# Restoration and Rehabilitation of Degraded Ecosystems in Arid and Semi-Arid Lands. II. Case Studies in Southern Tunisia, Central Chile and Northern Cameroon

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**Abstract**

**A model of ecosystem degradation and three possible responses to it—restoration, rehabilitation, and reallocation—is applied to ongoing projects in the arid mediterranean region of southern Tunisia, the subhumid mediterranean region of central Chile, and the semiarid tropical savannas of northern Cameroon. We compare both nonhuman and human determinants of ecosystem degradation processes in these contrasted regions, as well as interventions being tested in each. A number of quantifiable "vital ecosystem attributes" are used to evaluate the effects of ecosystem degradation and the experimental responses of rehabilitation on vegetation, soils and plant-soil-water relations. We argue that attempts to rehabilitate former ecosystem structure and functioning, both above- and below-ground, are the best way to conserve biodiversity and insure sustainable long-term productivity in ecosystems subjected to continuous use by people in arid and semi-arid lands of "the South." The success of such efforts, however, depends not only on elucidating the predisturbance (or slightly disturbed) structure and function of the consciously selected "ecosystem of reference," but also on understanding and working with the socioeconomic, technical, cultural, and historical factors that caused the degradation in the first place.**

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## Introduction

We have previously argued that for ecosystems subjected to long periods of prolonged human disturbance, three alternatives to continued degradation (or complete abandonment) can be defined: restoration, rehabilitation, and reallocation (Aronson et al. 1993*a*). *Restoration*, in the narrowest sense, aims at the complete return of a site to a pre-existing state from a taxonomic point of view—the reestablishment of all indigenous species and the extirpation of all exotics. By contrast, *rehabilitation*, in our sense, concentrates on repairing damaged or blocked ecosystem functions, with the primary goal of raising ecosystem productivity quickly yet sustainably. Rehabilitation normally implies a higher level of ongoing site management than does restoration, wherein it is hoped that innate ecosystem processes will eventually take over. For example, many restoration projects consist primarily of reducing or altogether removing existing human (or livestock) pressures on the site, while rehabilitation projects tend to intervene more massively, resorting to various techniques designed to "jumpstart" the recovery process. In economically depressed areas, ongoing management for sustained primary or secondary productivity will be the general rule.

These differences aside, however, restoration and rehabilitation both aim to re-establish self-sustaining ecosystems characterized by succession in plant and animal communities and sufficient resilience to repair themselves following natural or (moderate) human perturbations. Climatic and edaphic conditions, as well as socioeconomic and cultural conditions, will determine the rates at which either restoration or rehabilitation can proceed. Furthermore, it should be determined whether one or more "thresholds of irreversibility" have been crossed in the degradation process (see below).

The third response to ecosystem degradation, *reallocation*, usually disregards the indigenous ecosystem while imposing new uses on the site. Considerable ongoing input of energy and nutrients are generally required to keep reallocated sites productive in the long term. In most developing countries of "the South," moreover, it is rare that proper care is taken to identify the most appropriate sites for reallocation within a region or landscape.

In this paper, we compare three case studies in arid or semiarid regions of developing countries in light of the general model and four of the hypotheses presented in the previous contribution (Aronson et al. 1993*a*). In all three case studies presented here, structure and function of the original ecosystem have been seriously impaired, such that major interventions appear needed to halt further degradation. It is in such circumstances that the concept of "thresholds of irreversibility" seems particularly useful (Le Houérou 1968, 1993, Grouzis 1988; Aronson et al. 1993*a*). Physical, cultural, and socioeconomic differences among the sites require that different rehabilitation strategies must be applied, yet a certain degree of commonality exists.

We present data for a number of the vital ecosystem attributes presented previously, and introduce some new ones as well. We also discuss the history of land use and resource depletion that provide the background for present efforts in restoration or rehabilitation ecology, as well as contemporary factors that will determine the success or failure of technical "solutions" or proposed ecological approaches to continued degradation in each of the case studies.

**Vital Ecosystem Attributes.** A series of 18 "vital ecosystem attributes" (VEAs) was previously introduced (Aronson et al. 1993*a*) to compare the effects of nonhuman and human disturbances and of experimental interventions on different ecosystems, and to test hypotheses about ecosystems in a given phase of their trajectory. We defined VEAs as those variables that are correlated with, and can serve as indicators of an ecosystem's overall structure and functioning in a given stage of its development. We here employ some but not all of these VEAs, as well as three new ones. Our VEAs relate primarily to vegetation, soils, and microorganisms. A complementary list of VEAs related to fauna (insects, rodents, mammals, and other animals of importance as pollinators, seed dispersers, herbivores, decomposers, etc.) remains to be assembled.

In choosing the VEAs employed here, we seek above all to identify ecosystem-level traits that are reasonably easy to quantify and that can provide comparisons not only among various stages of a given ecosystem's trajectory but also among different ecosystems. Thus, a majority of the 24 attributes suggested by Odum (1969) to distinguish ecosystems in "developmental stages" and "mature stages" are difficult to record in the field. Moreover, research projects do not usually last long enough or have enough funding to allow comparisons of such variables as "gross production/community respiration ratio" or the "role of detritus in nutrient regeneration" at different successional stages. By the same token, some quantifiable community-level traits thought to change with succession may indeed reveal interesting trends among successional stages but cannot be used in comparisons among ecosystems.

Finally, we are unavoidably influenced by the type of environments in which we are working—ecosystems and landscapes heavily influenced by people for the last several centuries at least. In such situations, the history of land use at each individual site will have tremendous impact on extant ecosystem structure and functioning. Therefore, we are generally not able to use comparative data from different sites within a given ecosystem but are obliged to obtain all our data from single sites observed over time. We need VEAs that are not only applicable to all (or nearly all) ecosystems, but that are also sufficiently sensitive to human (and other) disturbances so as to show variations within a few years or decades. We thus speak at times of the relative response rates of VEAs to varying kinds of disturbance or intervention. All of these factors have gone into the construction of our revised list of would-be "universal" VEAs.

The first seven VEAs (perennial species richness, annual species richness, aboveground phytomass, total plant cover, life form spectrum, soil organic matter, and length of water availability period) to be used here are straightforward and were defined in detail previously (Aronston et al. 1993*a*). It is worth adding here that at times it may be necessary to obtain data for total plant cover separately for annuals and perennials, especially in arid and semiarid regions. Furthermore, in some situations it is useful to quantify cover for each class of plants represented in the life-form spectrum. Similarly, it can often be useful to distinguish between herbaceous and woody perennial plant cover.

The next three VEAs require a bit more explanation. *Coefficient of rainfall infiltration* (CRI) is defined as the amount of water infiltration into the soil, and is thus an indicator of solid-surface conditions and of absorption capacity of upper soil layers (Floret et al. 1981; Chaïeb et al. 1991). (We previously called this VEA "coefficient of rainfall efficiency (CRE)," but to avoid confusion with the term "rain use efficiency" (see below, we now prefer the term CRI.)

*Maximum available soil water reserves* (WR) indicate the naturally held reservoir of rainwater in the friable, upper soil layers. Given the unpredictability of rainfall in arid and semi-arid lands, this parameter is particularly important for plants (and many animals). Moreover, it integrates many different parameters of interest to the ecologist, such as the depth of soft, easily penetrated (and cultivatable) soil, and physical properties such as water holding capacity of the soil in that layer, organic matter content, and so forth. However, it should ideally be presented, whenever possible, with details concerning the depth of the soft soil layer.

*Rain use efficiency* (RUE) (Le Houérou 1984) is the ratio of kilograms DM aboveground phytomass produced per hectare per millimeter of rainfall. At times, this rough grain measurement may be replaced by the more precise *Water use efficiency*, expressed in terms of net $CO_2$ uptake per unit of transpiration only (mg $CO_2$ $gH_2O^{-1}$), and applicable at the level of the single leaf (Fischer & Turner 1978), of populations, or of crop stands (Floret et al. 1983; Seiny-Boukar et al. 1992).

Apart from these ten VEAs, three additional ones will now be introduced. The first is a structural attribute; the latter two are functional VEAs.

In many arid and semi-arid lands, the relative size and diversity of the *soil-borne seed bank* has been found to be a highly useful VEA of ecosystem structure, since the absence of certain keystone species in the seed bank may serve as a criterion by which to determine whether simple exclosures are adequate to achieve restoration or if interventions (such as artifical reintroduction or mechanical soil surface modifications) are necessary as well, in the name of rehabilitation. It may be worth emphasizing that by "seed bank" we speak only of *viable* seeds found in the soil.

A new functional VEA is soil-surface conditions (SSC). Various aspects of this broad parameter may, in different situations, be of great importance in determining water infiltration, run-off, and erosion factors in arid and semi-arid regions (Escadafal 1981; Casenave & Valentin 1989). SSC is particularly useful when large areas are considered with the aid of aerial photography and/or satellite imagery. In the field, it is necessary to define more closely which particular aspects are of primary importance; thus, surface sealing or encrustation can be a serious factor. Likewise, in our northern Cameroon case study, the presence or absence of deep soil cracks must be taken into account, because they play an important role in soil-water dynamics. Furthermore, the interrelations of SSC, CRI, and RUE should be noted. Data on the first two can help predict the latter for a given site.

Casenave and Valentin (1989) developed a preliminary key for determining the elemental types of soil surfaces found in arid and semi-arid regions. Ultimately, it would be useful to establish a hierarchical system for classifying SSC at any given state of an ecosystem's trajectory.

Finally, the well-known pedological variable *cation exchange capacity* (CEC) should also be considered as a functional VEA, because it is universally applicable, highly sensitive to degradation, and directly correlated with overall soil fertility. We present data for this VEA for the Cameroonian vertisols.

In summary, our full current list of VEAs is as follows (definitions for VEAs I:6–10 and II:1, 9–11 are given in Aronson et al. 1993*a*):

I. VEAs Related to Ecosystem Structure
   1. Perennial species richness
   2. Annual species richness
   3. Total plant cover
   4. Soil-borne seed bank
   5. Aboveground phytomass
   6. Beta diversity
   7. Life form spectrum
   8. Keystone species (presence or activity)
   9. Microbial biomass
   10. Soil biota diversity.

II. VEAs Related to Ecosystem Function
   1. Biomass productivity
   2. Soil organic matter (OM)
   3. Soil surface conditions (SSC)
   4. Coefficient of rainfall infiltration (CRI)
   5. Maximum available soil water reserves (WR)
   6. Rain use efficiency (RUE)
   7. Cation exchange capacity (CEC)
   8. Length of water availability period
   9. Nitrogen use efficiency (NUE)
   10. Microsymbiont effectiveness
   11. Cycling indices.

## Study Sites

**1. Southern Tunisia.** Over the past 40 years, a very rapid evolution has occurred in the interrelated factors of demographic and land tenure conditions, and increased investment possibilities on the part of returned immigrants. (Tunisia gained independence from France in 1956.) In the last 25 years, a dichotomy has become established between pastoralism and rainfed agriculture. Demographic pressures, and concomitant extensions of cultivated lands, will no doubt further diminish pastoralism. For the pre-Saharan (100–200 mm mean annual precipitation) region of Tunisia (the southern part of the country, excluding oases and the desertic Saharan region, the so-called "grand erg occidental"), human population density reached 24.2 inhabitants/km² in 1985, compared to only 5.3 inhabitants/km² 100 years earlier (DSA/CIRAD 1985).

Until quite recently, the very long history of human settlement and land use in the sandy plains of southern Tunisia proceeded virtually without change, largely due to the absence of permanent sources of surface water. It was only with the advent of mechanization, combined with growing demographic pressure throughout the region after World War II, that permanent settlements began to appear. Nowadays, people use tractors to transport lightweight metal tanks holding between 1000 and 2000 liters of drinking water, and are thus able to settle in areas that were formerly uninhabitable due to lack of water.

Nevertheless, some three-quarters of pre-Saharan Tunisia is said to still be used primarily for pastoralism, and some 70% of this area is still under collective ownership (Ministère d'Agriculture 1985). This situation has changed rapidly over the past 25 years, however, with some positive and some negative consequences. The process was as follows: all lands uncultivated when visited by the *Commission des Affaires Foncières* of the Ministry of Agriculture were declared to have collective status, even if locally they had long since been managed by a single family. To avoid this classification and obtain instead private-land status for their plots, many individuals borrowed or bought small tractors and plows and cultivated portions of land prior to the visit of the Commission. They then quickly sowed wheat or barley or planted olive trees, irrespective of the economic or ecological advisability of such activities at those sites. The essential thing was to establish a "presence" on the land so that the Commission would recognize owners for each "cultivated" field. It may be mentioned that in traditional Islamic law also, land "belongs" to those that work it. We seek to emphasize that a new situation was created by the Land Tenure Commission that led to rapid, widespread degradation of soils and vegetation in areas ill suited to cultivation. It is here that our experiments in rehabilitation have been centered.

The positive aspect of the now-outmoded process described above is that, once privatized (after three years of permanent "occupation"), these lands became eligible for mortgaging and served as leverage in obtaining bank loans, or cash in the case of outright sales. The negative aspect derives from the fact that most of the 30% of potentially privatized lands were in the least arid portion of the arid region. Degradation set in rapidly when these fragile lands were coveted by potential "owners" and the natural steppic vegetation was destroyed by them in anticipation of a visit from the government Commission. Fortunately, this process has been greatly decentralized and simplified in the past five years. But it will take many decades to undo the damage that has been done.

Our main study sites are near Menzel Habib, 50 km northwest of Gabes, in a 82,000-ha region of mainly sandy plains (37°80′–38°15′N, 8°00°–8°40′E). Average altitude is 80 m.a.s.l. (range 70–100 m). When intact or only slightly degraded, the vegetation on this sandy plain is now characterized by a subshrub layer 20–50 cm tall, consisting especially of *Rhanterium suaveolens* and *Artemisia campestris*, and accompanied by a num-

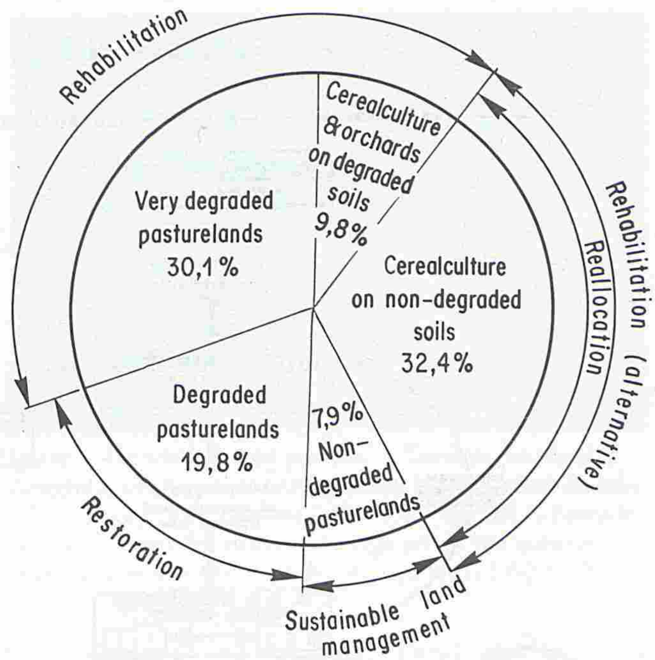



Figure 1. Land occupation in Tunisian study area as of 1975, with indications of which portions seem most suitable for restoration, rehabilitation, and reallocation.

ber of perennial grasses that are constantly overgrazed (Le Houérou 1959). A lower stratum (10–30 cm) contains perennial, rhizomatous, and bunch grasses, and a large number of annual grasses and forbs. This "shrub-steppe" occurs between the rainfall isohyets 100–200 mm, on plains below 200 m wherever sand has covered the silty substrate filled with calcareous nodules. This substrate in turn overlies a layer of Mio-pliocene gypsum (Floret et al. 1981, 1983, 1990).

It should be noted (Le Houérou 1959, 1969) that thousands of years of firewood collection, cereal culture, and chronic overgrazing have created this derived shrub-steppe from a more complex "arboreal steppe" that formerly supported a heterogeneous tree stratum over large areas, including a large, spiny legume tree, *Acacia tortilis* (Forssk.) Hayne subsp. *raddiana* (Savi) Brenan. There is only one relatively large remnant of this arboreal steppe in southern Tunisia, in the Bou Hedma National Park 80 km northwest of Gabes. In our study area, the last tall acacia tree fell to the axe about 15 years ago.

In the plain of Zougrata (about 82,000 ha), which includes the Menzel Habib area where we are working, land occupation as of 1975 is summarized in Figure 1. (More recent data are not available.) Lands cleared for cultivation on soils still in reasonably good condition represent about one-third of the area, while cleared, somewhat degraded lands, uncleared pasturelands not yet degraded, and those slightly degraded all

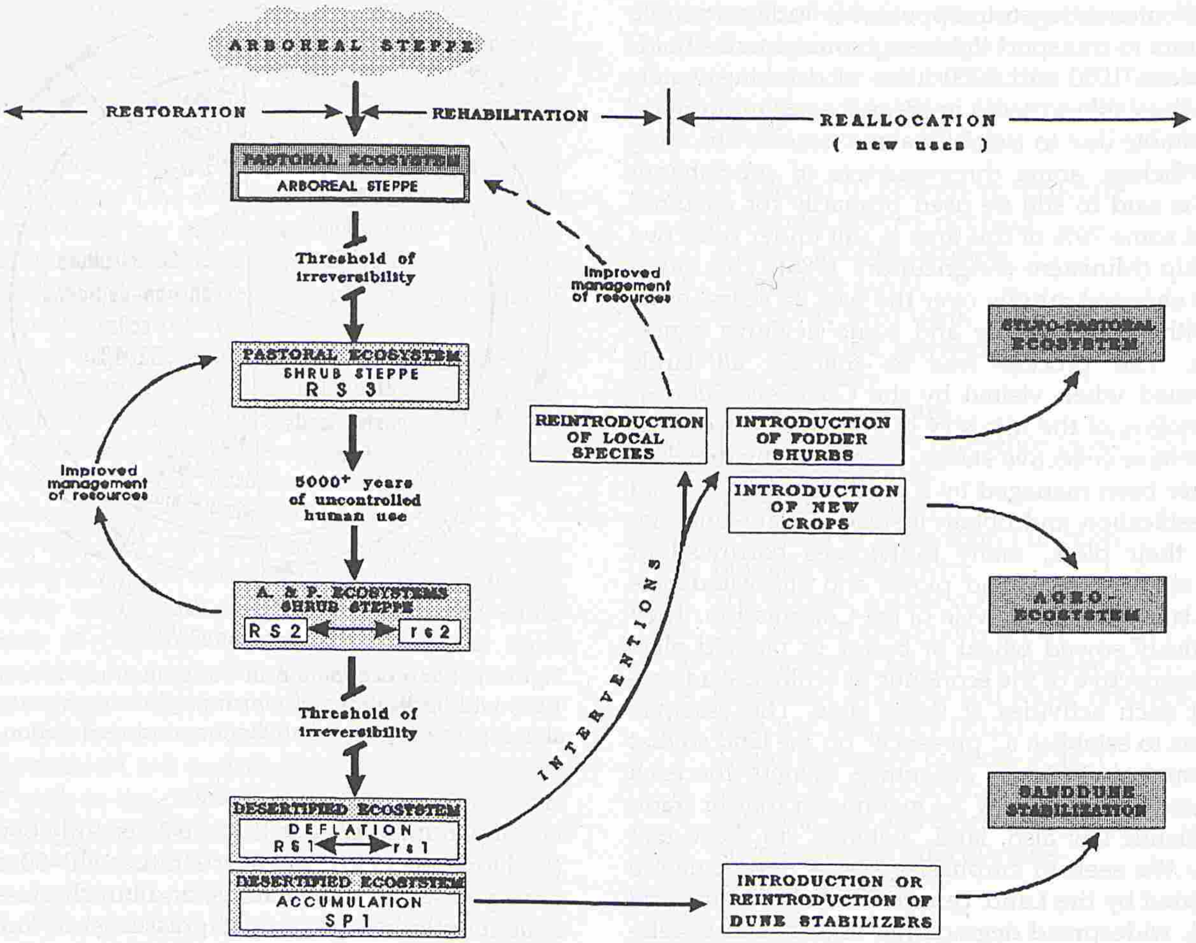



Figure 2. Application of the general model (Aronson et al. 1993*a*) to the southern Tunisia case study, showing the effect of aelion sand movements. $RS_3$ = *Rhanterium suaveolens*–dominated shrub-steppe on deep sandy sierozems in relatively undisturbed condition; $RS_2$ = *Rhanterium suaveolens*–dominated shrub-steppe on deep sandy sierozems somewhat degraded through overgrazing and woodcutting; $RS_1$ = *Rhanterium suaveolens*–dominated shrub-steppe on deep sandy sierozems badly degraded and with truncated soils; $SP_1$ = *Stipagrostis pungens*–dominated grass-steppe on aeolian sand deposits; A. and P. ecosystems = agro and pastoral ecosystems.

represent under 20% each. Badly degraded lands (30.1%) are nearly as prominent as those deemed in good condition (Floret et al. 1978). Figure 1 also shows how each area of land occupation can be classified according to whether it is most suitable for restoration, rehabilitation, or reallocation. Thus, it should be noted that about 8% of the pasturelands are in comparatively good condition as a result of underuse. For purposes of comparison, however, we can say that a sustainable form of land management is in practice here and no interventions are necessary. By contrast, restoration should be considered for some 19.8% of the slightly degraded pastures, while rehabilitation seems appropriate for no less than 39.9% of the study area. Although reallocation at present seems the most appropriate response for the 32.4% of the area currently occupied by cereal culture on degraded land that is unsuited for this activity, conditions may change in the future so that rehabilitation could be attempted here as well.

Degradation of cleared lands leads to a rapid reduction in the coefficient of rainfall infiltration (CRI). In pasturelands, CRI is also closely correlated with the state of the steppic vegetation. Pastoralism is increasingly restricted to smaller and smaller areas, and state subsidies for the purchase of feed supplements are declining. As a result, herd sizes decrease. The price of meat is expected to rise soon, however, and one hopes a new "economic steady state" for animal husbandry may arise.

Figure 2 shows in more detail the successive stages of degradation from the original arboreal steppe to shrub steppe and further desertified ecosystems, as well as the alternative pathways of restoration, rehabilitation, and reallocation as we see them. The shrub-steppe (RS3) stage shows relatively high total plant cover (35–40%), but gradually gives way to a depleted RS2 stage (Floret et al. 1978) due, in part, to poor germination of annuals on sealed soil-surface crusts or destabilized sands. Transformations due to overgrazing are more or less similar to those following prolonged plow agriculture. The removal of shrubs and perennial grasses by plowing causes the uppermost sand layer to disappear as well, and encroachment by the post-cultural shrub *Artemisia campestris* and the weedy perennial grass, *Cynodon dactylon* L. proceeds rapidly.

After the sand layer has been removed, a silty substratum with calcareous nodules is revealed. This is highly unfavorable for the growth and persistance of most native plants. We call the highly degraded steppe occupying such truncated soils "RS." When the steppe is placed under cultivation, the resulting fields are labelled according to the degree of soil degradation: rs2 = RS2 after cultivation: rs1 = RS1 after cultivation.

This study area's particular features require a deviation from the general model presented previously (Aronson et al. 1993*a*). Thus, at the bottom of Figure 2, two variants of maximum degradation are shown. Where the successive ablation (removal) of sandy soil from surface layers leads to the build-up of loose sands in nearby depressions, the native perennial grass *Stipagrostis pungens* rapidly colonizes the resulting sand drifts, where only a few native annuals can germinate and survive. We call this stage SP1.

In 1975, the portion of the study area originally occupied by shrub-steppe dominated by *Rhanterium suaveolens* was estimated as 42,8000 ha, of which 60% occurred as pastureland—shrub-steppe, or degraded derivatives thereof (RS3, RS2, RS1), and 40% was under cultivation in fallows or in preparation for cultivation in the near future: rs2 and rs1 (Floret et al. 1978). The highly degraded steppe (RS1) produced by overgrazing represented, in 1975, 31% of the steppes on sandy substrate, and severely degraded lands suffering from overcultivation (rs1) occupied about 8.9% of the same area. The remaining 60% of the area was less degraded (RS3, RS2, rs2). Since 1975, degradation has continued unabated (Figure 3).

Table 1 shows the effects of degradation of an ecosystem on sandy soil, and of initial stages of rehabilitation on various vegetation- and soil-related VEAs at the study site. The drastic diminution of species richness, total plant cover, and aboveground biomass are cumulative in succeeding stages of degradation. The combined effects of woody vegetation and topsoil removal cause reductions in both volume and duration of available water reserves. Organic matter in the upper soil layers also decreases by 75%. Consequently, CRI and RUE decline as well, although not as dramatically as the vegetation variables do. Such differences between rate of soil degradation and retrogression of vegetation are well-known in ecosystems of arid and semi-arid lands. Furthermore, the subsoil layers at the study site are quite sandy, so surface runoff has never been high and little change in CRI or RUE was to be expected.

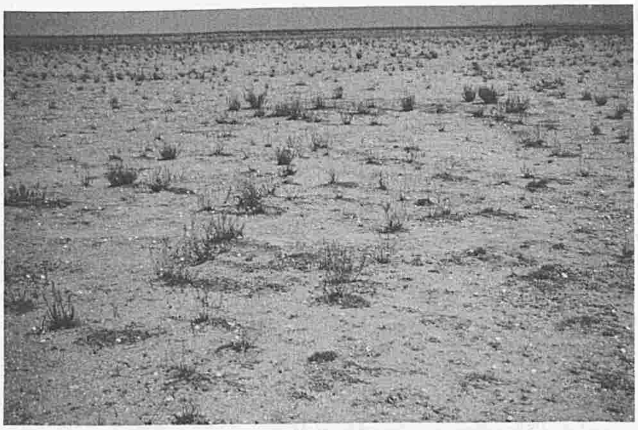


Figure 3. Unrehabilitated portion of Tunisian study site (Zougrata) on loamy crusted plateau locally called *Hamilet el Babouch* or "Snail plateau." The snail species is *Leucochroa candidissima*; the dried-out twigs are of the subshrub *Artemisia campestris*. (Photo by: E. Le Floc'h 1990.)

In early stages of a restoration project begun in the early 1950s (Long 1954), we found that the positive effects of simple exclosures to eliminate grazing pressure (Floret 1981) were aided by aeolion transport and localized accumulation of sand at the base of surviving shrubs and bunch grasses. Build-up of sand facilitated the germination and reestablishment of many native plant species. Such restoration processes proceeded most quickly in the RS2 sites, where mother plants and propagules of many species of the original shrub-steppe were still present, albeit in reduced numbers.

In the most degraded sites (RS1 and, mainly, degraded fallows on rs1), more important interventions were needed as well, in order to speed up the recovery process. Our rehabilitation studies thus began by seeking to identify a mixture of native herbaceous perennials that would be complementary, both in time and space, in their utilization of available ground water, and that were known to be productive and resistant under conditions of livestock grazing. A large body of experimental data was accumulated on these species, mostly perennial grasses, and many were successfully reintroduced on a small scale (Telahigue et al. 1987; Chaïeb et al. 1991; Neffati et al. 1991).

The right-hand column of Table 1 gives some preliminary indications of the dramatic changes that can be obtained through rehabilitation efforts involving reseeding of selected native species thought to be keystone species—*Rhanterium suaveolens, Stipa lagascae,*

**Table 1.** Vital ecosystem attributes, in late spring, of the typical steppe, 50 km northwest of Gabes, Tunisia, in various stages of degradation and three years after the beginning of experimental rehabilitation

| | *Stages of Ecosystem Development*[1] | | | | |
|---|---|---|---|---|---|
| *Vital Attribute* | $RS_3$ | $RS_2$ | $RS_1$ | $SP_1$ | *Rehabilitated Steppe*[2] |
| 1. Number of annual species | 41 | 13 | 6 | 2 | 21 |
| 2. Number of perennial species | 23 | 18 | 3 | 1 | 12 |
| 3. Total plant cover (%) | 35 | 20 | 2 | 0.5 | ca. 30 |
| 4. Average aboveground biomass in spring (kg DM $ha^{-1}$ $yr^{-1}$) | 1800 | 600 | 200 | 100 | ca. 1200 |
| 5. Soilborne seedbank | n.a.[3] | n.a. | n.a. | 32[4] | n.a. |
| 6. Soil surface conditions | Aeolian sands (mobile) | sand or loamy (crusted) | loamy (with crust) | sandy deposits (mobile) | mainly sandy (stable) |
| 7. Organic matter concentration in upper soil layers (%) | 0.42 | n.a. | n.a. | 0.1 | n.a. |
| 8. Maximum available water reserves in soil (WR) (mm) | 140 | 125 | 90 | 100 | n.a. |
| 9. Duration of available water for plants (months) | 6–7 | 5–6 | 3–4 | 5–6 | n.a. |
| 10. Coefficient of rainfall infiltration (CRI) (%) | 100 | 95 | 85 | 100 | n.a. |
| 11. Rain use efficiency (RUE) (kg DM $ha^{-1}$ $mm^{-1}$ of precipitation) | 5.0 | 2.8 | n.a. | n.a. | n.a. |

[1]$RS_3$ = *Rhanterium suaveolens*–dominated shrub-steppe on deep sandy sierozems in relatively undisturbed condition; $RS_2$ = *Rhanterium suaveolens*–dominated shrub-steppe on deep sandy sierozems somewhat degraded through overgrazing and woodcutting; $RS_1$ = *Rhanterium suaveolens*–dominated shrub-steppe on deep sandy sierozems badly degraded and with truncated soils; $SP_1$ = *Stipagrostis pungens*–dominated grass-steppe on aeolian sand deposits.
[2]See text for explanation.
[3]Not available.
[4]Total number of species as revealed by germination in a rainy year following superficial plowing.

Sources: Floret et al. 1978, 1982; Le Floc'h, unpublished data.

*Cenchrus ciliaris*, and *Plantago albicans*. Seeding was carried out following a superficial plowing (about 5–10 cm deep) similar to that practiced locally for sowing barley. Results of these experiments will be discussed in more detail in a forthcoming paper. Here we show simply that no less than five VEAs showed a positive response after only three years (Figure 4).

As mentioned, severe aeolian erosion in the region produces shifting sands that represent a serious environmental hazard for man, woman, and beast. Fixation of these dunes in our study area has been attempted through a technique that consists of identifying prevailing wind direction at a site and then erecting a tall, sturdy obstacle (made of dammed earth, fibrocement, or the like) against which shifting sands can accumulate. As needed, these artificial dunes are regularly increased in height to avoid sand spillover. At a certain variable point, these barriers are no longer submerged by sand and can be left unmanaged thereafter. The installation of artificial dunes can accompany or follow attempts at restoration (grazing exclosures), rehabilitation, or—most frequently—reallocation (fodder shrub plantations or fuelwood plantations of *Tamarix* spp.).

Recently, the reintroduction of the native *Acacia tortilis* subsp. *raddiana* and other nitrogen-fixing shrubs has begun as well, in efforts to recreate (or rehabilitate) a simplified arboreal steppe. Meanwhile, many local landowners are currently experimenting with a vast

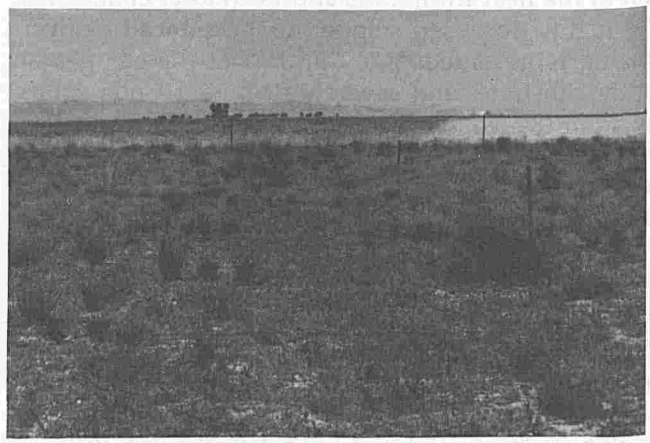


Figure 4. Same site as in Figure 3 plowed to a depth of 10 cm. Resulting plant cover consists mostly of annuals arising from the soil-borne seed bank. To the right and the left are two plots of rehabilitated steppe (plowed and sown), dominated by the subshrub *Rhanterium suaveolens*. (Photo by E. Le Floc'h.)

assortment of reallocation schemes. The future of landscapes and ecosystems in this area is thus very much in flux.

**2. Central Chile.** Over two million ha of the mediterranean climate region of central Chile (30–37° S) are dominated by the spiny legume tree *Acacia caven* and a wide variety of exotic (and some native) annuals (Ovalle et al. 1990). The resulting synanthropic formation (locally known as "espinales") occurs in an area where mean annual rainfall ranges from 1000 mm in the south to less than 100 mm in the north, and on virtually all gentle slopes and plains where the original sclerophyllous vegetation (matorral) has been cleared, and where overgrazing is a severe and chronic disturbance (Fuentes & Hajek 1979; Etienne et al. 1982). The introduction of exotic herbivores (cattle and European rabbits) also contributes to maintaining Chilean espinales in their present state (Jáksic & Fuentes 1989).

In particular, there were two specific periods during the post-Conquest history of Chile when the soils of central Chile were overexploited by large landholders to take advantage of temporarily booming export markets for wheat. The first of these began in about 1687 when the Spaniards in Peru began importing cereals from Chile (Gay 1865). Two centuries later, from 1848 until about 1890, the California Gold Rush created a huge market for wheat, a commodity that central Chile was then well-placed to supply (Domic 1979). The current state of exhaustion of espinal soils in the subhumid portion of central Chile, where rainfed cereal culture is far more practical than it is further north, can be directly linked to the unrestrained, short-term exploitation of limited resources through the "mining" of topsoil there in the past three centuries.

At present, the depleted soils and low productivity of most espinal soils are perpetuated by systems of low input and low output, exceedingly small farm units, animal traction, and a variety of other peasant practices. As a result of the prevailing land-use systems, some two-thirds of espinal soils are badly eroded (IREN 1964), and microbial activity is practically nil in many places. To this day, dryland farmers add little or no fertilizer or manure to their fields. Moreover, the nitrogen-fixing capacity and highly useful, out-of-phase phenology of *A. caven* are put to use by only a very small number of farmers in the region (Ovalle 1986; Ovalle & Avendaño 1987). Instead, the tree is treated, ineffectively, as a woody weed (Figure 5).

As currently managed by most local farmers, and even when totally abandoned, the espinales of central Chile appear to be an ecologically and economically blocked system showing little or no secondary succession or hope for increased agricultural or animal husbandry yields in the foreseeable future (for more details, see Ovalle et al. 1990, 1993). Unless, that is, major interventions are undertaken. As in the case of the Tunisian case study, our mandate is primarily to explore ecological means to reverse current trends. We have also recently begun to work more closely with rural economists, however, and the discussion that follows will thus reflect some of this interdisciplinary approach.

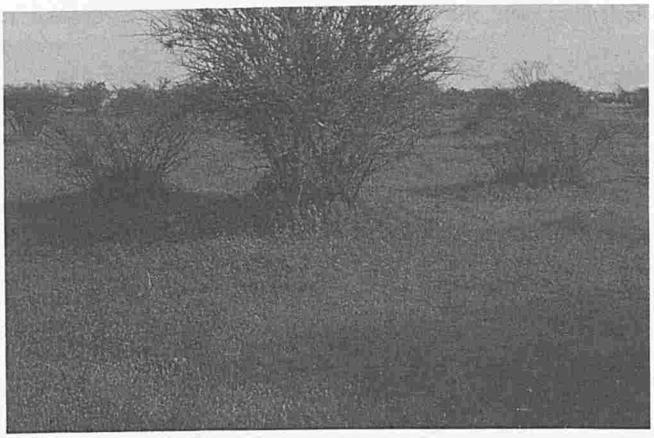
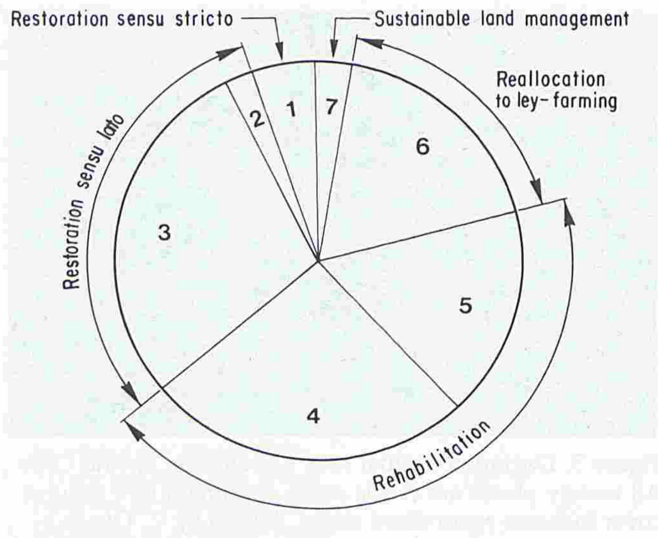


Figure 5. Degraded espinal near Cauquenes, central Chile. All woody plants are *Acacia caven*. Dominant forb ground cover includes naturalized aliens. (Photo by C. Ovalle.)

The sites where most of our studies have taken place in the Chilean espinal lie in the seventh administrative region, near Cauquenes (35°58′S; 72°17′W), where mean annual precipitation is 695 mm and the period of summer drought lasts 5–6 months. Soils in the area are loamy and of granitic origin, but badly leached, with pH 6.0–6.5 and very low nutrient levels. A long history of shallow cultivation, with animal traction only, and uncontrolled pastoral use have also led to severe compaction of most soils, some of which are therefore subject to periodic inundations in low-lying areas. Consequently, water infiltration is poor and rainfall is mostly ineffective for vegetation. *Acacia caven* forms a nearly monospecific shrub stratum, and most of the herbaceous understory consists of exotic annuals of little forage value (Ovalle 1986).

Evidence exists that revised management of *A. caven* in these espinales will positively affect pasture composition and productivity (Ovalle 1986; Ovalle & Avendaño 1987). But only about 2% of the Chilean espinales in the subhumid, mediterranean climate zone (about 37–35°S) possess adequate diversity and seedbanks to allow for fundamental structural and functional improvement through the revision of management techniques alone. Therefore, our efforts (Ovalle et al. 1990, 1993; Aronson et al. 1993*b*) concentrate on the artificial introduction and reintroduction of nitrogen-fixing legumes—both annuals and perennials, with associated




1 Degraded matorral (5%)
2 Very good espinales (2%)
3 Good espinales (28%)
4 Degraded espinales (27%)
5 Badly degraded espinales (17%)
6 Wheat and fallows (18%)
7 Vineyards (3%)

Figure 6. Land occupation in a 23,000-ha transect near Cauquenes (7th Region, Chile), as of 1988, with indications of which portions seem most suitable for restoration, rehabilitation, and reallocation. Evaluation of the state of the espinales ("good," "degraded," etc.) was on the basis of stem density and overall pasture value. The reallocations represented by vineyards and "very good espinales" appear more or less stable; all other land occupations at present are highly subject to change.

rhizobacteria for each candidate species in degraded and badly degraded espinales, which represent about half of the land surface in the Cauquenes district (Figure 6).

Based on many historical and first-hand accounts, the fertility and productivity of espinales of the region have declined steadily since the middle of the last century, and this no doubt has to do with the short-term mining of soils referred to above. Our goal is the rehabilitation of these espinales to a situation we call "mixed espinales" such as were common 100 years ago. This goal is also based on the existence of farmers who have chosen or have been convinced to manage their *A. caven* and other trees as a useful resource. One of these farms, La Estrella, serves as the source of data for what we call "rehabilitated espinal" in Table 2.

Table 2 shows that number of annual (mostly exotic) species rises dramatically when the matorral gives way to espinales of varying types, whereas perennial species richness drops off sharply. Total plant cover decreases dramatically with transformation to espinal, and the repeated burning and clearing leads to sharp reduction in woody plant cover. Belowground trends are similar: organic matter content, available water reserves in the soil, and duration of available water period for plant roots drop precipitously.

At La Estrella, by contrast, in the most outstanding example of a rehabilitated espinal, marked improvements in virtually all VEAs were observed following reorientation of espinal and livestock management as described above. The data included in the right-hand column of Table 2 are taken from this single farm, but it seems reasonable to assume that similar results could be achieved by other farmers in the region (Figure 7).

Ultimately, more contour plowing, revegetation, and bacterization (see Allen 1989; van Elsas & Heijnen 1990) will all be required in areas badly affected by erosion, as well as reorganization of small watersheds now under intensive farming pressure. Fortunately, in parts of the seventh Region of Chile, the National Forestry Corporation (CONAF) has begun several projects along these lines (V. Mourges, personal communication).

Despite their degraded state, the Chilean espinales provide the framework for subsistance-level agriculture and animal husbandry for approximately 300,000 people (Ovalle et al. 1990). Thus, ecological restoration or rehabilitation cannot be disassociated from agriculture and livestock management practices. On the contrary, it is only by incorporating economically beneficial changes in these human activities that increased biodiversity and other environmental desiderata can be achieved.

Figure 8 shows the succession from the original matorral to epinales of varying stages of degradation, as well as the alternative pathways of restoration, rehabilitation, and reallocation as we currently perceive them. In the long-term, we are convinced that fodder trees

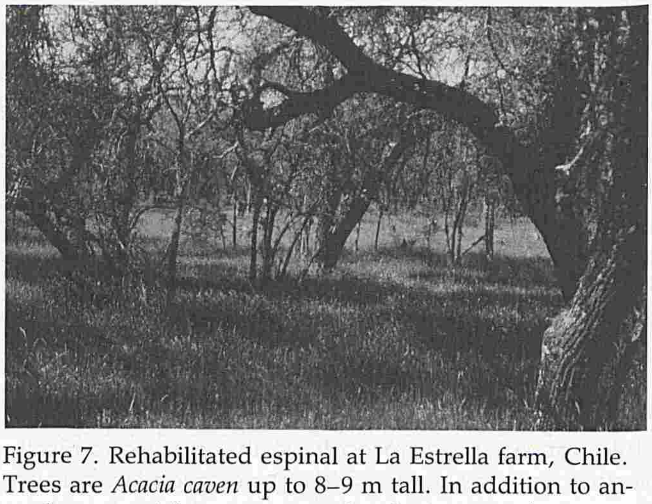


Figure 7. Rehabilitated espinal at La Estrella farm, Chile. Trees are *Acacia caven* up to 8–9 m tall. In addition to annuals, perennial grasses comprise dense ground cover. (Photo by C. Ovalle.)

**Table 2.** Vital ecosystem attributes, in late spring, of the Chilean matorral and the succeeding espinal in the subhumid zone near Cauquenes, in various stages of degradation and following rehabilitation.

| | *Stages of Ecosystem Development*[1] | | | | |
|---|---|---|---|---|---|
| *Vital Attribute* | *Matorral* | *Esp1* | *Esp2* | *Esp3* | *Rehabilitated Espinal* |
| 1. Number of annual species | 15 | 46 | 27 | 17 | 30–40 |
| 2. Number of perennial species | 30 | 16 | 8 | 4 | 5–10 |
| 3. Total plant cover (%) | 170 | 95 | 70 | 10–30 | 50–80 |
| 4. Number of *A. caven* stems $ha^{-1}$ | 0–10 | 1000+ | 600–800 | 300–500 | 630–830 |
| 5. Average height of *A. caven* (m) | —— | 5–7 | 3–4 | 1–2 | 3.5–5.5 |
| 5. Soil organic matter (%) | 3–5 | 2.7 | 1.0 | 0.2 | 3–4 |
| 6. Maximum available soil water reserves (WR) (mm) | 100–120 | 70 | 50 | 30 | 50–80 |
| 7. Length of water availability (months) | 8 | 7 | 6.5 | 5.5 | 6–7 |
| 8. Pastoral value[2] | 0–5 | 40–50 | 20–30 | 10–15 | 30–40 |
| 9. Aboveground phytomass (tons DM $ha^{-1}$ $yr^{-1}$)[3] | 0.1–0.5 | 3.5–4.5 | 2.0–3.0 | 0.7–1.0 | 2.5–4.5 |

[1]Uncut matorral near Cauquenes (7th Region) (approximations); Esp1 = mixed espinal, with *A. caven*, *Maitenus boaria*, and other tree species; Esp2 = degraded espinal (50–75% tree cover); Esp3 = badly degraded espinal (10–25% tree cover) (see text for explanation).
[2]Based on botanical and composition and fodder value of dominant herbaceous species of the understory (maximum = 100).
[3]Herbaceous plants only.

Sources: Ovalle 1986; Ovalle & Aronson, unpublished data.

and shrubs, especially those capable of biological nitrogen fixation, have an important role to play in both the ecological and economic rehabilitation of the Chilean espinales.

In the short-term, however, such revegetation will be too costly for most small land-owners to afford, especially in view of the degraded state of most espinal soils and the ubiquitous presence of voracious rabbits and hares, both of which render tree and shrub planting a somewhat risky and costly affair. Thus, concurrent with experiments with perennial plants, we are also selecting naturalized ecotypes (and rhizobial inoculants) of the annual legume *Medicago polymorpha* L., and testing alternative crop rotation systems designed to improve soil fertility while also increasing cashflow on medium-size farms.

In Figure 8, this approach is represented by the term "ley farming," which comes from the old English word 'ley' (= pasture). This method combines rainfed cereal culture and animal husbandry with the sowing of annual medics (*Medicago* spp.), subterranean clover (*Trifolium subterraneum* L.), or other annual legumes (Puckridge & French 1983; del Pozo et al. 1989). The rotation cycle varies from one to three years, depending on rainfall, local soil, and market conditions. As shown in Figure 8, ley farming can be seen either as a temporary "reallocation" on the path to full rehabilitation of degraded espinales or, preferably, as the first step toward the creation of multi-tiered agroforestry systems—to a permanent, economically and ecologically viable reallocation.

In 1983, the first experiment in ley farming was begun on a 4-ha plot near Cauquenes. In a four-year rotational scheme combining hualputra and winter wheat, annual yields of the naturalized medic *Medicago polymorpha* reached 8–11 D.W. tons/ha in good to very good rainfall years (mean annual precipitation at the site is 676 mm). Even in a year of lower-than-average rainfall (440 mm), hualputra yields were between 2 and 4 D.W. tons/ha. Wheat yields in the rotation ranged from 2.2 to 4.5 D.W. tons/ha, whereas average yields on nearby farms were only 1 to 1.2 D.W. tons/ha (Avendaño et al. 1990). Finally, ewes grazing on the annual medical pasture gained 150–351 kg/ha in periods ranging from 120–180 days, depending on the year. These are much higher live-weight gains than

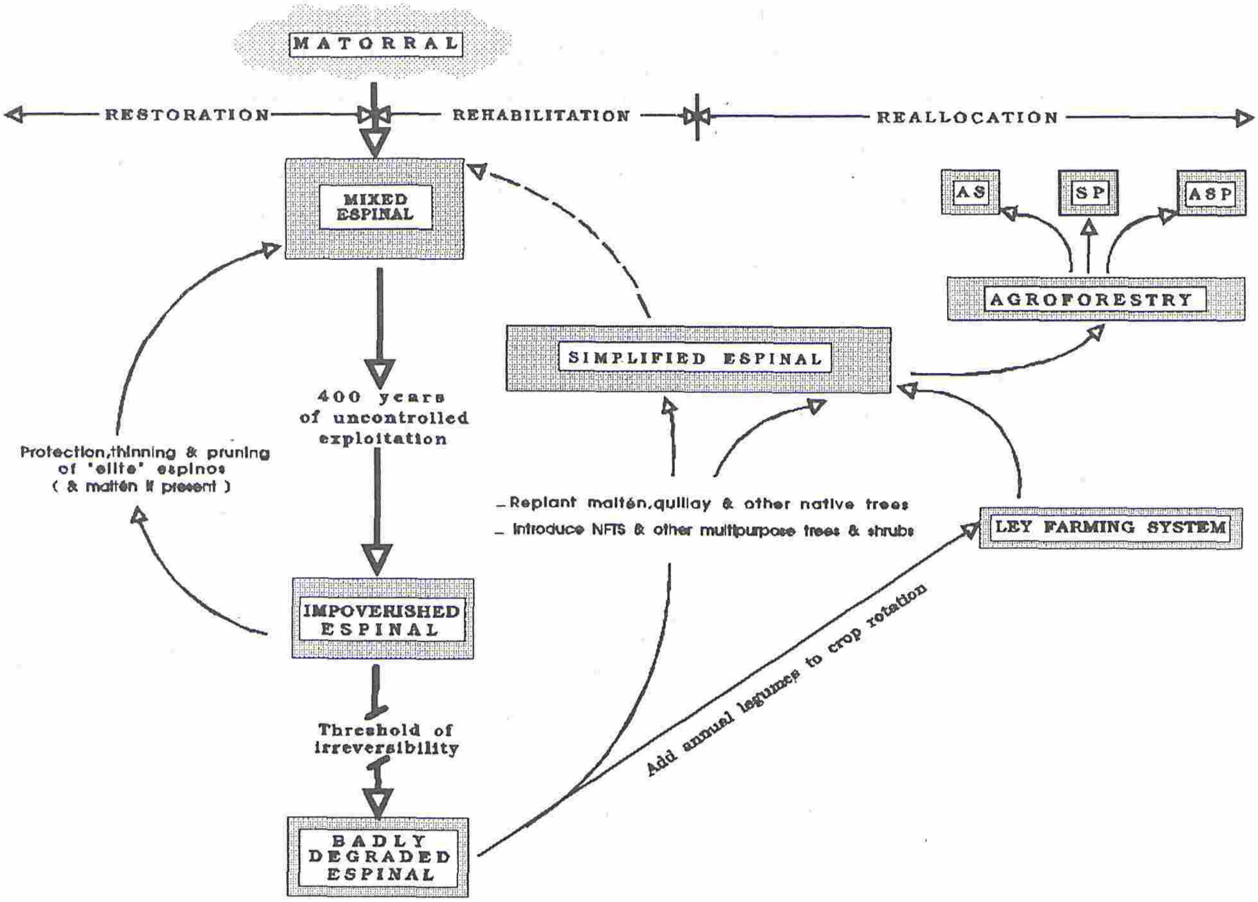



Figure 8. Application of the general model (Aronson et al. 1993*a*) to the Chilean rehabilitation study. NFTs = nitrogen-fixing trees; AS = agrosilvo; SP = silvopastoral; ASP = agrosilvopastoral.

those normally obtained in this region. Accordingly, it appears that ley farming is perfectly adaptable to espinal conditions and that it represents a rapid means of increasing soil fertility. A detailed account of this work and of preliminary results of tree and shrub planting can be found in Ovalle et al. (1993) and Aronson et al. (1993*b*).

**3. Northern Cameroon** The narrow northern arm of Cameroon (8°–13°N, 14°–15°E) has a tropical climate with rainy summers (600–1000 mm mean annual precipitation) lasting 4–6 months. Mean annual temperature is 28°C; mean minimum and maximum temperatures of the coldest and hottest month are 14° and 34.5°C, respectively. Soils are typically ferric on sandy substrates, and vertisols on clay substrates. Geomorphology results from peneplanation of ancient deposits of crystalline or metamorphic rock.

All three of our study areas experience a unimodal rainfall pattern and a prolonged annual period of drought. But in Cameroon the rains are more effective than in central Chile or southern Tunisia because they fall during the hottest season when plants make most active growth. Moreover, no cold season occurs in Cameroon as in the other two areas. Thus, absolute growth rates are highest in Cameroon, but given the tropical rain patterns and local land-use practices, northern Cameroon soils are especially liable to severe and rapid degradation. Moreover, several sahelian, spiny acacias tend to invade disturbed soils in this area, their seeds being carried by cattle and other livestock vectors.

Our studies to date (Seiny-Boukar 1990; Seiny-Boukar et al. 1992) have focused on forest regeneration during fallow periods on vertisols, where most clearing and planting takes place. The original vegetation was a relatively dense arboreal savanna (Letouzey 1978), probably richer in species and life forms than the arboreal steppe of southern Tunisia or the matorral of central Chile. The dominant formation today, however, is a secondary thornshrub savanna, with woody plant cover generally less than 25%. The dominant species are 2–4-m-tall thorny acacias, especially *Acacia seyal*. This situation is maintained in a relatively stable

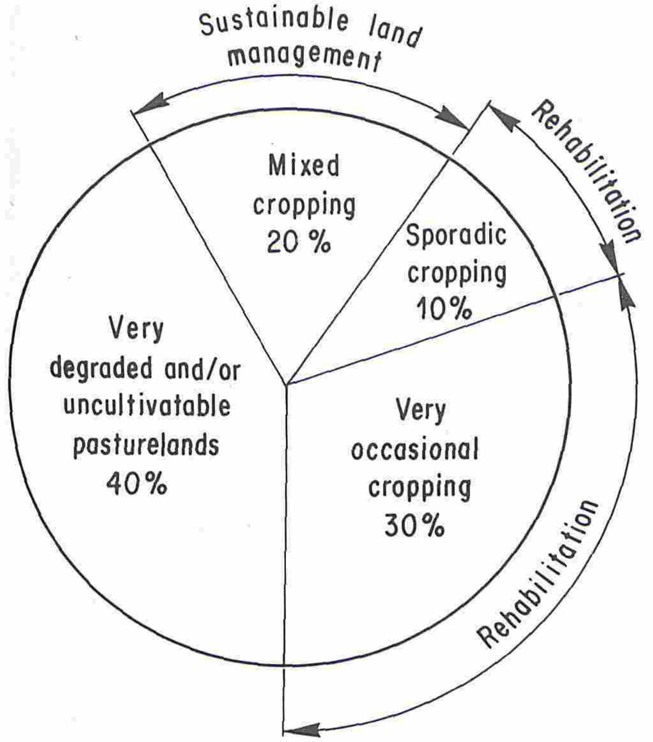



Figure 9. Land occupation in northern Cameroon study area, as of 1993, with indications of which portions seem most suitable for restoration, rehabilitation, and reallocation.

state by villagers dividing the landscape into *de facto* privately owned units corresponding to cultivated fields, new clearings, short- or long-term fallows (progressively colonized by acacias), and small fragments of arboreal savanna (0.1% of the area).

Figure 9 shows that approximately 20% of the study zone is occupied in what appears to be sustainable mixed field cropping. This occurs on privately owned plots lying mostly in the most humid, "run-on" sites where agriculture presents far less risk of failure than on higher ground subject to severe run-off after each rain. In contrast, 40% of the area consists of lands that have become so badly degraded that they are abandoned by farmers, and are subjected to only occasional grazing by herds of seminomadic pastoralists, or else by lands that have never been cultivated because the soil is too shallow or the slope too steep. At the present time, rehabilitation and reallocation are both virtually inconceivable on these collectively owned and ill-managed lands.

Of the remaining 40% of the study area, 30% is very degraded and 10% somewhat less so. It is in this critical latter portion that our rehabilitation efforts have been concentrated. It is here that an ongoing drama is now unfolding: in order to maintain legal rights on plots of land, farmers must continue to cultivate them, which is similar to the situation in southern Tunisia. In this region, however, a more-or-less important fallows period was traditionally observed between cropping cycles. This fallows is becoming increasingly shortened as a result of the demographic and socioeconomic pressures at play in the region, and ecosystem degradation is a clear result. On the other hand, local farmers are already catching on to the idea of rehabilitation as a way of improving crop yields, or of bringing under cultivation lands that have been abandoned in the recent past. Both these goals can be achieved through manual or mechanical interventions, as will be described below. Rehabilitation of croplands appeals to the peasant because the extra work implied will further reinforce his status as "proprietor" of the improved plots. By contrast, he is not interested in rehabilitation of the communal pasturelands because they belong to everyone.

Our study site is 30 km south of Maroua (10°30′N, 14°20′E). Mean annual precipitation (40-year average) is 781 mm, concentrated in the 4–6 month hot season (Olivry 1986). Average duration of fallow periods has decreased drastically in recent years, with the result that forest regeneration has been seriously impaired (Figure 10). In addition, escalating farm pressure on the land has led to accelerating degradation of vertisols. Following Seiny-Boukar (1990) and Seiny-Boukar et al. (1992), the sequential stages of soil degradation are as follows:

*Vertisol modal* (VM; typical or undegraded) is found in undisturbed or slightly disturbed savannas. They are characterized by the formation of numerous, wide (2–3 cm) surface cracks up to 80 cm deep, depending on the season. These cracks encourage water infiltration and

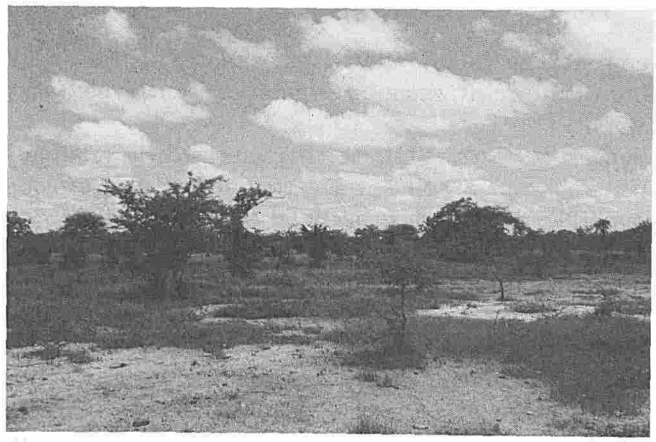


Figure 10. Degraded vertisols near Maroua, northern Cameroon. Extensive denuded area in foreground. In background dominant tree are *Sterculia setigera, Sclerocarya birrea,* and *Boswellia dalziellii* among the larger specimens, and *Acacia seyal* and *Lannea humilis* among the smaller ones. (Photo by Ch. Floret.)

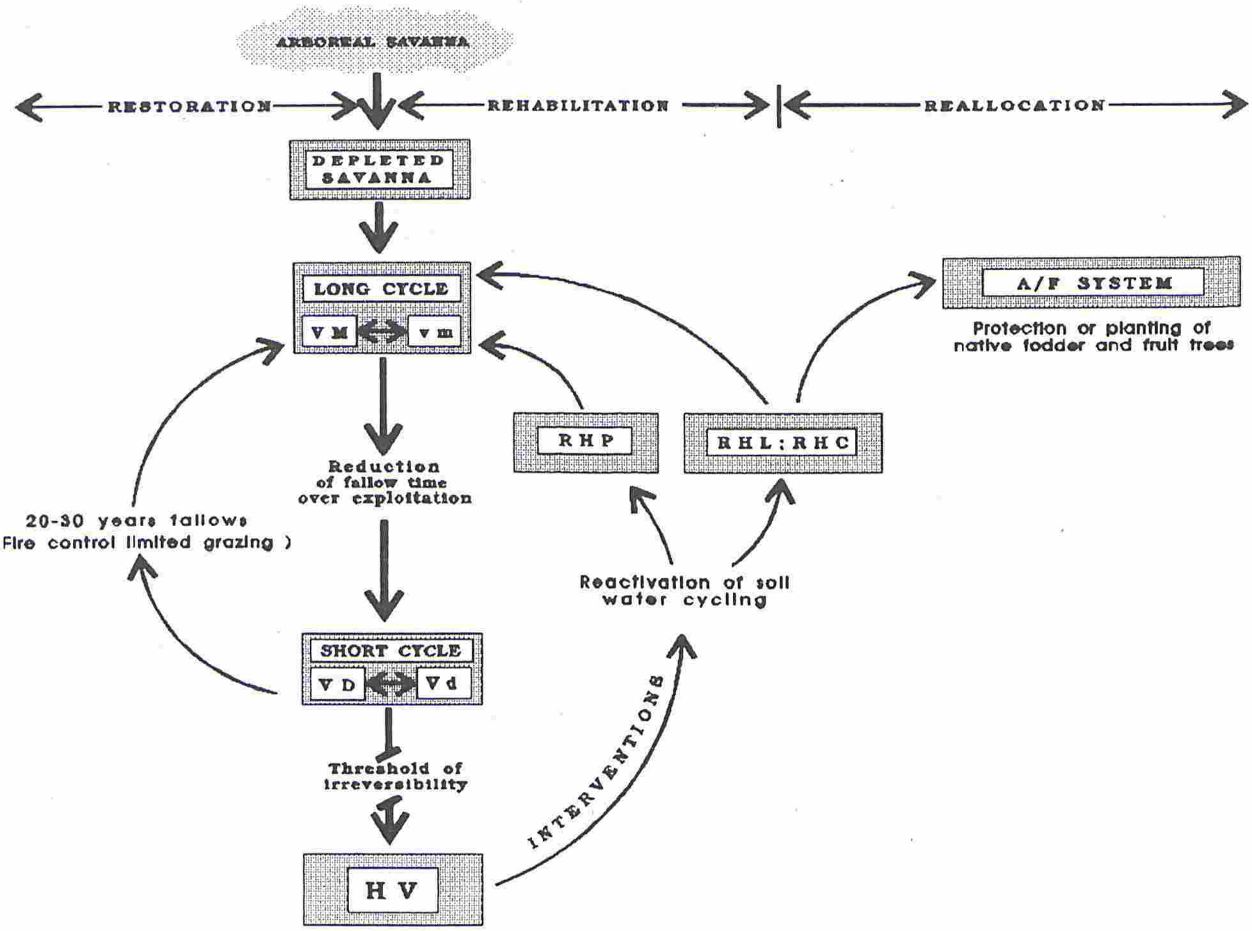



Figure 11. Application of the general model (Aronson et al. 1993*a*) to the Cameroonian case study, where growing land use for agriculture is assumed to be the dominant feature. VM = "modal" vertisol in good condition and uncultivated; vm = VM under periodic cultivation; VD = degraded, truncated vertisol that is uncultivated; vd = VD under periodic cultivation; HV = "hardé" (severely degraded and leached) vertisol; RHP = rehabilitated "hardé" by "pitting"; RHL = rehabilitated "hardé" by digging in of annual grasses; RHC = rehabilitated "hardé" by 50–200 $m^2$ water catchment basins.

deep storage (Masse 1992), particularly at the beginning of the rainy season. Organic-matter levels reach 1.65%.

*Degraded vertisols* (VD) are characterized by relatively rare cracks in a soil surface that forms an "erosion crust" 2–5 mm thick (Casenave & Valentin 1989), reducing water infiltration and inhibiting seed germination. This crust leads to increased surface erosion but does not affect lower soil layers.

*Hardé vertisol* (HV) is a local term adapted by soil scientists and ecologists working in this region to describe the last stage of vertisol degradation. The effects of degradation include a compacted profile and hard surface crust, without pores and without biological activity. There are virtually no surface cracks left, and up to 70% or 80% of precipitation can be lost in runoff as a result of the erosion crust forming at the surface. Lower soil layers are also considerably altered.

Figure 11 shows local degradation processes as well as the three mechanical rehabilitation techniques currently being tested for northern Cameroonian vertisols on which the traditional system of fallows is maintained. Reallocation by local people here takes the form of agricultural systems wherein fallows are no longer practiced at all. Finally, planting (or preservation in cultivated fields) of native nitrogen-fixing legume trees (such as *Faidherbia albida*) and/or fruit-bearing trees (such as *Tamarindus indica*), appears to be a promising pathway of reallocation toward a permaculture system, provided that fallow periods marked by burning and clearing are no longer employed.

At the intermediate stage of degradation (VD), degradation processes appear reversible through revised and improved management techniques that focus on maintaining adequate fallow time, during which fire control and reduced grazing and woodcutting allow

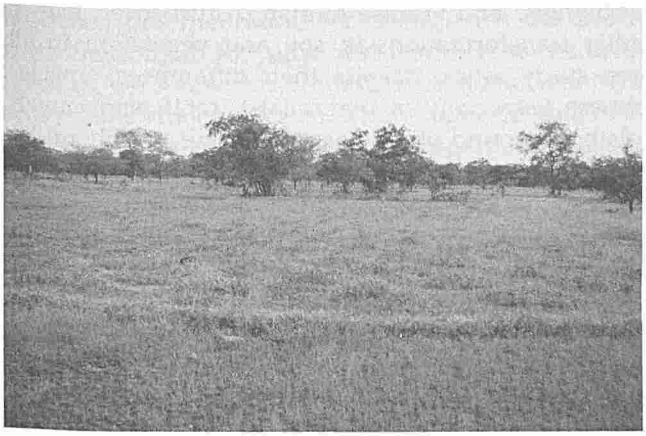


Figure 12. Same site as Figure 10 after mechanical soil emendation and RHC technique of 50–200 m² water catchment basins. Woody vegetation includes *Acacia seyal* and *Lannea humilis* within relatively continuous ground cover. (Photo by Ch. Floret).

more or less "passive" restoration to occur. By contrast, when the HV stage has been reached, recovery of native vegetation no longer occurs during fallows; intervention appears necessary to jumpstart a rehabilitation process.

For example, reactivation of the hydrological functioning of "hardé" vertisols may be triggered by mechanical interventions (Masse 1992). Three methods have now been tested thus far: "pitting," or the mechanical process of creating small pits in the soil surface to increase water penetration (RHP), plowing in of annual grasses (RHL), and creation of 50–200 m² water-catchment basins (RHC) that force water to infiltrate more or less evenly (Figure 12).

Table 3 presents vital ecosystem attributes at progressive stages of degradation and three years after the three soil reactivation techniques were introduced. As in the Tunisian and Chilean case studies, species diversity (in this case differentiated as herbaceous and perennial) as well as total cover and organic matter content decline drastically with degradation, from VM through VD (including vd), and still further at HV (including hv). At HV, as much as 40% loss of organic matter is recorded, as well as serious reductions in cation exchange capacity. Similarly, CEC, floristic diversity, and total plant cover decline.

After three soil treatments, distinct but variable improvements in several VEAs were obtained. In particular, OM and CEC recovered quickly after introduction of mechanical soil emendations to improve water infiltration. Also, surface cracks returned under RHC, but

**Table 3.** Vital ecosystem attributes of an ecosystem on vertisol near Maroua, northern Cameroon, in "modal" condition, two stages of degradation, and after three years experimental rehabilitation by three different techniques.

| | Stage of Development[1] | | | | | |
|---|---|---|---|---|---|---|
| *Vital Attribute* | *VM* | *VD,vd* | *HV,hv* | *RHP* | *RHL* | *RHC* |
| 1. Number of herbaceous species | 15–18 | 8–13 | 7–9 | 25–27 | 20–22 | n.a. |
| 2. Number of perennial species | 10–12 | 5–7 | 2–4 | 2–5 | 2–5 | 2–5 |
| 3. Total cover in summer (%) | 100 | 100 | 30 | 75 | 100 | 100 |
| 4. Organic matter content in upper soil layers (%) | 1.65 | 1.05 | 0.9 | 1.5 | 1.4 | 1.4 |
| 5. Cation exchange capacity (CEC) (mg 100g) | 35 | 25 | 17 | 20 | 25 | 20 |
| 6. Wide surface cracks | many | few | absent | absent | absent | few |
| 7. Available soil water reserves (WR) (mm)[2] | 40–45 | 30–35 | 20–25 | 25–30 | 25–30 | 25–30 |
| 8. Duration of available water for plants (months) | 6–7 | 2–3 | 1.5–2 | 1.5–2 | 2–3 | 5–6 |
| 9. Coefficient of rainfall infiltration (CRI) (%) | 75–85 | 50–60 | 30–40 | 40–50 | 60–75 | 100 |

[1] VM = "Modal" vertisol in good condition—uncultivated.
vm = VM under periodic cultivation.
VD = Degraded, truncated vertisol—uncultivated.
vd = VD under periodic cultivation.
HV = "Hardé" (severely degraded, leached) vertisol.
RHP = Rehabilitated "hardé" by "pitting."
RHL = Rehabilitated "hardé" by digging in of annual grasses.
RHC = Rehabilitated "hardé" by 50–200 m² water catchment basins.
[2] As the maximum WR is never obtained at this site, we present mean values for each 10-cm layer in the soft soil overlying bedrock (approximately 2 meters thick).

Sources: Masse 1992; Seiny-Boukar et al. 1992; Floret et al., unpublished data.

there was no difference in WR among the three treatments. By contrast, CRI and RS increased under RHC treatment, in contrast to the other two treatments. Under RHC, soil water storage was restored almost to the same levels as observed for VM. Unfortunately, these dramatic improvements are probably not sustainable in the long term unless organic matter content can also be significantly increased through a return to traditional management techniques (longer fallows and reduced intensity of fires, stocking rate, and woodcutting). A practice of digging green manure into fields following crop harvests would also help improve soils. Given current demographic trends, however, it seems doubtful that this will take place in the foreseeable future unless larger forces come into play.

## Discussion

The data presented reveal recurrent trends and suggest avenues for future research in ecosystems of arid and semi-arid lands. Trends in vital ecosystem attributes were measured at various stages of degradation and in the early phases of rehabilitation in three sites differing in environmental, historical, and contemporary human factors. Here we will first discuss the trends observed among the VEAs measured thus far, especially in terms of four of the ten hypotheses proposed by Aronson et al. (1993*a*). Then we will briefly discuss the socioeconomic, cultural, and geographic aspects of our three study regions, similar to those of many arid and semi-arid regions of the developing world today, that will inevitably influence the chances for success of any attempts at ecological rehabilitation or restoration.

*Hypothesis 1.* Beyond one or more "thresholds of irreversibility," ecosystem degradation is irreversible without structural interventions combined with revised management techniques.

In southern Tunisia, we have presented data from the portions of the study area with sandy soils that are more liable to physical than biotic modification. In this case study, "restoration" of a simplified version of a preexisting ecosystem's structure and biodiversity was pursued until the conclusion was reached that further progress was blocked by the depletion of local seed banks of perennial species and by the absence of nearby sources of propagules. This absence seems particularly critical for trees and shrubs whose biomass is exploited for energy, including the nitrogen-fixing *Acacia* trees that we assume were "keystone species" in the original "arboreal steppe."

*Hypothesis 2.* Water and nitrogen use efficiency and nutrient cycling times decrease with ecosystem degradation.

Prolonged and intense human disturbances lead to similar transformations in soil and vegetation in all three study areas, despite their differences. Species richness (especially of perennials), total plant cover, and aboveground phytomass all decline sharply under prolonged human use. Concurrently, soils undergo one or more of a series of transformations that impede proper functioning of the indigenous ecosystem of reference. Under these conditions, the depth and duration of soil water reserves available for plant growth diminish, and organic matter content in upper soil layers declines as well. It can also be presumed that nutrient cycling indices (Finn 1976), as well as the abundance, diversity and efficiency of soil biota, decline and ultimately reach a virtual state of sterility.

The data presented in Tables 1–3 directly support this second hypothesis, even though more detailed measurements are desirable. The Chilean project is one where we can easily pursue this question, since all three of the postmatorral stages included in Table 2 fall within a single degradation phase to the extent that *Acacia caven* continues to be the single dominant and keystone species. The next phase of degradation is marked by the complete disappearance of *Acacia caven*. Factors affecting the declining vigor of *A. caven* in the degradation process are not yet clear.

*Hypothesis 3.* The reintroduction of keystone species will accelerate rehabilitation by facilitating the reintroduction and establishment of additional native species.

Once a "threshold of irreversibility" has been crossed, the reintroduction of presumed keystone species may be the simplest method of "jumpstarting" a return to the higher phase. As mentioned above, many acacias, and other nitrogen-fixing legumes likely to be keystone species in arid and semi-arid ecosystems, are apparently pre-adapted to frequent, unpredictable disturbances. This feature makes them particularly useful for jumpstarting the rehabilitation process on degraded soils, even if some of them also pose the threat of escaping and becoming noxious weeds (Hughes & Styles 1987; Aronson et al. 1991).

There also remains the crucial question of how to proceed with proposed introductions so as to optimize the benefits and minimize the risks. It will be recalled that one of our VEAs for which we have not presented data concerns the presence or activity of keystone species (Aronson et al. 1993*a*). We reiterate here that certain species may act as keystone species in some phases of an ecosystem's trajectory but not in others. Moreover, as has often been pointed out, the resilience of ecosystems is not necessarily the kind that forces a trajectory following disturbance—secondary succession, restoration, or rehabilitation—that is the inverse of the degradation trajectory. Insofar as we are aware, however, this theoretical consideration is not backed with field data for any real-world ecosystem. At present, we know very little indeed about managing ecosystems' trajectories in what might be called the "upwards" direction, away from degradation and biotic impoverishment. We would postulate, however, that introducing a keystone species at the wrong stage of the rehabilitation process might in some cases be a mistake. The decision to reintroduce (or remove) certain species—especially keystone species, but others as well—must be evaluated from many angles. We recall at this point our use of the term "ecosystem of reference" to emphasize that a conscious decision must always be taken as to the historic (or presumed prehistoric or other) stage that is to serve as a model for the attempted restoration or rehabilitation of the ecosystem under investigation.

In the Tunisian case study, much of the extant shrub steppe (RS3, RS2) is dominated by *Rhanterium suaveolens*, a situation that appears to have arisen only after the formerly dominant acacias and numerous perennial grasses were eliminated. (This shrub, in turn, appears to require continual grazing pressure in order to maintain itself; where livestock are excluded, *R. suaveolens* eventually disappears. It is thus a keystone species for a limited range of conditions or trajectory phase only.) To achieve restoration or rehabilitation of the "arboreal steppe," it will be necessary not only to reintroduce at least some of the formerly critical grass and tree species but also, somehow, to effect a "switch" in nutrient cycling and other ecosystem functions such that these elements can regain their former roles as keystone species.

To illustrate a quite different situation, a spiny subshrub of low palatability, *Astragalus armatus* Willd. subsp. *armatus* (formerly *A. armatus* Willd. subsp. *tragacanthoides* [Desf.] Maire), is very abundant in the study area on badly degraded, overgrazed lands formerly dominated by *Rhanterium suaveolens* (RS1, rs1). This nitrogen-fixing perennial legume appears to act as a keystone species in the most advanced stage of degradation by colonizing denuded sites and presumably reactivating nutrient cycling in the soil. After several years of grazing exclusion, however, the *Astragalus* will often disappear by itself. It would be interesting to learn how efficiency of water and nitrogen use, soil surface conditions, CRI, RUE, and microsymbiont effectiveness, compare at the beginning and end of this keystone species' passage in the ecosystem.

Finally, in the Chilean situation, the non-native *Acacia caven* is without doubt a keystone species in the extant espinales, even though its presence in central Chile is probably relatively recent (Aronson 1992). The trouble is that in many sites we would like to alter the internal dynamics (or positive feedback cycles) of the espinales, but it is only when irrigated farming (reallocation) is introduced that the tree can be entirely removed on a cost-effective basis. We need to know much more about the actual role played by this and other keystone species at a given stage or stages of ecosystem development in order to know how best to manipulate them, physically, genetically, or otherwise. Certainly, candidate species need to be evaluated not only at the species level, but also at the levels of accession, provenance, or ecotype. Regardless of the plants or microorganisms under investigation, selections will be needed for local conditions, and mechanical or chemical interventions may well prove necessary for their establishment in early stages.

Moreover, as was mentioned regarding the Tunisian case study, we are developing complementary groups of species for simultaneous reintroduction. This approach appears promising because positive interactions do occur among co-occurring species, as well as competition for resources. For example, evidence exists that some (but not all) nitrogen-fixing legumes actually fix more atmospheric nitrogen, and grow larger in the presence of nonfixing herbs or grasses than in monocultures (see Patra et al. 1986). However, environmental factors such as light intensity, temperature regimes, and dryness also affect the fixation and transfer of nitrogen from fixers to nonfixers (Reynaud 1987; Ta & Faris 1988). Moreover, mycorrhizae and other soil-borne microorganisms apart from *Rhizobium* may play a larger role than previously suspected (see Haystead et al. 1988). Thus, much remains to be done before well-tailored species assemblages can be recommended for rehabilitation purposes.

*Hypothesis 4*. The rate of potential recovery by restoration or rehabilitation pathways is inversely related to the structural and functional complexity of the ecosystem of reference.

In the subhumid, mediterranean climate zone of central Chile, anthropic degradation has lasted for the relatively short period of 300–400 years. Nevertheless, the original complex, diverse matorral has been so badly degraded that its restoration appears impossible without massive interventions and expenditures. Moreover, even if it were possible, it is probably not worth considering in this area, given the ongoing rural presence of more than 300,000 people that depend upon imported European farming and animal husbandry techniques for their livelihood. By contrast, rehabilitation of the mixed espinales that existed 100–200 years ago would probably increase both primary and secondary productivity in the espinales and would pose no major conflicts with current land-use practises.

In Tunisia, severe ecosystem degradation has been underway at least ten times longer than in central Chile, and a shrub steppe has replaced the original arboreal steppe. Nevertheless, rehabilitation of at least part of the arboreal stratum consisting of *Acacia tortilis raddiana*, as well as most of the components of the other strata, appears feasible within a reasonable period, provided the technical problems that have often limited the value of nursery-grown acacias in the past can be resolved. This difference between Tunisia and central Chile is due to the combination of a more arid climate and sandy soils, and the fact that the ecosystem of reference appears to be simpler than it was in central Chile.

In northern Cameroon, human disturbances are presumed to be less longstanding and intense than those experienced by the first two study areas. Nitrogen-fixing and other native trees are still present, albeit in small numbers, and seed sources of most herbaceous as well as ligneous native plants are locally present. Few or no exotic invaders are present as yet. Except in special nature reserves, however, full restoration of the ecosystem of reference seems no more feasible in northern Cameroon than in central Chile due to the overwhelming presence of people and their traditional farming systems, not to mention the complexity of the original arboreal savanna. Thus, an agro-ecosystem intermediate in complexity and diversity between original and actual conditions seems the most realistic goal for northern Cameroon. For this to be achieved, disturbed vertisols must be reworked so as to force water to penetrate beyond the surface. This will take time because the soils are clay-like and have lost their cracks through the disruption of the former drying-wetting cycles.

Also, successful rehabilitation and reallocation in northern Cameroon depend not only on revised management of soils, but also on finding a substitute for the long fallows cycle (20–30 years) of earlier times. Since long fallows are no longer practical in this increasingly populous area, some other management technique will be needed to accelerate growth of natural vegetation during the short fallows currently practiced (5–10 years) and to maintain ecosystem resilience. One promising technique under investigation in northern Cameroon is to plant fast-growing and preferably nitrogen-fixing trees in the cultivated fields one or two years *before* their abandonment and the transition to the fallow period (Peltier 1991). These young trees are protected alongside the various crop plants and can be several meters tall at the time of field abandonment. This technique leads to the establishment of an "improved fallows" of short duration (about 5 years) but of high efficiency in restoring soil fertility.

The three techniques of soil reactivation tested in northern Cameroon appear to be promising complements to biotic interventions such as species reintroductions. They should be tried in central Chile and on the heavier soils in southern Tunisia, where processes other than those described for Cameroon have deteriorated soil structure and fertility to a high degree.

In discussing the fourth hypothesis, we briefly evoked the comparative structural and functional complexity of the three ecosystems of reference. It is as much the human determinants of degradation, however, as the nonhuman determinants of an ecosystem's structure and functioning that will determine the rate of potential recovery by alternative restoration or rehabilitation pathways at any given site.

In southern Tunisia, central Chile, and northern Cameroon, it could well be argued that if rehabilitation of a given area is to succeed, there must be a concurrent or corresponding increase in productivity of other areas, at least temporarily, so as to reduce human pressures on the study sites. Ideally, the inevitable reallocation to cereal culture or other forms of agriculture should be confined to lands best suited for this purpose, while other landscape units would be rehabilitated as quickly and as completely as possible. This would help advance the larger goals of landscape reintegration as well.

In order to convince local people to cooperate with such a program, they must be satisfied that it is in their own economic as well as environmental best interest. Given the uncompromising individualism of the average Chilean farmer and land owner, there is little sense in talking about community-wide or landscape-or watershed-level rehabilitation efforts there at the present time, unless the government (or the World Bank!) is prepared to foot most or all of the bills. Instead, all interventions must be conceived and undertaken at the level of the single farm. By contrast, both in southern Tunisia and in northern Cameroon, a long tradition of collectively owned lands allows at least the consideration of village or watershed-level projects that would theoretically benefit everyone. Even if privatization is taking place in both regions today, this tradition could theoretically be exploited in the interests of ecosystem rehabilitation and landscape reintegration.

To illustrate, consider a nearly universal activity of tremendous consequences throughout the arid and semi-arid regions of the "South," the unrestrained harvesting of standing woody biomass for fuelwood and charcoal. As has often been demonstrated, fuelwood plantations in highly localized sites near towns and villages can reduce pressure on trees and shrubs in pasturelands, fallows, and other sensitive areas. But the adoption of such practices in our three study areas does not seem likely to take place in the near future. In southern Tunisia, it must be said, the government has undertaken to improve the distribution of butane gas in small containers to all settlements. This alternative energy source should indeed reduce pressure on the standing woody biomass, but it does not contribute to a new way of thinking about natural-resource management. In northern Cameroon, as mentioned, farmers are enthusiastic about rehabilitating fields so as to improve crop yields. Yet community plantations of fuelwood trees are as yet of little interest to villagers because in their minds, there is still enough firewood and charcoal to be had from the standing woody biomass. In other climatically similar areas, however, such as around Bamako, such plantations are beginning to be accepted because there is virtually no "free" fuelwood to be had in the surrounding area.

Finally, in Chile, there do seem to be good prospects that a certain number of individual dryland farmers might soon start planting fast-growing trees on the most favorable of the nonarable sites of their lands due to the simple fact that a cubic meter of firewood now sells for the same price as a lamb in the cities of central Chile. Whereas most dryland farmers are currently obtaining no more than one marketable lamb per hectare per year, there is ample evidence that they could be achieving firewood yields well above 1 $m^3$ per ha per year on *parts* of the same land, at little additional start-up cost. (There even exist substantial government subsidies to support tree planting in central and northern Chile.)

In any case, it seems evident that if rehabilitation of degraded ecosystems in areas such as we have described here are ever to achieve lasting success, they must be accompanied by technical and economic reforms as well, such as improved animal husbandry techniques, better multipurpose use of landscape units well suited to reallocation (such as fodder shrubs grown in oases in southern Tunisia or firewood trees grown on steep slopes and gullies in central Chile), and better organization of commercialization and marketing methods for all products produced. For peasants and villagers to cooperate, in all three areas, it also appears necessary that private land tenure be carefully respected where it already exists, and nurtured where it is as yet ill-defined. This will help in two ways, by promoting acceptance of the idea of rehabilitation and by obtaining greater site-specific productivity—in terms of biomass, but also in terms of cash—in the most fertile and cultivatable portions of a given landscape segment. Without this increase in productivity on a landscape and microregional basis, there is little hope for restoration and rehabilitation of degraded ecosystems in the arid and semi-arid lands of the "South," be it on farmlands, in pasturelands, or even in national parks.

As shown, some improvements of agricultural productivity in central Chilean espinales can be achieved through revision of existing resource management practices. With revised herd management, a significant increase of sheep stocking rate was found possible without major investment. With the incorporation of selected ecotypes of the annual medic *Medicago polymorpha*, this secondary productivity should rise still further. This and other innovations are necessary and, as we have seen, technically feasible, despite the advanced state of soil degradation.

Results of soil emendation trials in southern Tunisia and northern Cameroon also support the argument that traditional systems of resource management in those regions can be made more productive through rehabilitation techniques, and that this represents a viable economic alternative for farmers and land owners. We also suggest that the relatively small changes these rehabilitation techniques can effect, unspectacular though they may be in the eyes of journalists or politicians, are far more likely to achieve the elusive goals of sustainable rural development and conservation of biodiversity than are the corps-of-engineers-style interventions that have predominated in the recent past in Africa, South America, and elsewhere.

Finally, in northern Cameroon, people are not at all likely to abandon their long tradition of intensive crop and fallows systems. This must be taken as a given. Thus, some way must be found, within the constructs of existing land-use systems, to compensate for the growing pressure on all arable lands that leads to drastic reduction of fallows time. In other words, it will be necessary to increase productivity in certain topographic and edaphic sites without sacrificing individual ecosystem stability or overall landscape integrity. In all three of our study areas, at present, a trend toward greater concentration of arable land in fewer and fewer hands may lead to a greater demand for ecological restoration and rehabilitation, because it is normally only the medium- to large-scale landowner that can afford the short-term risks that such new approaches to old problems entail.

## Conclusion

This contribution is based on the oft-repeated but seldom tested proposition that the best way to insure the conservation of well-functioning ecosystems, integrated landscapes, and biodiversity—at least in developing countries—is to develop their sustainable economic usefulness to local people. More specifically, we believe that the best way to insure sustainable agriculture and forestry in arid and semi-arid lands is to pursue rehabilitation based on the known or presumed structure and functioning of indigenous ecosystems, and to confine reallocation to very specific areas or, in some cases, to artificially desertified ecosystems that no longer serve any purpose to anyone. Such a policy, which at a higher level is aimed at the "reintegration of fragmented landscapes" (Hobbs & Saunders 1991) at local, regional, and national levels, seems the only way to reconcile long-term productivity, biodiversity conservation, and progressively improving ecosystem "health" in degraded ecosystems and landscapes in arid and semi-arid regions of the South.

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## LITERATURE CITED

Allen, M. F. 1989. Mycorrhizae and rehabilitation of disturbed arid soils: Processes and practices. Arid Soil Research & Rehabilitation **3**:229–241.

Aronson, J. 1992. Evolutionary biology of *Acacia caven*. I. Infraspecific variation in fruits and seeds. Annals Missouri Botanical Garden **79**:958–968.

Aronson, J., C. Ovalle, and J. Avendaño. 1991. Early growth rate and nitrogen fixation potential in 44 multipurpose legumes grown in an acid and a neutral soil in central Chile. Forest Ecology & Management **47**:225–244.

Aronson, J., C. Floret, E. Le Floc'h, C. Ovalle, and R. Pontanier. 1993*a*. Restoration and rehabilitation of degraded ecosystems in arid and semiarid lands. I. A view from the South. Restoration Ecology **1**:8–17.

Aronson, J., C. Ovalle, J. Avendaño, A. del Pozo, and A. Lavin. 1993*b*. Rehabilitation of degraded ecosystems and outmoded farming systems in central Chile. II. Tagasaste, multipurpose trees and a vision of the future. Agriculture, Ecosystems & Environment.

Avendaño, J., C. Ovalle, and A. del Pozo. 1990. Pastoreo de Hualputras con ovinos. Pages 302–307. in Informe Técnico 1989–1990. Area de Producción Animal, Estación Experimental Quilamapu, Chillán, Chile.

Casenave, A., and C. Valentin. 1989. Les états de surface de la zone Sahélienne. Influence sur l'infiltration. Editions de l'ORSTOM, Paris, France.

Chaïeb, M., C. Floret, and R. Pontanier. 1991. Réhabilitation d'écosystèmes pastoraux de la zone aride tunisienne par réintroduction d'espèces locales, vol. I. Pages 259–261 in Proceedings of the IVth International Rangelands Congress. Centre de Cooperation Internationale en Recherches Agronomique pour le Developpement, Montpellier, France.

DSA/CIRAD 1985. (Département des Systèmes Agraires/Centre de Cooperation Internationale en Recherches Agronomique pour le Developpement). Eléments de diagnostic sur l'Agriculture du Sud Tunisien. Centre de Cooperation Internationale en Recherches Agronomique pour le Developpement, Montpellier, France.

del Pozo, A., C. Ovalle, and J. Avendaño. 1989. Los medicagos anuales en Chile. I. Comparición con Australia. Agricultura Técnica (Chile) **49**:260–267.

Domic, L. 1979. Geodemografía. Perspectivas de desarrollo de los recursos de la VII Region. Intendencia de la Region del Maule/IREN-CORFU, Santiago, Chile.

Escadafal, R. 1981. Une méthode nouvelle de description de la surface du sol dans les régions arides. Sols **5**:21–27.

Etienne, M., C. Caviedes, C. Gonzalez, and C. Prado. 1982. Cartográfia de la vegetación de la zona arida de Chile. Terra Arida (Chile) **1**:1–78.

Finn, J. T. 1976. Measures of ecosystem structure and function derived from analysis of flows. Journal of Theoretical Biology **56**:362–380.

Fischer, R. A., and N. C. Turner. 1978. Plant productivity in the arid and semiarid zones. Annual Review of Plant Physiology **29**:277–317.

Floret, C. 1981. The effects of protection on steppic vegetation in the Mediterranean arid zone of Southern Tunisia. Vegetatio **46**:117–119.

Floret, C., E. Le Floc'h, R. Pontanier, and F. Romane. 1978. Modèle écologique régional en vue de la planification et de l'amenagement agro-pastoral des régions arides. Application à la région de Zougrata. Document Téchnique No. 2. Ministère de l'Agriculture, Tunis, Tunisia.

Floret, C., E. Le Floc'h, F. Romane, and R. Pontanier. 1981. Dynamique des systèmes écologiques de la zone aride. Acta Oecologica **2**:195–214.

Floret, C., R. Pontanier, and S. Rambal. 1982. Measurements and modelling of primary production and water use for a south Tunisian steppe. Journal of Arid Environments **5**:77–90.

Floret, C., E. Le Floc'h, and R. Pontanier. 1983. Phytomasse et production en Tunisie présaharienne. Acta Oecologica **4**:133–152.

Floret, C., E. Le Floc'h, and R. Pontanier. 1990. Principles of zone identification and of interventions to stabilize sands in arid mediterranean regions. Arid Soil Research & Rehabilitation **4**:33–41.

Fuentes, E. R., and E. R. Hajek, 1979. Interacciones hombre-clima en la desertificación del norte chico Chileno. Ciencia y Investigaciones Agrarias **5**:137–142.

Gay, C. 1865. Flora Chilena. Historia fisica y politica de Chile. Toma I. Botanica. Reissued in 1973 by ICIRA, Santiago, Chile.

Grouzis, M. 1988. Structure, productivité et dynamique des systèmes écologiques sahéliens (Mare d'Oursi, Burkina Faso). Université de Paris-Sud. Etudes et Thèses. ORSTOM, Paris, France.

Haystead, A., N. Malajczuk and T. S. Grove. 1988. Underground transfer of nitrogen between pasture plants infected with vesicular-arbuscular mycorrhizal fungi. New Phytologist **108**:417–423.

Hobbs, R. J., and D. A. Saunders, 1991. Re-integrating fragmented landscapes—a preliminary framework for the Western Australian wheatbelt. Journal of Environmental Management **33**:161–167.

Hughes, C. E., and B. T. Styles. 1987. The benefits and potential risks of woody legume introductions. International Tree Crops Journal **4**:209–248.

IREN (Instituto Nacional de Investigaciones de Recursos Naturales), Suelos. 1964. Descripción proyecto aerofotogramétrico. Publication No. 2. OEA/BID, Santiago, Chile.

Jáksic, F. M., and E. R. Fuentes. 1989. Ecology of a successful invader: The European rabbit in central Chile. Pages 273–284 in R. H. Groves and F. di Castri, editors. Biogeography of Mediterranean invasions. Cambridge University Press, Cambridge, England.

Le Houérou, H. N. 1959. Recherches écologiques et floristiques sur la végétation de la Tunisie méridionale. Mémoire hors série. Institut de Recherches Saheriennes, Algiers, Algeria.

Le Houérou, H. N. 1968. La désertisation du Sahara Septentrional et des Steppes Limitrophes (Libye, Tunisie, Algerie). Annales Algériennes de Géographie **6**:2–27.

Le Houérou, H. N. 1969. La végétation de la Tunisie steppique (avec références au Maroc, à l'Algérie et à la Libye). Annales Institut National de la Recherche Agronomique de Tunisie. **42**:1–629.

Le Houérou, H. N. 1984. Rain use efficiency: A unifying concept in arid-land ecology. Journal of Arid Environments **7**:1–35.

Le Houérou, H. N. 1993. Land degradation in Mediterranean Europe: Can agroforestry be a part of the solution? Agroforestry Systems **21**:43–61.

Letouzey, R. 1978. Etudes phytogéographiques due Nord Cameroun. Editions Le Chevalier, Paris, France.

Long, G. 1954. Contribution à l'étude de la végétation de la Tunisie Centrale. Annales Service du botanique et agronomique de Tunisie **27**:1–228.

Masse, D. 1992. Amélioration due régime hydrique de sols dégradés en vue de leur réhabilitation: Cas des vertisols du Nord-Cameroun. Ph.D. dissertation. Institut National Polytechnique, Toulouse, France.

Ministère de l'Agriculture. 1985. Strategies nationales de la lutte contre la désertification. Ministère de l'Agriculture, Tunis, Tunisia.

Neffati, M. N. Akrimi, C. Floret, and E. Le Floc'h. 1991. Stratégies germinatives de quelques espèces pastorales de la zone aride tunisienne. Conséquences pour le semis des parcours, vol. I. Pages 281–284 in Proceedings of the IVth International Rangelands Congress. Centre de Cooperation Internationale en Recherches Agronomique pour le Developpement, Montpellier, France.

Odum, E. P. 1969. The strategy of ecosystem development. Science **164**:262–270.

Olivry, J. C. 1986. Fleuves et rivières du Cameroun. Collection Monographie Hydrologique ORSTOM Paris, France.

Ovalle, C. 1986. Etude du système ecologique sylvo-pastoral à *Acacia caven* (Mol.) Mol.: Applications à la gestion des ressources rénouvables dans l'aire climatique méditerranéenne du Chili. Dissertation. Université des Sciences et Techniques du Languedoc, Montpellier, France.

Ovalle, C., and J. Avendaño. 1987. Interactions de la strate ligneuse avec la strate herbacée dans les formations d'*Acacia caven* (Mol.) Mol. au Chili. I. Infuence de l'arbre sur la composition floristique, la production et la phénologie de la strate herbacée. Oecologia Plantarum **8**:385–404.

Ovalle, C., J. Aronson, A. del Pozo, and J. Avendaño. 1990. The espinal: Agroforestry systems of the mediterranean-type climate region of Chile. Agroforestry Systems **10**:213–239.

Ovalle, C., J. Aronson, J. Avendaño, A. del Pozo, and A. Lavin. 1993. Rehabilitation of degraded ecosystems and outmoded farming systems in central Chile. I. Historical overview and early results with an annual medic, *Medicago polymorpha*. Agriculture, Ecosystems & Environment.

Patra, D. D., M. S. Sachdev, and B. V. Subbiah, 1986. $^{15}$N studies on the transfer of legume-fixed nitrogen to associated cereals in intercropping systems. Biology & Fertility of Soils **2**:165–171.

Peltier, R. 1991. Les jachères à composante ligneuse. Carecterisation, conditions de productivité, gestion en vue d'une agriculture durable. Pages 67–87 In Ch. Floret and G. Serpan, editors. La jachère en Afrique de l'Ouest. Collection Colloques et Séminaires, ORSTOM, Paris, France.

Puckridge, D. W., and R. J. French 1983. The annual legume pasture in cereal-ley farming systems of southern Australia: A review. Agriculture, Ecosystems & Environment **9**:229–267.

Reynaud, P. A. 1987. Ecology of nitrogen-fixing cyanobacteria in dry tropical habitats of West Africa: A multivariate analysis. Plant and Soil **98**:203–220.

Seiny-Boukar, L. 1990. Régime hydrique et erodibilité des sols au nord du Cameroun. Proposition d'aménagement. Dissertation. University of Yaoundé, Yaoundé Cameroon.

Seiny-Boukar, L., C. Floret, and R. Pontanier. 1992. Degradation of savanna soils and the reduction of water available for the vegetation: The case of northern Cameroon vertisols. Canadian Journal of Soil Science **72**:481–488.

Ta, T. C., and M. A. Faris. 1988. Effects of environmental conditions on the fixation and transfer of nitrogen from alfalfa to associated timothy. Plant and Soil **107**:25–30.

Telahigue, T., C. Floret, and E. Le Floc'h. 1987. Succession postculturale en zone aride de Tunisie. Acta Oecologica **8**:45–58.

van Elsas, J. D., and C. E. Heijnen, 1990. Methods for the introduction of bacteria into soil: A review. Biology & Fertility of Soils **10**:127–133.