

Restoration Ecology: Limits and Possibilities in Arid and Semiarid Lands

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Abstract—Most attempts at repairing the damage caused by anthropogenic disturbance in arid and semiarid lands of the western United States have historically consisted of revegetating with monocultures or simple mixtures of mainly exotic species. Most revegetation was done for utilitarian purposes, typically to increase forage. The realization that biodiversity has been lost in many arid lands because of grazing, agriculture, and mining has prompted an interest in restoration which has conservation goals. Because of the extent of damage, restoration has limitations to simulating the original ecosystem before disturbance. Three major ecological limitations are discussed. The invasion of exotic weeds has reduced the diversity of native species, and they can be controlled with variable success. Where topsoil has eroded or been altered by compaction or other means, the community that develops is often floristically dissimilar from the original, as is its functioning. Restoration of biodiversity may be our greatest challenge, as even the best examples of restoration have been able to reintroduce only a fraction of the plant species richness, and natural recolonization is slow at best. Water for plant establishment in arid lands is discussed as well, but this is a limitation that can be overcome with ingenuity and patience. The costs for restoration can be borne if society decides that restoration is important enough. Even with our best efforts we cannot simulate what was once there, but we can improve the habitat value for many declining species.

Revegetation has been practiced for many decades in arid and semiarid lands of the western United States, but most often the goal has been to improve grazing for domestic animals. In the past few years the emphasis in North America has shifted from a strictly utilitarian purpose, to revegetation for conservation of plants and animals (West 1993). A great deal is known about range improvements for livestock, which typically consists of revegetating depleted rangelands with exotic monocultures or low diversity mixtures (e.g., Johnson 1986; Pendery and Provenza 1987), or of removing unpalatable species, such as shrubs or weeds. Less is known about restoring a diverse, native vegetation. However, there have been recent attempts at restoration for the sake of conservation, with varying degrees of success. The decisions that need to be made before restoration is undertaken are social, economic, and

ecological. The social and economic decisions involve the valuation of conservation, which can sometimes include multiple use of lands. Here I will describe the ecological limitations that managers must consider to decide what lands can be restored, and to what degree.

The goals that can be achieved with different revegetation practices are captured in the definitions of restoration, reclamation, and rehabilitation that were put forth by the National Academy of Sciences (NAS 1974). Restoration means to reproduce the ecosystem structure and functioning that existed prior to disturbance, assuming the site was a relatively undisturbed, late successional or otherwise desirable native ecosystem. This was seen by the committee as the most difficult goal to achieve, if not impossible, because the entire diversity of native species would need to be reintroduced, and the late seral soils would need to be preserved or restored. Reclamation could, however, be achieved, because exotic species might be used that establish more readily than native species, and a lower diversity of native species would be acceptable. Reclamation still requires a high level of functioning of ecosystem processes, as the new ecosystem would also need to be self-organizing and stable, capable of existence with a minimum of human input. Reclaimed land would, however, be structurally less complex than restored land.

The third goal is rehabilitation, which implies that the land has been made productive again, but that an alternate ecosystem has been created, with a different structure and functioning from the original system. A rehabilitated system might be low in diversity, include only exotic species, and require continual inputs, such as fertilizer or irrigation, to exist. Improved rangelands that have been converted to monocultures are an example of rehabilitation. These three goals of revegetation were considered a continuum by Bradshaw (1984, Fig. 1), although I have changed the definitions from his model to reflect the NAS (1974) definitions.

The concern for rangeland restoration, rather than rehabilitation, has arisen as a loss in biodiversity and in habitat has occurred. For instance, mammal species diversity was lower in a monoculture of western wheatgrass than in adjacent Great Basin shrub grassland (Smith and Urness 1984) and bird species diversity was greatly reduced in lovegrass pastures in Arizona (Bock and others 1986). Even forty years of natural succession did not increase the plant or animal diversity of these pastures (Bock and others 1986). Biodiversity in rangelands has recently become a management goal (West 1993; Pyke and Borman 1992). As a result of these concerns for biodiversity, we can begin to expect changes in management practices that address the multiple use of arid and semiarid lands for people, livestock, and native organisms.

The values of arid land management vary with the revegetation goals (Table 1). Restoration is the highest

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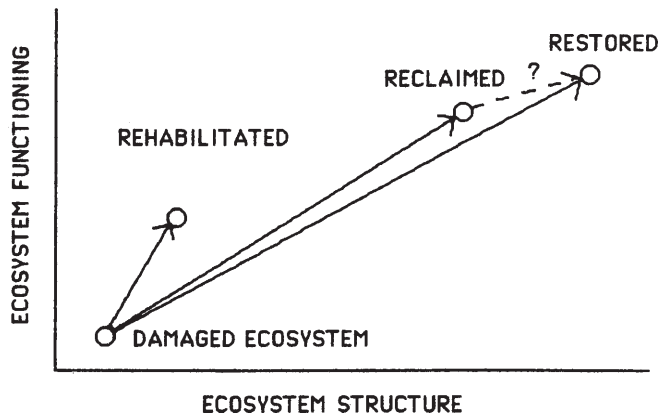


Figure 1—Model of goals to achieve restoration, reclamation, or rehabilitation. Restoration replicates the structure and functioning of the original system, or nearly so. Reclamation still requires a high level of functioning, but is structurally less complex. A reclaimed site may, through natural succession, approximate the original ecosystem, depending on species and treatments used. A rehabilitated ecosystem has little similarity to the original ecosystem in structure or functioning. Revised from Bradshaw (1984), definitions from National Academy of Sciences (1974).

conservation goal that can be attained. Reclaimed land still has some conservation value, but rehabilitated land is used for utilitarian purposes entirely. The economic value of these goals is at odds with the conservation value, as high input, rehabilitated lands yield more income than restored lands. The true economic value of restored lands is determined, of course, by how society views the cost of protecting rare species, and the indirect return of the “ecosystem services” (Westman 1977) restored lands will provide. However, the cultural, scientific and intrinsic values of restored lands are highest, where intrinsic value means value for its own sake, not associated with any other human benefit (Naess 1986). Restoration may be more costly to implement initially, but it results in ecosystems that require less maintenance input in the long term, are more stable, and have higher species diversity.

In some instances restoration may be accomplished economically by allowing natural succession to return the original ecosystem, but this is typically a slow process that is

not accomplished within human lifetimes. Where lands have been severely disturbed, as after mining, overgrazing, or severe erosion, or when rare species are at stake, restorationists hasten the rate of succession by introducing late successional propagules of plants, animals, and microorganisms. Thus we may speak of “active” and “passive” restoration, where passive restoration consists of removing the stresses that caused the original degradation, such as heavy grazing, air pollution, and so forth, and then allowing natural succession. Active restoration means applying a number of management techniques, such as introducing propagules of organisms, weeding, burning, alleviating compaction, improving soil moisture, nutrients, or organic matter, and so forth. Typically, restorationists rely on a combination of both active and passive approaches, depending upon the severity of the disturbance. For instance, restoration may only require weeding and reintroduction of a few species if the seed bank is largely intact.

A strict reliance on succession will also not be sufficient to achieve restoration after many kinds of disturbances, if succession will no longer restore the original ecosystem. This seems to apply especially to arid and semiarid rangelands, where numerous observations on release from grazing or other disturbance have not shown a return to the original vegetation (Laycock 1991; Allen 1988; Westoby and others 1989). Succession may result in a new vegetation type, either consisting of exotic species or of new combinations of native species that were not formerly part of the landscape. For instance, mechanical disturbance that disrupted the soil and seed bank in southern California shrublands resulted in stands dominated by exotic annual grasses up to 70 years after abandonment (Davis 1994). Burned blackbrush (*Coleogyne ramosissima*) shrublands in southern Utah gave way to different kinds of shrublands, each dominated by different native shrub species in different areas (Callison and others 1985). Rest from grazing in sagebrush (*Artemisia tridentata*) steppe did not result in reduced shrub density with improved grass productivity in the understory, as classic succession theory would predict (West and others 1984). The native plant species that recolonized naturally after mining in Alberta, Canada, formed such an unusual community that no similar native communities existed (Russell and La Roi 1986).

In these and numerous other cases, disturbance resulted in a new trajectory of succession that involved both native and exotic species. Multiple stable states of vegetation types may coexist after disturbance, such that succession will not return to the original vegetation, but may result in one of several types. The concept of global stability has implications for the practice of restoration. The vegetation may return to a native type depending upon the kind of disturbance, but it may change to another vegetation type with another disturbance regime or with exotic introductions (Laycock 1991). Arid lands are especially subject to changes in trajectory of succession when the interval of disturbance becomes too short for recovery, as with increased anthropogenic disturbance (Turner and others 1993). In such cases, restoration may be the only solution to restoring an ecosystem that approximates the original. However, restoration may not always be the solution either, as there are limits to restoration.

Table 1—Comparison of values of restoration with reclamation and rehabilitation.

	Rehabilitation	Reclamation	Restoration
Conservation value	low	medium to high	higher
Economic value	high	medium	low
Intrinsic value	none	medium	high
Cost to implement	low	medium	high
Species diversity	low	medium	high
Maintenance input	high	low	lower
Stability	low	high	higher

I will discuss three primary ecological limits to restoration: invasion of weeds, loss of topsoil, and biodiversity. A fourth limitation that applies especially to arid lands, moisture, will be discussed as well. However, moisture is not a serious limitation, as water or a water harvesting system can be provided for most restoration efforts. The first three are limitations that usually cannot be cured by technological means.

The Limits to Restoration

Moisture for Plant Establishment

Water has frequently been studied as a limitation to arid land restoration and reclamation. Before the Surface Mine Control and Reclamation Act of 1977 was passed, many arid land scientists feared that the western rangelands would become "national sacrifice areas" because we did not have, and possibly could not develop, the technology to reclaim after mining. Western researchers at that time showed that we could, indeed, reclaim using native species, although these studies did not necessarily include all or even mostly native species (McArthur and others 1978; DePuit and Coenenberg 1979; Aldon 1981). These studies often used irrigation, sometimes at a higher level than necessary for establishment. High levels of initial irrigation may improve productivity, but at the cost of species diversity (DePuit 1988). The most drought-adapted species are lost under high irrigation levels, while species that respond to high soil moisture dominate. There were many early anecdotal reports of loss or decline of plant communities after irrigation was removed. While the literature is replete with reports of the effects of moisture on plant productivity, especially on crop plants, there is little information on the minimum critical moisture required for establishment of arid land plants.

Precipitation is more variable in deserts and semiarid lands than any other ecosystem, so plant establishment often occurs in pulses that are related to high precipitation years. Natural plant establishment varies from year to year in hot deserts, because the plants may need a wet year, or a series of wet years, for establishment to occur (Jordan and Nobel 1981; Romney and others 1987). Conversely, West and others (1979) did not detect any evidence for a pulse phenomenon of establishment in the cold desert of Idaho. Restorationists working in the hot deserts might be able to take advantage of the pulse phenomenon of desert plant establishment, either by waiting for the appropriate sequence of high precipitation years or by irrigating.

Based upon infrequent establishment of many species in hot deserts and the fact that precipitation is highly variable, MacMahon (1981) observed that succession is slow in deserts not because desert plants grow slowly, as was previously thought, but because of delayed establishment if an appropriate precipitation year occurs infrequently. There has been relatively little work on growth rates of native desert shrubs. However, several hot desert species exhibited rapid growth rates once they were transplanted as seedlings, even with only a minimum of water once at the time of planting (Bainbridge and others 1993). After only three years, species such as mesquite had achieved a

volume as large as that of many shrubs from adjacent undisturbed areas. Thus, irrigation may be necessary to overcome a series of dry years, and the only problem is determining the amount. A rule of thumb is to irrigate no more than the average amount of precipitation, or no more than a wet year if it is known that certain species require higher than average moisture for establishment. However, in recent studies, irrigation to make up the deficit of a dry year did not improve the establishment of Indian rice grass in Utah (Belnap and Sharpe 1993), nor did irrigation improve the establishment of shrubs in the Wyoming Red Desert (Powell and others 1990). Thus, it is necessary to know the individual moisture requirements of species for establishment.

In a study on off-season irrigation, I studied the effects of dry season irrigation on purple needle grass (*Stipa pulchra*). This grass has had poor establishment success in revegetation trials in native California grasslands where it was once abundant, probably because of competition from exotic annual grasses (Nelson and Allen 1993; Bartolome and Gemmill 1981). The Mediterranean climate grasslands where purple needle grass occurs will probably have enough winter/spring precipitation to allow germination in most years, but seedlings must be able to survive a six-month dry summer for establishment. Thus I experimented with a low level of off-season (summer) irrigation. The site was in Santee, southern California, and the experiment was performed during the below normal precipitation spring of 1991. I irrigated at two levels from mid-March through early June, either 1 cm daily or 1 cm weekly, to assure adequate germination in four 0.5-m² replicate plots. Species other than purple needle grass were weeded. After June 11 half of each of the high and low level irrigation plots were irrigated with 1 cm water weekly, the other half were not irrigated further. The high spring irrigation level resulted in a higher proportion of large plants with crown diameter greater than 3.0 cm, while the low level had a higher proportion 2.9 cm or smaller (Fig. 2). However, leaving the

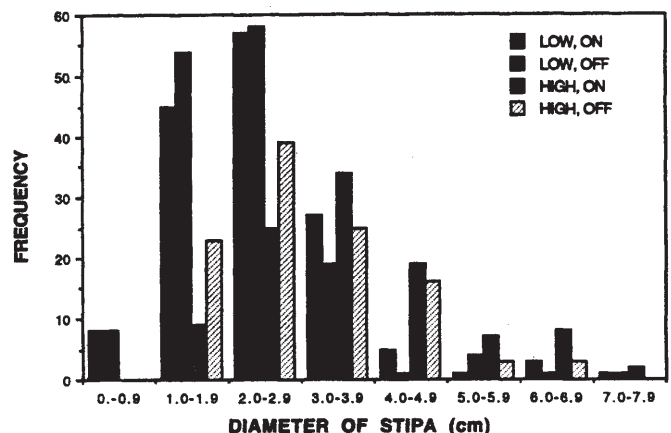


Figure 2—Frequency of diameter classes (crown cover) of purple needle grass subjected to four irrigation treatments. Low and high are spring (March through June) irrigation levels, on and off refer to presence or absence of summer (June through August) irrigation. Data collected in August.

Table 2—Density of purple needle grass in plots with high and low irrigation levels. Irrigation was left on or turned off in one-half of the plots on 11 June.

Treatment	No./m ²		% mortality
	5 June	7 August	
high on	82.5	75.5	8.4
high off	97	73	24.7
low on	70.5	52	26.2
low off	67	54.5	18.7

water on during the summer only resulted in reduced mortality of the high spring irrigation plots (Table 2), where most of the plants were already quite large. In the two low spring irrigation treatments, the mortality was equally high whether there was summer irrigation or not. In effect, the many small plants of the low spring irrigation treatment died whether they had summer irrigation or not. The results suggest this grass must have seedlings that are in the range of 2 cm diameter or greater for survival during the dry summer. Spring irrigation was important to assure that the plants were large enough to survive the normal summer drought.

If water is a limitation to arid land restoration in the short term, in the long term moisture will eventually become available for natural establishment via the pulse phenomenon, or moisture can be provided by irrigation or surface treatments such as pitting, furrowing, imprinting, and so forth. More imaginative methods, such as deep pipe irrigation or water catchment systems, have also been used (Bainbridge 1992). These methods have proven capable of increasing establishment even in hot deserts. Thus I will not further consider water as a limitation to restoration, because it is one that can most often be overcome with management, technology and imagination.

Exotic Plant Competition

The arid and semiarid lands of the western U.S. are experiencing an unprecedented invasion by exotic plant species that threatens native ecosystems and reduces the success of restoration. Disturbance is often considered necessary for plant invasion to the extent that natives will be entirely replaced, but invasion is occurring even in lands that are subject to relatively little anthropogenic disturbance. Fox and Fox (1986) list a number of characteristics of ecosystems that make them subject to invasion, and open vegetation structure with large interspaces, such as is typically found in the desert, explains why exotics have dominated to a much greater extent than they have in forested ecosystems, for instance. Even a natural disturbance such as fire has become an agent for opening native vegetation to invasion in the semiarid shrublands of southern California and the Great Basin, especially when the frequency exceeds that occurring naturally (Freudenberger and others 1987; Billings 1990).

These exotics are almost certainly reducing the diversity of native plant communities, as a number of them form persistent near-monocultures. Some of them occur on a large scale, such as the Mediterranean annual grasses that have

displaced the native Central Valley grasses in California, or cheatgrass (*Bromus tectorum*) that has invaded Great Basin shrublands (Billings 1990; D'Antonio and Vitousek 1992). These are the better known examples, but more recent invasions cause range managers to long for the days when the dominant invaders were at least palatable forage grasses. For instance, several species of knapweed are replacing large acreages of annual grasslands in California and northwestern North America, as are artichoke thistle and mustards in southern California shrublands. Some of these invasions are occurring so recently that even sites that were dominated by natives in the last few decades are now dominated by exotics. For instance, the exotic annual grass *Schismus barbatus* and the mustard *Brassica tournefortii* have been invading the southwestern deserts since about the 1930's, and are spreading into areas with relatively little recent disturbance such as the Anza Borrego Desert State Park.

Controlling these weeds and the restoration of these sites has become a major problem. Range managers continue to use productive, exotic perennial grasses and shrubs to reclaim the land from annuals because the seedlings of native species typically are not able to compete with the aggressive annuals (Johnson 1986; Nelson and Allen 1993). The "greenstripping" program, which consists of planting strips of less flammable vegetation in a matrix of cheatgrass, has been initiated to begin controlling wildfires in cheatgrass grasslands (Pellant 1990). However, the planted vegetation is typically a mix of exotic plus native species, and does not constitute restoration. A successful effort at perennial grassland restoration has been provided by the Nature Conservancy at the Santa Rosa Plateau Preserve in southern California. This site had past grazing, but the grasslands have persistent native perennial grasses with the colonizing exotic annuals. Burning was timed to destroy the annual seed crop in June when the seeds were mature but before they had shattered. In the next growing season, burned sites were dominated by the native purple needle grass (*Stipa pulchra*) (Gary Bell, The Nature Conservancy, pers. comm.). Since the seeds of these grasses, members of the genera wild oats, wild barley and brome, do not have a long-lived soil seed bank (Marshall and Jain 1967), fire is an effective restoration technique. In the case of cheatgrass, the seed bank is more persistent (Hassan and West 1986), so multiple spring fires would likely be required to deplete the seeds. I am not aware of such an effort, which would likely need to be supplemented by seeding and planting of native species that have been eliminated in cheatgrass monocultures.

The cheatgrass problem is one of the largest and perhaps most difficult to solve for restoration purposes. Others have been solved, if not to result in perfect restoration, at least to recreate communities dominated by native species. Natural succession in disturbed sagebrush grasslands of north-eastern Wyoming that were dominated by Russian thistle (*Salsola kali*) resulted in a return to native vegetation, whether Russian thistle was controlled or not (Allen 1988). The difference was that plots dominated by Russian thistle had reduced sagebrush seedling establishment in the early years following discing. Even though Russian thistle disappeared naturally after 3 to 4 years, it had a persistent depressive effect on sagebrush, so Russian thistle plots were

grass dominated, while weeded plots were sagebrush-dominated (Allen and Knight 1984; Allen 1988). In another experiment on mined land reclamation, Russian thistle was a nurse plant in upland sites because its litter catches snow and shelters grass seedlings (Allen 1992). More recently we have learned that the soil under Russian thistle is elevated in phosphorus because of the high levels of oxalate in its tissue, so this weed may facilitate establishment of the next stage of seral plants where phosphorus is limiting (Cannon 1993). Thus, not all weed problems of arid lands are insurmountable, and in the case of Russian thistle there may even be some benefits.

In general, restoration has not been practiced on a large enough scale to eliminate large scale weed problems, or has not necessarily had the goal of reducing exotic species. Oak savanna "restoration" in southern California consists of replanting oaks and leaving the exotic grass understory. Even after fire has greatly reduced the exotic grasses, exotic forbs such as storksbill cannot be eliminated because of their deep roots (Gordon and Rice 1992) and possibly persistent seed bank. In spite of our best efforts, we will need to accept that a certain percentage of the plant community will consist of exotics, and some lands may be so weedy that they are beyond our abilities to restore.

Loss or Alteration of Topsoil

Restoration often consists of planting late successional species into early successional soils that have had topsoil removed, eroded or compacted. Even with the best restoration efforts, it takes time for soil genesis, and a return to a soil that will support the previous community may take decades to millennia. Many species of soil microorganisms are slow to recolonize impacted soils, and inoculation is not possible for most of them. The soil surface hydrology may be altered, as after leveling for agriculture or heavy grazing, limiting plant establishment (Anderson and others 1976; Allen and Jackson 1992). The mining laws in this country have assured that, at least for coal mining, the topsoil is replaced. Even here, the resultant "topsoil" is a mix of topsoil, subsoil, and parent material that is lower in nutrients, organic matter, and mycorrhizal fungi than undisturbed soils (Allen and Allen 1980). Another problem exists for degraded rangelands, where topsoil has eroded away leaving the B horizon exposed (Schlesinger 1985). The problem of eroded rangelands is typically solved by planting exotic species that both stabilize the soil and provide forage (Johnson 1986; Vallentine 1989), but restoration is seldom a goal. To assess the effects of soil material on vegetation establishment and succession, it is necessary to compare restoration on topsoiled and untilled sites. Although native species are often included in range improvement mixtures, I am only aware of experiments that compare untilled and untilled sites in mined land reclamation and restoration experiments. Therefore, I will restrict the examples to illustrate my points to mined soils. Only those examples will be discussed that studied soils that were not toxic or otherwise different from the original subsoil or parent material, so the conclusions drawn from these studies can be extrapolated to non-mine soil disturbances.

The first bit of evidence that untilled soils are unsuitable for restoration comes from studies on natural succession of mine spoils. For instance, even after 30 years the grasslands that developed in Oklahoma mine spoils did not resemble the species composition of native prairie (Johnson and others 1982), and in Alberta, Canada, the plant community that developed naturally on mine spoils did not resemble any native communities as shown by ordination (Russell and La Roi 1986). These sites were dominated by early and mid-seral species, but also by species that came from different habitats entirely. Restorationists would of course hasten succession by using later seral species, but even here the evidence shows that untilled and untilled sites have very different vegetation. For instance, in a simple community dominated by four species of planted wheatgrasses and a few dozen invasive species, western wheatgrass was more abundant on subsoil, thickspike wheatgrass on topsoil, and the invasive community was different on the two soils (Waaland and Allen 1987; Waaland 1985). In a study that compared succession of native sagebrush-steppe, the weed community, including shrubs such as rabbitbrush, was more persistent on subsoil while the native grasses and late seral shrubs were more abundant on topsoil (McLendon and Redente 1990). These studies indicate that, even where the same mix of native species was planted in untilled and untilled plots, the effect of soils was key in sorting out the species composition.

The problem of restoring degraded soils is one of soil genesis. It may take centuries to millennia for natural soil building processes to restore the soil naturally. Soil genesis has been studied in mine soils, and where vegetation has been successfully reestablished, the formation of horizons may occur more quickly than many researchers had suspected (Schafer and others 1980). Soil amendments, primarily nutrients and organic mulches, are used to hasten the rate of soil genesis, and can help build a soil that has similar chemical and biological characteristics to undisturbed soils (Whitford 1988). However, there is no substitute for time in rebuilding certain features of the soil, especially soil structure.

Undisturbed desert soils that are already low in organic matter and nutrients may not be very different from disturbed soils from the standpoint of soil chemistry, although microorganisms are still impacted (Allen 1988). The native species are probably already adapted to early successional soils with poor chemical qualities, and in fact, deserts exhibit autosuccession where the late stage seral vegetation is also the colonizing vegetation. The loss of topsoil may not be as much of a problem for desert plants, where there is no true topsoil. The exceptions to this would be where there is no nearby source of natural inoculum of soil microorganisms, and where the topsoil has eroded away leaving a hardpan such as a caliche layer, that would either require millennia of weathering or artificial treatment for any plant to grow on it at all. Where topsoil has been lost and is essential to the reestablishment of native vegetation, restoration is not a realistic goal, but reclamation may certainly be practiced to assure that many of the values of the land are reestablished for protection of some of the wildlife and plants.

Biodiversity

The greatest conservation threat that we face is the loss of biodiversity, but restoration cannot be viewed as the cure. Of the many restoration projects in the western U.S. and elsewhere, none have achieved the goal of bringing back the diversity of species that once existed at a particular site. Most prescriptions for revegetation include only a few dozen species at most. Usually these are the most dominant species, and the goal in choosing species is to include those that represent vegetation life forms and structural layers, such as trees, shrubs, forbs, and grasses, where those life forms are part of the natural vegetation. Once these dominant life forms are reestablished, restorationists hope that rare plant species and animals will recolonize. However, the number of rare species in any natural community is always so much greater than the number of abundant species, that the task seems hopeless. We are slowly learning to propagate many rare species, but we do not yet have knowledge or economic resources to introduce them all.

Learning the rates of natural plant recolonization into disturbed areas is important to predicting whether restoration efforts may someday result in vegetation with its original diversity. Large tracts of abandoned farmland in the Sonoran Desert may require 200 years or more just for the recolonization of the dominant creosote bush (Jackson 1992). Obviously, restorationists can hasten the rate of dispersal by artificial introduction of creosote seeds, but we must still wait long time periods for the rare species, hopefully less than 200 years if creosote bush attracts animal seed dispersers or acts as a nurse plant. The scale of the revegetation and the proximity of adjacent disturbed vegetation is critical to the rate of recolonization, and these abandoned farmlands certainly constitute some of the largest restoration problems.

There are still relatively few restoration efforts that are long-term enough to have measured species recolonization. The restored Curtis Prairie in Wisconsin is over 50 years old and probably has most of the species indigenous to native prairies, but these were largely planted by researchers working over many years (Cottam 1987). In a survey of highway revegetation in southern California with sites up to 18 years old, a maximum of 15 native species colonized in the oldest sites (Fig. 3, Allen and others 1993). These native species colonized only where the roadside was adjacent to native shrubland, while exotic Mediterranean annual grasses and forbs colonized everywhere. Generally no more than a dozen mostly native species were planted at any one site along a 50 mile stretch of Interstate 15, so the maximum richness obtained was some 40 species, including the exotics, in plots of some 200 m². The local coastal sage shrubland may have, by contrast, some 70 species in an equal area (Allen, unpublished data).

The most ambitious restoration effort I have seen to reestablish diversity was on a bauxite mine in SW Australia, where some 80 species were seeded that were collected from adjacent jarrah (eucalyptus) forest (Nichols and others 1991). This forest type actually contains some 200 species, and the numbers of species that established, survived and perhaps colonized after 16 years was considerably fewer than those seeded (Gardner pers. comm.).

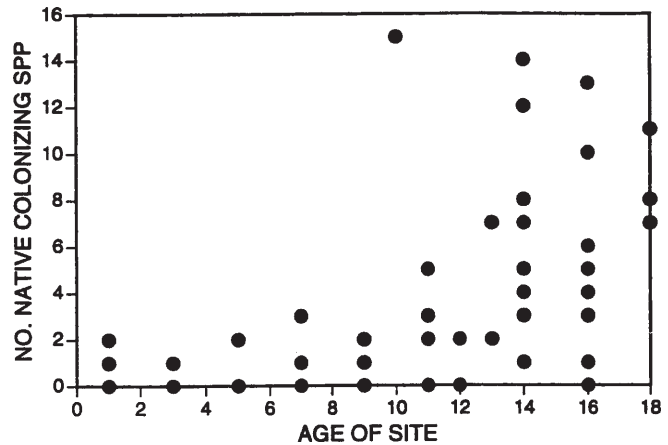


Figure 3—Age of revegetated site vs. number of colonizing native species along Interstate 15. No more than 15 native species colonized any one site, which each had a dozen or fewer planted species. Older sites with few colonizing species were always adjacent to urbanized areas; those with more colonizers were adjacent to native shrublands.

A number of restoration projects have been done with the express purpose of saving a threatened or endangered species, but these usually focus on one or two species and are not done for the purpose of reestablishing biodiversity. For instance, fiddleneck (*Amsinckia grandiflora*) was seeded into areas where it had become locally extinct in California annual grassland (Pavlik and others 1993). This has been done now for a number of other rare plant and animal species in California and elsewhere, although the prospects for long-term survival are still variable (see Pavlik and others 1993 for a brief review). Vegetative restoration was done to attract the endangered least Bell's vireo, which lives in riparian habitat, but the vegetation itself was relatively low in diversity (Baird 1989). Thus, single rare species restoration efforts to date have typically not included a high diversity in the restoration plant mix, but they have rather focused on the needs of the rare species, such as appropriate shrub architecture to attract an endangered bird.

Conclusions

There are other ecological limits to restoration of arid lands in addition to those discussed. Herbivory is known to reduce establishment success in natural and restored communities (McAuliffe 1986), and without protection of seedlings, restoration in some areas may be impossible. This, like lack of moisture, is a limitation that can be overcome with hard work and imagination. An irreversible situation is created by the many dammed rivers and trans-basin water diversions in the western U.S., that have changed surface and groundwater hydrology and assured that restoration cannot take place. This brief paper has dealt with the uplands, but riparian areas in arid lands are also in great need of restoration. The successful restoration of arid landscapes may depend on restoration of

both aquatic and terrestrial systems, as these are interrelated by movement of water, nutrients and biota.

Because of these limitations, our restoration efforts will not be as successful as we would like them to be. The limitations show how important it is for us to conserve those remaining lands that do have high biodiversity, as restoration is not a substitute for conservation. Restoration efforts will still play an important role in conservation even if they are not perfect, because so many lands have been impacted by humans and are in need of restoration. Those lands that are hopelessly weed infested or have very poor soil quality should be identified, and might not be important candidate sites for restoration. Aronson and others (1993) have discussed the concept of "thresholds of irreversibility" beyond which restoration is no longer possible, but revegetation efforts will still be fruitful to improve the ecological and economic value of the land. The cost of restoration may vary from a few hundred to tens of thousands of dollars per acre, depending upon the degree of disturbance and the restoration effort needed. It is up to society to decide to what extent these costs are justifiable. But even with the best restoration efforts, the result will not be vegetation precisely like it was prior to human disturbance. Restoration becomes a valuable exercise even with these limitations if we can increase the habitat value for declining species that would otherwise not be able to exist in an ever more impacted western landscape that is dominated by agriculture, livestock, and urbanization.

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