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Perspectives on desertification: western Mediterranean

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In the western Mediterranean desertification is triggered by climatic variability and demographic disequilibrium, both of which directly and indirectly affect water budgets and land degradation through associated changes in land use patterns. This paper gives a historical perspective by reviewing major findings in climate and land use changes in the area, including information from tree ring, palynological, sedimentological, archaeological and archive analysis, with special emphasis on the past 500 years. This paper discusses the synergies between these changes and their implications to the most vulnerable ecosystems, such as mountain and semi-arid ecosystems, and compares current desertification processes in the area's north and south. In both cases rangelands and irrigated zones are the most affected land use systems. In the Maghreb (Algeria, Morocco and Tunisia), rangelands are being destroyed by overgrazing and agricultural encroachment. In northern countries rangelands are increasing at the expense of marginal agriculture. This paper discusses some controversial implications of rangeland vegetation recovery on fire and water regimes and reviews information on the steppes of *Stipa tenacissima*, paying attention to changes and degradation patterns, irreversible thresholds and implications of their spatial structure. Finally, this paper discusses western Mediterranean irrigated lands as hot spots of desertification; their vulnerability to rainfall variability; the difficulties of relieving them from overexploitation of water resources; and their terminal symptoms, such as soil salinization, exhaustion and deterioration of aquifers, and damage to downstream fluvial and wetland systems.

Introduction

In spite of the wide range of causes and effects often used to describe it, desertification is a well defined process. It is triggered by changes in climatic and socio-economic boundary conditions of affected dryland systems. These changes cause the systems to enter an irreversible positive feedback loop of overexploitation of land (Puigdefábregas, 1995). The final outcomes are land degradation and disruption of local

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economies. Desertification is an acute process that occurs at rates several orders of magnitude faster than purely climate-driven land responses.

Mediterranean desertification, particularly in the European zone, has been discussed in a number of comprehensive works and workshops (Fantechi & Margaris, 1986; Albadalejo *et al*., 1990; Rubio & Rickson, 1990; Sala *et al*., 1991; Perez-Trejo, 1992; Fantechi *et al*., 1995; Brandt & Thornes, 1996; Rubio & Calvo, 1996) which show how research has progressed from the descriptive approaches that prevailed in the 1980s to the more functional approaches that have prevailed in the past years. Fewer efforts, however, have been made to bring together information from both northern and southern Mediterranean countries (Le Houérou, 1992; Puigdefábregas & Garca Lorca, 1995).

The western Mediterranean zone (Fig. 1) has suffered several desertification episodes throughout its history. Reviewing them will help to widen our perspective on this process and on mitigation strategies. To this purpose, the following section focuses on the synergies between climatic and socio-economic forces driving desertification over the past centuries, the contrasts between desertification processes in northern and southern countries and features of rangeland dynamics relevant to understanding rangeland vegetation changes and degradation patterns.

Relevant features of the evolution of Mediterranean landscapes

The Mediterranean landscape was generated at the end of the Tertiary during a large-scale climatic transition that led to the area's characteristic summer drought. Since then this landscape has evolved under the stresses of major geological and climatic changes. Most orographic systems are young and often tectonically active. Hillslopes and drainage networks are far from equilibrium and support high rates of erosion and sedimentation. During the Holocene arid conditions consolidated in the basin, with particularly dry periods from 6000 to 4000 years B.P. and humid fluctuations between 3000 and 250 years B.P. (Rognon, 1987). Recent findings using carbon isotope discrimination in grain cereals across the Iberian peninsula show that the north–south

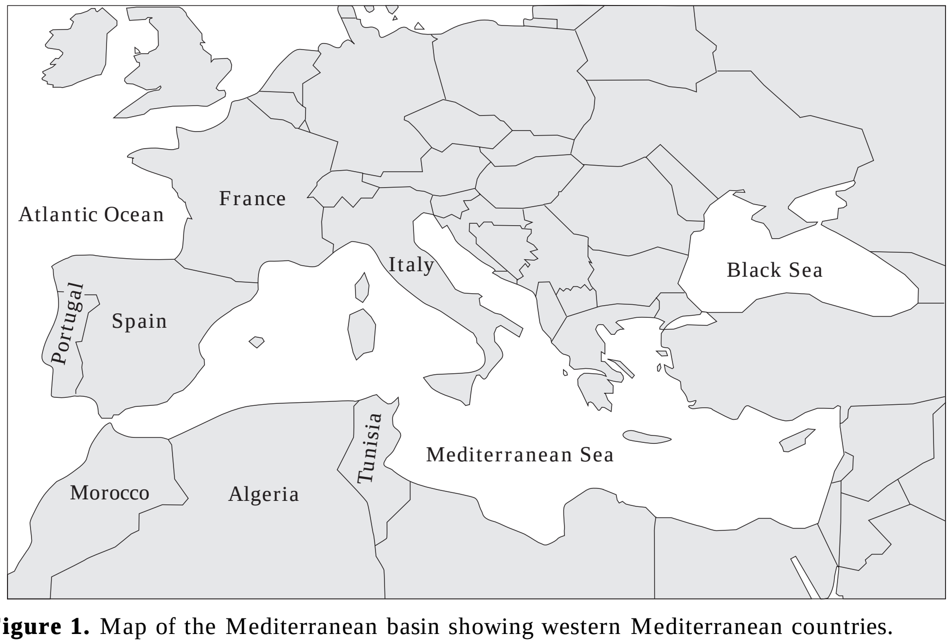

Figure 1. Map of the Mediterranean basin showing western Mediterranean countries.

aridity gradient has tended to become steeper during the past 5000 years (Araus *et al.*, 1997).

A growing amount of palaeobotanical evidence shows that xeric vegetation types have been pushing mesic woodlands to the north and up into the mountains (Parra, 1993). The Mediterranean vegetation as it is found today has doubtless been shaped by human activity (Pons, 1981; Pons & Reille, 1988). Humans helped to extend sclerophyllous vegetation, evergreen oaks and pines from their original areas in dry soils at the expense of deciduous and semi-deciduous oaks. In so doing, humans have created a mosaic-like landscape, with a very rich spatial structure and high levels of biological diversity (Naveh, 1991).

Two conclusions can be drawn from the above-mentioned trends. The first is that Mediterranean landscapes have evolved under strong environmental changes and disturbances that have been occurring up to the present time. Information is needed to assess the relative importance of those forcing factors *vs.* internal regulatory mechanisms (i.e. density dependence, competition) in controlling the ecosystem trajectories. The second conclusion warns against the idea of a pristine or primeval Mediterranean landscape, free from human intervention, as an unrealistic assumption. We should not try to understand the landscape without the presence of humans but to understand it as it is. This conclusion has strong implications for conservation policies because if we want to conserve our landscape, we are obliged to live in it.

Climatic–social synergies during the past 500 years

The above-mentioned changes provide a framework of long-term trends, but they can hardly be considered desertification episodes. They happened at rates several orders of magnitude slower than those happening today. Human influence was lacking or very limited. And their irreversibility cannot easily be established, considering low population densities. As an outcome of climatic and social driving forces operating synergetically, desertification is not new in the Mediterranean. Well documented cases involving fast landscape changes can be found in the Iberian peninsula over the past centuries.

Major climatic and land use fluctuations during the past 500 years

Dendroclimatological information from the Iberian peninsula allows us to characterize the Little Ice Age (LIA) fluctuation as a cooler and more humid although highly variable climatic period. Its maximum occurred in the second half of the 17th century, with around –0·5 °C as the medieval optimum (Creus, 1992). Most of the analysed series converge in that this maximum was characterized by dry summers, moist autumns and mild springs (Creus & Puigdefábregas, 1976, 1983). Since then temperatures have been slowly increasing while autumn–winter rainfall has declined (except for a small rise at the beginning of this century) as recorded in tree ring indices from Sierra de Cazorla in the south and the Pyrenees in the north of Spain (Fig. 2).

Since 1900 two opposite secular trends have been reported (Oñate Rubalcaba, 1993) from instrumental records. In the south-east, mainly between 1890 and 1940, annual rainfall decreased by about 3 mm year^{-1} (Thornes, 1991), whereas annual rainfall increased in the north-west. During the last 15 years a general increase of temperature and decrease in rainfall has been recorded (Oñate Rubalcaba, 1993).

These facts do not allow us to reject the hypothesis of the climate still being in the warming limb of the 17th century fluctuation. The short-term variations may be interpreted as an east–west oscillatory behaviour of the 500 hPa field, described by Italian authors in the Mediterranean (Palutikof *et al.*, 1996).

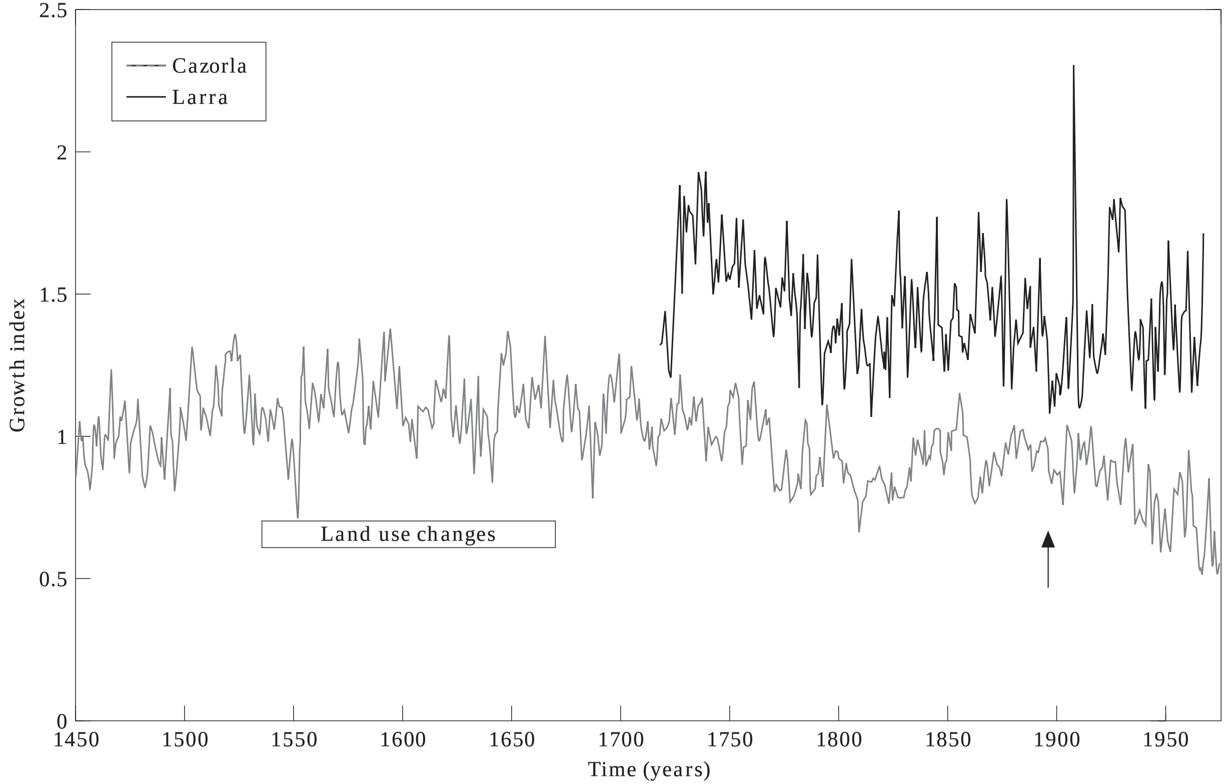

Figure 2. Two series of standardized tree ring growth indexes from Spain. Cazorla: 37° 55' N; 3° 0' W, on *Pinus nigra* Arnold, at 1850 m altitude, $N = 7$. Larra: 42° 57' N; 0° 48' W, on *Pinus uncinata* Ramond, at 1800 m altitude, $N = 8$. The occurrences of the main land use changes and demographic peaks (arrow) are shown. Source: Creus & Puigdefábregas (1976, 1983).

Socio-economic systems experienced three critical phases reflected in land use changes during the last half millennium. The first occurred during the 16th and 17th centuries as a result of the definitive establishing of Christian rule over the whole peninsula and the consolidating of colonization in America. Both factors caused a southward expansion of the dryland agriculture that prevailed on the inner Iberian high plains as well as a high demand for wool and wood products to meet the needs of the American settlements (Elliot, 1965) and the growing shipbuilding industry (Dupre Ollivier, 1990). The result was a spread of grain crops and large-scale sheep raising, as well as a thinning of forests. The second critical phase was caused by the population increase that started in the 18th century (Dupre Ollivier, 1990) and overcrowded rural areas during the first half of the present century. The third critical phase has been produced by the major technological, social, cultural and economic changes that pervaded rural life in the 1960s.

Landscape responses to combined climatic and anthropogenic stressors

Heavy impacts on the landscape resulted during the 16th–18th centuries from the simultaneous occurrence of cold and humid climatic fluctuations (Little Ice Age peak) and extensive land use changes that left large areas exposed to erosion, as has been shown by sedimentological, archaeological and ecological evidence. Two types of such evidence may be outlined.

In hilly country accelerated erosion caused by new agricultural systems resulted in increased sedimentation rates recorded in the deltas of eastern Andalucia rivers: 17 mm year^{-1} in the Adra and 80 mm year^{-1} in the Andarax during the 18th century (Hoffmann, 1988). Changes in river regimes to higher flood frequencies have also been recorded in sedimentary sections of the Jucar River in Valencia (Butzer *et al.*, 1983) and the Guadalmendina River in Málaga (Sermet, 1969).

In mountain areas the conversion of forest to grassland in the subalpine belts disturbed a particularly sensitive zone. In the Pyrenees natural timberlines have been lowered by some 500 m, from around 2100 to some 1600 m altitude (Barrio *et al.*, 1990; Garcia-Ruiz *et al.*, 1990). This anthropogenic change, coupled with the Little Ice Age, produced two main effects.

The first effect was the downward extension of the solifluction limit, with an increase of mudflows that strip away soils previously occupied by forests. Late spring frozen layers in topsoil are more common in grasslands than in forests. In such conditions, the positive hydraulic pressures that build up near the snowpack margins often give way to mudflow outbursting (Barrio & Puigdefábregas, 1987).

The second impact has been documented by comparing present day responses of the sensitive zone with nearby remaining forest areas in the Pyrenees (Puigdefábregas & Alvera, 1986). The results show that the conversion of forest to grassland may double the specific runoff and increase by 16 times the specific sediment yield. Solute responses are variable: yield increases of 300% have been recorded for dissolved phosphate, and decreases of 150% have been recorded for nitrate. These effects are likely to be exacerbated in the colder and more humid conditions that prevailed during the Little Ice Age.

The second socio-economic disturbance, overpopulation in Spanish rural life during the first half of the 20th century, was associated with the encroachment of agriculture on rangelands and the increase of stock densities, followed by grassland exhaustion and soil loss by erosion. The crisis seems to have particularly affected the Ebro Valley, as shown by the aggradation of its delta (Dupre Ollivier, 1990).

The impacts of the third group of stressors mentioned above — the technological, social, cultural and economic changes that pervaded the rural life in the 1960s — are discussed in the following section.

Current desertification in the west Mediterranean zone (WMZ)

North–south contrasts

In the southern part of the west Mediterranean zone (WMZ) current desertification is driven by population growth, whereas in its northern counterpart the main factor is the intensification of agriculture. Both stressors operate synergetically with the variability of rainfall, but their outcomes greatly differ.

The Maghreb's (southern WMZ) population has increased by 300% since 1950, whereas the northern WMZ's population has increased by only 30% (Fig. 3). In most countries current growth rates are very low or even negative. But large-scale trends in population reallocation are also occurring in northern WMZ countries. Here population lost from inland areas is concentrating in coastal zones. This 'littoralization' of the economy is particularly acute in some countries, such as Spain, where the population density in the coastal Mediterranean zone is double the country's average population density (Grenon & Batisse, 1988).

Evolution of the arable land surface traces the population trends on both shores (Fig. 3). In the south, agriculture is encroaching on steppes and rangelands to meet the needs of growing populations. In the north, agriculture is being abandoned in most marginal areas, where it spread during the first half of this century, and is concentrating in the more fertile areas with irrigation facilities, particularly on the littoral plains. Rangelands and irrigated lands are the present day hot spots of desertification in the WMZ. Both give rise to important management challenges that are discussed below.

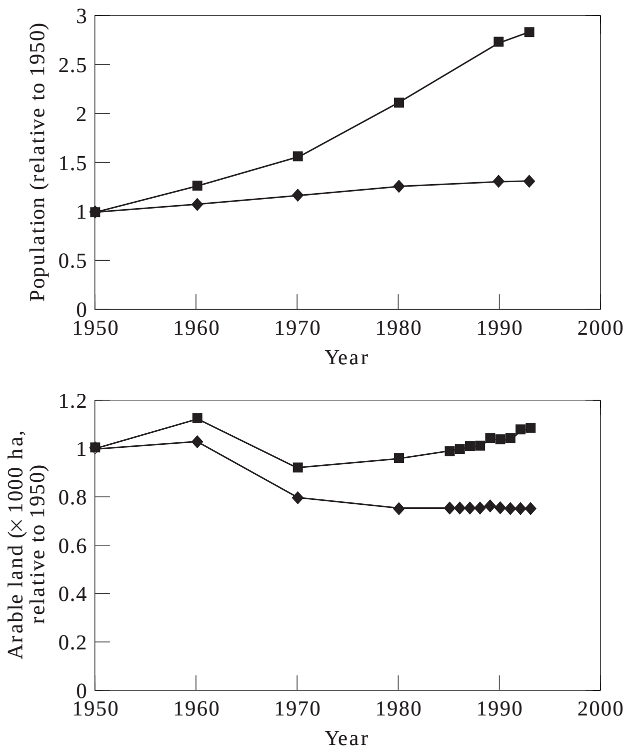

Figure 3. Demographic and land use trends in western Mediterranean countries, relative to 1905 values. (◆) = N-MED (Portugal, Spain, France, Italy); (■) = S-MED (Morocco, Algeria, Tunisia). Source: FAO (1950–1993).

Desertification in rangelands

Most of the changes in the area of arable land during the past 50 years occurred at the expense of the rangelands and shrublands. In the south WMZ the steppes are reportedly (Le Houérou, 1995) being destroyed by human pressure at a rate of 1% per year. The primary production of the steppe has declined to 40–20% of their values 50 years ago, and the contribution of the steppes to meeting nutritional requirements of livestock is less than 50%. The remainder of nutritional requirements are being provided by concentrate fodder supplements.

In the north WMZ the area of rangelands and forests is increasing at the expense of marginal agriculture. The process is being driven by the above-mentioned littoralization of the economy and by the agricultural policies at the European Union and country levels. This process may be accelerated by prolonged droughts that bring the ratio of precipitation/potential evapotranspiration (P/PET) below 0·5, an admitted threshold under which rain-fed grain crops become uncertain (Le Houérou, 1992). It is estimated that more than 80,000 km^2 of grain crops on red Mediterranean oxysoils overlying shallow limecrusts in Greece, Spain, Portugal and southern Italy could become untenable with a worsened water balance and could revert to shrubland (Le Houérou, 1992).

The present rangeland and forest area, particularly in the north WMZ, includes a wide range of past degradation and desertification impacts. Some of these impacts exceeded the land's resilience thresholds and today remain essentially unchanged. This is the case of some badlands in southern Spain that were already in this condition some 5000 years ago as shown by archaeological evidence (Wise *et al*., 1982). In other areas active desertification lasted until quite recently. This is the case of southern Portugal (Roxo, 1995), where a combination of population growth and a cereal self-sufficiency policy caused the second desertification episode to last from the beginning of this century until the present time.

Now agriculture is being abandoned, and the soils left behind are exhausted and degraded. In some areas desertification impacts were shaded out by natural or artificial vegetation and soil recovery. In many regions attempts of rural populations to promote soil and water conservation in harsh environments created potentially unstable conditions as hillslope terraces and runoff harvesting structures. Such structures cannot be maintained without local rural workers, who are leaving these areas as populations decline.

Distinguishing active from inherited or relict desertification is crucial to setting up mitigation programmes. First, stressors must be relieved. Then the threatened areas must be rehabilitated or restored. Even doing nothing would be sufficient.

Some implications of vegetation recovery in rangelands

Despite the benefits of vegetation recovery to soil conservation, two potential side effects may result, particularly in the north WMZ: an increase of wildfires and a decrease of runoff. Both of these effects are still poorly understood and controversial, but they present fundamental challenges to land management policies in semi-arid areas.

In Spain the annual area burned by wildfires increased by 600% from 1960 to 1990 (Prieto, 1993), and the question arises about the possible association between fire and the increase of vegetation cover. Fuel accumulation in large tracks increases the occurrence of big fires, while fine-grained mosaic landscapes with small patches of forest, shrub and cultivated fields or grassland suffer from more localized and easy-to-control fires (Trabaud, 1991). Palynological evidence (Pons, 1981) shows that during the Holocene humans and fire have been shaping Mediterranean landscapes into fine-grained mosaic structures. But new relationships between humans and their environment may lead to coarse-grained landscapes that are more and more susceptible to fire.

The expansion of forest and shrubland also has hydrological consequences that are mainly related to the increase of evapo-transpiration and the reduction of total runoff. In southern France, for an annual rainfall of 1200 mm, the vegetational changes that occurred between 1946 and 1979 associated with agricultural abandonment are responsible for a decrease of 80 mm in specific runoff (Rambal, 1987). In south-west Spain the conversion of grassland to dehesa or open forest leads to increased evapo-transpiration from 250 mm to 500 mm (Joffre & Rambal, 1993). Data from southern France and Spain (Fig. 4) show that if the whole land were covered with forest and shrubs and had less than 400 mm annual rainfall there would be virtually no drainage. This means that drainage is actually supported by degraded areas that function as runoff sources. On the other hand, it is also well established that forest and shrubland favour infiltration and base flow at the expense of rapid runoff (Piñol *et al*., 1991; Domingo *et al*., 1994).

Rangelands often occupy headwater domains. Changes in their area or structure, therefore, have heavy impacts on flow regimes, sediment transport and the water quality of rivers. Less evaporation and more fast runoff are the main hydrological issues of rangeland degradation. Increased risks of flooding and sedimentation damage are their immediate consequences. Rangelands are often at the crossroads of integrated watershed management in arid climates. Some degradation is needed to feed aquifers, while too much degradation has harmful downstream impacts.

The case of steppes

In the WMZ, steppes stretch between P/PET ratios of 0·25 and 0·05, which correspond roughly to annual rainfall values of 400 mm and 100 mm, respectively. In the southern WMZ stock breeding is the main support of the population, and rain-fed grain cropping is risky. With harvest expectancies lower than 0·5 (Le Houérou, 1992, 1995), tree crops using runoff harvesting techniques are becoming popular.

Most of the steppes in the Maghreb and south-east Spain are dominated by alfa grass (*Stipa tenacissima*) or plant communities associated with it. As revealed by present day remains and by local people, in the past, arboreal steppes of alfa mixed

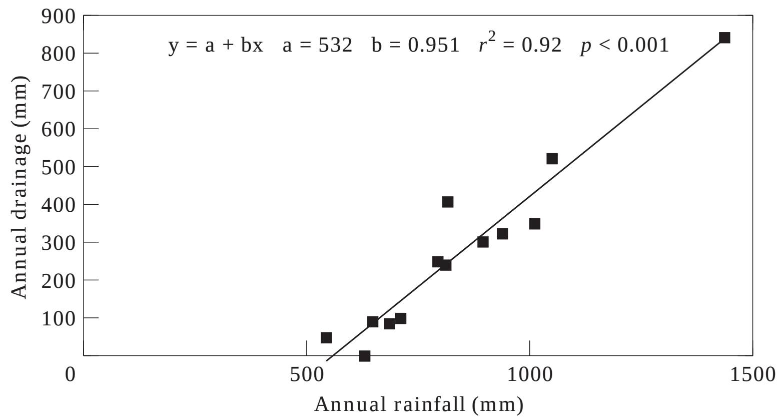

Figure 4. Field measurements of annual drainage against annual rainfall from western Mediterranean forest and shrubland catchments. Sources: Domingo *et al.* (1994); Piñol *et al.* (1991); Rambal (1984); Joffre & Rambal (1993).

with open woods and shrubs were widespread (Aidoud, 1989; Aronson *et al*., 1993; Le Houérou, 1995). Their woody components are Mediterranean woodland residuals (i.e. *Pinus halepensis*, *Quercus coccifera*, *Juniperus phoenicea*, *Rosmarinus officinalis*) with an increasing share of thermophyllous (i.e. *Tetraclinis articulate*, *Chamaerops humilis*) and tropical species (i.e. *Acacia tortilis* spp. *radiana*) toward the south (Le Houérou, 1995).

In the Maghreb the aboreal steppe is being converted into a grass steppe by firewood harvesting. Only 6% of its area 50 years ago remains (Le Houérou, 1995). In south-east Spain destruction of the arboreal steppe has probably been going on much longer and has been worsened by associated activities such as mining and harvesting alfa for cellulose and the demographic saturation during the second half of the 19th century.)

Overexploitation of the grass steppe by grazing and harvesting, both for cellulose and fuel, has caused the steppe's progressive enrichment with 'steppic' shrubs, such as *Artemisia herba alba*, *Artemisia campestris*, *Retama sphaerocarpa*, *Anthyllis cytisoides* and *Rhanterium suaveolens*. This 'lignification' or shrub encroachment, hastened by cultivation and abandonment of degraded soils, may eliminate alfa.

Compared to grass steppes, shrub steppes have lost soil organic matter. At the soil surface of intershrub areas shrub steppes often develop physical or biological crusts that increase runoff at the expense of infiltration (Chaieb, 1991; Le Houérou, 1995).

Shrub steppes may be further degraded by water or wind erosion, but their changes are more controlled by site characteristics (Aidoud, 1989). Shedding sites become enriched with rock fragments, exposed rocks or limestone crusts, while accumulation sites tend to be enriched with sand or silt. Such environmental changes strongly influence the development of vegetation. Typical shedding site specialists are *Atractylis serratuloides*, *Helianthemum* spp., *Thymus* spp. and *Noaea mucronata*. *Aristida pungens* is a widespread specialist at sand accumulation sites.

The above described states are interpreted by most authors as a degradation 'cascade.' Many of the changes are considered reversible, but there is a general agreement that the shrub steppe state cannot return to a grass steppe state, even if the

Table 1. *Rainfall and annual biomass budgets in* Stipa tenacissima *stands at the Rambla Honda field site (south-east Spain). Source: Puidgefábregas* et al. *(1997)*

		1992		1993		1994		1995	
Rainfall (mm)	Autumn	55		107		45		139	
	Winter	165		124		50		40	
	Spring	31		61		27		15	
	Summer	78		0		0		17	
	Annual	329		292		122		211	
Stipa tenacissima		Mean	SE	Mean	SE	Mean	SE	Mean	SE
AGB	Living	356	44	483	48	442	37	456	56
AGNPP	Total	326	26	400	71	329	67	197	18
LFALL		387	39	411	55	224	41	279	55
BINC		127	18	–42	15	14	19	25	16

AGB = above-ground biomass (g m^{-2}); AGNPP = above-ground net primary productivity (g m^{-2} $year^{-1}$); LFALL = litterfall (g m^{-2} $year^{-1}$); BINC = above-ground biomass increment (g m^{-2} $year^{-1}$).

stressors are relieved. Supported more by field observations than by direct experimental evidence, this threshold of irreversibility is explained by the loss of organic matter and physical degradation of intershrub soil surface and by the failure of alfa to regenerate from seed in open areas without the protection of scattered trees (Aronson *et al*., 1993; Le Houérou, 1995).

The contrast between exploited and non-exploited alfa steppes can be documented by comparing two field studies carried out in the Rogassa field site (250 mm annual rainfall) in north-west Algeria (Aidoud, 1996) and in the Rambla Honda field site (280 mm annual rainfall) in south-east Spain (Puigdefábregas *et al*., 1996, 1997).

In Rogassa the above-ground green biomass of alfa stands was monitored for almost 20 years (1975–1993). Its starting biomass was 146 g m^{-2}, 50–60% of which was lost during the study period, both in the grazed and ungrazed condition. The growth response to rainfall decreases in grazing exclusions and moderately grazed areas and becomes virtually absent in freely grazed stands (Fig. 5).

In Rambla Honda a 4-year record (1992–1995) of above-ground biomass in alfa stands that have been unharvested and ungrazed since the 1950s shows a steady-state condition (Table 1). Biomass values fluctuated without obvious trend between 356 g m^{-2} and 483 g m^{-2}, and annual biomass increments varied between 127 g m^{-2} and –42 g m^{-2}. Above-ground net primary productivity is particularly sensitive to variations in spring rainfall.

These results cast some doubt on the degradation cascade described above. The Rambla Honda data suggest that the upgrading of alfa steppe to arboreal steppe, if possible at all, may be extremely slow. Data from Rogassa suggest that in exploiting alfa steppes some threshold may be reached, beyond which degradation proceeds irreversibly, even if the stressor is relieved.

Field observations at Rambla Honda and simulations of up and down cycles of alfa tussocks using cellular authomata models (Sanchez & Puigdefábregas, 1994) show that at low growth rates tussocks are caught by their own debris, and the whole stand may stay without noticeable changes for a long time. Moreover, field experiments and

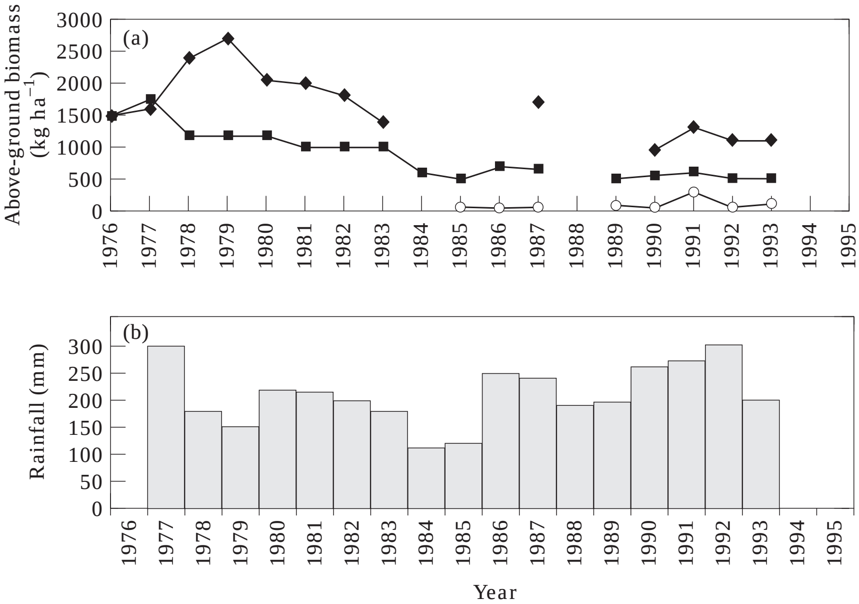

Figure 5. (a) Trends of above-ground biomass (kg ha^{-1}) in *Stipa tenacissima* stands under three grazing levels, at the Rogassa field site (western Algeria). (◆) = grazing excluded; (■) = moderate grazing; (○) = free grazing. (b) Annual rainfall (mm). Source: Aidoud (1996).

geostatistical measures on alfa-covered hillslopes (Puigdefábregas & Sanchez, 1996) have shown that (a) tussocks harvest runoff from uphill bare patches and (b) the spatial pattern of the tussocks is controlled by topography and the associated sediment flow. This means that tussocks and bare patches are complementary elements and that the fraction of plant cover allowed by water availability tends to adopt spatial structures that minimize the lengths of runoff and sediment movement along slopes.

Research conducted in shrub steppe stands of *Retama sphaerocarpa*, also in Rambla Honda (Pugnaire *et al.*, 1996*a*,*b*; Moro *et al.*, 1997), found that shrubs build up resource islands (Garner & Steinberger, 1989) that are self-reinforced by powerful mechanisms of mutual facilitation between shrubs and their ground layer of winter annuals.

Desertification in irrigated lands

The WMZ irrigated area amounts to 105,000 km^2, 80% of which is in the northern countries (Portugal, Spain, France and Italy) and the remaining 20% in the Maghreb (Morocco, Algeria and Tunisia). Since 1970 the irrigated area has grown fast, particularly in the south WMZ and in Spain, where it has increased by 300% and 150%, respectively (Fig. 6).

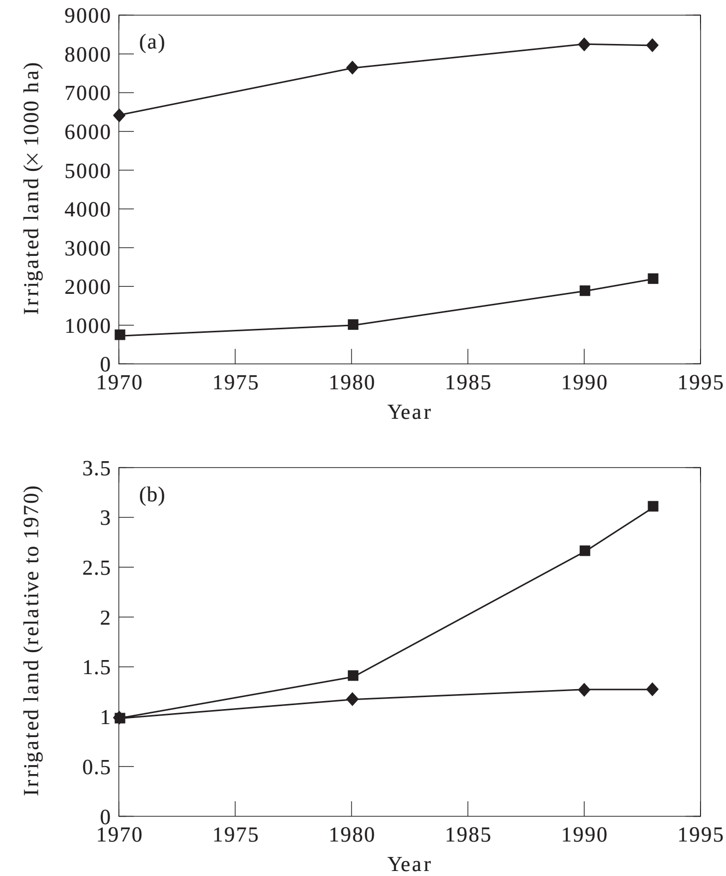

Figure 6. (a) Absolute and (b) relative trends of the irrigated area in western Mediterranean countries. (◆) = N-MED (Portugal, Spain, France, Italy); (■) = S-MED (Morocco, Algeria, Tunisia). Source: FAO (1970–1993).

Given the amounts of available water resources, this trend makes the WMZ irrigated systems particularly vulnerable to climatic fluctuations. Regular or ground-water resources can provide only small-scale buffer supplies because they account for only a small percentage of the total resources — Spain 24%, France 56%, Italy 17%, Greece 19% — and total consumption relies largely on them — Spain 163%, France 6%, Italy 49%, Greece 48% (Table 2).

As a result of this situation, in many irrigated areas rates of water consumption are largely above the sustainable availability of the resource. Another limitation is that most of the irrigation developments are supported by large investments and population inflows. Such economic systems are not easily adaptive. If the water supply is reduced by climatic causes or demand pressures, there is no other choice than to go into a positive feedback loop of overexploitation, which further deteriorates aquifers and soils by marine intrusion and salinization. In fact, around 25% of the irrigated land in Mediterranean Europe suffers from soil salinization (Szabolcs, 1990). In Spain salinization affects particularly the south-east and some inland regions in Castilla and the Ebro Valley, as well as estuaries and deltas (Ebro, Guadalquivir).

On the other hand, mean net water consumption of irrigated land in Spain is around 600 mm year^{-1} (Chabart *et al.*, 1996), an amount equal to the mean effective annual rainfall collected over an area more than 10 times as large as the irrigated area, if 10% of rainfall is assumed to be left to drain. Under such conditions local tensions are inevitable. Once irrigated areas reach a certain density, they consume the ground-water resources of their whole district and affect the health and conservation status of fluvial and wetland ecosystems (Llamas, 1992), which are important to the global functioning of the landscape.

WMZ irrigated areas include hot spots for current desertification. Most of these spots have developed recently, stimulated by technologies for obtaining water and by the European Union's growing demand for horticultural products. This development results in an inflow of people and investments to the affected areas. Overexploitation soon follows without possibility of release because of the large investments that have been made. Unless social and economic measures are taken to diminish the pressure on water resources, this positive feedback loop destroys the affected agricultural systems.

Two examples of advanced desertification triggered by irrigation developments are well documented: the Souss Valley in south-west Morocco and La Mancha in central Spain. In the Souss Valley over pumping lowers the water-table at a rate of 2 m year^{-}, and soil salinization reaches untenable levels. Already land is being abandoned, and sand is encroaching in some areas (Ait Tirri, 1995). In La Mancha water-tables are being lowered at rates of around 1 m year^{-1}, and irrigated agricultural systems have extended beyond their sustainability thresholds. The aquifers cannot continue to feed

Table 2. *Water resources in western Mediterranean countries in 1985 (m^3 $year^{-1} \times 10^9$). Regular resources include ground-water and regulated surface water supplies. Source: Blue Plan (Grenon & Batisse, 1988)*

Country	Total resources (a)	Regular resources (b)	% (b/a)	Total extraction (c)	% (c/a)	% (c/b)
Spain	31	8·0	26	14·0	38	184
France	74	35·0	47	16·0	21	45
Italy	187	31·0	17	46·0	25	152
Morocco	4	0·9	23	1·1	29	122
Algeria	11	2·5	23	1·7	26	68
Tunisia	3	1·5	50	2·0	65	133

the fluvial systems, and in their downstream wetlands (the National Park of Tablas de Daimiel) peat oxidiation and vegetation collapse lead to the destruction of key ecological functions (Romero *et al*., 1997).

Conclusions

Mediterranean landscapes evolved under strong environmental changes and disturbances that have been occurring from their differentiation in the late Myocene until the present time. Information is needed to assess the relative importance of those forcing factors *vs*. internal regulatory mechanisms (i.e. density dependence, competition) in controlling ecosystem trajectories.

The idea of a pristine or primeval Mediterranean landscape, free from human intervention, is unrealistic. We should not try to understand the landscape without the presence of humans but to understand the landscape as it is. If we are to conserve our landscape, we must live in it.

In the west Mediterranean zone (WMZ) desertification is triggered by the synergetic operation of climatic and socio-economic driving forces that affect water budgets and land degradation through associated changes in land use patterns. In the Iberian peninsula during the past 500 years three critical phrases were reflected in land use changes. The first phase, in the 16th–17th centuries, is associated with the joint effects of the Little Ice Age, political changes and requirements of American colonization. The second phase, at the beginning of the 20th century, can be associated with the demographic saturation of rural areas. The third phase started in the 1960s and has been driven by social and technological changes in rural life.

The main driving forces of current desertification are population increases in the south WMZ and population reallocation to littoral and irrigated areas in the north WMZ. As a consequence, rangelands and irrigated lands have become hot spots of desertification. While Maghreb rangelands are being destroyed by overgrazing and encroachment of agriculture, north WMZ rangelands are increasing at the expense of marginal crops. In spite of the local positive consequences on soil and water conservation, vegetation recovery has some controversial implications for wildfire and hydrological regimes. Rangelands are at the crossroads of integrated watershed management. Too much fuel across large surfaces may favour big fires and large water losses by evapo-transpiration. Some degradation is needed to feed aquifers, but too much degradation increases the downstream risks of flooding and sedimentation damage.

The steppes of alfa (*Stipa tenacissima*) are widespread in the WMZ. Field evidence suggests a degradation cascade, from arboreal steppes to grass steppes and shrub steppes. Most authors agree on the irreversibility of the second step.

In alfa steppes tussocks tend to adopt spatial structures that minimize runoff lengths and sediment movement along slopes. In *Retama sphaerocarpa* steppes mutual facilitation between shrubs and their ground layer of winter annuals helps build up high fertility spots or resource islands.

Degradation of irrigated lands by overexploitation of water resources includes all the features of desertification. Its ultimate symptoms are exhaustion and deterioration of aquifers, soil salinization and damage to downstream fluvial and wetland systems.

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