

## ECOLOGICAL IMPACTS OF GLOBAL CHANGE ON DRYLANDS AND THEIR IMPLICATIONS FOR DESERTIFICATION

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### ABSTRACT

Potential impacts of global change on dryland ecosystems are reviewed from four perspectives. First, results are reviewed from recent research on vegetation change, land degradation and desertification. The role of disturbances and low recurrence events in triggering non-linear changes by driving ecosystems beyond their resilience thresholds is outlined. Particular attention is paid to the development of spatial structures as feedbacks that tend to buffer degradation. Secondly, the synergetic operation of climatic and anthropogenic forcing factors of dryland degradation is discussed in the light of some case histories. Changes in one group of those factors often render the ecosystems particularly sensitive to changes of those in the second group. Thirdly, off-site effects of dryland degradation are summarized, including feedbacks to the atmosphere, changes in biological diversity and downstream impacts in river catchments. Finally, the implications of global change for land degradation control policies are outlined. The importance of prevention is emphasized, as well as the need to carefully consider where to apply rehabilitation and restoration. Prevention includes soft and cheap measures based on management practices, while restoration calls for massive and expensive interventions on soil or vegetation. © 1998 John Wiley & Sons, Ltd.

KEY WORDS: climate change; desertification; dryland; dynamics; ecosystem; land use

### INTRODUCTION

Drylands include areas where rainfall does not meet the evaporative demand. They occupy around 45 per cent of the land surface, contain about 30 per cent of the world's total carbon in above- and below-ground biomass (Allen-Diaz, *et al.*, 1996), and include grasslands, shrublands, savannas, xerophytic woodlands, and hot and cold deserts. Their rangelands support approximately 50 per cent of the world's livestock and provide forage for both domestic animals and wildlife. This survey on the expected ecological impacts of global change on drylands will be divided in four sections. First, recent findings and ideas on ecosystem dynamics and land degradation in drylands will be reviewed, with emphasis on the role of disturbances in ecosystem change and of feedbacks triggered by spatial organization. Secondly, forcing factors of dryland degradation will be discussed with particular attention given to climatic and anthropogenic synergies. Thirdly, a summary of the off-site effects of dryland degradation is presented, including global feedbacks to the atmosphere, biological diversity and downstream impacts in river catchments. Finally, some suggestions are made on the implications of the expected impacts of global change for land degradation control policies, such as prevention, rehabilitation and restoration.

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## ECOSYSTEM DYNAMICS AND LAND DEGRADATION IN DRYLANDS

Rainfall variability and depletion of above-ground biomass by herbivores or fire have been forcing factors in the evolution of dryland ecosystems. These are, in a sense, pre-adapted to be exploited and people have taken advantage of this feature throughout history (Margalef, 1974). In most cases, drylands are far-from-equilibrium systems, because their trajectories, in terms of carbon cycling, productivity and species composition, are controlled by the aforementioned forcing factors or disturbances, rather than by internal regulatory mechanisms such as density dependence or competition (Ellis, *et al.*, 1993).

*Disturbances and Ecosystem Change in Drylands*

Linear succession models, in which vegetation states move back and forth under the effect of perturbations, going forward asymptotically to a particular reference type when they approach carrying capacity upon cessation of a disturbance, apply only to subhumid conditions. Such models are meaningless in the unsteady climates of the arid zones, where ecosystem dynamics is essentially event-triggered (Lavee, *et al.*, 1998, this issue). In these zones, cessation of a disturbance does not necessarily lead to recovery, several states may coexist in a stable way, a particular event may lead to different responses depending upon antecedent local conditions, and episodic changes are interspersed between long, stable intervals (Stafford Smith and Pickup, 1993; Wiegand, *et al.*, 1995).

Research conducted on arid grazing systems in New South Wales, Australia (Caughley, *et al.*, 1987) shows that where the coefficient of variation of annual rainfall is larger than about 30 per cent, long-term ecosystem behaviour is better explained by rainfall variability than by mean values. In Africa, the work carried out in the southern Ethiopian (Coppock, 1993) and northern Kenyan rangelands (Ellis, *et al.*, 1993) shows that 400 mm annual rainfall forms the threshold between equilibrium-type dynamics and non-equilibrium or event-triggered dynamics.

In the Ethiopian highlands, with 400–700 mm rainfall, cyclic patterns occur, driven by herd management. Cyclic sequences, of overgrazing, shrub encroachment, destocking, grass recovery, restocking and again overgrazing, prevail in the region. Carrying capacity, successional trends and density-dependent population dynamics are consistent concepts in that environment (Coppock, 1993). On the contrary, the dynamics of the Turkana grazing system in northern Kenya are essentially driven by drought events. Herbivores and herds respond opportunistically to climate driven changes in the rangeland (Ellis, *et al.*, 1993). They exploit the ranges during 'good times' and go away looking for other resources when drought arrives. This strategy is also practised and recommended in Australian arid rangelands (Westoby, *et al.*, 1989). In such conditions, the ecosystem is always in a transient, far-from-equilibrium condition.

Most disturbances (i.e. rainfall fluctuations and fire) have been embodied into dryland ecosystems during their evolution. However, some of these disturbances are new or not yet incorporated and may drive the ecosystem to qualitatively different new states along irreversible trajectories. In a thorough study, conducted in the southern edge of the Great Karoo, South Africa (Wiegand, *et al.*, 1995), the former type of disturbances have been called 'integral events' and the latter 'transition-triggers'. The research was carried out in an area with a mean annual rainfall of 167 mm and a high interannual variability (50–400 mm). Major recruitments of shrubs were found to be linked to gap availability and winter rainfall above a certain threshold. As gaps depend on the death of adults, which is triggered by drought, recruitment is finally controlled by the occurrence of a sequence of events, i.e. a drought followed by a wet winter. These climatic features may be considered as 'integral events' that keep the ecosystem far from equilibrium, in a permanent transient condition. The authors were able to simulate the behaviour of this system in a more predictable climate, by increasing rainfall by 5 mm–10 mm per month. Results show a shift to equilibrium dynamics in these new conditions and a more linear and predictable ecosystem behaviour, with saturation features when approaching carrying capacity.

Of the changes generated by transition triggers, a subset includes desertification and land degradation, being those that lead to a decline in productivity and a reduction of below-ground carbon storage. In most

cases, transition triggers result from combinations of climatic and anthropogenic events. Among the large-scale and best-documented examples are the vegetation changes that occurred in central Australia, after the European settlement, due to the introduction of livestock and the European rabbit, and to the modification of fire patterns, which became less frequent and more widespread. The impacts of these changes were particularly dramatic when coinciding with climatic fluctuations (Griffin and Friedel, 1985).

#### *Spatial Organization and Land Degradation*

A very important class of feedback that arises from dryland ecosystems in response to disturbances is linked to spatial organization. Disturbances stimulate spatial heterogeneity and their negative effects are buffered by the latter in two ways: (a) through spreading of extinction risks by developing patches with desynchronized life cycles and different physiological responses at the specific and intraspecific levels (Wiegand, *et al.*, 1995); and (b) through development of spatial structures in vegetation and soils that help non-uniform infiltration and increase soil water retention (Bergkamp, *et al.*, 1996).

Asynchronous dynamics among patches is the spatial organization required to compensate for buffer crashes in fluctuating environments (Bascompte and Solé, 1995). Heterogeneity among patches provides a source of asynchrony and this may be an underlying factor of the high levels of intraspecific diversity in dryland ecosystems. Genetic heterogeneity has been postulated to explain the observed differences in water relations among replicate bushes of *Anthyllis cytisoides* in the south of Spain (Puigdefábregas, *et al.*, 1996). Gene migration, via pollen, from the inner regions towards the more stressed borders of specific ecosystems has been proposed as a mechanism which increases resilience to climatic fluctuations (Sage, 1996). In chaotic systems, slight differences in initial conditions are another source of asynchrony. Although the probability of local extinction increases in the chaotic domain, the probability of global (metapopulation) extinction decreases, because the higher the dependency on initial conditions, the faster the desynchronizing rate by divergency of initially close trajectories in the space phase (Bascompte and Solé, 1995).

Spatial mosaics of vegetated and bare patches develop through powerful self-reinforcing mechanisms. Vegetated patches increase their fertility by catching runoff, sediments and nutrients that are supplied by the neighbouring bare areas. The former have been called 'resource islands' that build up 'hot spots' in the land mosaic and support the highest rates of soil and vegetation change (Schlesinger, *et al.*, 1990).

Differences between vegetated and bare ground patches have been extensively described in the literature and have been reviewed by several authors (Schlesinger, *et al.*, 1990; Puigdefábregas and Sanchez, 1996). Vegetated patches show an increase in soil-water storage capacity, higher soil carbon and nitrogen inputs, as well as increased sediment accumulation. Resource islands themselves are complex systems that integrate different plant biotypes. Building blocks are perennial plants, with long response times and reacting to low recurrence disturbances. Subordinate communities include annual plants with shorter response times, that react to high recurrence fluctuations. It has been shown that mutual facilitation mechanisms develop between both vegetation types, that are reinforced in time and increase the resilience of the whole assemblages (Pugnaire, *et al.*, 1996a, 1996b).

Up and down cycles of growth and senescence in patchy vegetation were described years ago (Watt, 1947) and their significance has been recognized through the incorporation of cyclical or non-directional change in succession theory (Horn, 1981; O'Neill, *et al.*, 1986). Nevertheless, it is only recently that the hydrological implications of these processes began to attract the attention. It is acknowledged, for example, that leaf area index and canopy projective cover are controlled by water availability (Specht and Specht, 1989) and hence by climate and soil properties. However, the same leaf area can be distributed in different ways, forming mosaics of clumps and bare patches of different sizes and with different patterns. It has been suggested that such spatial structures evolve as an adaptation, to harvest as much water as possible in the vegetated clumps (Puigdefábregas and Sanchez, 1996). The outcome of these adaptive mosaic structures is that plant clumps have to develop mechanisms to harvest water from bare patches, and that the dimensions of both phases of the mosaic are not independent from each other. Both hypotheses are supported by field observations.

Water harvesting from contiguous bare areas can be achieved by extending root systems into these, and by catching runoff. The first strategy has been demonstrated in sparse shrublands of *Retama sphaerocarpa* (Haase, *et al.*, 1996) by adding labelled water at different depths (up to 28 m) in bare patches. After 24 h, the label was identified in the canopies. The second has been shown in *Stipa tenacissima* communities (Puigdefábregas and Sanchez, 1996) by excluding tussocks from the overland flow generated in uphill bare patches. Average annual soil moisture and plant growth were significantly lower in excluded tussocks than in controls. The significance of runoff harvesting by vegetated patches has also been shown in field studies using rainfall simulation experiments (Bergkamp, 1996; Bergkamp, *et al.*, 1996).

If vegetated and bare patches are hydrologically linked to each other, it can be shown that for a given leaf area index or plant cover, optimum mean lengths of clumps and bare areas depend on: (a) the overland flow harvest efficiency of the former, and (b) the parameters of overland flow decay with slope length in the latter. Reasonable agreement has been found between predicted and observed lengths in old *Stipa tenacissima* stands (Puigdefábregas and Sanchez, 1996).

One of the major trends in the degradation of dryland ecosystems is the replacement of grass by shrubs. Two main processes associated with this change are a reinforcement of vegetation patchiness and an increase, by several orders of magnitude, in the spatial scale of the redistribution of water and resources. This was also evidenced by experiments that show an increase of interrill runoff (Abrahams, *et al.*, 1995), and by the increase in spatial variability of soil or surface properties upon degradation (Figure 1) (Schlesinger, *et al.*, 1990; Pickup, *et al.*, 1994; Imeson, *et al.*, 1996; Cammeraat and Imeson, in press; Aronson, *et al.*, 1995).

Plant cover is not a passive semipermeable umbrella, but a living structure that reacts to hillslope fluxes of water and sediments by changing its shape and its spatial organization. Such changes have been reported for tussock grasses (Sánchez and Puigdefábregas, 1994) and explained as an interaction between sediment fluxes and plant growth. In flat areas, tussocks are circular, while on hillslopes, particularly at low growth rates, tussock shapes become elongated in the direction of contour curves.

At the hillslope level, sediment fluxes are not only regulated by sediment availability and sediment transport capacity, but also by a set of feedbacks from plant growth that may be considered as an analogue

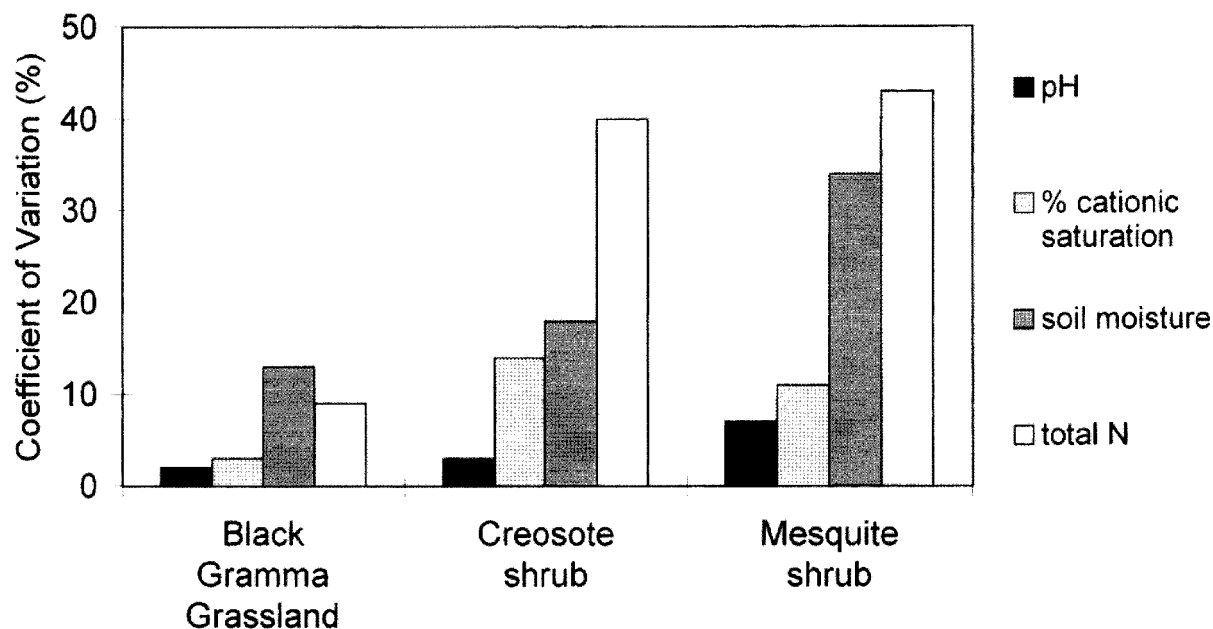


Figure 1. Coefficient of variation associated with mean values for soil properties measured in 200 samples taken in each of three habitats in the Jornada experimental range. Source: Schlesinger, *et al.*, 1990.

of a tuning mechanism. This behaviour has been described for tussock grasses (Gallart, *et al.*, 1993; Puigdefábregas and Sanchez, 1996). If the sediment movement rate remains within the tuning range, parallel stripes along contours develop in response, which maximize water availability to plants and soil storage. At sediment fluxes below the tuning threshold, as happens where slope gradient is low, the plant clump-bare patch assemblages distribute at random. At sediment fluxes above the tuning threshold, plant clumps break off and originate rills that hinder plant installation in these rills. In this way, finally banded patterns are produced with stripes in the slope direction.

Classical exponential decay curves which are widely used to describe the effect of decreasing plant cover on sediment yields and runoff outputs are likely to be influenced by spatial patterns. It can, for example, be predicted that contour line stripes have larger exponential decay parameters and more concave curves (Gallart, *et al.*, 1993), while downhill stripes have smaller decay parameters and less concave curves or even convex curves, with peaks in output at low plant covers as a result of rill enhancement (Rogers, 1989).

Examples of feedbacks from spatial patterns of vegetation to runoff fluxes can be found in arid climates, on gentle slopes and fine textured soils. In such cases, vegetation often develops banded patterns at decametre scale, which are known as 'tiger bush' in the Sahel (Valentin and Herbes, 1996), but have also been described in Australia (Tongway and Ludwig, 1990) and in the Mexican Chihuahuan desert (Janeau, *et al.*, 1996). Tiger bush has been interpreted by these authors as a structure that develops by interaction between vegetation and runoff processes, as an adaptation to harvest the overland flow generated in bare stripes. It has also been shown that the tiger bush pattern is rather unstable, since it can be destroyed through slight changes in rainfall or land use (Rossetti, 1996).

#### FORCING FACTORS OF LAND DEGRADATION

Available information hardly supports the hypothesis of climate change as a direct forcing factor or transition trigger in dryland degradation. Some simulations, based on doubled CO<sub>2</sub> climate change scenarios, predict the expansion of warm grasslands and shrublands, rather than that of deserts (Figure 2) (Allen-Diaz, *et al.*, 1996). Temperate and cold grasslands are supposed to become net sources of carbon, while warm grasslands and savannas could become net sinks (Scholes and Hall, 1996). In the former, a shift of C-storage from the soil to above-ground biomass is predicted (Ojima, *et al.*, 1993) with possible implications for soil stability and water storage capacity. The reliability of these predictions is still medium to low, given the uncertainties in current models.

Effects of small changes in the temporal patterns of rainfall on event-triggered dryland ecosystems may surpass those resulting from changes in average values, as used in current climate change scenarios. However, if looking at the level of a landscape, dryland ecosystems are more resilient to current climate variability than it was thought years ago (Puigdefábregas, *et al.*, 1996). This can be ascribed to the combination of an opportunistic behaviour of its species and a wide range of buffering mechanisms, which were already discussed in the previous section.

Almost all reported case studies on dryland degradation show that transition trigger events are the result of some combination of anthropogenic and climatic factors, which reinforce each other to induce such events. This synergy between both types of factors, which act as driving forces for ecosystem change, exhibits the same process structure that characterizes desertification.

In a fluctuating and drought prone environment, systems exploited by humans can be driven to extinction or desertification through disturbances, connected with transition triggers and due to changes in, e.g. climatic, soil, social, cultural and economic factors. These disturbances are such that boundary conditions are changed and the system is overexploited, i.e. beyond its resilience thresholds (Figure 3). In a very simplified way, this system behaviour may be considered as an analogue to that of predator-prey systems, in which humans stand for the predator and natural resources for the prey (Puigdefábregas, 1995). In such systems, changing boundary conditions often imply some modification of system parameters. For

example, climatic changes may affect parameters such as carrying capacity or fertility, while technological changes may modify the per capita intake or survival rates.

The synergy among anthropogenic and climatic forcing factors arises from the fact that they are linked together concerning their effects on system behaviour. A change in a forcing factor modifies some parameters and, as a consequence, the system behaviour in relation to the remaining factors will also be modified. Most

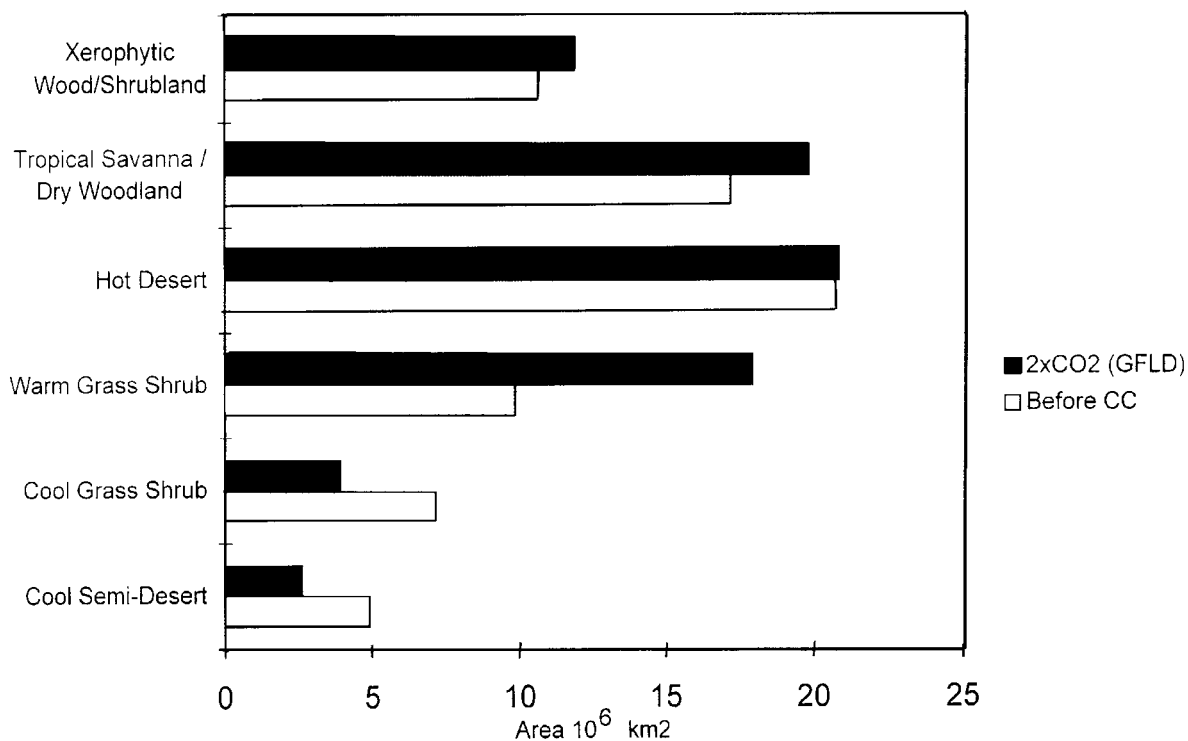


Figure 2. Potential changes in area of dryland types from BIOME 1.1 model (Prentice, *et al.*, 1992) after temperature and CO $_2$  increases under the Geophysical Fluid Dynamics Laboratory (GFDL) scenario (Greco, *et al.*, 1994). Source: Allen-Diaz, *et al.*, 1996.

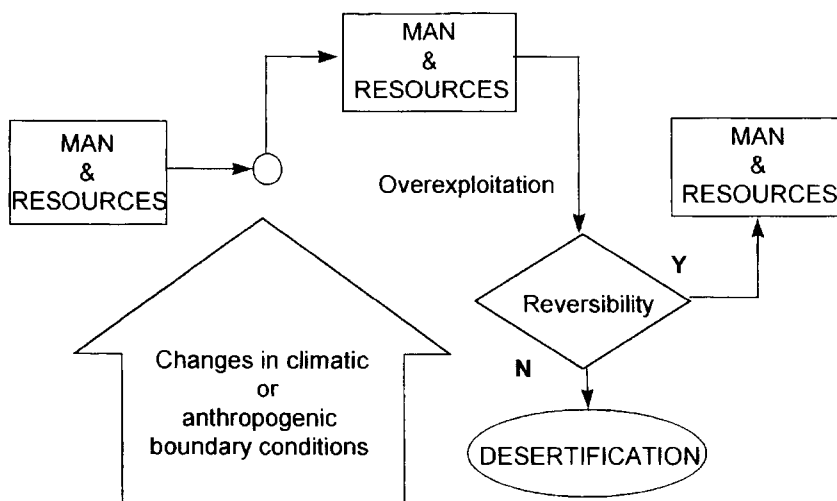


Figure 3. Outline of the desertification process. Square boxes represent the system formed by the human population and its local resources. The system is driven to an overexploitation regime by changes in its boundary conditions. Source: Puigdefábregas, 1995.

of such synergetic combinations fall into three categories: (a) synchronic climatic and anthropogenic changes, (b) anthropogenic changes that lead to climatic vulnerability, and (c) climatic changes that lead to socio-economic unsustainability.

#### *Synchronic and Independent Climatic and Anthropogenic Changes*

An example of this situation may be found in the Iberian Peninsula during the 16th and 17th centuries. At that time, two large-scale perturbations occurred almost simultaneously: the peak of the Little Ice Age (Creus and Puigdefábregas, 1983) and the major political, cultural and economic changes associated with the consolidation of Christian rule over the Peninsula (Figure 4). The consequences of the political transformations were large-scale land-use changes, such as the agricultural encroachment of woodlands in the south and the lowering of timberlines in the northern Iberian mountains. In both cases, the simultaneous occurrence of a climatic fluctuation caused soil erosion to increase and, hence, significant changes in some system parameters, such as carrying capacity or fertility in the resource subsystem.

The new agricultural systems that were established in the Iberian southeast were brought by colonizers from the Iberian inland high plains and were based on grain crops and sheep. Such land use systems left the soil bare during long periods of the year and it became vulnerable to autumn rains, which were more abundant during the Little Ice Age (Creus and Puigdefábregas, 1983). The result was a dramatic increase of sedimentation rates in the deltas, ranging from 17–80 mm  $y^{-1}$  (Hoffmann, 1988). Those figures are of the same order of magnitude as those which are recorded today in the most unstable Alpine and Andean regions. Fluvial regimes became more torrential, as has been shown in the sediment profiles from river channels (Butzer, *et al.*, 1983; Sermet, 1969).

Grassland conversion of subalpine forests left a zone particularly sensitive to climatic fluctuations. In the Pyrenees, natural timberlines were lowered with about 500 m (Garcia Ruiz, *et al.*, 1990; Del Barrio, *et al.*, 1990). This land-use change started around the 13th century (Montserrat, 1992) and consolidated thereafter as a result of the need for the timber of the shipyards. When combined with the Little Ice Age peak, around the 17th century, the descent of the timberlines triggered a downwards extension of solifluction with associated mudflows that stripped away large tracks of soil previously occupied by forests (Puigdefábregas, 1988) and caused significant increases of fast runoff ( $\times 2$ ) and sediment yield ( $\times 16$ ) (Puigdefábregas and Alvera, 1986).

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Figure 4. Two series of standardized tree ring growth indexes from Spain. Lower series: Cazorla: 37°55'N; 3°0'W, on *Pinus nigra* Arnold, at 1850 m altitude,  $n = 7$ . Upper series: Larra: 42°57'N; 0°48'W, on *Pinus uncinata* Ramond at 1800 m altitude,  $n = 8$ .  
Source: Creus and Puigdefábregas, 1983.

*Anthropogenic Changes Leading to Climatic Vulnerability*

Two kinds of anthropogenic triggers can be mentioned: population increase and changes in market conditions. Both factors act as transition triggers that increase the vulnerability of land to climatic stresses and ultimately lead to land degradation and desertification. This process is well documented in the Maghreb, where the area used for grain crops increased from 8.6 million ha in 1950 to 9.7 million ha in 1985, and sheep stocks expanded from 18 million in 1950 to 33 million in 1984 (Le Houérou, 1991). In Western Algeria, for example, this resulted in a decrease of the standing biomass of *Stipa tenacissima* rangelands from 1460 kg ha<sup>-2</sup> of dry matter to less than 100 kg ha<sup>-2</sup> in the period from 1976 to 1992 (Aidoud and Touffet, 1996). Similar situations occurred at the beginning of this century in European Mediterranean countries such as Spain.

Changes in market conditions can also trigger land-use changes that increase land vulnerability to climate fluctuations. System parameters, such as the per capita exploitation rate of natural resources, often exhibit dramatic modifications. Per capita resource availability also changes and triggers in or out fluxes in population. One of the best examples stems from the United States, where at the beginning of this century farmers cultivated grain crops on the best lands of the Great Plains. World War I caused a cereal shortage in Europe and prices went up sharply. This event had a quick transcontinental effect: in the southern plains, grain crops doubled in area between 1910 and 1920, encroaching over rangelands (Blakenburn, 1993). At the end of the 1920s, a major drought expanded from the east, resulting in the well-known 'dust bowl'. A fall in prices often has contrary effects. In Mediterranean Europe, for example, this led to a reversion of around 6000 ha of marginal crop land into shrubland and forest between 1965 and 1984 (Le Houérou, 1991).

*Climatic Changes Leading to Socio-economic Unsustainability*

A very common class of transition trigger arises from the alternation of humid and dry periods. The former leads to an increasing pressure on the resources, while the latter results in irreversible degradation, if it is not possible to release this pressure before resilience thresholds have been exceeded.

The Sahelian zone provides one of the best examples of this sequence of events (Puigdefábregas, 1995). Rainfall in the region was well above the mean during the period 1945–68 (Rognon, 1991). An inflow of people coming from the south substituted the seminomadic pastoralism with agriculture and sedentary husbandry. The drought of the 1970s trapped those people between the desert to the north and the already settled agricultural land to the south (Thebaud, 1993). No choice was left to them other than full exhaustion of resources.

Similar histories are reported for the arid zones of Australia and Patagonia where tracking rainfall with stock numbers is common practice for pastoral industries. These strategies often fail because destocking cannot be achieved fast enough when drought arrives. In such conditions, livestock concentrates in bottom lands with higher soil water storage, and degrades these by overgrazing (Pickup and Stafford Smith, 1993; Bonvissuto, *et al.*, 1992).

Anticipated impacts of climate change provide further examples of these kinds of synergies. A 2°C increase in mean annual temperature for the next 50–60 years, as assumed by most scenarios, would cause the ratio of rainfall to potential evapotranspiration to fall below 0.5 or even down to 0.25, in areas with about 400 mm annual precipitation within the semihumid zone (Le Houérou, 1991). It has been estimated that, as an outcome of these changes, more than 80 000 km<sup>2</sup> of grain crops on red soils with petrocalcic horizons (Xeralfs) across Portugal, Spain, Italy and Greece would turn into shrubland and forest (Le Houérou, 1991). The consequences are not only net increases in above-land carbon storage, but also higher fire risks.

It is well established that large-scale fuel accumulation encourages large-scale fires, while in fine grained mosaic landscapes, with small patches of forest, shrub and crops or grasslands, fires are more localized and easy to control (Trabaud, 1991). In drylands of developed countries, climate changes and market conditions may foster new relationships between humans and their environment, leading to coarser grained landscapes, which are more liable to fire. It is also well established that a lower moisture content of the fuel and higher

temperatures may dramatically increase flammability (Christensen, 1985) and thus enhance fire risks. How the expected relative increase of structural compounds in the vegetation (trunks and stems), associated with enhanced atmospheric CO<sub>2</sub> concentrations (Overdieck and Forstreuter, 1991), will affect fire risks, has still to be verified.

### OFF-SITE EFFECTS OF LAND DEGRADATION IN DRYLANDS

Drylands occupy huge areas and throughout these, the climatic and anthropogenic factors that work together as triggers for dryland degradation, are active. Therefore, significant off-site effects of their degradation can be expected. Three kinds of effects will be considered: (a) global feedbacks with the atmospheric system, (b) consequences for the biological diversity, and (c) downstream effects.

#### *Global Feedbacks of Dryland Degradation*

Dryland degradation leads to an increase of the albedo and decrease of the latent heat flux into the atmosphere. The effects of these changes on the climate system, although still controversial, are likely to be significant (Schlesinger, *et al.*, 1990), particularly in large flat inland areas where land–atmosphere exchanges are more driven by surface properties than by oceanic or topographic controls. Long-range transport of dust link dryland degradation to global biogeochemical fluxes. Climate-driven changes of land use, as has already been mentioned, may override the predictions on carbon emission/sequestration values that derive from pure climate change scenarios. Thus, for the Mediterranean region, carbon storage predictions for combined climate change scenarios and doubled atmospheric CO<sub>2</sub> (Ojima, *et al.*, 1993) predict a diminution of soil organic carbon (SOC) of about 5 per cent, while the agricultural release of marginal lands, which is currently happening, may increase SOC up to four times (Table I).

Table I. Biomass, productivity and organic matter in the upper 5 cm of soil in some north Mediterranean ecosystems (C g m<sup>-2</sup>)

| Ecosystem type                           | Soil 0–5 cm | AGB  | BGB  | NPP | SDEAD | Author |
|--|-------------|------|------|-----|-------|--------|
| Shrub ( <i>Retama</i> sp.)               | 649         | 176  | 1198 | 85  | 111   | 1      |
| Shrub ( <i>Anthyllis</i> sp.)            | 933         | 189  | 325  | 107 | 150   | 1      |
| Steppe ( <i>Stipa tenacissima</i> )      | 1796        | 287  | 515  | 147 | 1054  | 1      |
| Shrub ( <i>Quercus coccifera</i> )       | 1579        | 1057 | 2070 | 49  | 103   | 2      |
| Shrub ( <i>Quercus coccifera</i> )       | 10 272      | 1417 | –    | 370 | –     | 2      |
| Shrub (maquis)                           | 2453        | 4350 | 2480 | –   | –     | 3      |
| Shrub ( <i>Pinus pinaster</i> afforest.) | –           | 2230 | –    | 441 | –     | 4      |
| Shrub ( <i>Pinus nigra</i> afforest.)    | –           | 2509 | –    | 534 | –     | 4      |
| Shrub ( <i>Rosmarinus</i> sp.garrigue)   | 1921        | 344  | –    | 45  | –     | 5      |
| Shrub ( <i>Quercus coccifera</i> )       | –           | 2564 | –    | –   | –     | 6      |
| Shrub ( <i>Cistus ladanifer</i> )        | –           | 1651 | –    | –   | 32    | 7      |
| Shrub ( <i>Cistus ladanifer</i> )        | –           | 738  | –    | –   | 127   | 7      |
| Phrygana                                 | –           | 493  | –    | 91  | –     | 8      |
| Cereal                                   | 587         | –    | –    | –   | –     | 5      |
| Cereal                                   | 678         | –    | –    | –   | –     | 9      |

Notes: C (biomass) = 0.45 × dry weight; Soil C values are derived from C concentrations by weight and bulk density estimates when necessary; AGB = above-ground biomass; BGB = below-ground biomass; NPP = net primary productivity (C g m<sup>-2</sup> yr<sup>-1</sup>); SDEAD = standing dead. Blanks in the Table mean either, not estimated or not mentioned. Sources: (1) Puigdefábregas, *et al.*, 1996; (2) Rapp and Lossaint, 1981; (3) Arianoutsou and Paraskevopoulos, 1992; (4) Moro, 1992; (5) Lopez Bermudez, *et al.*, 1996; (6) Merino, *et al.*, 1990; (7) Nuñez, 1989; (8) Margaritis, 1975; (9) Kosmas, *et al.*, 1996.

### *Effects on Biological Diversity*

In the past, civilizations relied heavily on the biological diversity of drylands. Most of our important crops, such as wheat, barley, sorghum, millet and cotton, originate from these lands (Bie and Imevbore, 1995). As mentioned earlier, spreading of risks and mutual facilitation are mechanisms that enhance specific and intraspecific diversity in drylands. The effects of dryland degradation on diversity depend on their climatic conditions and evolutionary background. In arid climates, disturbances often lead to diversity losses, while in semiarid to subhumid climates maximum diversity values are associated with moderate disturbances. Furthermore, disturbances which are foreign to the evolution of drylands have a far more dramatic impact on biological diversity (West, 1993). This is exemplified by the differences between the impacts of large mammal grazing with European herbivores in the Argentine pampas and in Australia (Pickup, *et al.*, 1995), and of land use in the circum-Mediterranean drylands. The first resulted in heavy diversity losses, while the latter is marked by intermediate levels of exploitation, associated with peaks in diversity (Naveh and Whittaker, 1979).

### *Downstream Effects*

Drylands often occupy headwater areas, for which reason their degradation has a heavy impact on flow regimes, sediment transport and water quality of rivers. Less evaporation and faster runoff are the main hydrological issues and these result in increased risks of flooding and damage caused by sedimentation. Integrated watershed management in arid climates has to aim at a certain balance in degradation. Some degradation is necessary to feed aquifers, while too much degradation leads to harmful downstream impacts. Rainfall-drainage curves derived from field data in shrubland and forest catchments in the western Mediterranean (Figure 5) show that in areas with a rainfall of less than 400 mm, evapotranspiration is so high that virtually no water is discharged. Field records from southern France, with 1200 mm annual rainfall, show that the increase in vegetation biomass that occurred between 1946 and 1979, due to agricultural abandonment, was responsible for an 80 mm decrease in runoff (Rambal, 1987). The extent and

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Figure 5. Plot of annual streamflow against annual rainfall from west Mediterranean forest and shrubland field data, showing the fitted linear regression line.  $y = a + bx$ ;  $a = 532$ ;  $b = 0.951$ ;  $r^2 = 0.92$ ;  $P < 0.001$ . Sources: Domingo, *et al.*, 1994; Piñol, *et al.*, 1991; Rambal, 1984; Joffre and Rambal, 1993.

spatial distribution of degraded areas are therefore crucial issues to ensure runoff sources and recharge of aquifers in drylands.

### IMPLICATIONS FOR DESERTIFICATION CONTROL POLICIES

Given the extensive land use in drylands, effective measures to control desertification should rely more on prevention and rehabilitation than on restoration. The latter includes direct interventions in soils or vegetation, that are often too expensive for widespread implementation. Long-term self-maintenance of restoration actions should be ensured and they should be based on ecological and economic arguments. Prevention requires cheap, quick and reliable indicators for early assessment of desertification risks and trends, and for differentiating areas where recovery or rehabilitation is possible using soft technology from those beyond resilience thresholds, which would require restoration programmes (Aronson, *et al.*, 1995). In any case, prescribed measures must have appropriate spatial and temporal dimensions, in order to ensure that they will be taken at the right place and at the right time.

Rehabilitation includes soft management-based measures (i.e. grazing, watering, fire), which restrict agriculture to areas capable of sustaining it, and accompanying policies to increase local economic complexity in order to alleviate the burden of humans on natural resources. Rehabilitation, particularly in the case of rangelands, requires the adoption of appropriate management strategies to cope with rainfall variability. Measures in this line include the use of an appropriate (large) size of units in management and the development of viable marketing systems capable of absorbing opportunistic variations of stock numbers (Allen-Diaz, *et al.*, 1996; Pickup and Stafford Smith, 1993). Furthermore, it is essential that the fitness of traditional opportunistic behaviour of pastoral populations (i.e. nomadism, migration) is assessed and its behaviour adapted to current economic conditions, before mining-type exploitation of rangelands becomes the rule.

### CONCLUSIONS

Most dryland ecosystems, particularly in arid climates, are far-from-equilibrium systems. Their trajectories are more controlled by changes in their boundary conditions than by their own internal regulatory mechanisms, such as density dependence or competition. They are in a quasi-permanent transient condition under the influence of events that arise from particular combinations of forcing factors, and their response to them is often non-linear.

Most of the existing variability in boundary conditions has been incorporated into dryland ecosystems during their evolution. However, there is a class of events (transition triggers) that brings the ecosystem beyond its resilience thresholds and drives it to qualitatively different new states. Desertification is one of such trajectories leading to land degradation, which includes productivity decline, reduction of below-ground carbon storage and changes of several orders of magnitude in the characteristic scale of spatial redistribution of water and nutrients. In these event-triggered dryland ecosystems, effects of small changes in time distribution patterns of rainfall or other boundary conditions may surpass those of changes in average values, as applied in most climate change scenarios.

An important class of feedbacks developed by dryland ecosystems in response to disturbances relates to their spatial organization. These buffer effects operate at two levels: (a) by spreading extinction risks, and (b) by increasing the efficiency of water and nutrient use, counteracting the trend to increase the scale of their spatial redistribution.

Large-scale climatic and anthropogenic forcing factors often have synergetic effects on dryland ecosystems. Changes in one of these render the ecosystem more sensitive to changes in the others. In most cases, transition triggers start from an alternation of humid and dry periods. The former induce an increasing human burden on the resources, while the latter leads to irreversible degradation, if steps are not taken to release the pressure before resilience thresholds have been exceeded.

By considering the effects of global change on dryland ecosystems, not only local impacts, but also off-site influences or global feedbacks need to be taken into account. Among the latter, there are three significant groups: (a) feedbacks to the climate system by modifying earth-atmosphere fluxes, (b) impacts on biological diversity patterns, and (c) downstream effects on rivers, pertaining to flow regimes, water quality and sediment transport.

Given the extensive land use in drylands, effective measures to control degradation and desertification should rely more on soft techniques, such as prevention and rehabilitation through management practices, than on hard methods, such as restoration using direct interventions on soils or vegetation. The latter is often too expensive for widespread implementation. In most cases, the choice between both types of approaches should be based on the assessment of resilience thresholds.

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