 60% as snow. The landscape is heterogeneous,
varying from patches of taller woody vegetation on east facing convex slopes to 'cushion' plants on wind-scorched ridge tops. Mining tends to homogenize the landscape, creating slopes of equivalent 1:5 pitch, equal depths of replaced topsoil, and similar mixtures of plant species that are seeded everywhere. The homogeneity of the mine soil nutrients and microorganisms, compared to the heterogeneous native soils, has been characterized (Allen & MacMahon, 1985).

The remaining elements of heterogeneity on the mined site are due to microclimatic effects such as ridge tops vs. slopes and replacement of sub-soils rather than topsoils in some areas. The sub-soils are lower in nutrients, have virtually no organic matter, and are higher in clay than the topsoils (Waaland & Allen, 1987). Because the topsoils were stored for seven years prior to their replacement, neither the sub-soils had sufficient mycorrhizal spores to cause greater than 1% of the root length of planted grasses and shrubs to become infected in the first growing season. The plants in undisturbed areas may have up to 75% infection (Waaland & Allen, 1987).

Because mycorrhizae improved the growth of several of the species which were planted, and increased their ability to compete with colonizing non-mycorrhizal annuals (Allen & Allen, 1984), we hypothesized that the addition of mycorrhizal inoculum should increase the rate of mycotrophic plant establishment and succession on this site. In order to test the hypothesis, we inoculated five different sites on the mine with a thin layer (1–2 cm) of topsoil from the native area containing a high density of mycorrhizal spores, and removed non-mycorrhizal annuals by hand in a 2 × 2 factorial experiment (with and without mycorrhizae, with and without annuals). Further details of the experimental design are published elsewhere (Allen & Allen, 1986; 1988). The five sites included two on stockpiled topsoil that were located mid-slope, and three on sub-soil that were either on the winds-scorched upper portion of the slope, on the lower portion of the slope, or mid-slope.

The data after three years of growth are presented here to compare results for sites established in different years (Fig. 1). Only the results for the planted Agropyron species (A. dasystachyum, A. smithii, A. trachycaulum) are reported, as they were by far the most abundant species. Seeded shrubs (Artemisia tridentata and Atriplex canescens) established sparsely, with only 1% cover on any of the sites. Colonizing annuals persisted into the third year in only two of the sites, the subsoil upper where the non-mycorrhizal species Salsola kali was still abundant (Allen & Allen, 1988) and the 1984 topsoil site where Bromus tectorum still formed 15% of the ground cover (unpublished data).

The most notable result from these five sites is that no two sites had the same outcome with respect to Agropyron density and cover (Fig. 1). The order of abundance of the grasses in the four treatments was different on each site. The predictions that mycorrhizal inoculation would increase growth of the planted grasses, and that removal of non-mycorrhizal annuals would improve grass establishment, were in general not borne out (Table 2). If mycorrhizal and annual plant removal had been the most important factor in causing increased grass establishment, the MNA (mycorrhizal, no annuals) treatment would have had the greatest grass abundance on all sites. However, this treatment had the greatest benefit on only one site, the topsoil site established in 1984. In fact, in some cases a treatment which caused the greatest percent cover and density on one site caused the lowest cover and density on another site (see, for instance, the non-mycorrhizal, no annual treatment—NMN—in Table 2).

Sorting out the causes of these different results requires an examination of individual species interactions and characteristics of the sites, or in other words, a reductionistic coupled with a landscape-scale approach. The fact that mycorrhizal inoculum did not increase grass growth to all of the field treatments, and in fact decreased their growth in some cases, was contrary to our preliminary greenhouse experiments (Allen & Allen, 1984). The relatively high nutrient soils coupled with high precipitation may in part be responsible, as mycorrhizae may act as a carbon drain under these conditions. In spite of their reduced growth, the grasses had other physiological benefits, including decreased stomatal resistance to water vapor, decreased leaf mortality, and delayed phenology (Allen & Allen, 1986).

One very surprising result was the number of times the treatments with annuals had the greatest grass cover and density (see NMA, Table 2). This implies that the annuals were facilitating grass establishment, rather than competing with the grasses. In fact, removal of Salsola kali did cause a reduction in grass establishment, but only on the harshest site where the soil was poor, plant establishment was low, and wind was important (Allen & Allen, 1988). The removal of S. kali reduced the amount of snow trapped, especially on the upper site, and reduced moisture input. In addition to increasing moisture, the
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References


standis. er of S. kalli, which persisted during the spring germination period, enhanced sealing establishment (Allen & Allen, 1988).

Another surprising result was that utilization with mycorrhizal fungi directly reduced the growth and density of S. kalli and Atriplex rosea, both non-mycorrhizal species (Allen, 1984; Allen & Allen, 1984, 1988). This apparently 'pathogenic' effect of mycorrhizal fungi on non-mycorrhizal species may help explain why non-mycorrhizal species are so abundant in severely disturbed sites where inoculum density is low, and at the same time unaided in soils of high inoculum. (Pendleton & Smith, 1983; Allen & Allen, 1988). The loss of the S. kalli 'nurse crop', due to either mycorrhizal inoculation or experimental removal, reduced grass establishment. However, S. kalli facilitated grass establishment on only some of the five sites (two sites the grass exhibits is considered, three if grass density is considered). The important point here is that the harshness of the site, which was determined by its position in the landscape and whether topsoil was used, determined whether the annuals acted as facilitators or as competitors. A high initial density of mycorrhizal inoculum may either hinder or promote the early stages of successions, and an understanding of the heterogeneity of the landscape is critical before making decisions such as whether to inoculate, or whether to rely on natural dispersal of the fungi.

The natural dispersal of spores of VA mycorrhizal fungi in mined sites appears to be great enough to begin detecting infection in the roots within one to three years (e.g., Waaland & Allen, 1987). These sites are characterized by small patches of disturbance, usually no larger than 100 ha, with nearby undisturbed native areas which serve as sources of spores. Spores from these native areas are moved primarily by wind (via soil erosion) and animals which ingest the spores, although wind is most important in arid regions (Warner et al., 1987). In a much larger disturbance, the Mt. St Helens volcano in the state of Washington, U.S.A., plant species which had established c. 5 km from the undisturbed forest edge still had no VA mycorrhizal inoculum after 7 years (Allen, 1987, and unpublished). The scale of desertification, which includes the erasure of topsoil and loss of inoculum, is larger still. Whether islands of vegetation which are re-established in a sea of desertified soils will become naturally inoculated must still be examined.

In a 10-year anniversary review on the progress since the 1977 United Nations Conference on Desertification, the authors agreed that the lack of political, social, and economic support are the greatest limitations to arid land restoration (see Desertification Control Bulletin No. 15, 1987). In keeping with the theme of the workshop, "What is Special about Desert Ecology?" I assert that destruction and the need for restoration are not unique to deserts. However, more productive lands can be economically restored while deserts continue to lie barren. Continued scientific advancement in landscape-scale approaches to restoration, as exemplified in the discussion above, may eventually lead to more rational restoration plans which are compatible with the low economic returns from arid lands.

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