# The Effects of Drought on Plant Communities in the Desert Rangelands of Tunisia

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**Abstract**

Plant communities of the Dahar plateau are characteristic of the Tunisian desert both in structure and in dynamics. Although a number of plant communities can be differentiated, four major vegetation types are often distinguished that differ in plant species cover and composition, as well as other factors, such as soil types and elevation. Among the abiotic factors which affect the vegetation structure and diversity of plants, climate is probably the most relevant. The desert vegetation in southern Tunisia is in a state of change and the most debilitating risk is that of drought in these desert areas. Under protection from grazing the dynamic nature of this vegetation is affected by such conditions such as drought. Precipitation probably explains part of the vegetation response to drought. The effects of drought stress on vegetation were tested for four plant communities who are *Stipagrostis pungens*, *Anthyllis sericea*, *Helianthemum kahiricum,* and *Hammada schmittiana*. Patterns of plant response to drought differed among the four vegetation types considered. Vegetation cover, species richness, and diversity were used for the characterization of the considered vegetation. Main results show that plant cover, richness, and diversity change with vegetation type and rainfall variations. Vegetation cover on *H. kahiricum* steppe is more affected by drought than the other steppes. Plant diversity is affected by drought in all plant community and mainly on the *A. sericea* and *H. kahiricum* steppes.

**Keywords**

Drought stress • Plant communities • Management • Desert rangeland

## Introduction

Desert lands throughout the world are often degraded or increasingly at risk of degradation. These lands, including those at the border of arid

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regions, commonly exhibit accelerated soil erosion, losses of productivity, and impaired economic potential to support human populations. Desertification involves human and environmental drivers of change but is a regional symptom that emerges from degradation at finer spatial scales (Reynolds and Stafford 2002). Desertification does not describe cyclic phenomena, as when decadal variations of precipitation lead to periods of drought and to losses of vegetation cover that are fully restored when rains return (Tucker et al. 1994). Natural disturbances and often previously unappreciated human disturbances became a serious challenge to traditional succession beginning as early as the 1940s (e.g., Stearns 1949; Raup 1957), but really solidifying in 1970s. The increasing recognition of the pervasiveness of disturbances (White 1979) led to idea that ecological systems consisted of patches of different times since the last disturbance.

Dynamic changes in vegetation level are controlled by the balance between climatic and anthropogenic factor. Human activity has greatly accelerated the rate at which species have disappeared from the earth (May et al. 1995). The anthropogenic origin of phenomenon was underlined by many studies (Auclair et al. 1999): demographic pressure, overgrazing, and clearing of the ligneous family.

The primary causes of vegetation destruction on arid land and indirect causes of desertification are the clearing of land unsuitable for cultivation, prolonged overstocking and overgrazing, and the destruction of woody species by excessive firewood (Le Houérou 2009). In addition to the anthropogenic factors, climate influences a variety of ecological processes (Stenseth et al. 2002). These effects operate through local weather parameters such as temperature, wind, rain, snow, and ocean currents and the interactions occurring among them. There have been several recent studies on the impact of large-scale climatic force on ecological systems.

The relationships between climate, soil, vegetation, and other components of arid ecosystems have been described by many authors, including Shreve (1942), Hassib (1951), Tadros (1953), Vernet (1955), Kassas (1955), Chapman (1960), Zohary (1962), Batanouny (1973), Younes et al. (1983), Ayyad and El-Ghareeb (1984), Evenary et al. (1985), Zahran and Willis (2008), and Le Houérou (2009).

The Mediterranean bioclimates are characterized by winter rains and summer drought (Le Houérou 2005a, b). However, droughts can occur anywhere, and this simple statement tells us nothing about what constitutes a drought (Allaby 2003). In parts of North Africa, a drought occurs when no rain has fallen for at least 2 years. Perhaps, then, a drought is a period during which rainfall is insufficient to meet the needs of plants. Droughts meeting this definition have led to ongoing debates about desertification (Mainguet 1995; Thomas 1997).

Millions of people in rangelands depend directly on livestock for their livelihoods, but the management of these regions remains mired in controversy (Gillson and Hoffman 2007). The sustainable management of natural resources demands controls negating land degradation and desertification (Ganry and Campbell 1993). Although restoration models and practices have previously been applied to ecosystems, there is now a more recent focus on the "landscape perspective" of ecosystem restoration to improve nature conservation and management effectiveness (Moreira et al. 2006).

Among rare rehabilitation experiments, the one carried out by Le Floc'h et al. in 1999 in southern Tunisia enabled the reconstitution of a badly degraded steppe. In addition to this technique being generally beneficial to vegetation cover and species diversity (Gamoun et al. 2010a), it was found here that although sandy soil is more productive than limestone soil, the latter is more resistant to animal trampling (Gamoun et al. 2010b).

This chapter describes the effects of drought on the vegetation of the major plant community types of the desert rangelands in Tunisia from 2007 to 2009, with emphasis on cover, species richness, and diversity.

## Study Area and Data Collection

Studies reported here were carried over 3 years on four protected rangelands, which are located in southern Tunisia, forming a collective steppe unit of the plate of Dahar to the south of Tataouine. The climate is characterized by hot, dry summers and cool, mild winters, so according to Emberger (1954), it has a Saharan superior Mediterranean bioclimate.

The studies reported here can be divided into four different types of plant communities designated since 2007. Soil type influences brush growth in the distribution and determines the type of plant community. These soils were grouped into sand accumulation, limestone, loam, or sandy soil. Consequently, soil type is the key determinant in defining plant community. These plant communities are distributed as follows:

– *Stipagrostis pungens* [(Desf.) de Winter] community on sand accumulation
– *Anthyllis sericea* [Lag. subsp. *henoniana* (Coss.) Maire] community on limestone soil
– *Helianthemum kahiricum* [(Desf.)] community on loamy soil
– *Hammada schmittiana* [(Pomel) Ilji] community on sandy soil

These plant communities are exploited by individual, and although overgrazing can occur, they are largely composed of all plants that have resisted or benefitted from grazing and drought (Gamoun 2005). On these rangelands each individual had an incentive to increase the number of animals, and no individual was entitled to prevent access to others (Hardin 1968).

Since each plant communities, three 20-m-long transects, were set up to measure Total Plant Cover using the quadrats point method (Jauffret and Visser 2003). Observations were made every 20 cm, providing a total of 100 points in each transect. The total plant cover is determined by the formula TPC = (n/N) × 100, with n: number of points where the vegetation is present and N: numbers of total points in each transect (100 points in our case).

Shannon information index (H′; Shannon and Weaver 1948) is the most widely used measure of diversity in plant communities (Stirling and Wilsey 2001; Pélissier et al. 2003). Shannon-Wiener diversity index combines richness and relative abundance of plant species. It is calculated from centesimal frequencies of species (*fi*): $H' = -\Sigma\ ((fi/N) * \log2\ (fi/N))$, where *fi* is the number of *i*th species in the samples and N is the total number of species in the plant community.

A two-way ANOVA was used to test differences in the effects of vegetation types and years. The obtained data is subjected to several statistical analyses using SPSS for windows software v. 11.5 (SPSS Inc. 2002).

## Results and Discussion

Vegetation cover, diversity, and species richness were assessed in March each year (2007, 2008 and 2009). The following classes of data were analyzed for each vegetation type: (1) cover, (2) species richness, and (3) diversity.

### Weather Conditions

Rainfalls during the period from 1987 to 2009 averaged 75 mm/year (Fig. 1). The temporal rainfall distribution in Tataouine is eminently variable, with annual rainfall irregularly being either side of the norm. Various anomalies include considerable reduction in rainfall in 1987, 1998, 1998, 1998, 1999, 2001, and 2009 and contrasting periods of excessive humidity (1990, 1994, 1995, and 2002).

From 2003, there was a marked tendency in annual rainfall to constantly remain lower than the average precipitation established over one long period.

These rainy events in southern Tunisia are characterized by their great variability and very irregular distribution. During the past 20 years, several droughts have affected the south and caused significant losses, mainly in agricultural sectors, and these have endangered farms specializing in livestock. The research community perceives the apparent desertification as a transient response to reduced rainfall (Tucker et al. 1991, 1994). A number of studies have

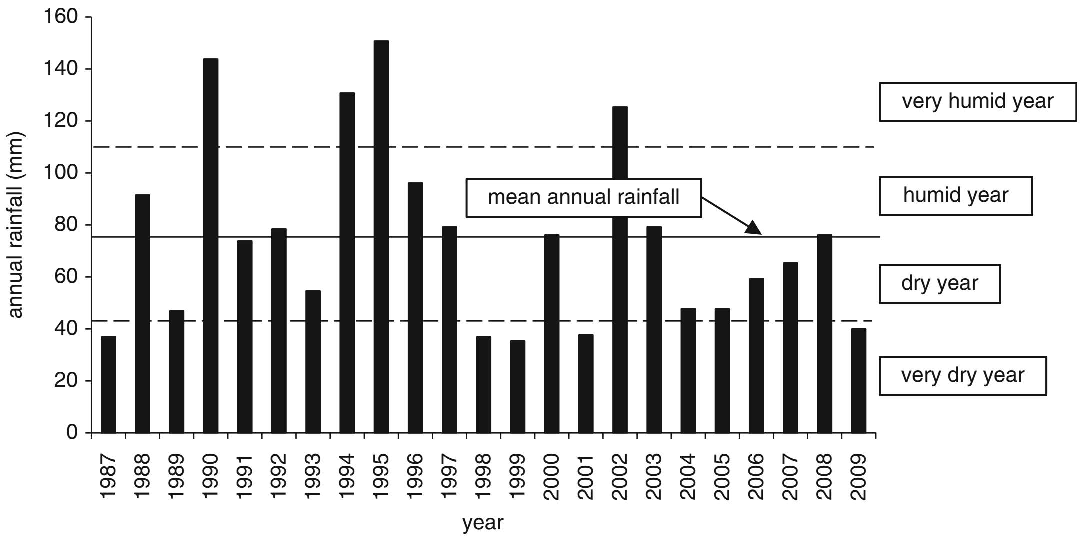
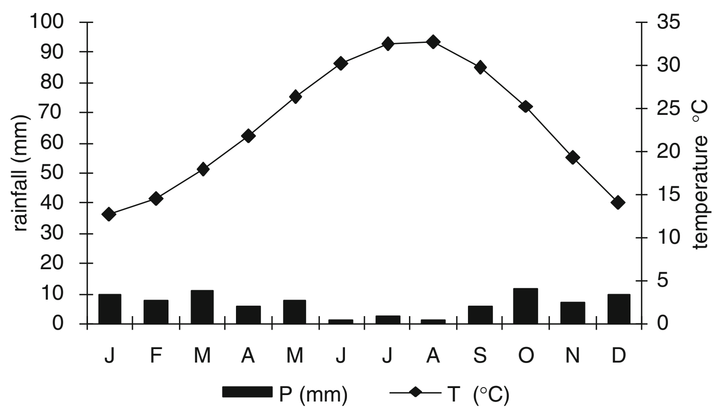



**Fig. 1** Average annual rainfall in Remada region between 1987 and 2009

**Fig. 2** Climatogram plans showing temperature and precipitation in Remada




shown that degradation to this land is reversible: with vegetation restored when rain returns.

Temperatures exceeded 30 °C for 60 days, and the maximum exceeded 40 °C for 10 days. The average annual heat-under-shelter varied between 22 °C and 23 °C, while the winters were cold registering between 11 °C and 12 °C in January.

The rains are spread over part of the year, with a relative variability of rainfall above 95 % and a Martonne index of aridity of approximately 2–3. The most important rains generally fell in winter, spring, and autumn, with the heaviest at 11.57 mm in October. The minimum rainfall of 0 mm has occurred in summer, in July (Fig. 2).

During summer, temperatures are pleasant with 32 °C in July and August and 30 °C in June. Maximum temperatures are expected to increase with the growing season which is segmented into rain events separated by drought periods. If two successive years are dry, the probability of a third dry year in the south ranges from 14 % to 17 % (Benzarti and Habaieb 2001).

## Trend in Vegetation Cover

Based on the ANOVA analysis, results showed significant vegetation type effect ($F = 9.14$; $p < 0.001$) and year effect ($F = 9.362$; $p = 0.001$) for vegetation cover. The vegetation type and year interaction was also significant ($F = 2.713$; $p = 0.037$). This interaction

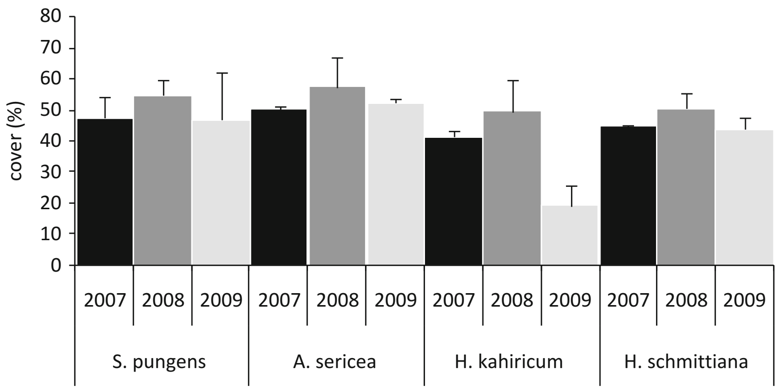



**Fig. 3** Mean of vegetation cover of four plant community types in southern Tunisia. Cover is significantly different among vegetation type and year ($p < 0.05$). *Vertical bars* represent standard errors

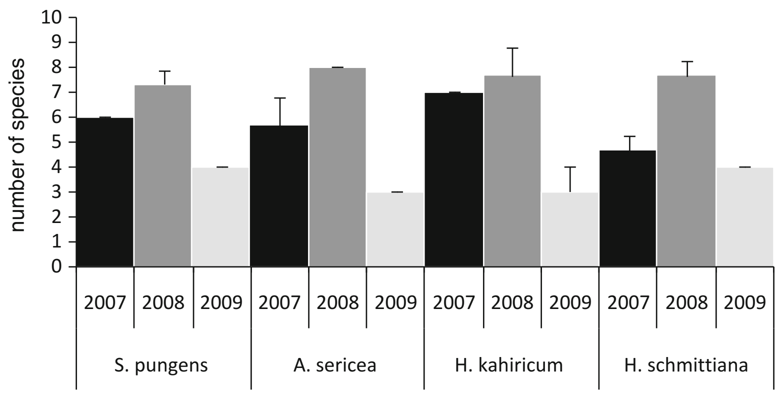



**Fig. 4** Mean of plant species richness of four plant community types in southern Tunisia according to time after protection (2006). The effect of year is highly significant ($p < 0.05$), but the effect of vegetation type is not significant ($p > 0.05$). *Vertical bars* represent standard errors

suggests that available water content depends on the soil type.

Figure 3 gives the participation in vegetation composition of the four plant community types from year 2007 to year 2009 after protection. The vegetation covers measured on the *S. pungens* and *A. sericea* steppes are the highest followed by the *H. schmittiana* and the *H. kahiricum* steppes. The impact of drought on plant cover is important and varied with plant community type. For example, on the *H. kahiricum* steppe, the vegetation covers increase from 42 % to 50 % in wet year (2008) and then dropped to 19 % in dry year (2009), whereas on the other plant community types, the decrease does not exceed 9 %.

## Trend in Species Richness

Community composition is further modified by the occurrence of rainfall through time and vegetation type. In this chapter, ANOVA results showed significant year effects ($F = 134.571$; $p < 0.001$) and no significant vegetation type effects ($F = 0.952$; $p = 0.431$), but showed significant interaction between the two factors tested year*vegetation type ($F = 4.667$; $p = 0.003$). During the first year of protection (2007) with low rainfall, plant species richness ranged from five species on the *H. schmittiana* and *S. pungens* steppes to seven species on the *H. kahiricum* steppe. During the rainy year (2008), species richness increased and reached eight species on all vegetation community. In 2009 species richness decreased with decreased rainfall, but *S. pungens* and *H. schmittiana* steppes remained richer than *H. kahiricum* and *A. sericea* steppes (Fig. 4). The distributions of species suggest that plant species respond to the variation of available water which in turn depends on vegetation type. Table 1 gives the total species richness in the four plant community types.

**Table 1** Plant species occurrence in four plant community types during the 3 years of protection (Presence of each species in each sampling plant community during the three years of treatment is symbolized by (*), absent species is symbolized by (–))

| | *S. pungens* | | | *A. sericea* | | | *H. kahiricum* | | | *H. schmittiana* | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 2007 | 2008 | 2009 | 2007 | 2008 | 2009 | 2007 | 2008 | 2009 | 2007 | 2008 | 2009 |
| *Anthyllis sericea* Lag. subsp. *henoniana* (Coss.) Maire | – | – | – | * | * | * | – | – | – | – | – | – |
| *Anabasis oropediorum* Maire | – | – | – | – | – | – | * | * | – | – | – | – |
| *Argyrolobium uniflorum* (Decne.) Jaub. & Spach | – | – | – | – | * | – | * | * | – | – | – | – |
| *Atractylis serratuloides* Sieber ex Cass | – | – | – | * | * | * | * | * | * | – | – | – |
| *Calligonum comosum* L'Herit | – | * | – | * | * | * | – | – | – | – | * | |
| *Cutandia dichotoma* (Forssk.) Trab. | * | * | * | – | * | – | – | – | – | * | * | – |
| *Daucus syrticus* Murb. | – | – | – | * | * | * | – | – | – | – | – | – |
| *Fagonia glutinosa* Delile | – | – | – | – | – | – | – | * | – | – | – | – |
| *Gymnocarpos decander* Forssk. | – | – | – | – | – | – | * | * | * | – | – | – |
| *Hammada schmittiana* (Pomel) Ilji | * | * | * | * | * | * | * | * | * | * | * | * |
| *Helianthemum kahiricum* (Desf.) | – | – | – | – | – | – | * | * | * | – | – | – |
| *Herniaria fontanesii* J. Gay | – | – | – | – | – | – | * | * | * | – | – | – |
| *Koelpinia linearis* Pall. | – | – | – | * | * | – | – | * | – | – | – | – |
| *Launaea resedifolia* (L.) O. Kuntze | – | – | – | – | – | – | – | * | – | – | – | – |
| *Plantago albicans* L. | – | – | – | – | – | – | – | * | – | – | – | – |
| *Polygonum equisetiforme* Sibth. & Sm. | * | * | – | – | – | – | – | – | – | – | – | – |
| *Retama raetam* (Forssk.) Webb | * | * | * | – | – | – | – | – | – | – | – | – |
| *Salsola vermiculata* L. | – | – | – | – | – | – | – | – | – | * | * | * |
| *Savignya parviflora* (Del.) Webb | – | – | – | * | * | * | * | – | – | – | – | – |
| *Schismus barbatus* (L.) P. Beauv. | * | * | – | – | – | – | – | * | * | * | * | * |
| *Stipa lagascae* Roem. & Schult. | – | – | – | * | * | – | – | – | – | – | – | – |
| *Stipagrostis pungens* (Desf.) de Winter | * | * | * | – | – | – | – | – | – | * | * | * |

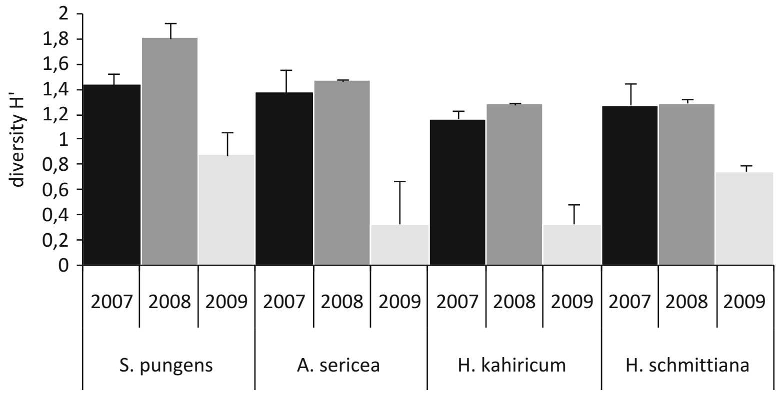
## Trend in Diversity Index

There are many different types of diversity indexes used by plant ecologists. Plant species diversity refers to number of species and their relative abundance in a defined area. Diversity measurements incorporate species richness, where S = the number of plant species in a community. The spatial scale strongly influences biodiversity (Crawley and Harral 2001; Symstad et al. 2003). Similarly, drought represents a determining component of arid ecosystem dynamics and biodiversity maintenance. Climate constitutes a crucial feature in desert rangelands, and drought often seriously affects the structure and diversity of vegetation communities, especially those on this steppe.

One key indicator of rangeland degradation is the diversity and its trend. In this chapter, the widely used Shannon-Weiner diversity index was used to assess the diversity of plant community in different years.

The diversity was significantly different between the four plant community types, and it was changing between years of study. In this chapter, ANOVA showed that diversity was significantly different between vegetation type ($F = 14.502$; $p < 0.001$) and year ($F = 122.932$; $p < 0.001$). Also vegetation type*year interaction is significant ($F = 3.215$; $p = 0.018$), indicating

**Fig. 5** Mean H′ diversity of four plant community types in southern Tunisia according to year. The effects of year and vegetation type are highly significant ($p < 0.05$). *Vertical bars* represent standard errors




that the effect of climate on plant diversity depended on steppe vegetation type.

The results obtained illustrate an increase in diversity in 2008 for *S. pungens*, *H. schmittiana*, *A. sericea,* and *H. kahiricum* steppes. Then it decreased in 2009 on all sites. In 2008, diversity reached 1.8 on the *S. pungens* steppe and decreased to 0.3 on the *A. sericea* and the *H. kahiricum* steppes in 2009 (Fig. 5). This may be explained by that the plant species on the *H. kahiricum* and *A. sericea* steppe would be more exposed to edaphic drought than on *H. schmittiana* steppe.

The effects of climate and vegetation type on plant have been studied by numerous researchers, and drought remains a major factor, if not the most difficult problem, in maintaining efficient, economical animal production enterprises on dryland pasture (Valentine 2001).

Grime (1979) reported that the environment can control species richness in two distinct ways, by regulating the expression of dominance and by affecting the potential richness (pool of suitable species). Previous studies in southern Tunisia have demonstrated relationships between vegetation type (Floret and Pontanier 1982) and rainfall (Le Houérou and Hoste 1977). Different resilience levels are intertwined with different vegetation type and intensity of natural disturbance. According to Floret and Pontanier (1984), climatic aridity can be increased or decreased if considering soil conditions and their utilization by man.

As was shown by Ludwig and Tongway (2000), once rangeland has been degraded, it is often possible to rehabilitate it and thus restore it to a level of utility, possibly not as good as its original state, but better than it was in its damaged state.

Desert rangelands have frequent droughty periods that have a marked effect on vegetation. Precipitation is extremely variable temporally. Variations of vegetation cover exhibited an overall ascending tendency at beginning then decreasing, but significant spatial contrast exists. Vegetation cover increased in 2008 where precipitation is strong. But in 2009 where it has relatively weak precipitation, vegetation cover deteriorated.

The dynamic variability of climatic parameters can also have significant implications for vegetation cover, species richness, and diversity. Changes in the vegetation diversity can be one of the most sensitive indicators of climatic variability, and they can be used to monitor the climate changes on different spatial scales. Hence, it is necessary to identify the different relationships existing between different climatic variables and vegetation dynamics. These relationships can have different patterns and magnitude on different spatial and temporal scales.

The soils in arid areas have characteristics that put limitations on the ecosystems (Lundholm 1976). As indicated by Whitford (2002), soil is the most important factor affecting vegetation structure in desert ecosystem. Sparsely distributed vegetation results in a heterogeneous horizontal pattern of vegetation patches alternating with areas of bare soil (Noy-Meir 1973). This spatial heterogeneity is now more broadly interpreted as cumulative outcome of the processes affecting the spatial and temporal distribution of vital resources such as water, topsoil organic matter, and propagules individually and collectively (Tongway and Ludwig 2005).

In this chapter we have shown that the changes in vegetation are affected more by the local climatic effects prevailing in the region. It is reasonable to assume that these changes will manifest themselves in the frequency of warmer months and warmer seasons. I show that the close relationship that exists among the vegetations and climatic variables is reflected by the impacts of the 2009 droughts. They also reveal that the dry season rainfall in the country will change by 25 % of vegetation cover on *H. kahiricum* steppe. These results would predict that these droughts would have resulted in substantial declines in vegetation.

The number of species is highest on *H. kahiricum* steppe, while *A. sericea* steppe reveals a slightly lower average. However, the number is lowest on *S. pungens* and *H. schmittiana* steppes.

The impact of species diversity on ecosystem functioning has generated considerable research and tremendous debate in view of accelerated depletion of biodiversity worldwide (Singh et al. 2005). In particular, many recent advances have indicated that diversity can be expected, on average, to give rise to ecosystem stability (Wolfe 2000; Chapin et al. 2000; Tilman 2000; McCann 2000). Although, for same period of drought stress, the real drought for plants in loamy soils is 60 % (*H. kahiricum* steppe) longer than in sandy soils (*H. schmittiana* steppe) (Floret and Pontanier 1984). These results show that *H. kahiricum and A. sericea* steppes are most affected by drought. This may be attributed to soil sealing, which plays a key role in crusts formation. In arid environments, sealing and crusting surfaces play major roles in ecosystem processes, particularly in water and soil flows, and are therefore critical for landscape structure and function (Eldridge et al. 1995; Zaady et al. 1997; Eldridge et al. 2000). They also play a major cause of decreased infiltration, increased erosion, and runoff, as well as low seedling emergence, on agricultural fields (Morin et al. 1989; Zhang and Miller 1996). Therefore, the reducing species richness and biodiversity in this study is said to crust, which is a physical barrier to plant germination, mainly annual species.

According to Noy-Meir (1998), the Mediterranean region includes a wide range of climatic and edaphic conditions so that it is difficult to generalize the results from one region or one ecosystem for all the Mediterranean areas. However, this chapter suggested that the stage of desert rangelands succession is an important factor which influences the evolution of vegetation composition and diversity after cessation of grazing in desert area.

Clearly, in these situations, the response to variation in species diversity cannot be separated from the response to environmental variation. This relationship is a central but contentious issue within ecology (Schmid 2002).

Today, the risks of desertification are substantial and clear. Under present scenarios of population growth, drought, and loss of ecosystem services, even within a stricto sensu classic sustainable development approach, the challenges posed by desertification are enormous and should therefore be easily comprehended. The desertification process embodies a strong disruptive potential in terms of rangelands stability. In desert rangelands, we suggested that there was an increase in soil vulnerability to wind erosion associated with fragile ecosystems resulting in desertification of the site (Fig. 6).

## Conclusion

The desert area in southern Tunisia is characterized by average annual rainfall limits (isohyets) of 75 mm years$^{-1}$. It is clear that desertization is the consequence not only of mismanagement of the environment but also of droughts. The natural vegetation is in accordance with rainfall and vegetation type, mainly via the soil drought regime, since soils are enormously different from place to place. And that, in dry

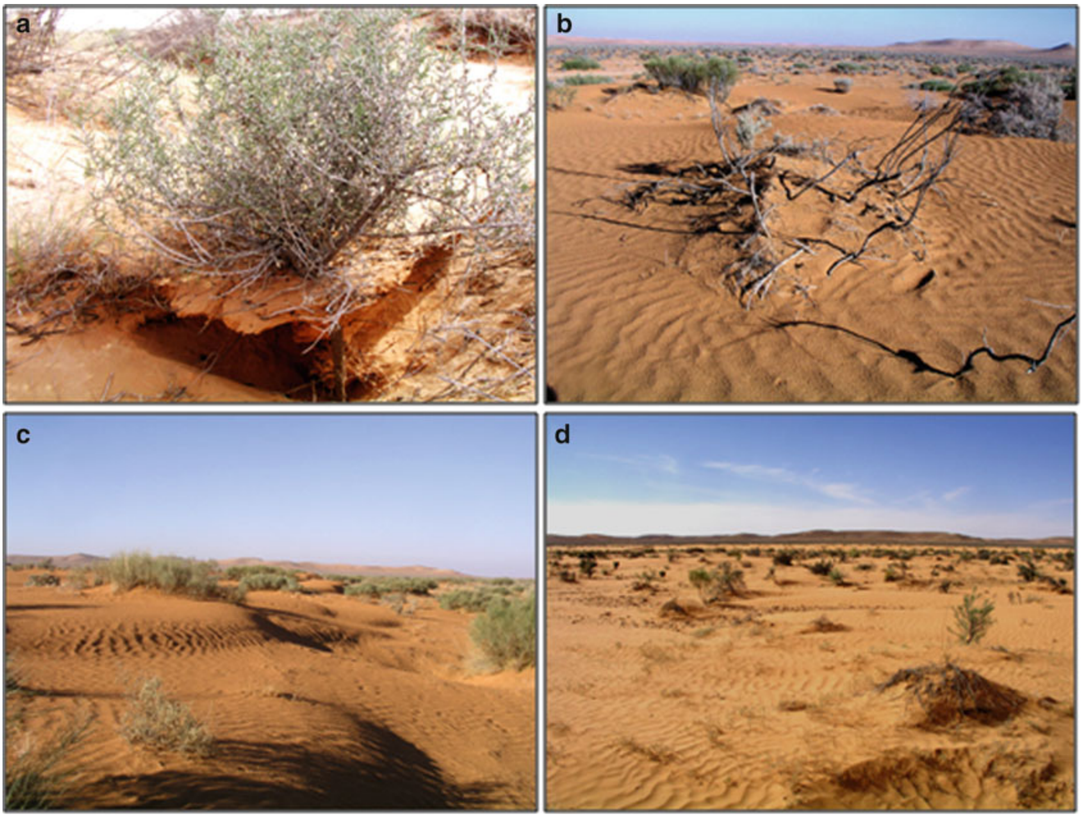



**Fig. 6** The desertification processes in protected desert rangelands; (**a**) wind erosion, (**b**) plant mortality, (**c**–**d**) sand accumulation

land, growing plants require particular soil textures and appropriate amounts of water.

We tested whether particular patterns of variation in cover, diversity, and richness could be applied generally to distinguish between mechanisms responsible for organizing vegetation.

The rangeland vegetation is highly dynamic due to climatic variability and the extensive ecosystem degradation resulting from increased population pressure brought about by both humans and animals. Spatial scale strongly influences vegetation, but, drought disturbance here was more evident. In the same way drought represents a determinant component of the arid ecosystem dynamics and for the maintenance of their biodiversity.

This chapter shows mainly that cover, species richness, and species diversity changes depending on spatial scale and rainfall. Vegetation cover on *H. kahiricum* steppe is very affected by drought than the other steppes, while diversity has been affected by drought on all steppes and mainly on *H. kahiricum* and *A. sericea* steppes.

Water condition is a limiting factor for vegetation, and precipitation takes a key role in the ecological distribution of vegetation. Variations of the annual precipitation from increasing, vegetation cover was improving, and conversely, variations of the annual precipitation from decreasing, vegetation cover was degraded. Moreover, the relation between precipitation and vegetation rapidly reached a significant level and steadily keeps at this level, indicating that vegetation change is very sensitive to precipitation change. Thus one can deduce that vegetation type affects the amount and structure of associated cover, and consequently the infiltration rate differs among vegetation types. The effects of cessation of grazing vary with vegetation composition, precipitation, and soil condition. The time needed for degraded overgrazed rangelands to recover and reach an excellent condition through natural succession differs depending on the climate and vegetation type.

In the light of our findings and also considering the short time of our experimentation, it is difficult to recommend optimal year duration of protection to regenerate the desert rangelands of southern Tunisia. Three years protection cannot increase more stability and biodiversity of desert rangelands.

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