5. IMPACTS ON PLANT BIODIVERSITY

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ABSTRACT

The direct impacts of climate change on plant biodiversity will occur through two antagonistic effects: warming, which lengthens the period of activity and increases plant productivity, and the reduction in water availability. The projections of the model Promes indicate the first will prevail in the north of the Iberian Peninsula and in the mountains, while the second will affect mainly to the southern half of Spain. Hence, the “mediterraneisation” of the North of the Peninsula and the “aridification” in the South are the most significant tendencies for the next century. The more severe scenario involves a climatic displacement of almost a whole bioclimatic belt, particularly in the South, whereas in the more moderate scenario the change is equivalent to half of the altitudinal interval of a belt. Such displacements exceed the migration thresholds of most species.

The biggest indirect impacts are those deriving from edaphic changes, changes in the fire regime and the rise in sea level. Interactions with other components of global change (land use changes, modification in atmospheric composition) constitute another important potential source of impacts for which evidence is now beginning to accumulate. The modification of the interactions among species (competence, asynchronies, herbivory, pests, invasions) is also being documented, although the dimensions of these kind of impacts is still uncertain. High-mountain vegetation, forests and deciduous thickets sensitive to summer drought, sclerophyllous and lauroid forests in the South and Southwest of the Peninsula and coastal vegetation are among the most vulnerable types. Structural simplification of plant communities and the predominance of local extinction over re-colonisation constitute recurring tendencies identified for most of the expected impacts. The ecological role of plants as primary producers determines that floristic and vegetational changes have direct or indirect influences on most sectors commented in this Report. The loss of floristic diversity is of particular relevance in Spain, given that our country contains a high proportion of Europe’s plant diversity.

Avoiding losses of biodiversity caused by the impacts of climate change requires global responses. The sectorial strategies designed require a broader geographic framework than that of regional or local administrations, upon which they currently depend. The network of protected areas, conservation strategies, ecological restoration, forest management, the regulation of livestock farming and hunting uses, land planning, environmental evaluation and education constitute the policies most involved in the challenge of providing responses to the impacts of climate change.

Future research on the impacts of climate change on vegetation can be structured in three main and interconnected lines: monitoring of current changes, species and community responses to change, and predictive modelling, which should be based on the information reported by the former and on climate projections, to allow the adoption of mitigation measures for the anticipated impacts. Concerning monitoring activities, support for the participation in phenological networks, including aerobiology, for development of applications based on dendrochronological databases and remote sensing approaches is advisable. It is also needed the implementation of long-term monitoring programs aimed to the quantification of floristic and vegetational changes. These programs could be implemented in the network of protected areas and should be representative of the Spain’s biogeographical and habitat diversity. Threatened species and vegetation dynamics after disturbances are issues to be integrated in these programs. Concerning responses to climate change, the effects of the modification of species interactions, identification of functional groups of plants with similar responses and assessment of indicators of climate change impacts on biodiversity are the main issues for research.

Predictive models of the dynamics of the floristic diversity under the pressures of climate change will be the best tools for generating projections, adopt mitigation measures and evaluate their efficiency. An effort is still needed for computerising the data available on Spain's floristic and vegetational diversity. Models must progressively incorporate resolution at landscape and regional scales, fragmentation effects, dispersal and migration species thresholds, indirect effects of climatic change and interactions with other components of global change.

5.1. INTRODUCTION

The objective of this chapter is to evaluate the impacts of climate change upon the plant components of biodiversity in Spain. The dimensions of these climate modifications are based on the projections appearing in Chapter 1. The concept of biodiversity considered is the most common one, which was proposed in 1992 in the Rio de Janeiro Convention on Biological Diversity: "biological diversity is the variety and variability of all types of living organisms and of the ecological complexes they form part of; it includes the diversity within each species, among species and that of ecosystems" (CBD Secretariat 2001), and redefined by Gitay *et al*. (2002) in terms of "the relative numbers and abundance of genes, species and communities or ecosystems". Knowledge of this variety and variability, even in territories as specific as a country with the dimensions of Spain, is far from satisfactory, and neither is that of their different components and levels of complexity (Noss 1990, Heywood and Baste 1995, Purvis and Hector 2000), at the different spatial scales to be analysed. For this reason, an evaluation of the characteristics required by this Report should focus on those components and levels, the degree of knowledge of which allow for a reasonable prediction of responses to climate change like the ones analysed here. With regard to the plant components of biodiversity, we will comment upon aspects related, on one hand, to vascular flora, paying particular attention to endemic and threatened flora, and on the other, to vegetation as a more complex and integrating level of plant diversity.

Plant diversity in Spain: vascular flora

Spanish plant heritage has a very relevant species diversity in the European and Mediterranean context. Around 80% of the flowering plant species in the European Union are found in our country. This richness is due not only to the geographic dimensions of the territory, but also to a combination of factors that include an intricate orography, the variety of climates and microclimates, the mosaicism of lithologies and soils and geographic location. The strategic position of the Peninsula has historically favoured floristic richness therein, due to the fact that, in the different eras in the past, it has served as a migratory junction in the northward movement of North African flora towards Europe, as the western limit in the spread of flora from Southeast Asia and the eastern Mediterranean, or as the southern refuge, acting as a cul-de-sac, when climate changes associated with glaciation periods forced flora towards the warmer southern climes. The accumulation of floristic contingents of different origin has found a suitable context in which to subsist in the noteworthy spatial and temporal heterogeneity of Mediterranean environments (Blondel and Aronson 1999), which, besides, are characterised by the recurrent disturbances, both natural and anthropic, which have favoured the co-existence of species with differentiated responses (Cowling *et al*. 1996, Lavorel 1999).

In spite of the relevance of Spanish flora compared with the surrounding areas, there is still a lack of accurate figures in relation to the species it consists of. The most recent estimates situate the vascular flora (ferns, gymnosperms and angiosperms) at around 8,000 or 9,000 species. Of these, there are around 2,200 in the Canary Isles (Santos 2001, Izquierdo *et al*. 2001), around 1,500 in the Balearic Isles and over 7,000 in the Spanish Peninsula (Médail and Quézel 1997).

The map in Fig. 5.1 illustrates the patterns of floristic richness in the Iberobalearic territory. It is based on the distributions of 1,400 taxa expressed in 10 x 10 km cells of the UTM grid, constituting between 15-20% of the flora in the territory, estimated at between 7,500 and 8,500 taxa (Castroviejo 1995, 1997). The Iberian mountains are seen to have the greatest level of diversity, which is not surprising if we consider the habitat diversity existing therein and the lesser degree of anthropic transformation (Castro *et al*. 1996). Sierra Nevada is the most noteworthy, with 14% of species mapped, followed by several enclaves in the Pyrenees, the Cantabrian, Iberian and Central mountain ranges, the Catalonia-Valencia coastal mountains and the remaining Betic ranges. The plateaus and the large inland endorheic and river catchments tell another story, presenting areas that have been transformed since ancient times and which sustain a lesser variety of habitats for the flora. The analyses carried out on the regional patterns of floristic richness (Lobo *et al*. 2001, Rey and Scheiner 2002, Pausas *et al*. 2003) show that the main determining factors are those related to environmental heterogeneity (relief, substrates and climates). Plant species richness is also positively related to temperature and to water availability (Lobo *et al*. 2001, Pausas *et al*. 2003).

This image of total floristic diversity becomes modified when its different components are separately examined. Fig. 5.2 shows the richness patterns of ferns, a group of vascular plants suffering from particular conservation problems in the Ibero-balearic territory. In this case, added to the importance of mountain areas, among which we can highlight the Pyrenees, is the oceanity factor, which conditions geographic richness patterns.

Plant diversity in Spain: endemic, rare and threatened vascular flora

Apart from its total floristic richness, Spain has been highlighted as the Mediterranean and European country with the highest number of endemisms (Médail and Quézel 1997, Gómez Campo 2002; table 5.1). There is a direct relationship between the endemicity percentage and the degree to which a determined flora is threatened. Indeed, the levels of endemicity combined with the sensitivity to diversity loss were the criteria used to designate biodiversity *hot-spots* at a worldwide scale (Médail and Quézel 1999, Myers *et al*. 2000), among which the Mediterranean basin or the Canary Isles were included. Although there are Spanish plant endemisms that occupy large areas and are not threatened at all, the high endemic densities present a clear correspondence with those areas richest in threatened flora.

The figures for Iberobalearic endemic vascular plants, including species and subspecies, are at around one thousand five hundred (Pita and Gómez-Campo 1990; Moreno Saiz and Sainz Ollero 1992, Sainz Ollero and Moreno Saiz 2002). Although this number will increase as the drafting of *Flora iberica* progresses, the endemicity percentage of the flora could be expected to remain at around the known value of 15-20%, one of the highest exclusivity figures in the Mediterranean environment (Table 5.1). The Canary isles, with around 550 endemisms (20% of total flora and almost 30% of the native flora), are in a privileged position, due to their historic biogeographic isolation, their little involvement in the pleistocene glaciations, and to an abrupt relief which provides varied habitats and climates.

Analysis of the areas with most stenochorous species once again highlights the grid for Veleta, in the Sierra Nevada, as the one with the highest level of diversity, followed by other from Ibiza, the Sierra de Algeciras mountains, the Segura-Cazorla massif and a whole series of enclaves scattered throughout the Betic ranges (Castro *et al*. 1996).

A total of two thirds of the species included in the Spanish Red Book of vascular flora (Bañares *et al*. 2003) are endemisms (321 over 478; Moreno Saiz *et al*. 2003). The distribution of these indicates the large amount of stenochoric species existing on the islands or in the mountain areas. The map in Fig. 5.3 shows the 10 x 10 km cells of the UTM grid in which populations of some of the taxa included in the Red Book have been detected. Of approximately 5,600 grids of this size covering Spain, a total of 727 contain at least one of the taxa dealt with. The distribution of these shows clear similarities with the provincial map of the most threatened species presented on the Red List (Aizpuru *et al*. 2000).

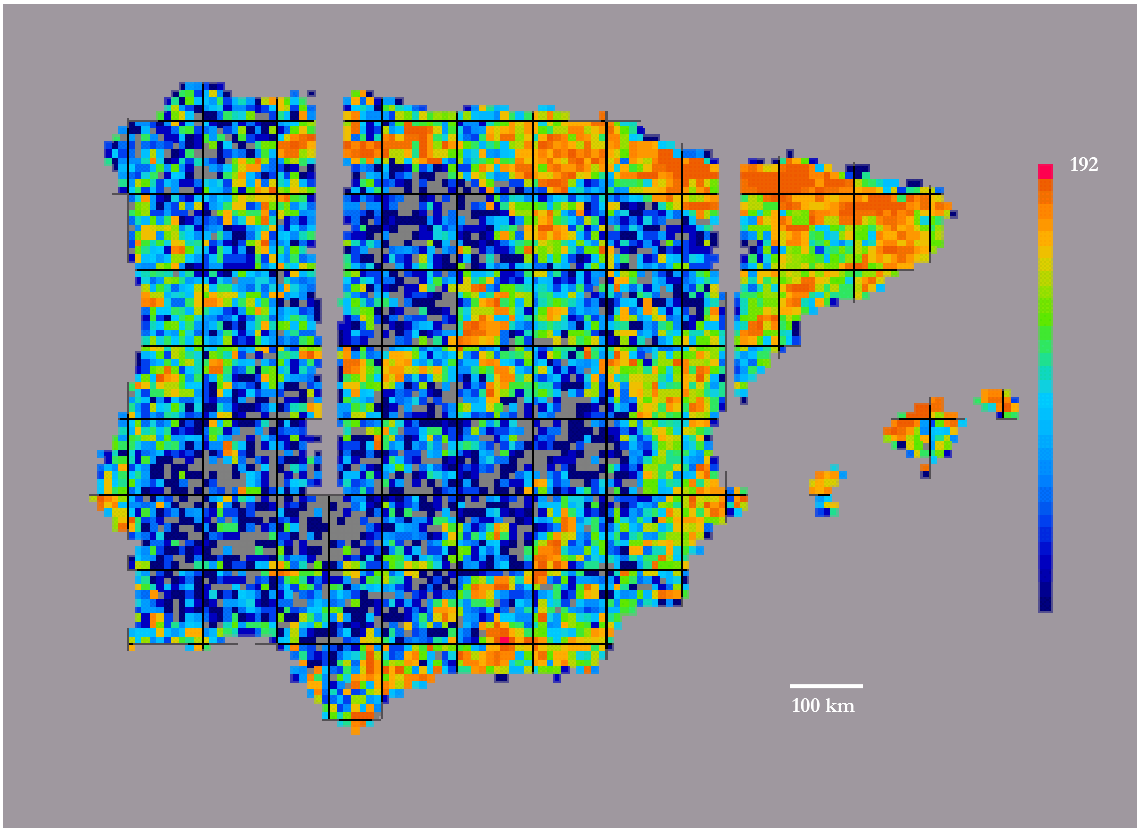

Table 5.1. *Number of species of vascular plants and endemisms in Spain and neighbouring countries (modified from Médail* and *Quézel 1999)*

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Country	***Number of species***	***Number of endemisms***
Ex–Yugoslavia	5000	320
Algeria	3150	320
Spain	8000-9000	1300-1500
France	5000-6000	900
Greece	5500	1100
British Isles	1443	17
Italy	5600	730
Morocco	4200	900
Portugal	2600	114
Tunisia	1800	40

With regard to floristic originality, the Canary Isles have been considered as a biodiversity "*mini-hotspot*" (Mittermeier *et al*. 1999), with endemicity percentages at around 30% if we consider only the native flora (Santos 2001; Machado 2002) and the vast majority (58%) of endemisms restricted to one single island (Humphries 1979, Izquierdo *et al*. 2001). The map of the 171 species studied on these islands shows that there is practically no grid on the Canaries without at least one endangered plant species (Moreno Saiz *et al*. 2003).

The Balearic Isles also has endangered and critically endangered species on the five biggest islands, although with a lower occupation density. The Pythiusian islands (Ibiza and Formentera) are relatively well represented, as are the Serra de Tramuntana mountains and certain points of the coast of Majorca. This pattern coincides with those identified in recent studies of the Iberobalearic flora threatened or protected by the Habitats Directive (Domínguez Lozano *et al*. 1996, Sáez and Roselló 2001, Bañares *et al*. 2003). Clear evidence of the importance of the islands in general and of the risks their flora is faced with, is what we can see on the map 5.1.3, in spite of the small area, the Columbretes Isles, Alborán Island and the National Park of the Atlantic Islands of Galicia, all of which contain endangered species.

With regard to the Spanish Peninsula, Fig. 5.3 highlights the importance of the Betic ranges and the Levant for threatened flora. Not only do the Betic ranges have the highest percentage of peninsular endemicity, but also the highest levels of stenochory, with numerous differential species among their main massifs (Sainz Ollero and Hernández Bermejo 1985, Castro *et al*. 1996, Blanca *et al*. 1998, 1999). There is a notable number of grids with threatened flora on the coast and in the mountains of Galicia, the density of which had not been estimated through previous analyses (Moreno Saiz and Sainz Ollero 1992), and which has not been proportionally reflected in Spanish or European conservation legislation (Domínguez Lozano *et al*. 1996). The other mainland Euro-Siberian territories are less clear, or are represented only by vulnerable taxa, as occurs in some sections of the Cantabrian and Pyrenees ranges.

The central Iberian plateaus constitute the areas with the lowest number of threatened plants. This is very much accounted for by the fact that they have been altered since ancient times, and have therefore lost their naturality, but also partly because these are areas with low endemicity levels. (Moreno Saiz and Saiz Ollero 1993). Together with the extraordinary accumulation in the Canaries, the grids show a considerable and noteworthy dispersion of endangered species in many parts of the country. Up to 80 taxa of the Red Book are present in one single 1 km^2 UTM grid throughout Spain; for many of them, this constitutes their whole worldwide distribution, given that they are endemic. Only 138 taxa have a distribution area divided among ten or more grids of these dimensions.

Ibero-balearic endemisms as a whole are more related to intermediate successional stages, and to the particular substrates and biotopes typical of mountain reliefs, than with late-successional communities (Montserrat and Villar 1973, Gómez Campo *et al*. 1984, 1987; Laguna 1999, Melendo *et al*. 2003).

Plant diversity in Spain: vegetation types

Spain's vegetation is also among the most varied on the European continent. Around 90% of the vegetation classes included in European classifications currently in use are represented in Spain (Mucina 1997, Rivas-Martínez *et al*. 2001, 2002). Recent compilations recognise over 2,500 plant associations on the Iberian Peninsula and its archipelagos, the diversification of which allows for the evaluation of the internal diversity of the territories and the geographic differentiation of the respective floristic ensembles.

Among the main patterns recognised in the distribution of most types of peninsular terrestrial vegetation, in first place appears the North-South summer drought gradient, which marks the separation between Euro-Siberian vegetation, which requires summertime water availability, and Mediterranean vegetation. The North and Northwest of the Peninsula, including the Pyrenees and Cantabrian ranges and Galicia, are currently dominated by the former, which spreads, in a fragmented way, throughout the mountain ranges of the North half of the Peninsula. Five sixths of Spain's territory is biogeographically included in the Mediterranean region. A second determinant factor for the vegetation is lithology, which separates the siliceous substrates predominant in the West from the base-rich soils existing in most of the Levant and the South of the Peninsula (Loidi 1999). The occurrence of particular, minority substrates (gypsum, dolomites, ultrabasic rocks, etc.) is among the most important factors promoting the endemicity of the flora (Gómez-Campo *et al*. 1984, Gómez Campo 1985, Heywood 1995). The complex relief, with the associated altitudinal climatic gradients and barrier effects, introduces another level of biogeographic differentiation, in which the transversal orientation of the main mountain ranges must be highlighted. Although these may have hindered meridian migration, they have also favoured accumulation of species along ecotones and biogeographic borderlines. Among the main effects caused by relief are the two main peninsular semi-arid areas, the Ebro basin and the Southeast of the Peninsula; the latter contains a noteworthy plant richness of Iberian-North African range (Alcaraz *et al*. 1991). Lastly, the insularity reaches its maximum expression in the originality of the flora and vegetation on the Canary Isles; this includes several classes of vegetation which is either endemic or is shared with the neighbouring archipelagos (Rivas-Martínez *et al*. 1993). The biogeographic sector delimitation of Spain's territories adjusts to these factors (Rivas-Martínez and Loidi 1999a, b; Rivas-Martínez *et al*. 2002).

The main structural types of vegetation on the Peninsula will be summarised in section 5.3.3, along with an evaluation of the corresponding impacts of climate change. Riparian vegetation and wetlands are dealt with in Chapter 4 of the Report.

Recent impacts on plant diversity

At present, Spain's vegetation, except in a very limited number of small enclaves, is, like in most of Europe, the result of a long history of human intervention which has been partially superimposed over climate changes in the recent past, of which the most recent episode is the Little Ice Age (LIA). Among these interventions are extensive deforestation processes, either mass or selective ones, which, together with the extraction of firewood or timber, reached their height at the beginning of the last century, along with the establishment of agricultural or livestock farming uses and the associated transformations. The composition of landscapes shaped by these land uses, interfering with a complex vegetation which only partially adjusted to the latest climate changes, must surely have suppressed those less tolerant components of diversity, but must also have contributed to the joint maintenance of other components that now characterise much of the biodiversity of European countries in general and the Mediterranean ones in particular.

Over the last few decades, other impacts on vegetation have been recorded, which are part of the so-called global change and which are actively contributing to the deterioration of biodiversity. With regard to Spain, on one hand we can highlight changes in land uses, partly caused by rural abandonment, and on the other, the intensification of farming systems. The abandonment of the agriculturally less productive land (almost 2% in the last decade, a sixth of which has been changed for urban construction and infrastructures), together with the reduction or abandonment of extensive livestock farming, and firewood and charcoal extraction, have favoured the spread of shrublands and semi-natural pastures, and the growth of forest, all of which has also been promoted by reforestation policies (over 4 Mha in the last 40 years). The consequent accumulation of fuel in woodlands has led to the occurrence and spread of fires, as is described in Chapter 12. In turn, intensified uses affect biological diversity through promoting processes of eutrophication, pollution, erosion and desertification, the reduction and fragmentation of certain habitats in some cases and the homogenisation of landscapes in others, etc. Furthermore, anthropic transformation and the development of trade and transport are believed to be favouring the introduction of exotic species, thus providing opportunities for biological invasions. This concurrence of factors has not yet led to the extinction of vascular flora to such a considerable extent as in other biological groups, but it does probably contribute to increasing the proportions of flora threatened (Greuter 1991, 1994).

In a parallel fashion, in the last few decades, there has been an increase in the number of protected natural areas, which currently represent around 6% of the national territory (Gómez-Limón 2000, Reyero 2002), and this area will rise to 23% with the Special Conservation Areas established within the framework of the Habitats Directive (Orella 1999, Morillo and Gómez-Campo 2000). The geographic distribution of this future network of protected areas (Orella 1999) is concentrated in the mountain areas, leaving big gaps on inland plains and depressions, and in many coastal areas; the imbalances among territories are equally notorious.

The coincidence in time of the changes in land uses with the first symptoms of climate change makes it particularly difficult to delimit the corresponding impacts. The changes in uses and especially abandonment, favour in some cases the spread of more competitive types of vegetation, and in others, there is evidence of adjustments by the vegetation to recent climate change, which could not be expressed while the pressure of the respective land use was maintained. In the case of flora and vegetation, we must also consider the superposition of other effects deriving from changes in atmospheric composition, which have a direct influence on plants, such as increases in CO_2, tropospheric ozone, aerosols, atmospheric deposition of nitrogen oxides, etc.

In the drafting of the Atlas and Red Book of Threatened flora (Bañares *et al*. 2003), an individualised evaluation was made of the risk factors for each of the 2,223 populations studied, which allowed for analysis of the threats recognised for the most vulnerable component of the flora. The results obtained from the field surveyers are synthesised in Table 5.2. Although the level of detail is now greater (populations instead of taxa), the most commonly indicated risk factors generally coincide with the previously described ones (Gómez Campo 1996; Domínguez Lozano *et al*. 1996). Whether it be a verified threat or a potential risk, overgrazing is the most quoted factor (almost 40% of the populations), closely followed by human transformation of the land, traditional gathering and plant collecting, or competitive displacement by other plant species. Likewise, results indicate that the reduction, fragmentation and degradation of natural and semi-natural habitats constitute the main risks for species persistence. What is noteworthy, however, is that no consideration has been given to serious threats such as competition by introduced exotic plants or climate change. The latter, however, has been identified as a threat in Sierra Nevada (Blanca *et al*. 2002) and in the Canary Isles (Marrero *et al*. 2003).

With regard to the potential risks dealt with in the Red Book, the table shows a high level of coincidence on indicating many of the disturbances associated with climate change or susceptible to being exacerbated by it (Houghton *et al*. 2001): droughts, fires, storms, floods and other geological risks. Although these disturbances are inherent to most of the ecosystems comprising the endemisms and threatened flora, their presumable greater incidence in the future under the available scenarios of climate change constitutes a certain risk factor for floristic diversity.

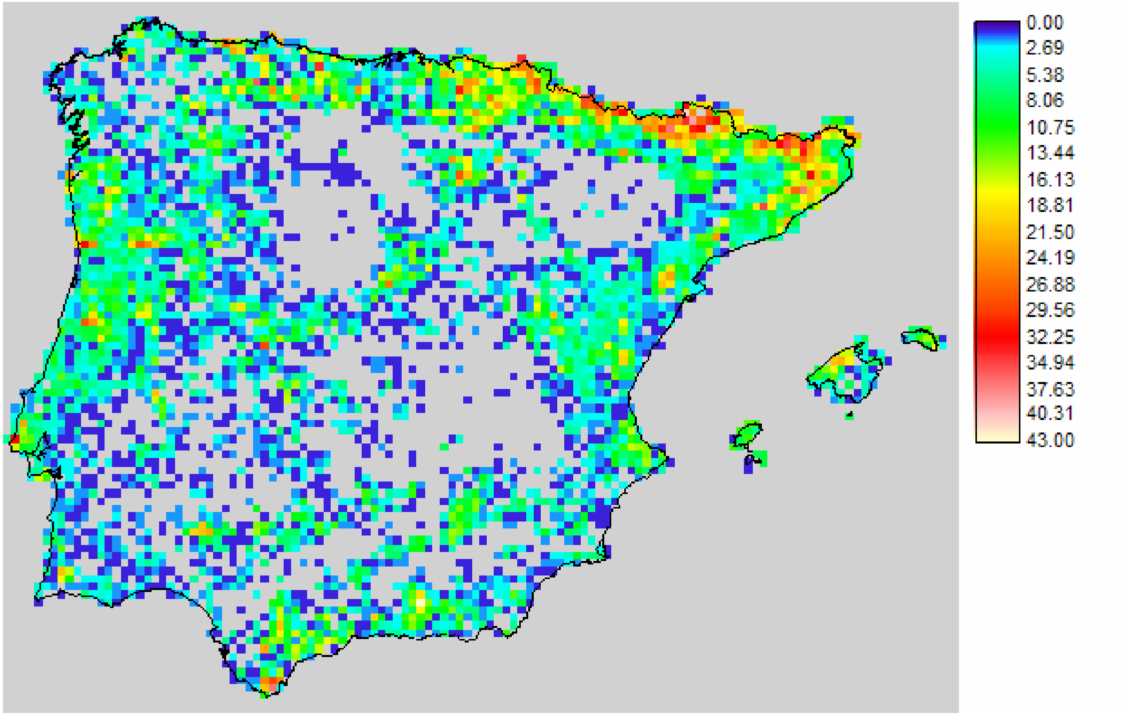


Fig. 5.2. *Patterns of floristic richness of ferns in 10 x 10 km grids for the Iberian Peninsula (Moreno Saiz* and *Lobo in prep.).*

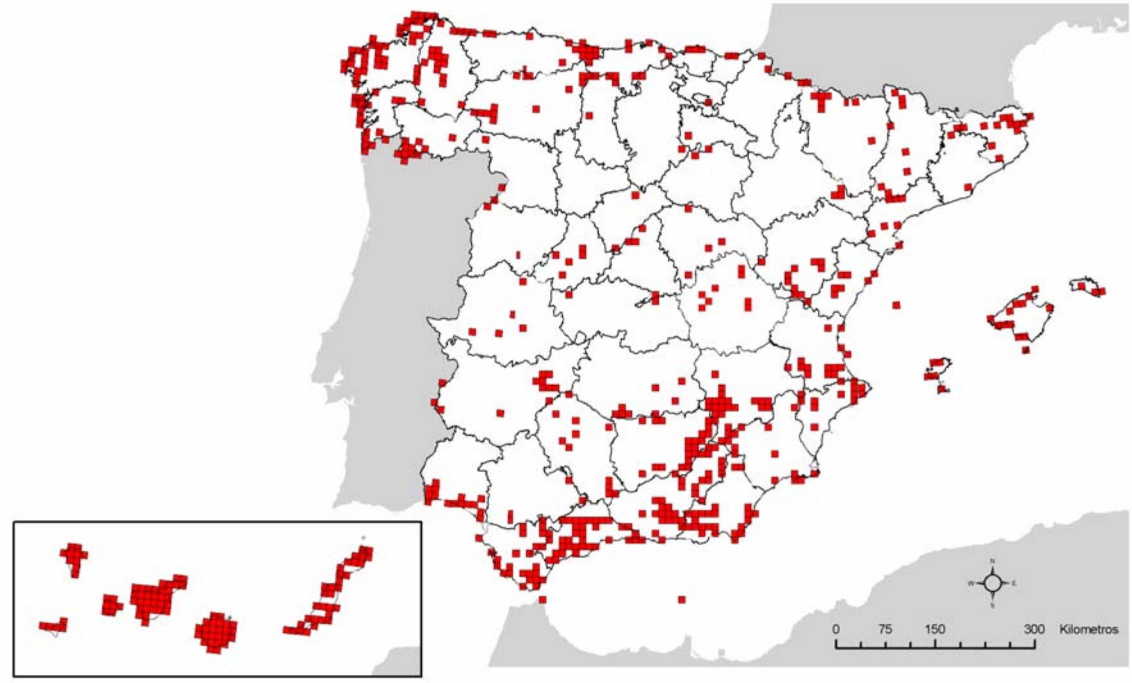


Fig. 5.3. *UTM 10x10 km grids in which populations of threatened species have been detected.*

Table 5.2. *Most mentioned potential threats and risks for the threatened vascular flora included in the Spanish Red Book, expressed as the proportion of the total number of populations sampled (2223) of the 478 species considered (Moreno Saiz et al. 2003).*

Threats	Number of populations affected	% of the total of populations studied (2223)
Threats resulting from human action		
Shepherding	851	38
Trampling and *artificialisation*	656	30
Traditional collecting or gathering	426	19
Threats of biotic origin		
Natural plant competition	493	22
Depredation	475	21
Little ecological plasticity	441	20
Poor reproductive strategy	224	10
Threats resulting from the type of development		
Improvement of roads network	527	24
Construction of new roads	324	15
Housing development	210	9
Indirect threats		
Improvement of access by vehicles	398	18
Improvement of access by the population on foot	328	15
Improvement of access to neighbouring lands	239	11
Potential accidents due to:		
Droughts	1192	54
Rockfalls	995	45
Fires	806	36
Storms	696	31
Floods	499	22
Landslides	475	21
Volcanic activity	448	20
Avalanches	426	19

5.2. SENSITIVITY TO THE PRESENT CLIMATE

5.2.1. Current relationships between climate and vegetation

The relationships between vegetation structure, primary productivity and plant distribution, and climatic elements have constituted one of the classical themes of geobotanical literature (Woodward 1987, Blasi 1996, Fernández-González 1997). Numerous authors have proposed climate classifications that attempt to fit the former to values of the latter, which have generally been derived from standardised weather records. Almost all these classifications coincide in the importance of temperature variables (mean or minimum temperatures, positive temperature or temperature sums, duration of the period of vegetative activity, etc.) on one hand, and variables related to annual or seasonal water availability (precipitation, water balance, ombrothermal indices) on the other, on climatically fitting the distribution of plants or vegetation types. Multivariate analyses of these relationships reflect similar results (Moreno *et al*. 1990, Gavilán and Fernández-González 1997, Gavilán *et al*. 1998). Although the fits obtained are mainly correlational, general models have been designed which are capable of predicting at a global level the distribution of the main physiognomic vegetation types (Box 1981, 1996; Woodward 1987, 1992; Prentice *et al*. 1992, Haxeltine *et al*. 1996), which prove the existence of causal relationships, among which the most obvious ones derive from the tolerance of plants to extreme temperatures, minimum duration of the vegetative period, or the fit of the life cycle and dependence on productivity in relation to water and energy availability.

In Spain, one of the most detailed climatic classifications available, especially with regard to its relationship with vegetation types, is the one developed by Rivas-Martínez and collaborators (1997, 1999, 2002, www.globalbioclimatics.org). This classification makes use of different summertime ombrothermal indices to separate Mediterranean climates from the temperate ones (without summertime water deficit). The bioclimatic belts corresponding to each type of macroclimate are defined with the use of positive temperature summations (thermotypes) and annual ombrothermal ratios (ombric types). The units of this classification will be applied in the following section (5.3.2 and 5.3.3) to the evaluation of the projections of climate change.

Whereas the classifications mentioned are based on climatic regularities, irregularities or interannual climatic fluctuations (Rodó and Comín 2001), merit special attention because of their effects upon reproduction (Herrera 1998), recruitment rates and plant mortality (Moreno *et al*. 1999, Quintana *et al*. 2004), in the disturbance regimes (Chapter 12) and in the dynamics of community diversity (Figueroa and Davy 1991, Peco *et al*. 1998). This type of effects, documented in different types of habitats and species, although with much less detail, are superimposed over average climate tendencies, and could interfere with these or even camouflage them (Hulme *et al*. 1999).

5.2.2. Past climate changes and changes in vegetation

Since the appearance of Mediterranean type climates, between 3,2 and 2,3 MaBP ago (Suc 1984), climate change have become the norm during the Pleistocene, which, during the two million years it lasted registered between four and six main glaciation cycles, interrupted by several other interglacial periods, which were warmer, generally more humid, and, as a whole, over ten times shorter than the former. It has been estimated that, during the glacial periods, global temperature fell by between 6 and 8 ºC, more sharply in the boreal and polar latitudes than in the tropical ones. These periods, however, appear to have been characterised by great climatic irregularity. Apart from the development of big icecaps at high and medium latitudes, the sea level fell (180 m in the Mediterranean at the height of the last glaciation, around 18,000 years ago), as did global precipitation, due to the decrease in air humidity, although the evidence in this respect is less clear, and there may have been considerable regional variation.

The effects of glaciation periods on the flora and vegetation were seen in migration towards lower latitudes and local or regional extinctions. During the interglacial periods, the flora, quartered in southern refuges, advanced latitudinally, at different speeds according to species and periods, of between, for holarctic trees, 5-50 km/century for most zoochorous species and up to (10)50-200 km/century for the anemochorous ones (Huntley and Birks 1983, Huntley 1991). *Pinus, Corylus* (150 km/century) and *Alnus* (up to 200 km/century) are among the most rapid genera. Vegetation re-composition had different dimensions according to latitude, as in the boreal and temperate northern territories, re-colonisation occurred following the withdrawal of the ice, whereas in medium temperate latitudes, there was a predominance of displacements (latitudinal and in many cases also longitudinal), and in the southern temperate latitudes there were processes of accumulation and species packing, as well as altitudinal movements of the flora, favoured by the rejuvenation of the relief of these territories during the alpine orogeny. The extinction mainly affected the flora of tropical affinities that had predominated in medium latitudes until the end of the Tertiary. Following the first two glaciations, practically all this contingent had become extinct in Europe, where the transversal orientation of the main mountain ranges had hindered meridian migration more than on other continents. Little is known of the changes that took place in the previous interglacial periods, but in some of these, the Mediterranean pulses appeared to have had a great influence (Pons and Quézel 1985). There is evidence that during the previous interglacial period, temperatures were reached of up to 4 ºC higher than at present.Although the Quaternary did not last long in evolutionary terms, speciation processes must have played an important role, and were promoted, apart from climate variations, by the fragmentation of the species' refuge areas, the orographic diversity and lithology of these, and processes of poliploidy, which operated effectively in different groups of angiosperms during interglacial re-colonisation. With regard to the Mediterranean basin, speciation was much more active among the "modern" Mediterranean element than among the "pre-Mediterranean" one (see 5.3.3.3). Thus, the glacial periods were more effective with regard to determining extinction, and those that provided the necessary evolutionary time. During the interglacial periods, there were mainly processes of floristic redistribution, in which coasts, valleys and mountains played an important role as refuges and migration routes.

Since the height of the last glaciation, around 18,000 years ago, the climate has become progressively more temperate, but in a non-uniform fashion both in the temporal and spatial sense. In the initial phases (14,000-10,000 years BP) acute warming episodes took place, during some of which the big spread of sclerophyllous vegetation can be appreciated (Pons and Reille 1988, Burjachs and Juliá 1994), which was interrupted by sudden cooling periods such as the old and recent Dryas. The final deglaciation began immediately afterwards, through the Holocene, with moderate temperature changes, but big changes in rainfall (Pérez-Obiol and Julià 1994, Burjachs *et al*. 1997, Jalut *et al*. 1997, Jalut *et al*. 2000, Davis *et al*. 2003). During the first quarter of the Holocene, there was progressive warming, but within a framework of moderate rainfall, at least in the western Mediterranean. In the second quarter (approx. 7,500-5,000 years BP) there was a considerable increase in rainfall, while temperatures could have been up to 2ºC higher than at present during the warmer phases. In the third quarter, up to 2,500-2,000 years BP, rainfall decreased, especially in the southern and eastern half of the Peninsula, and in North Africa (Cheddadi *et al*. 1998); to the contrary, in Europe and the North of the Peninsula, a moderate cooling period was only detected. From half way through the Holocene, in particular in the coastal regions, the first impacts upon vegetation by Neolithic humans can be noted, which will also become generalised inland during the last two thousand years, while the aforementioned tendencies continue with lesser oscillations, such as the medieval cold and warm episodes or the Little Ice Age.

The changes in vegetation, documented mainly with the use of paleopalynological records, have responded with diverse phase differences among territories to these climatic changes, which were probably not uniform. In Europe, the first post-glacial warming periods caused the displacement of the tundra regions, theretofore dominant, northwards and towards the high mountain, and the spread, first of pines and birches, and then of hazel and other deciduous fagaceae. Similar tendencies have been recorded in the North of the Peninsula (Peñalba 1994, Allen *et al*. 1996); to the contrary, in the western Mediterranean, the steppes and open conifer forests (*Pinus* and *Juniperus*) that dominated the lateglacial landscapes (Pons and Reille 1988, Pérez-Obiol and Julià 1994) remained relatively unaffected during the first quarter of the Holocene. Only then did progressive migrations begin, of both deciduous and sclerophyllous elements, slower and later in the South, until, with increased rainfall, there was a big spread of the different deciduous forest formations. Mesophytic forests began to withdraw in favour of sclerophyllous, and in general xerophytic, vegetation half way through the Holocene, when, especially in the South, aridification was accentuated and the present Mediterranean climate became established in the basin. In the North of the Peninsula, however, there was little or no

Mediterraneisation, given that the spread of the beech was prolonged up to the last quarter of the Holocene (Peñalba 1994).

Paleopalynological analyses only reveal a small fraction of the species involved in the changes, but they show that the responses by the vegetation to climate change have relatively slow phases, which can be prolonged for several centuries, and in which the migrations take place and mechanisms for resisting the change are activated by the pre-existing vegetation. But there are also alternating phases of changes in dominance that can be very rapid, especially when synergies occur with certain disturbances or other causes that can break the resistance of the theretofore dominant species. For example, changes in the fire regime associated with intensified aridity and with anthropic intervention in some cases, have accelerated the substitution of deciduous forests by sclerophyllous vegetation or conifer forests, or the alternation among the latter two in different parts of the South and East of the Peninsula (Riera and Esteban 1997, Carrión *et al*. 2001, 2003). These rapid changes in dominance can take place in less than one century, once the immigration has occurred. Equally rapid is the spread of the herbaceous and woody vegetation associated with anthropic changes in the territory as a result of livestock farming or agricultural uses, which have been superimposed over the environmental changes of the last few millennia, thus complicating the interpretation thereof (Pons and Quézel 1985, Reille and Pons 1992, Carrión *et al*. 2001). The migration processes probably were more limited in the Mediterranean territories than in more northern latitudes (Huntley 1991), due to the abundance of refuges and to physiographic obstacles. For these reasons, and within the framework of the individualistic responses by the species to climate changes (Moore 1990, Graham and Grimm 1990, Huntley 1991), there has been a redistribution of the flora that has determined the current rigid geographic compartmentation observable in many types of plant communities.

5.3. FORESEEABLE IMPACT OF CLIMATE CHANGE

5.3.1. Types of foreseeable impacts upon flora and vegetation

As was documented in Chapter 1, meteorological records from the last few decades are beginning to show clear signs of climate change in Spain, although these are still difficult to resolve with the necessary spatial and temporal clarity. The thermic signal is clearer, with an estimated increase for the last third of the century of between 0.3 and 0.6ºC per decade, with notable differences between territories, seasons of the year and maximum and minimum temperatures. The signs of pluviometric changes are less evident due to their complex patterns of spatial and temporal variability (inter and intra-annual). There appears to be evidence of reduced rainfall throughout most of the Peninsula, except in the North and Northwest, where winter rainfall might also have decreased. It cannot be said, however, that interannual variability is globally increasing, although symptoms have been detected both of greater pluviometric variability in the South and Southeast and a higher seasonal concentration of rainfall. The analysis of dendrochronological series suggests an increase of thermal and pluviometric variability during the last century (Manrique and Cancio 2000). Local reductions of snowfall and of snow cover have also been indicated. As a whole, this weather signal can be seen in an average warming rate of between 1-1.5ºC, equivalent to an altitudinal displacement of around 200 m in the limits of many species.

5.3.1.1. Direct impacts

Projections of climate change for the end of the century in south-western Europe, documented in Chapter 1, indicate a generalised temperature increase, with certain significant seasonal differences; reduced water availability, caused by warming and by the reduction and seasonal redistribution of rainfall; and increased climatic variability, above all in relation to temperature regime and to extreme heat events, although this is not so evident in the rainfall regime.

The direct consequences of temperature increase for plants will initially involve an extension of the vegetative activity period, on reducing the restrictions caused by low temperatures. In the absence of hydric limitations, biological activity will increase, which will lead to increased potential productivity. The adjustment of the phases of the plants' development cycle to the new temperature regime will bring about phenological changes, both in wild and agricultural species, and those that are to be most expected hereinafter will refer to foliation, flowering and fruiting dates, or delays in leaf abscission in deciduous species in winter. We could, however, expect different responses according to species and processes, given that the degree of thermoperiodic control compared with other types of control (photoperiod, hydroperiod) is variable, as well as the phenotypic plasticity (Kramer 1995). Thus, not all species will be able to make use of the extension of the vegetative period in the same way (Körner 1994, 1995), which would involve modifications in competitive relations. To the contrary, this phenological advance could give rise to a greater risk of exposure to possible late frosts, which would only have an impact in mountain areas, given that the general occurrence of this phenomenon would be reduced. Furthermore, warming provides the possibility, for the species that are capable of capitalising it, of entering territories from which they had previously been excluded because of temperature conditions. This latitudinal or altitudinal spread will become more effective if the new habitats are not already occupied by other species, or if they are occupied by competitively inferior species. For certain plants, however, especially cold climate ones (high latitudes or altitudes), warming could cause the inhibition of the life cycle phases induced by low temperatures (Körner 1995, Larcher 1995), thus affecting the viability of the populations involved, which might be subjected to serious retreat. Evidence of phenological changes caused by climate change will be commented upon in section 5.8.

The direct consequences of a reduction of water availability are partially opposed to the previous ones, given that they involve reduced potential productivity, which corroborates climate change in the same projections (Chapter 1). The parallel temperature increase will increase net primary productivity even more, on increasing respiration rates. This tendency will favour drought-tolerant plants or those that avoid it, compared with other, potentially more productive ones, and, in extreme cases, it will also favour the structural simplification of the vegetation through reduced height of plant cover. Unlike warming, reduced hydric availability is much more effective with regard to excluding less tolerant species, through impacts on reproduction and recruitment rates, lesions in adult specimens or debilitation and susceptibility to predators and pests. Indeed, in the distribution of Mediterranean flora and vegetation, hydric relations often show greater discriminant power than the purely thermic ones (Gavilán 1994, Gavilán *et al.* 1998).

An increase in climatic variability and in the incidence of extreme events is one of the aspects of climate change that involves the highest degree of uncertainty. If, as the projections appear to indicate, the main tendency involves more frequent late spring heat waves, then the effects of these could exclude the less drought-tolerant species.

In relation to preceding and documented climate changes, including the sharp oscillations of the Alleröd and the last Dryas, the one that is currently occurring is of an extraordinarily abrupt nature. The response capacity of species through micro-evolutive processes will be very limited (Bradshaw and McNeilly 1991), although in certain relatively short-lived genera of plants which have shown great capacity for recent evolutive radiation, responses of this type could be detected. The magnitude of the changes projected indicates that the limits of phenological plasticity of much of the flora will be surpassed in many territories (Holt 1990); furthermore, this plasticity could be restricted in unfavourable environments, such as the Mediterranean and high-mountain ones (Valladares 2001). For the more long-lived plants, apart from the impossibility of producing any micro-evolutive response, we must also consider the difficulty involved in producing acclimatisation responses, at least in adult specimens. Thus, latitudinal and altitudinal displacements will constitute a fundamental response in the readjustment of flora to new climatic conditions, and could attenuate the proportions of foreseeable extinction (Bakennes *et al*. 2002, Thomas *et al*. 2004).

The capacity to migrate under the pressure of a climate change is documented in the Pleistocene for many species (see 5.2.2), but only the fastest ones will be able to deal with the scope of future changes in this way, and only after a variable delay period. Latitudinal displacements will be easier for short-lived species, with good dispersal skills and few demands in relation to appropriate habitat. The latter condition will be a serious limiting factor for many species in current, profoundly transformed and fragmented landscapes (Pitelka *et al*. 1997). Altitudinal migration requires movement of less range, but involves reductions of the area available for those species whose climatic space rises in altitude. There is much evidence of altitudinal and latitudinal movement of flora attributable to climate change (Hughes 2000, McCarthy 2001, Walther *et al*. 2002, Parmesan and Yohe 2003), including lichens (van Hark *et al*. 2002). Whatever the case may be, these movements will involve a radical re-composition of communities. Apart from their capacity for dispersal and their plasticity, other factors that will favour the survival of species in relation to climate change are related to their altitudinal, geographic and habitat amplitude and to the genetic diversity of their populations. Prolonged *in situ* persistence could be viable in certain long-lived species able to take advantage of the temporal windows of opportunity favourable for reproduction and recruitment, or characterised by metapopulation or source-sink dynamics allowing the periodic recolonisation of available habitats (Eriksson 1996).

5.3.1.2. Indirect impacts

Apart from these direct effects, climate change will also have an influence on other factors which in turn will affect vegetation. Among these indirect effects are those involving soil as a physical and nutritional support for plants, those related to the disturbance regime (Pickett and White 1985) and the ones associated with a rise in sea level.

Soil organic matter content is a key factor with regard to the edaphic availability of water and nutrients. The available projections of climate change indicate a generalised reduction of the organic carbon content in soils, as a consequence of reduced rainfall combined with warming. This tendency could become synergically accelerated through interaction with the accentuation of the fire regime, caused by the new climatic conditions, and through erosion, caused both by disturbances and by the reduction of soil organic matter. In the short term, mineralisation of organic matter could rapidly affect certain dependent systems, such as bogs, which would be easily displaced by other types of more productive vegetation. Increasing aridity of the climate and loss of edaphic organic matter would also promote desertification processes, which at present seriously affect one third of Spain’s territory, especially in the South and the East. Lastly, soil salinisation can also occur rapidly, especially on irrigated agricultural land. Increased salt content in surface horizons constitutes a very selective factor for the flora, which responds rapidly in terms of impoverishment because of the low number of species capable of tolerating these conditions. In the areas affected by aridification, the reduction or even the inversion of leaching and the consequent pH elevation could favour the expansion of basophile or indifferent plants, to the detriment of the acidophile flora.

With regard to the disturbance regime, the possibility of a greater frequency of heat waves has already been mentioned. In relation to our latitudes, one of the consequences will be the changes in the fire regime commented upon in Chapter 12. Greater incidence of fires could accelerate replacements among species or vegetation types with different degrees of adaptation to the new climatic conditions, as these will govern the dynamics of post-fire regeneration (Quintana *et al*. 2004, Rodrigo *et al*. 2004). Big or recurring fires could contribute to making the resulting landscapes more homogeneous (Pérez *et al*. 2003), and to initiating degradative dynamics that accelerate the structural simplification of the vegetation, or increasing erosion, which in turn impoverishes seed banks (García-Fayos *et al*. 1995) and soil nutrient availability.

Another consequence of warming will be a rise in sea level, caused by the thermal expansion of the ocean and by the fusion of icecaps and glaciers. Changes in the coastline, in the geomorphological processes of this, in the water table and the associated coastal systems (coastal pools, dunes) will affect the flora and the plant communities depending on them.

5.3.1.3. Interactions with other components of global change

One of the greatest sources of uncertainty in relation to the effects of climate change involves possible synergies and antagonisms with other components of global change, and in particular, with regard to plants, with changes in atmospheric composition and with changes in land use (Moreno and Oechel 1992, Moreno 1994, Moreno and Fellous 1997, Lavorel *et al*. 1998), which should play a modulating role with regard to the direct and indirect effects of strictly climatic changes (Peñuelas 1996, 2001).

Changes in the atmosphere are the main cause of climate change, but these also have independent effects on the functioning of plants. Atmospheric enrichment in CO_2 and the deposition of nitrogenated compounds have a fertilising effect that increases photosynthesis and biological activity in general (Woodward *et al*. 1991, Ceulemans and Mousseau 1994, Strain and Thomas 1995, Körner 2000). The former also improves the efficiency of the plants with regard to water and nitrogen consumption (Peñuelas and Matamala 1990, Peñuelas and Azcón-Bieto 1992), which could attenuate the impacts of aridification, especially from the agricultural perspective (Hulme *et al*. 1999). It has also been shown that it favours advances in flowering and increases flower and fruit production (Peñuelas 2001). Although they were tested in experimental chambers, these effects appear to have limited temporal efficiency due to acclimatisation, to the attenuated responses by adult plants and to the limiting role of other factors, such as water availability (Reichstein *et al*. 2002); their magnitude is diluted on magnifying the temporal and spatial measurement scales, and it is possible that they may have started to enter a stabilisation phase (Peñuelas 2001). Nitrogen deposition is lower in the Mediterranean than in other temperate latitudes, but it could hare a more important effect on plant growth than CO_2. As with the previous case, effects differ depending on species. An example of the antagonisms involved in global change is that whereas CO_2 enriched atmosphere will favour C_3 plants, warming or aridity will favour the C_4 ones, which function better in conditions of high irradiation, high temperatures and low hydric availability (including saline soils).

Warming and increasing concentrations of CO_2 are also known to favour emissions of volatile organic compounds (VOCs) by the vegetation. VOCs have different functions in plants: defence against predators and against certain effects of climate change, interactions with other plants, etc. (Peñuelas *et al*. 1995). They have an important influence in atmospheric balances of C (Peñuelas 2001), especially in areas with Mediterranean-type vegetation. They also show several types of interaction with greenhouse gasses and are, together with N oxides and hydrocarbons of anthropic emission, among the precursors of formation of tropospheric ozone, which is favoured by the high temperatures and irradiation typical of the Mediterranean climate. Ozone has negative oxidising effects for organisms, which are accentuated in environments with high levels of atmospheric humidity, such as coastal ones, and in turn, it stimulates the defensive emission of VOCs by the plants. The final balance of this synergy is unclear, because the emission of VOCs could be limited in conditions of water stress, and its effects would at least amplify some of the aforementioned direct impacts, and would also influence the balance of interactions among species.

The increased UV radiation associated with atmospheric changes is considered to be a less important factor in Mediterranean areas, because the flora therein is endowed with great physiological protection against over-radiation. Possible effects, however, have been indicated in relation to the chemical composition of the phytomass (Gehrke *et al*. 1995) and to edaphic micro-organisms (Caldwell *et al*. 1999).

Changes in land use, with regard both to type and intensity, are considered to be the factor with the greatest current impact upon biodiversity in general and that of Mediterranean ecosystems in particular (Sala *et al*. 2000). This circumstance, which greatly hinders the detection and interpretation of effects strictly attributable to climate change, will bring about new modifications in distributions and types of uses (Parry 1992), with tendencies towards the abandonment of agriculture and livestock farming in the territories in which climatic conditions become adverse, the displacement of these uses to new territories with more favourable climatic conditions, an increase in intensive use modalities as a way of attenuating climate changes, changes in forestry uses, crises in certain traditional land use systems, new locations of urban housing and infrastructures, etc. The distribution of uses determines other factors, such as the fragmentation of landscapes, which will condition and limit the migration possibilities of species. Furthermore, the characteristics of the fire regime are closely related to the composition of the landscape and of fuels, and therefore to land uses. The synergic effects of climate change and of changes in uses will constitute one of the main threats for biodiversity in the coming decades.

5.3.1.4. Impacts upon inter-species interactions

Apart from the aforementioned direct and indirect effects, the behaviour of species in relation to climate change will be governed by the resulting modifications in the interactions among species. Climate change will locally modify the competitive relationships among plant species. Wherever the reduction of hydric availability is moderate, warming will intensify the aerial competition among plants, which is mainly asymmetrical. Species displacement and local extinction will also modify the role of competitive relationships. In places with a predominance of aridification and other processes reducing productivity, aerial competition will become less important, giving way to subterranean competition (Tilman 1988). Species with greater tolerance or avoiding ability to stress will be favoured.

Both intensified competition and increased stress usually lead to a decrease in floristic diversity (Grime 2001). Processes favouring the co-existence of species, and therefore floristic richness, will also be subjected to changes, which are difficult to evaluate. In general terms, the disturbance regime (fires, droughts, etc.) will probably be intensified, although varying from one territory to another, so that it will foreseeable have a greater influence in some and a lesser one in others. These changes in the disturbance regime could favour the exclusion of species less tolerant to the new conditions and the subsequent spread of other better adapted ones. Species displacement will alter the separations between established niches, and there will be periods of instability while the new interactions become established. The new climatic conditions will also affect, among others, facilitation processes, although the direction of the changes appears to depend on the different situations (Pugnaire *et al*. 2001, Callaway *et al*. 2002). Goldberg *et al*. (1999) found no consistent intensification patterns of these interactions along productivity gradients.

In spite of their apparent self-sufficiency as primary producers, plants depend on many other organisms for nutrition (fungi and symbiotic bacteria), reproduction and dispersal (pollinating and dispersing animals). Furthermore, they serve as food (Harrington *et al*. 1999) and house and protect a multitude of dependent organisms. Local displacements and extinction will affect the species involved in these interactions. Of particular relevance will be the impacts upon those species that play important roles in ecosystems (keystone and engineer species), due to cascade effects upon other species.

The occurrence of different phenological responses among species with relationships of a mutualist, parasitic or competitive nature (for example, plants and pollinating insects, dispersing animals, phytofagous) could cause maladjustment in the relationships, thus altering the dynamics of the corresponding populations. Trophic asynchronies among birds, insects and plants have been detected in Europe (Visser *et al*. 1998, 2001; Chapter 6). Maladjustment in mutualist relationships will have negative effects on the species involved; these might be partially attenuated by the functional redundancy among species, which appears to be frequent in Mediterranean communities. Zoophilous and zoochorous plants will be more sensitive to this asynchronies, this sensitivity increasing with dependence on pollinators or dispersers. Available studies of Mediterranean vegetation indicate that these relationships often involve sets of species, both of plants and animals (Herrera 1995, 2001; Zamora 2000; Zamora *et al*. 2001), so that the mutual dependence is quite lax, which tallies with an environment characterised by fluctuations. Phenological maladjustment could constitute an added disadvantage for the species less tolerant to climate change, but it could also interfere with the possibilities of the species more favoured by climate to spread. We can therefore expect ecologically less complex structures, that is to say, with less interactions among species, in the new plant communities that will become established as climate change progresses.

Herbivory has both positive and negative effects on floristic diversity (Zamora *et al*. 2001). The negative ones will generally be accentuated with increasing aridity, especially for species sensitive to browsing or seedling depredation, which is the case of many woody species of the Mediterranean woodlands. In a reciprocal manner, the palatability of many species has worsened due to atmospheric CO_2 enrichment (Peñuelas and Matamala 1990, Peñuelas 2001).

Climate change will also favour the spread of parasite species to new territories, as well as the severity of the parasitism process upon individuals weakened by accentuated stress or even competition. This seems to be the role of certain parasite fungi in the oak decline (Brasier 1992, 1996; Brasier *et al*. 1993, Montoya and Mesón 1994). Hódar *et al*. (2003) documented a greater incidence of pine caterpillars in relict forests of Scots pine in Andalucia; the sensitivity of this type of pine forest to drought had been demonstrated in the Northeast of the Peninsula (Martínez-Vilalta and Piñol 2002). Increased mortality, especially among adults, could favour the creation of niches that could be occupied by other species, either local or immigrated ones, which are better adapted to the new climatic conditions.

A relevant role will be played in these interactions by the numerous plant and animal species that humans have introduced, accidentally or deliberately, away from their native habitats. The invasion processes by these exotic organisms are considered to be a component of global change as important as habitat destruction or fragmentation (Vitousek 1994, Vitousek *et al*. 1997). Exotic species modify the ecological conditions and processes (nutrient recycling, soil structure and properties, disturbance regime) of the communities into which they are introduced, and it has been shown that they can affect the genetic diversity of related native species, influence interactions among native species, cause extinction (Vitousek 1990, D'Antonio and Vitousek 1992, McNeely *et al*. 1995, Mack and D'Antonio 1999, Parker *et al*. 1999) or contribute to the homogenisation of the landscapes invaded (Dukes and Mooney 1999). Although invasion processes in Europe are severe, plants are not yet among the worst invasive organisms. It has been estimated that European flora contains around 5% of introduced species, mostly from the Eurasian continent (Lövei 1997, Stanners and Bourdeau 1998). On the Spanish Peninsula, the figures rise to almost 10% (Vilá *et al*. 2002). Excluding the exotic plants restricted to artificialized environments, 75 invasive species have been identified affecting to natural or seminatural habitats, and 45 of them show recent expansive tendencies (Dana *et al*. 2003).

Invasibility, the susceptibility of a community to invasion is a controversial theme, in which a key factor appears to lie in the existence of fluctuations in the availability of resources (Davis *et al*. 2000, Grime 2001). Disturbances, natural or artificial, but also other events that temporally increase the availability of resources, such as irrigation, fertilisation or eutrophication, initiate periods with a propensity for entry of invasive species, until the subsequent recovery of the community reduces the excess of resources. Invasive events are therefore of a clearly intermittent nature. As climate change causes population decline and local extinction, or accentuates the regime of certain disturbances, there will be periods characterised by environmental fluctuations in which the exotic species will find opportunities to spread. Sobrino *et al*. (2001) described signs of this type of spread, although it is still difficult to delimit the influence of other components of global change. Insularity is considered as an added risk related to invasive plants. This threat is particularly severe in the case of Canary Isles, with 45 species endangered or critically endangered for which competence with exotic species has been indicated as a risk for survival of some or even all of their populations (Dana *et al*. 2003).

5.3.1.5. Changes in vegetation and climate change: feedbacks

Although vegetation may appear to be a passive witness to climate changes, changes in the structure and composition of plant cover have an influence on climate, through the role they play in the radiative balances derived from the albedo, in evapotranspiration and water balances, in the production of organic matter and the composition of soil properties and in the spatial configuration and characteristics of fuels, which determine important properties of fire regimes, in emissions of VOCs, etc. Phenological changes and in particular the associated lengthening of the period of vegetative activity, will also affect determinant factors of climate change, such as carbon cycle and balances, and will also modify the water and nutrient flows of ecosystems (White *et al*. 1999), interfering, in turn, with some of the indirect effects of climate change.

We can conclude from this section that the predictable effects of climate change on plants will affect, directly or indirectly, practically all of their demographic and ecological processes: productivity, growth, chemical and genetic composition, nutrition, phenology, pollination and reproduction, dispersal, germination and recruitment, mortality and interactions with other species. These impacts will be dealt with through changes in the dynamics of the populations affected, including local migrations and extinction, which will generate changes in the composition, structure, distribution and functioning of the resulting communities. Most of the impacts indicate the structural simplification and reduced cover of the new communities, which will contain less interrelated species, with an abundance of those of great ecological amplitude, wide geographic and altitudinal distribution, tolerance to water stress, good dispersal ability, great phenotypical plasticity and genetic variability, etc. As current vegetation distribution follows biogeographic patterns associated with climatic ones, and as the projected climate change is not spatially uniform, the foreseeable impacts will be dealt with in different ways, depending on territories and vegetation types. Although the possible floristic changes will be governed by the individual responses of species, individualised predictions for the flora as a whole are beyond the reach of current scientific knowledge. An analysis is therefore provided of the projections by the Promes model, along with the consequences derived from this for each one of the main types of vegetation represented in Spain. This type of approach, although imperfect, is useful for identifying potentially significant effects in the different territories and habitats, insofar the reduction of the areas climatically appropriate for a determined type of habitat determines the magnitude of the risk of extinction of the typical species therein (Thomas *et al*. 2004).

5.3.2. Analysis of the projections by the PROMES model

In order to develop the assessment of the climate changes most significant for different types of vegetation, the projections by Promes (see Chapter 1) have been transformed to the units of the bioclimatic classification most commonly used in Spanish studies of vegetation (see 5.2.1). This classification (Rivas-Martínez 1997, Rivas-Martínez *et al*. 1999, 2002) makes use of summertime ombrothermal quotients in order to separate Mediterranean climates from Temperate ones (Fig. 5.4), an annual temperature sum (positive temperature) as a delimiting variable of the thermic belts or thermal types (Fig. 5.5), and an annual ombrothermal quotient (ombrothermal ratio) as a delimiter of the ombric belts or ombrotypes (Fig. 5.6). The results are mapped in figures 5.4-5.6 with the resolution of the 50 x 50 km cells generated by the model. The transition matrices are also shown between the units of climatic classification used in the 1961-1990 period, and the two SRES scenarios (B2 and A2) projected for the 2070-2100 period, based on demographic, socioeconomic, and technological tendencies similar to the currently predominant ones. In these matrices (Tables 5.3-5.5), each bioclimatic classification unit has been subdivided into two sub-units (upper and lower) in order to improve the resolution of the transitions.

The projection of the same model for the 1961-1990 period was taken as a reference framework for the present climate, as no other one with the same scale of geographic resolution was available. This projection, transferred to the units of the climatic classification used, reflects quite appropriately the relationships established between the vegetation and the present climate, although it exaggerates certain contrasts which ought to be taken into account in the comparisons. For example, the temperate climates are superimposed throughout the Iberian Range, as are the semiarid ombrotypes in the South. With regard to the latter, it seems that part of the lower dry ombrotype is reflected as upper semiarid by the model. To the contrary, the present area of the thermomediterranean belt is underrepresented, and perhaps also that of the thermotemperate one on the Northwestern coast.

Fig. 5.4 shows the change in the Mediterranean-Eurosiberian climatic boundary in the two scenarios. The “mediterraneisation” of the Peninsula, which is the most notable tendency, is more evident in the Northwest than in the Northeast, and it advances along three wedges: the Cantabrian coast, the Ebro valley and perhaps the northern slopes of the Pyrenees. The temperate climates are confined in the Cantabrian Range and the Pyrenees, and, although they still maintain certain continuity in scenario B2, they show clear evidence of fragmentation in A2.

The modifications in the distribution of the bioclimatic belts (Fig. 5.5) highlight the big spread of the thermomediterranean belt in the Southwest of the Peninsula and even the appearance of the inframediterranean belt, the closest current representation of which is at relatively distant latitudes in North Africa and in the Canary Isles. As a whole, the area of these warm Mediterranean climates is multiplied by six in scenario B2 and by eight in scenario A2. Furthermore, this spread over one century surpasses the maximum migration distances documented, and warm Mediterranean flora could therefore be expected not to reach the new northern distribution limits in this space of time. The spread of the thermomediterranean along the eastern coast is more moderate and restricted to the coastal area, so that, in scenario B2, it only slightly surpasses its real present limits towards the North and in the Ebro Valley. The distribution of the mesomediterranean belt, currently the most widespread, suffers a big displacement towards the North and the West of the Peninsula, but maintains a similar total area.

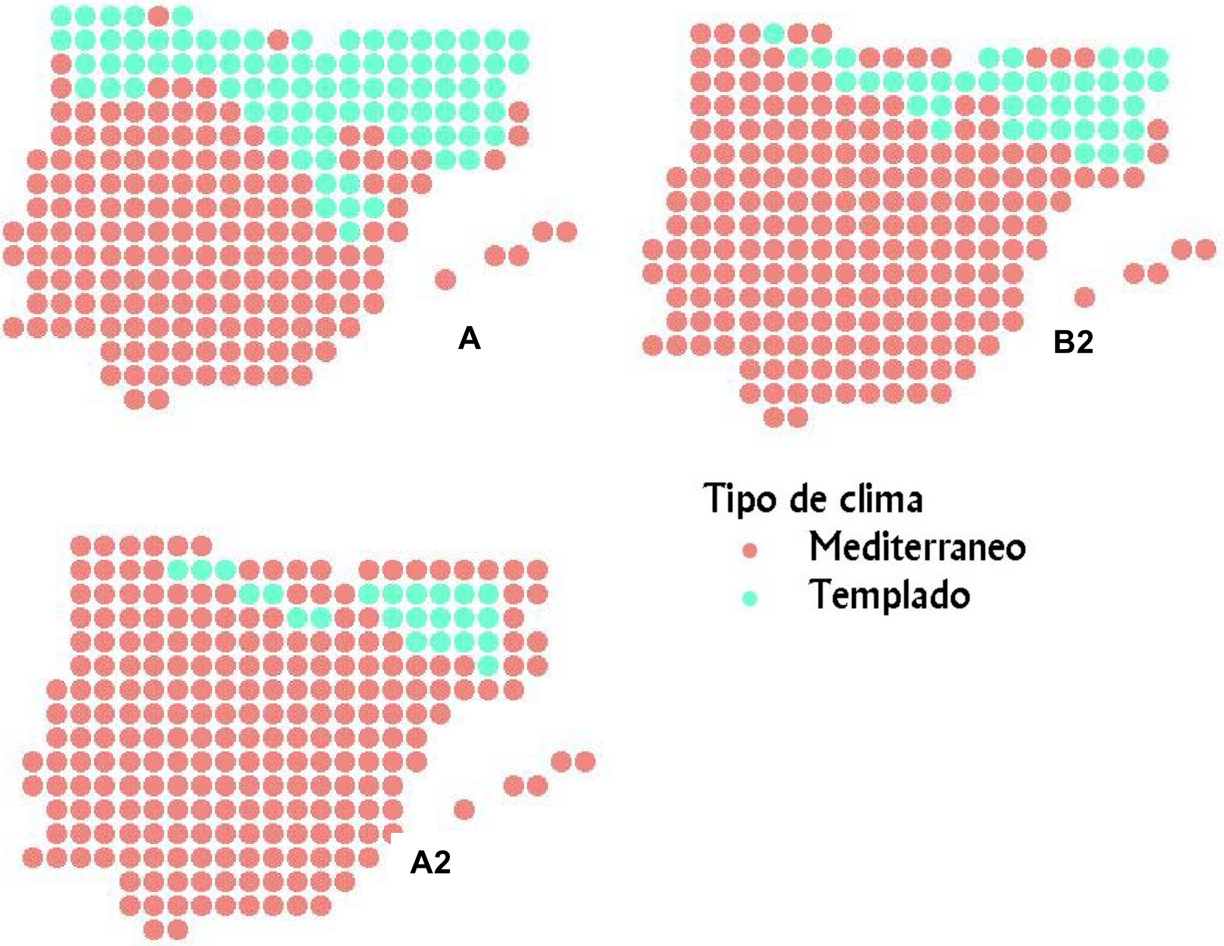


Fig. 5.4. Changes in the distribution of Mediterranean and Temperate climates according to the projections by Promes (scenarios B2 and A2; A: present climate).

Table 5.3. *Transition matrices (number of 50 x50 km grid cells) between Mediterranean and Temperate climates in the two scenarios.*

Scenario B2	**Med.**	**Temp.**
Mediterranean	175	
Temperate	45	41

Scenario A2	**Med.**	**Temp.**
Mediterranean	175	
Temperate	63	23

Table 5.4. *Transition matrices between bioclimatic belts (thermotypes) in the two Promes projections (B2 and A2). The rows indicate the number of grid cells corresponding to the current thermotypes and the columns, the number corresponding to each of the projections. Two subdivisions have been considered for the main thermotypes: lower or inferior (inf.) and upper or superior (sup.).*

SCENARIO B2	**IM**	**TMi**	**TMs**	**MMi**	**MMs**	**SMi**	**SMs**	**TT**	**MTi**	**MTs**	**STi**	**STs**	**OT**	*Total*
Inframediterranean														*0*
Thermomediterranean inf.	3													*3*
Thermomediterranean sup.	7	4												*11*
Mesomediterranean inf.	3	33	27	2										*65*
Mesomediterranean sup.			5	42										*47*
Supramediterranean inf.				9	27									*36*
Supramediterranean sup.					6	7								*13*
Thermotemperate														*0*
Mesotemperate inf.			2	5				1						*8*
Mesotemperate sup.				7	6				9					*22*
Supratemperate inf.					10	1				15	1			*27*
Supratemperate sup.						11	3				9			*23*
Orotemperate											1	4	1	*6*
Total	*13*	*37*	*34*	*65*	*49*	*19*	*3*	*1*	*9*	*15*	*11*	*4*	*1*	*261*

SCENARIO A2	**IM**	**TMi**	**TMs**	**MMi**	**MMs**	**SMi**	**SMs**	**TT**	**MTi**	**MTs**	**STi**	**STs**	**OT**	*Total*
Inframediterranean														*0*
Thermomediterranean inf.	3													*3*
Thermomediterranean sup.	11													*11*
Mesomediterranean inf.	38	25	1	1										*65*
Mesomediterranean sup.		10	22	15										*47*
Supramediterranean inf.				32	4									*36*
Supramediterranean sup.					13									*13*
Thermotemperate														*0*
Mesotemperate inf.		2	2	4										*8*
Mesotemperate sup.			2	17	1			2						*22*
Supratemperate inf.				2	17				6	2				*27*
Supratemperate sup.					5	11				4	3			*23*
Orotemperate											2	4		*6*
Total	*52*	*37*	*27*	*71*	*40*	*11*	*0*	*2*	*6*	*6*	*5*	*4*	*0*	*261*

The mesotemperate or coline suffers a loss of territory as a result of the Mediterraneisation process, which is only partly compensated for by a gain in territory which is currently supratemperate. The mountain belts will experience a notable reduction. The current area of the supramediterranean is halved in B2 and reduced to a quarter in A2, and also suffers an almost complete displacement towards areas that are supratemperate at present. The supratemperate, that is to say, the Euro-siberian middle mountain, is reduced to one third in B2 and to one fifth in A2. Mediterraneisation is responsible for most (50% in B2, 65% in A2) of this reduction of the territory, the rest corresponding to warming (transformation to mesotemperate). Due to these changes, Euro-siberian mountain climates become fragmented into one nucleus in the Cantabrian region and another in the Pyrenees; the former is reduced to only one cell in scenario A2. This sharp drop in mountain thermotypes should be interpreted as a rise in altitudinal limits between belts, resulting from warming; the lesser proportion of area within the cell means that the predominant belt therein changes. On average and in accordance with the transition matrices, scenario B2 involves a displacement of almost half of the altitudinal interval of most of the belts, whereas in scenario A2, the change is equivalent to almost a whole belt, particularly in the South. It is interesting to note the more conservative thermal behaviour of the coast, and the accentuation of continentality (annual thermal amplitude) throughout the inland area of the Peninsula.

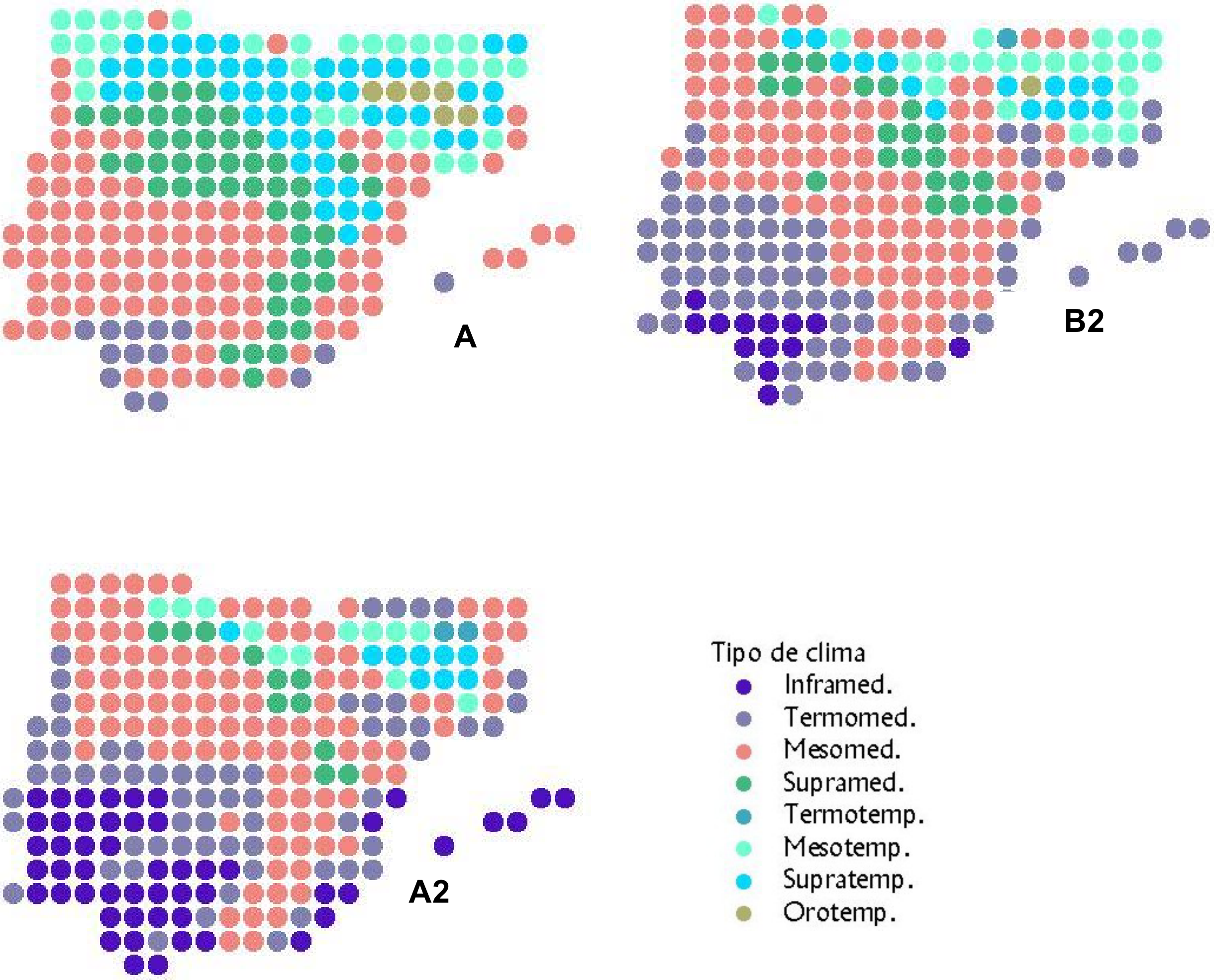


Fig. 5.5. *Changes in the distribution of the bioclimatic belts (thermotypes) by Rivas-Martínez according to projections by Promes (scenarios B2 and A2; A: present climate).*

The projection of a sharp decrease in precipitation in much of the Iberian Peninsula is one of the innovative contributions of Promes and determines that, in combination with warming, the scenarios present a considerable spread of semiarid and arid ombrotypes (Fig. 5.3.3), the latter heretofore restricted to very small enclaves on the coasts of Murcia and Almería. The semiarid ombrotypes enter well into the Ebro valley and also reach the Duero basin in scenario B2. The upper semiarid of the current projection by Promes (A) becomes lower semiarid in scenario B2, and practically the whole southern part of the Peninsula is framed within lower semiarid and arid climates in scenario A2. The appearance of the arid ombrotype in the Southwest of the Peninsula represents a considerable change, if we taken into consideration the greater stability of the semiarid-arid areas of the Southeast of the Peninsula, which only show an evident spread towards the South-eastern border of La Mancha in scenario A2.

Table 5.5. *Transition matrices between bioclimatic belts (ombrotypes) in the two Promes projections (B2 and A2). The rows indicate the number of grid cells corresponding to the current ombrotypes and the columns, the number corresponding to each of the projections. Two subdivisions have been considered for the main ombrotypes: dryer or lower, and more humid or upper.*

SCENARIO B2	**A**	**SAi**	**SAs**	**Si**	**Ss**	**SHi**	**SHs**	**Hi**	**Hs**	**HH**	*Total*
Arid	3										*3*
Lower semiarid	9	4									*13*
Upper semiarid		42	1								*43*
Lower dry		5	40	7							*52*
Upper dry			3	33	2						*38*
Lower sub-humid				9	20	1					*30*
Upper sub-humid					6	21	1				*28*
Lowe humid						7	15	9			*31*
Upper humid								17			*17*
Hyperhumid									3	3	*6*
Total	*12*	*51*	*44*	*49*	*28*	*29*	*16*	*26*	*3*	*3*	

SCENARIO A2	**A**	**SAi**	**SAs**	**Si**	**Ss**	**SHi**	**SHs**	**Hi**	**Hs**	**HH**	*Total*
Arid	3										*3*
Lower semiarid inferior	13										*13*
Upper semiarid	16	27									*43*
Lower dry		33	19								*52*
Upper dry		2	16	20							*38*
Lower sub-humid				22	8						*30*
Upper sub-humid				5	17	6					*28*
Lowe humid						17	13	1			*31*
Upper humid							9	8			*17*
Hyperhumid								2	3	1	*6*
Total	*32*	*62*	*35*	*47*	*25*	*23*	*22*	*11*	*3*	*1*	

In the North of the Peninsula, the dry ombrotypes recovers much of the area lost in the South. The sub-humid and humid ombrotypes undergo clear fragmentation processes throughout the Central and Iberian ranges in scenario B2, which are accentuated in A2. As a whole, the upper subhumid or more humid ombrotypes are reduced in area by 40% in scenario B2 and by 60% in A2. The northern third of the Peninsula continues to be dominated by sub-humid climates, with humid nuclei located in Galicia, the Cantabrian range and the Pyrenees, which even subsist in scenario A2. Rainy Mediterranean climates (sub-humid and humid) are therefore the ones that become predominant in the North at the expense of the temperate ones.

The increasing aridity in the South is more marked than in the North, and, once again, it seems that the changes in the West of the Peninsula will be more accentuated than in the East. Therefore, apart from the generalised “mediterraneisation”, the rainy Mediterranean enclaves become drastically reduced in their present locations and are displaced northwards. Contrary to what was stated in previous reports (Parry 2000), which predicted moderate impacts for Mediterranean vegetation, the currently available projections indicate big changes in the main climatic limits for vegetation, at least in the western Mediterranean.

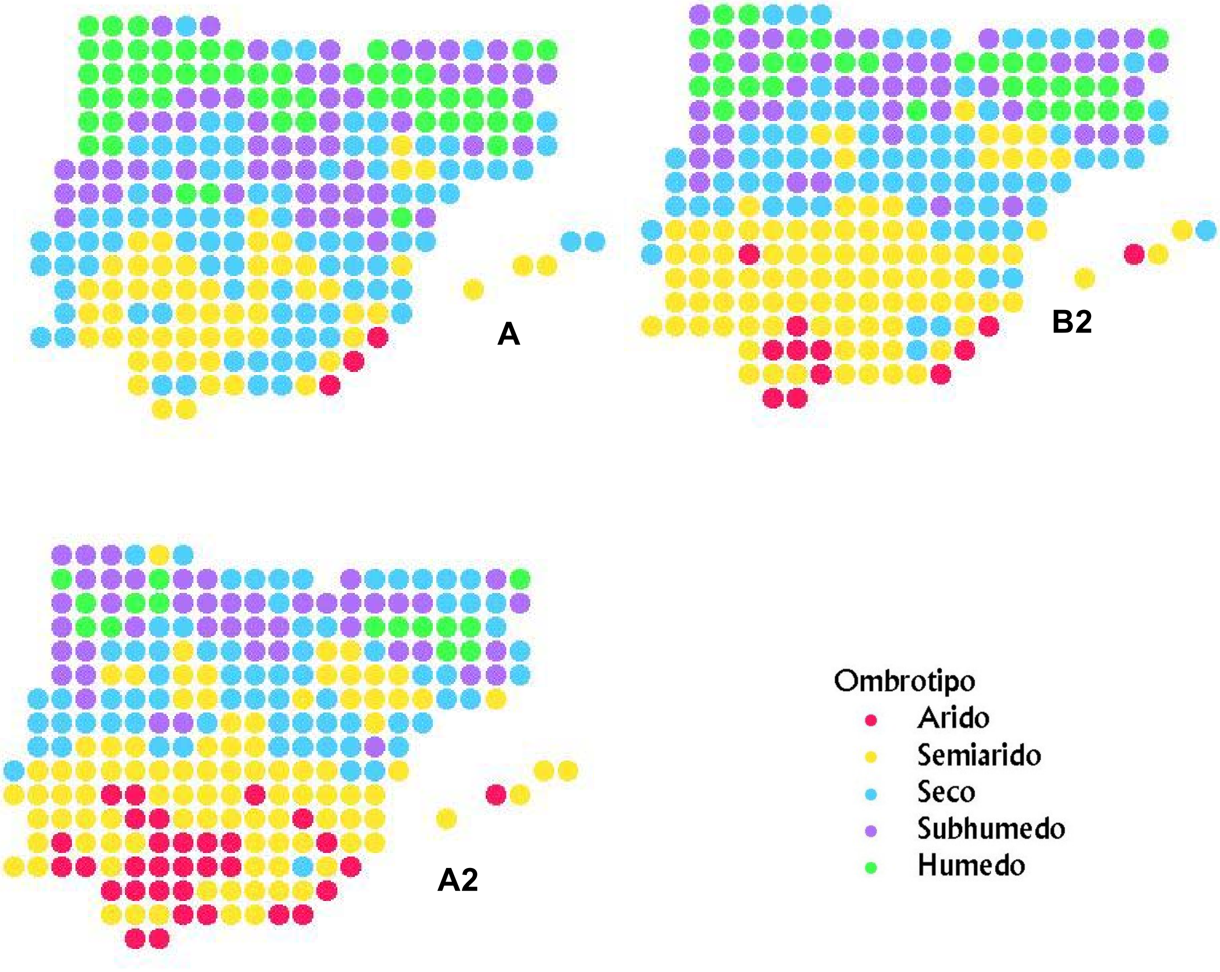


Fig. 5.6. *Changes in the distribution of the bioclimatic belts (ombrotypes) by Rivas-Martínez according to projections by Promes (scenarios B2 and A2; A: present climate).*

Regardless of their reliability, the implications of the aforementioned climate change scenarios may be very different for flora and vegetation depending on the trajectories that these changes really follow. In the case of temperatures, the projections indicate more or less linear change tendencies. In the case of precipitation, however it seems the fluctuations will be considerable, which would hinder possible movements of vegetation due to the antagonistic effects between warming and aridification. Several projections at mid term suggest in some peninsular territories that aridification could be compensated by a seasonal redistribution of rainfall (del Río 2003, del Río *et al*. 2004), which would act favouring vegetation changes opposed to those announced by Promes for the long term.

5.3.3. Analysis of impacts according to vegetation types

5.3.3.1. High mountain vegetation

High mountain habitats above the timberline comprise diverse types of herbaceous and shrub vegetation adapted to short periods of vegetative activity, intense cold, variable snow cover and soils characterised by the high erosive energy of the relief (Billings and Mooney 1968, Beniston 1994). The geographic isolation and specificity of these environments have promoted their richness in endemisms, and, in the case of Iberian mountains, in taxa at the southern limit of their distribution. It is therefore not surprising that many high mountain habitats are included in the Habitat Directive (Romao 1996, European Commission 2003).

Their close relationship with high mountain climates and cryoturbated soils means that generalised climate warming would be especially unfavourable for them. The spread of woody vegetation (bushes in the cryorotemperate and cryoromediterranean belts, trees at the lower limits and intermediate levels of the orotemperate and oromediterranean), accompanied by herbaceous flora which is currently at its optimum at lower altitudinal levels, would reduce the area available for high mountain vegetation and alter its composition, especially in relation to psychoxerophile and chionophile pastures. Although the climate change projected suggests that between half and practically all of the altitudinal interval currently assigned to the cryorotemprate and cryoromediterranean belts could be affected, it is unlikely that in one century there could be an altitudinal spread of several hundred metres of the current upper limits of woody vegetation. Estimates for the Alps indicate that increases in mean annual temperature of over 3ºC would have serious impacts, but over a period of various centuries (Körner 1995, Theurillat 1995). However, on many mountains below 2,300-2,400 m altitude, very considerable reductions and denaturalisation of mountaintop habitats are to be expected. The flora of these habitats would avail of refuges in particular types of topography: crests, snowfields, scree or abrupt and exposed slopes, depending on the ecological requirements, as can be seen at present on more modest mountains (between 1,400 and 1,800 m), where similar phenomena must have occurred following the last cold postglacial pulses. But the reduced area and the competition between woody plants and herbaceous ones from lower levels will reduce the high mountain floristic ensemble and increase the fragmentation of populations. Warming will attenuate the differentiating effects of relief and microtopography, reducing the diversity of the high mountain microhabitats. Although floristic diversity can increase locally, regional richness will decline, due to the loss of flora exclusive to these habitats. In the Pyrenees, for example, the more orophile vegetation of *Loiseleuria procumbens* or *Arctostaphylos alpina*, currently relictic, will be probably jeopardized.

The foreseeable decrease in snowfall, both in terms of precipitation in the form of snow and permanent snow cover, will determine the recession of all types of chionophile vegetation and particularly of the *Nardus stricta* swards and chionophile pastures of calcareous substrates, which will disappear from their current lower elevations in the surroundings of Cantabria and the Pyrenees. They may possibly persist at higher elevations, but in small areas within the high mountain mosaic. Furthermore, the spread of woody and herbaceous vegetation from lower elevations will be easier in soils long covered by snow, which are deep, cool and more productive. The relative proportions of xerophile and chionophile pastures will be modified in high mountain areas; the former might benefit from the aridification tendencies of the climate. The changes in the albedo caused by the decrease in snow cover will accelerate warming in the high mountain; a similar effect has been attributed to the proliferation of woody vegetation (Betts 2000). The snowfield habitat in the higher zones will suffer even more, with local extinction in the mountains more to the South (Cordillera Central, Sierra Nevada). Similar risks threaten peaty *Nardus* swards and high mountain bogs, which, apart from depending on soil water regimes, will undergo an acceleration of the mineralisation of organic matter caused by warming.

Big changes in land uses are not to be expected in high mountain areas. The profitability of many winter sports facilities will be negatively affected, although their conversion to other leisure and tourism activities could be an alternative. The exploitation of springs for production of artificial snow would alter severely soil hydrology. Livestock carrying capacity could be increased on some mountains as the availability of summer pastures at middle elevations decreases. A reversion of the predominant tendency in livestock during the last century could be used to control the proliferation of woody vegetation, but it would favour the spread of herbaceous plants coming from meadows at lower elevations.

Floristic changes in high mountain vegetation and in the altitudinal limits of certain species are among the first ones documented and attributed to the effects of climate change. Grabherr *et al.* (1994, 1995, Gottfried *et al.* 2000; Pauli *et al.* 2001) detected changes of this type in herbaceous species in the Alps throughout the last century, although with mean climbing rates of 1-4 m/decade as opposed to the 10 m/decade that might be expected for an estimated warming rate of 0.7ºC. Sanz-Elorza *et al.* (2003) indicate densification processes in scrub plants (gorse and dwarf juniper) at cryoromediterranean altitudes in the Central range for the 1957/1991 period. These rises in elevation are moderate and the influence of reduced pressure by livestock cannot be fully ruled out (Archer *et al.* 1995). A rise in the upper forest limit would probably require longer time periods (Burga 1988, Ammann 1995) and might react negatively to climatic variability, as seen in studies of *Pinus uncinata* in the central Pyrenees (Camarero and Gutiérrez 2003). Peñuelas and Boada (2003), however, detected an ascent in the upper altitudinal limit of beech forests in Montseny, evaluated by at 70 m for the last 55 years (13 m/decade), although this was not independent from the abandonment of livestock farming uses. Rises in the upper forest limit have also been detected in Scandinavia (Kullman 2001), where the signs of climate change are more notorious, in the Balkans (Meshinev *et al.* 2000) and in New Zealand (Wardle and Coleman 1992).

5.3.3.2. Forest vegetation: forests

Forests constitute the natural potential vegetation of most of Spain, with the exception of high mountain areas, the semi-arid parts of the Peninsula and the arid and desertic areas of the Canary Isles (Rivas-Martínez 1987, Blanco *et al.* 1997). The diversity of our forests at present and their diverse bioclimatic spaces mean that we can expect very varied responses to climate change.

Deciduous forests predominate in the Euro-siberian area, but there are also large areas in the Mediterranean mountains. The more ombrophile deciduous forests, such as beech, oak and fir forests, mixed forests, etc., will be negatively affected by the aridification of the climate and the accentuation of summer drought (Mediterraneisation). In some woodlands in the Pyrenees, the pre-Pyrenees, the Castilian-Cantabrian transition, the northern Iberian range and the Central range (Ayllón, Somosierra), beech forests will suffer severe recessions with regard to area which will cause them to become extinct or merely residual, as it also will with the oak forests of *Quercus petraea*. In other territories, they will experience a noteworthy retrocession. Something similar is expected for the Carpetan birch forests of *Betula celtiberica,* the oretan of *B. parvibracteata* and the ones in the sub-Betic and Sierra Nevada ranges involving *B. fontqueri*.

Other deciduous forests with bigger areas at present and less hydric requirements will suffer big losses of some of their territories, but they will have possibilities to remain in others and even to spread, at the expense of more ombrophilous forests, such as beech or oak forests. It is the case of the *Quercus robur* mixed forests of the Galicia-Cantabria regions, *Quercus pubescens* Pyrenean forests from Pyrenees, and *Q. pyrenaica* and *Q. faginea* forests. The latter are best represented in sub-humid or humid supramediterranean climates, in which they will become reduced, while its role will become more prominent in the neighbouring supratemperate territories. To the contrary, the forests of *Quercus pyrenaica* and *Q. faginea* subsp. *faginea,* as well as the Andalucia *Q. canariensis* forests and the Portuguese oak forests (Q*. faginea* subsp. *broteroi*) of the Southwest of the Iberian Peninsula, will suffer a severe crisis, with local extinction due to a lack of refuge altitudes and the impossibility of migrating northwards. The crisis will be more acute in the Southwest of the Peninsula than in the Ibero-Levantine area.

The responses of these forests negatively affected by the projected climate change are complex and will be verified at temporal scales that are surely longer than those of the projections. In those species that have the chance to spread at the expense of others, it must be kept in mind that the recession in areas affected by climate change could be quite rapid, as recurring drought, the associated plagues and other disturbances decimate the adult trees and reduce recruitment. The spread, however, will be much slower, due to the longevity of the dominant species and to the fact that several generations are needed for a tree species to significantly amplify its territory in a natural manner. It does not seem possible that the migration speeds documented in the Quaternary post-glacial periods to be reached, due to the degradation of present landscapes and to the fact that the destination habitats will, in many cases, be occupied by other species. Disturbances could accelerate the change dynamics, but the foreseeable balance indicates that the spread will not compensate for losses of territory, at least in the medium term. The forests with the best chances of persisting will be those with sufficient altitudinal or latitudinal continuity. In the forests located in territories subjected to retreat, refuges might subsist in areas with favourable topography (ravines, shaded areas, northern exposures), and topographic diversity will therefore be another important factor in the change dynamics. The retreat processes governed by disturbances could give rise to the temporary disappearance of the tree cover until new cover is generated which is in accordance with the characteristics of the climate. In these periods, many species associated with the forest environment may become jeopardized or rapidly disappear. Furthermore, the substitution of a deciduous canopy by a perennial one will lead to changes in the light regime, which will affect the rich deciduous underbrush. Peñuelas and Boada (2003) described the progressive substitution of beech forests by Holm oak forest in a relatively large altitudinal interval (700-1200 m) on the southern slopes of Montseny during the last half century. The replacement dynamics appear to be quite effective in the ecotone zones between both types of forest, and are supported by differences in the recruitment by both species and by a greater incidence of defoliation or decoloration symptoms in the beech trees situated in the retreat areas of the beech forest.

The casuistics of conifer forests are also varied. Generalised warming and aridification will cause area reductions in high mountain species, such as forests of mountain pine (*Pinus uncinata*), of Scots pine (*Pinus sylvestris*) and even of black pines (*Pinus nigra s.l.*). In certain currently rainy areas, however, such as the Pyrenees, the Scots and black pine forests could make use from the decreased water availability to gain ground from deciduous forests requiring rainfall. The lower covering of snow will cause modifications in the composition of high mountain forests; in the Pyrenees, for instance, a decrease is to be expected in forests of *Pinus uncinata* with rhododendrons in favour of pine forests with *Arctostaphylos*. The increasing incidence of plagues has already been documented in southern relictic enclaves of Scots pine (Hódar *et al*. 2003), as has been the sensitivity of these to drought in northern enclaves on the Peninsula (Martínez-Vilalta and Piñol 2002).

The behaviour of Spanish juniper woods (*Juniperus thurifera*) is more difficult to predict. Their present territories, mainly supramediterranean and inland ones, will be exposed to warming but also to decreased rainfall, which, in combination with edaphic characteristics, could constitute an obstacle for them in relation to the spread of sclerophyllous or deciduous species of *Quercus*. Furthermore, the Spanish juniper tolerates semiarid mesomediterranean climates, as can be seen in their populations in the Ebro basin and in the Southeast of La Mancha. To the contrary, they have little resistance to fire disturbances, and, indeed, their present distribution is associated with regimes of low incidence and recurrence of fires (Vázquez *et al*. 2002). Something similar can be said of *Juniperus oxycedrus* forests, currently very fragmented in abrupt enclaves in the centre and west of the Peninsula. Both types of forests must have occupied bigger areas in more arid but colder eras of the recent Pleistocene. The Arbor-vitae woods (*Tetraclinis articulata*), now relictic in the Sierra de Cartagena hills, could, however, be favoured by climate change, in view of its climatic preferences in Northwest Africa. The opposite occurs with the pinsapo (*Abies pinsapo*), an endemic fir to the mountainous regions of Malaga and Cadiz (Arista *et al*. 1997), where, in most of its current altitudinal interval, conditions will become too hot and dry for its present requirements.

The response by Mediterranean pine forests, which currently constitute most of the forested area of the Peninsula, will also be varied. We have already mentioned the foreseeable decrease in area of maritime and black pine forests, at least in some of the present enclaves, as a consequence of aridification. Furthermore, the latter will become more susceptible to fire in its present territories, as this type of disturbance leads to regenerative failure (Trabaud and Campant 1991, Escudero *et al*. 1997, 1999; Rodrigo *et al*. 2004). Maritime pine forests respond well to fire, although with big differences from one territory to another (Faraco *et al*. 1993, Pérez *et al*. 2003). The Aleppo pine (*Pinus halepensis*) could benefit, at least in the East of the Peninsula, as it resists semiarid climates relatively well, is capable of becoming established in open areas, furthermore, and its competitive capacity in relation to Holm oak increases with drought and with recurring disturbances (Zavala 1999, 2003; Zavala *et al*. 2000).

Sclerophyllous forests are exposed to diverging tendencies. The mediterraneisation of the northern third of the Peninsula will favour the spread of Holm oak and cork oak forests –above all in the Northwest- at the expense of other deciduous forests. Peñuelas and Boada (2003) documented a process of this nature at the mid-mountain level in Catalonia. The spread should be more rapid in the lowlands, although the higher degree of deforestation would counteract this. The fragmented enclaves of sclerophyllous forests on the coast and in inland valleys of Galicia, on the coast and in the mountains in the Basque Country and Cantabria and on the southern slopes of the Pyrenees, will constitute efficient nuclei for this spread. In the Mediterranean region, however, although aridification will favour altitudinal rises of sclerophyllous vegetation also at the expense of mainly deciduous forests, the potential losses of territory will be greater, due to the spread of semiarid and arid climates, especially in the South. The sensitivity of sclerophyllous forests has been confirmed in recent drought episodes, due to their poor hydric efficiency in extreme conditions and to their slow subsequent recovery, and these effects are accentuated on sun-facing slopes and in well-drained soils or soils with little retention capacity (Peñuelas *et al*. 2000, 2001; Martínez-Vilalta *et al*. 2002a). The accentuation of symptoms such as the oak decline in holm and cork oaks, currently widespread in the Southwest (Montoya and López Arias 1997), will be one of the mechanisms intervening in the retreat of sclerophyllous forests (Brasier 1992, 1996; Brasier and Scott 1994, Montoya and Mesón 1994). Increasing aridity will have a great impact on the main nucleus of Iberian *dehesas,* an agro-forestry system of great biodiversity in which the regeneration of the trees is problematic because of interactions with the livestock farming use (Pulido *et al*. 2001). Both the structure of the *dehesas,* including the maintenance of the trees, and the livestock farming use itself, will be negatively affected by the projected climate change.

The territory that might be lost by the holm oak forests in the South will be compensated for to a great extent by territories gained in the North, to which they have easy access due to the widespread distribution of this tree on the Peninsula. The recession of the cork oak will be more irreversible, due to its greater hydric requirements, in the South and especially in the Southwest, (western Andalucia, Sierra Morena, Extremadura and Montes de Toledo), where high-altitude refuges are scarce and limited in extent.

5.3.3.3. Shrub vegetation

Tall shrublands and scrub involve extraordinarily varied types of vegetation in Spain, and contain a notable floristic diversity. Indeed, in spite of their relative structural simplicity, their floristic composition responds with high turnover rates to climatic, edaphic and geographic gradients. As with forests, most types of shrubland vegetation are included in the Habitats Directive, and some of these have been given priority interest.

The responses by shrubland vegetation to climate change will depend in each territory on the predominant direction of the antagonistic change tendencies. The increased productivity promoted by warming and reinforced by growing CO_2 concentrations will favour the tree development and the successional displacement of shrubland communities, but also colonisation by shrublands from pastures or abandoned croplands, as has been occurring in the last few decades. To the contrary, aridification will favour the spread of shrubby structures, which are simpler and have less water requirements than forests. The intensification of certain disturbances such as fire, the abandonment of agricultural and livestock farming uses on land that suffers from decreased productivity as a result of climate change, and the collapse of certain forests affected by drought, will be parallel processes favouring the spread of shrublands, which, in any case, will undergo big readjustment processes in their floristic composition and distribution.

Deciduous shrublands comprise different types of deciduous thickets and forest-fringes (with perennial species in some cases, such as boxwood forests of *Buxus sempervirens*), which are widespread as a pre-forest stage in many Euro-Siberian forests, in mountainous areas of the Mediterranean and along river valleys. As a result of their hydric requirements, they could be expected to suffer recessions in most of their present distribution areas, which could be particularly acute in their southernmost representations in Andalusian ranges.

The current distribution of sclerophyllous shrublands mainly comprises the thermo and mesomediterranean belts, as well as some enclaves isolated in southern Euro-Siberian territories, which will act as expanding nuclei as mediterraneisation becomes accentuated. Thus, climate warming and aridification will promote their spread, both in altitude and towards more northern latitudes. Aridification will also favour the spread of formations typical of semiarid and arid climates (kermes oak garrigues, thorn brushes in the Murcia and Almeria regions, etc.), which will at least increase their potential distribution areas. Their possible spreads will not be homogeneous because resistance to drought differs among species, both in adult individuals (Martínez-Vilalta *et al*. 2002, Ogaya *et al*. 2003), and in seedlings (Ogaya *et al*. 2003, Vilagrosa *et al*. 2003). Indeed, some types of sclerophyllous shrublands with high hydric requirements could be expected to suffer losses of territory, especially in the South and Southwest of the Peninsula: tree strawberry, (*Arbutus unedo,* see Martínez-Vilalta *et al*. 2002, Ogaya *et al*. 2003), heathlands of *Erica arborea, Quercus lusitanica* scrub, etc. To the contrary, these same types, or other similar ones (laurel groves, for example), will increase their territory in the mediterraneised areas in the northern third of the Peninsula.

Whereas sclerophyllous shrublands are dominated by the so-called pre-Mediterranean element, mostly consisting of resprouting species, with root systems that are relatively deep, long-lived, zoochorous and demanding with regard to germination and establishment conditions, in the remaining types of Mediterranean shrublands, there is a predominance of shrubs and scrub which were more recently differentiated from the evolutive point of view, smaller and/or less long-lived, with shallower roots, often deprived of the capacity to resprout, with non-specialised or passive dispersal and diaspores that germinate easily in clearings created by disturbances (Herrera 1992, 2001). Part of this flora also presents flexible leaf architectures, including the capacity to lose some of the foliage during periods of greater drought (Valladares 2001). Although adult individuals of certain species show vulnerability to drought, particularly due to their shallow rooting systems, populations are capable of rapid recovery through germination (Peñuelas *et al*. 2001). Responses of this nature are, however, closely dependent upon climatic fluctuations (Quintana *et al*. 2004). The flora of these shrublands therefore present some of the most typical symptoms of adaptation to drought and to other characteristics of Mediterranean environments, and increasing aridity of the climate can therefore generally be expected to favour their spread. However, the relationships between the different types of shrublands and climate are very variable, and as a result, so are their responses to the projected change.

Among silicicolous shrublands, the more ombrophilous types will suffer loss of area and retreat at their current distribution limits, although probably in more moderate degrees than other, more sensitive formations. This is the case of heaths ("jaral-brezales") of Atlantic and Iberian-Atlantic area, which are currently distributed throughout sub-humid or rainier climates. This retreat will be more accentuated in the Southwestern quadrant: Cádiz sierras, Sierra Morena, Montes de Toledo and Montes de Extremadura. Further north, in the Montes de León and the Iberian and Central ranges, and on the coasts of the Basque Country and of Cantabria, the reduction will be less drastic but with changes in floristic composition caused by mediterraneisation. Brañas or peaty heaths, given priority in the Habitat Directive, will be most severely affected and serious local extinction processes can be expected. Similar tendencies can be expected in gorse shrublands, especially in the more ombrophilous types occupying large areas in supramediterranean and supratemperate belts. Broom fields (*Retama sphaerocarpa*), however, will have possibilities to spread, due to the considerable expansion of the thermo- and mesomediterranean belts and to their tolerance to drought.

The recessions suffered by other more ombrophilous silicicolous shrublands will favour *Cistus*-shrublands ("jarales"), which will also be favoured by possible regional intensification of the fire regime. Within these, the more thermophile types will show a notable increase, particularly the types of *Cistus ladanifer* and other *Cistus* species capable of losing leaves in the dry season (*C. monspeliensis, C. salviifolius*, etc.), to the detriment of the relatively more mesophytic species such as *Cistus populifolius* o *C. laurifolius*.

Shrublands typical of particular lithologies, such as ultrabasic rocks (peridotite, serpentine), dolomites or gypsum, contain a high proportion of endemisms associated with this kind of substrates, which are selective for flora, and these will probably subsist therein with minor adjustments, considering that they have resisted, *in situ* preceding climate changes of greater magnitude. Their tolerance to stress induced by the unfavourable chemical composition of the soils they inhabit, and the low level of competition that characterises these environments, will be an advantage in the event of increased aridity. Nevertheless, monitoring of the future evolution of these habitats is advisable because the combination of the projected warming and aridification to which they will be exposed might be unprecedented in certain enclaves. In the case of the Iberian gypsum steppes, given priority in the Habitat Directive, increasing aridity could spread to gypsum outcrops in the Pyrenees and the North of the Iberian range, the present climate of which is too rainy for any manifestation of the selective effect of this kind of substrate on plants.

Shrublands typical of calcareous soils (rosemary, sage, lavender, gorse, thyme, etc), very widespread in central and eastern Spain, will undergo an advance in general terms. Floristically, they are richer than their silicicolous homologues, and contain more plants adapted to xericity. Within these, the more xeric types, currently dominant in the semiarid Southeast, will be more favoured and could spread to the inner La Mancha, South of Valencia, the Baza basin and eastern Andalucia. The more thermophilous types, now mainly restricted to coastal regions, will, spread in altitude and inland, although these spreads will filter only the elements that are less dependent on oceanity and will be moderated by the barrier effect of the coastal mountain ranges. The meseta types will not undergo big changes, except for peripheral reductions and altitudinal rises at the expense of the mid- and high mountain types. In the Ebro valley, the "romerales" (rosemary scrub) of the basin could spread at the expense of the peripheral types (Gavilán 2003). Mid- and high mountain shrublands, rich in stenochorous endemisms and cushion plants, will suffer a reduction in climatic space, although the displacement of these by more thermophilous shrubland components will be slow and limited. However, the foreseeable warming in these areas will increase productivity and could facilitate more effective displacement of shrublands by trees and tall shrub vegetation. This problem will be more marked in the Betic and sub-Betic ranges, subjected to greater warming and clearly richer in endemisms with little dispersal capacity, than in the Iberian range and in the coastal mountains of the East coast, which offer greater possibilities for migration and which will not suffer such a marked reduction in the mid- and high mountain areas.

Changes in shrubland types and areas involve other important relationships with climate change. Mediterranean shrublands release considerable amounts of VOCs into the atmosphere. Likewise, it is characteristic of many types of Mediterranean shrubland to rapidly accumulate necromass, which favours the risk of ignition and above all the spread of fires (Moreno *et al*. 1998).

5.3.3.4. Herbaceous vegetation (meadows and pastures)

Herbaceous plants constitute around four fifths of the total amount of vascular flora species in Spain. In the temperate climates in the northern third of the Peninsula, the herbaceous flora is dominated by perennial plants, and in the Mediterranean climates annual species dominate, their life cycle enabling them to avoid summer drought while resisting disturbances. Although the greater abundance and richness of herbaceous flora is found in the communities in which woody cover is lacking or reduced, and especially in those shaped by practices associated with grazing, large contingents of herbaceous flora are associated with forest and pre-forest environments, and to other large types of habitat mentioned in this section, and their responses to climate change will depend, to a greater or lesser extent, on the aforementioned tendencies of each of these habitats. The predictions for flora and vegetation in relation to climate change are varied, and will closely depend on what happens with land uses, and in particular with livestock farming, which is the one that has the greatest control over woody plants and which favours the development of different types of pastures.

The most noteworthy change tendencies for grassland vegetation will be the following ones. A general reduction of mesophile and hygrophile meadows can be expected, including mid-mountain *Nardus* swards and many forb mountain communities, dependent both on hydric availability and on the abundance of organic matter. This reduction will be moderate in the territories with temperate climates and greater in Mediterranean mountains. The representations of certain particular types at the ends of their area, such as mown meadows in the Iberian and Central ranges, might be almost completely substituted by meadows more tolerant to summer scorching, such as “ballicares” (*Agrostis castellana*), “fenalares” (*Brachypodium phoenicoides*), “gramales” (*Cynodon dactylon*), “juncales churreros” (*Scirpus holoschoenus*), etc.

To the contrary, different types of grassy steppes will gain ground, both in Euro-Siberian and Mediterranean mountains and in the Mediterranean areas most affected by aridification. Therein, there could be a spread of “espartales” (*Stipa tenacissima*), “albardinares” (*Lygeum spartum*), “cerverales” (*Brachypodium retusum*), “lastonares” (*Stipa spp., Helictotrichon spp.*), “berceales” (*Stipa gigantea*), etc. These pastures are quite tolerant to fire disturbance and habitually contain great richness and abundance of therophytes. However, certain steppes requiring relatively rainy climates, like the “cerrillares” of *Festuca elegans* in Ibero-Atlantic Mediterranean mountains, will suffer reductions of potential area and might even disappear from their present Oretan, Sierra Morena and Sierra Nevada refuges. We might also expect a recession, at least in part of their present territory, of different psycroxerophile Mediterranean pastures, currently abundant in the Betic ranges, and clearly associated with climates that promote processes of soil cryoturbation. In this case, the recession would be governed by competitive displacement by other types of vegetation favoured by climate warming, more than by aridification.

The vegetation typical of more anthropised habitats (ruderal and weed communities) will be displaced, but extinction is not to be expected, given the large climatic space typical of most of the species involved. To the contrary, in an unstable environment in which disturbances and gaps in the vegetation cover become more common, these species can be expected to have greater possibilities of spreading.

In short, the balance of the climate change projected will generally be favourable for herbaceous vegetation, although there will be important conditioning factors derived from possible changes in land uses. Climate change will generally favour xerophytic types of vegetation, annual species and, in many cases, the relatively opportunist ones, with great climatic (and therefore geographic) amplitude and agile dispersal mechanisms. Due to the short generation cycle of most species, and the facility for anemochorous and zoochorous dispersal of many of these, the changes could be relatively rapid, although the response to climate change will be more resistant in relation to water availability than to temperature regime. It has been speculated that the positive response by C_4 plants, more demanding with regard to temperature and light requirements, and with more efficient transpiration, will be more marked than that of C_3 plants, although the carboxylation capacity of these will be favoured by a CO_2 enriched atmosphere.

5.3.3.5. Rock and scree vegetation

All the vegetation typical of fissures in rocks, stony ground and scree comprises a highly specialised flora with a large amount of endemisms. Practically all these types are included in the Habitat Directive. The peculiar conditions of these plant environments combine soils with little capacity for hydric retention and poor in nutrients, with particular disturbance regimes associated with periodic earth slides or with the mobility of substrates, and they determine very open pant structures with little competition for light. The tolerance to stress of rock flora includes the capacity to resist long periods of physiological cessation or attenuation, and to concentrate activity in the short and often irregular favourable periods. Thus, this type of adverse habitat is among those less directly vulnerable to the projected climate change (Theurillat 1995).

The harshness of winter, however, is another adverse factor for these plants, and evident altitudinal zoning can be observed in their communities. Under circumstances of climate warming, the higher-altitude species (oro-cryorotemperate and oro-cryoromediterranean) could therefore be affected by the immigration of species typical of lower elevations. In any case, this process would be very slow, given the high staying capacity of this type of vegetation, as can be seen in the presence in low-altitude rocky habitats of species whose optimum elevation is at higher altitudes, reminiscent of previous cold periods.

Epiphytic vascular vegetation has certain floristic relationships with rock vegetation. It is poorly represented at our latitudes, mainly by ferns and mosses. Its dependence upon climates with high atmospheric humidity and rainfall seems to indicate a decrease in its optimum territory, with recessions towards coastal oceanic areas.

5.3.3.6. Coastal vegetation (dunes, beaches, cliffs and salt marshes)

The vegetation dealt with in this section comprises different types of habitats conditioned by coastal geomorphology and the influence of the salt spray. They are well represented in the Habitat Directive, and many of them have been given priority status due to the clear retreat of coastal ecosystems caused by the age old concentration of the human population in these areas, and to the addition, in recent times, of housing and the construction of infrastructures associated with tourism.

Climate changes should have quite moderate direct effects on coastal habitats, because the warming process will surely be gentler on the coast, and aridificaiton will be partly compensated for by the high level of atmospheric humidity caused by the vicinity of the sea. The foreseeable rise in sea level, however, could greatly reduce the areas occupied by these habitats, and there could be other changes in the geomorphologic processes that maintain dune systems and marshlands and coastal pools. Housing and infrastructures will make it difficult for these systems to gradually retreat from the advance of the sea. This restriction will be even more serious in areas like Galicia and especially Cantabria, where in certain areas, the dune and post-dune zones have been occupied for quite some time (Basque Country) or will be occupied quite soon (Cantabria). Mediterranean coasts are also affected by these processes, which contribute to fragmenting coastal ecosystems and hinder possible migration routes. The current locations of coastal wetlands will be affected by increased salinity, which will profoundly alter the composition of the mosaic of communities in these systems; in extreme cases, the sea could directly invade them. Dune vegetation and in particular woody plants, given priority in the Habitat Directive, are also among those vulnerable to these processes. In this sense, vegetation living on cliffs might prove to be the most resistant to changes.

Some psammophilous herbaceous and shrubland communities have interesting representations in inland sandy areas. As with other types of vegetation associated with unfavourable soils, the impacts to be expected are moderate, although, as these representations are very scant, any tendency towards population decrease would give rise to quite rapid extinction processes. In the case of brine basins, increased temperatures accompanied by decreased rainfall, or wintertime concentration of this, would favour their advance, especially in the Southeast of the Peninsula, the La Mancha plateau, the Ebro basin and the Duero basin.

5.3.3.7. Island vegetation

There has been speculation over whether the impacts of climate change will be greater on the islands, due to the fact that the flora (and fauna) typically present, which has become impoverished, with interactions among more fragile species and lower rates of genetic diversity of species, both endemic and non-endemic (Frankham 1997). However, at least on the smaller islands, the tempering and humidifying effect of the sea could attenuate climate change tendencies. The projections by Promes (Fig. 5.5-5.6) for the British Isles show the effects both of warming and of reduced hydric availability, although somewhat more moderate than in the East of the Peninsula. Thus, the foreseeable impacts for this archipelago will be similar to those that have been dealt with for the habitats represented therein, with the exception that there will be a higher proportion of coastal refuges. The most vulnerable points appear to be mountain vegetation in Majorca and increased aridity in Menorca.

The resolution of the projections of the Promes model for the Canarian archipelago are insufficient due to the dimensions of the islands with regard to the grids, and to the difficulty involved in interpreting the changes in the varied pluviometry of the bigger islands. The western isles will be more affected by warming, but as a whole this will be more moderate than in the mainland and has been evaluated in less than half of a bioclimatic belt for the scenario B2 and just over one half for A2. The reduction of rainfall will imply a moderate spread of the arid and semi-arid ombrotypes currently dominant in the lower altitudes of the isles. The conical topography of the isles will determine the decrease in area of the different ombrotypes and thermotypes to progressively increase in those at greater altitude. According to the peculiar climatic and altitudinal zonation of the Canarian vegetation (Rivas-Martínez *et al*. 1993, Del Arco *et al*. 2002, Reyes-Betancort *et al*. 2001, Rodríguez Delgado *et al*. 2004), it is foreseeable an expansion of the inframediterranean arid and semi-arid vegetation types -“tabaibales” (*Euphorbia balsamifera)* and “cardonales” (*Euphorbia canariensis*)-, whose more pioneer and dynamic species could play the role of vanguard: “tabaibas amargas” (*Euphorbia lamarckii, E. regis-jubae, E. berthelotii*), “aulagas” (*Launaea arborescens*), “salados” (*Schizogyne sericea*), “vinagreras” (*Rumex lunaria*), “cerrillos” (*Hyparrhenia sinaica*), “panasco” (*Cenchrus ciliaris*), and even invasive species like the “rabogato” (*Pennisetum setaceum*). On the contrary, the thermo-sclerophyllous woodlands with Canarian junipers (*Juniperus canariensis*), Canarian olives (*Olea cerasiformis*), “almácigos” and “lentiscos” (*Pistacia atlantica, P. lentiscus*), whose remnants are strongly affected by urban and agricultural land uses, will be negatively affected by aridification and will not get much chance to reach refuges at higher altitudes. Concerning the laurel forests or “monteverde”, dependent upon the mist swept by trade winds, a reduction in area could be expected due to the warming of the ocean air masses and the lowering of the upper subsidence layer of warm air. The shrublands with *Hypericum canariense* and *Rhamnus crenulata* and the “fayal-brezal” of *Myrica faya* and *Erica arborea* could benefit from the retreat of laurel forests, whose persistence in the summits of some eastern isles would be problematical. The lowering of the mist-bound would favour to the Canarian pine forests (*Pinus canariensis*) and their associated broom shrublands. Negative impacts on riverine vegetation, and in particular “sauzales” (*Salix canariensis)* and palm groves (*Phoenix canariensis)*, are also foreseeable. These vegetation types are nowadays very affected by water extraction practices. Coastal habitats are also decimated by touristic infrastructures and will suffer for the remodelling of the coastline associated to the rise in sea level.

5.4. MOST VULNERABLE AREAS

The previous section highlights the fact that practically all Spain’s habitats and territories will undergo relevant impacts as a consequence of the projected climate change. If we consider that the processes induced by climate change will involve pressure that will exceed the plasticity and acclimatisation capacity of many species, that the evolutionary responses are unfeasible in the short term in which the changes will occur, and that the efficiency of species migration as a mechanism of persistence will not be sufficient to compensate for local extinction, the only conclusion is that climate change will determine considerable losses of floristic diversity, added to those that are now being caused by other components of global change. This section highlights the groups of habitats and territories the plant diversity of which could be most rapidly or intensely affected, in accordance with what has heretofore been described.

High mountain

The decrease in the space climatically suitable for the flora typical of many high mountain habitats will cause reductions in their floristic ensembles, particularly relevant because of the high endemicity rate that characterises them. The risks are greater in those mountains that are not excessively high, because the altitudinal interval currently occupied by typically high mountain habitats is smaller. For the same reason, and because of the differential tendencies of the warming projected, the problems will be more serious in southern mountains than in northern ones, and more in Cantabrian Mountains than in the Pyrenees.

Mid-mountain mesophytic habitats

Most supramediterranean and supratemperate mesophytic habitats will also be subjected to the pressure of altitudinal displacement, although in this case, they will avail of higher elevations to spread to, at least in the main mountain ranges. Once again, the main risk for these ecosystems is found in the more modest mountains, such as Sierra Morena, Montes de Toledo and many foothills of other orographic systems and of the southern mountainous regions. While altitudinal displacement is an alternative for subsistence for the flora of these habitats, latitudinal migration will be very problematic due to the transversal orientation of most of the mountain ranges and to the fragmentation and alteration of the intermediate lowlands. Reduction in area and the jeopardization and local extinction of floristic elements therefore constitute foreseeable impacts. This situation is aggravated by the fact that these habitats are characterised by containing taxa at their southern distribution limit that make a substantial contribution to territorial floristic rarity (Fernández-González 1999). The recession of these mesophytic mountain habitats in the South of the Peninsula will also involve a decrease in the abundance of certain woody species with fleshy fruits, with possible repercussions for their dispersing animals. The recession and consequent impoverishment of mesophytic flora in mountain ranges will reduce one of the components of their current floristic richness; to the contrary, they will serve as refuges for the flora of habitats that are now spread to piedmonts and which are sensitive to aridification. Lastly, changes in vegetation in mid- and high mountain areas could involve serious additional risks if the local collapses of certain populations cause temporal decreases in cover, due to the intense erosive energy characterising mountain reliefs.

Warm mesophytic habitats

All Mediterranean vegetation rich in deciduous and lauroid elements associated with rainy climates in winter and spring will be subjected to opposed pressures in the North and the South of the Peninsula. In the North, the accentuation of summer drought will favour their spread from the more or less relictic enclaves they currently occupy, although this advance will be strictly conditioned by the degradation and fragmentation of the territory, forestry uses, competition with invasive species and the extremely high incidence of deliberate fires in some areas. In the South and above all in the Southwest, the lengthening of the summer drought period, combined with an insufficient soil water recharge in wintertime, will be a disadvantage and evidence exists of the sensitivity of Mediterranean deciduous and lauroid flora to the latest drought episodes. The distances are too great for the natural migratory processes of flora in the southernmost habitats. Impoverishment of the warm Mediterranean and mesophytic ensemble can therefore be expected in the South and Southwest of the peninsula. Similar comments apply to the Canarian laurel forest or “monteverde”. Invasive species will also be an important pressure for island communities.

Mediterranean forests and southern dehesas

Large areas in the South of the Peninsula currently occupied by forest will experience a considerable reduction of water availability, which will even fall below the requirements of the least demanding forests. The effects of chronic droughts in these territories have been appreciated in the last few decades. The crisis could be even more acute in the case of *dehesas*, where the problem of tree regeneration will be exacerbated by aridification, a situation that will also question the viability of its traditional agricultural and livestock farming uses.

In the South and above all in the Southwest of the Peninsula, climate change projections indicate the appearance of new types of climate, such as the inframediterranean or the continentalised versions of the thermomediterranean, and of combinations of substrates and climate types heretofore unheard of in Iberian vegetation, such as semiarid and arid ones on siliceous substrates. These new conditions will have a selective effect on the present flora, which will be partially excluded due to intolerance. As the immigration of flora adapted to these situations is unlikely, a “voiding” phenomenon may occur in regional floristic richness, along with impoverishment of communities. In such circumstances, invasibility by exotic species would be exacerbated, and these would include the cactus family and other xerophytic elements, such as certain acacias, which would find space to spread.

Coastal habitats

We have already mentioned the particular risks affecting certain coastal systems, like dunes and coastal pools, which have been accentuated by the long history of human occupation of the coast and the intensification of certain uses in the last few decades. During previous climate changes, the coasts have played an important role as refuges, and they could also play this role at present because the degree of warming is lower and the impact of aridification will be partially compensated for by the greater atmospheric humidity. To the contrary, ozone toxicity is potentially greater in coastal environments. The interactions between climate change and land-use changes will therefore be especially decisive in the conservation of coastal habitats.

5.5. MAIN ADAPTATION OPTIONS

Due to their magnitude and scope, the impacts of climate change upon plant diversity will be difficult to attenuate through the application of local measures. We will now enumerate some of the options to be considered in this respect, encompassed within the framework of the corresponding sectorial policies.

Review of the network of protected areas and of conservation policy

The present network of protected areas, including those proposed for the integration of the European network Red Natura 2000, encompasses a considerable area, although this is mainly concentrated in mountain areas and has big gaps in the intermediate plains and piedmont areas. The conservation of mid- and high mountain areas is a recommendable measure for dealing with the impacts of climate change, as it would favour the altitudinal displacements of flora. However, it is also recommendable to review the delimitation of the protected areas in order to ensure that these cover complete altitudinal intervals, up to the summit lines, particularly in mountains of moderate altitude where the upper vegetation belts have narrow elevation ranges. Furthermore, in these circumstances, the continuous and extensive representation of natural and semi-natural habitats is of special importance, especially if they are located along a broad altitudinal interval, because they will have greater possibilities for conservation, at least in part of the territory. Attention should be paid on the conservation of corridors and the connectivity between protected areas that are too far from each other for the average dispersal thresholds of plant species. In situations of warming and aridity, northern slopes and shaded areas will play an important role as refuges, and will therefore require special attention with regard to conservation. In relation to lowlands, forest habitats should be better preserved, as these will play an important role as nuclei of re-colonisation or as milestones in migratory processes. Mixed forests, with a multi-specific tree layer, will have better chances of conserving their forest structure throughout future climate changes.

With regard to other actions, an effort should be made to harmonise current figures relating to protected natural areas, which are excessively numerous (Gómez-Limón 2000), especially in relation to management, given that conservation of biodiversity will become an increasing trans-border problem. The implementation of the Natura 2000 network of protected areas could constitute an opportunity for advance in this sense. The generalisation of procedures of adaptative management of protected areas (see subsequent section 10) will constitute another vital requirement for the follow-up of the effects of climate change.

As a consequence of climate change, species protection will be subjected to a long readjustment process, which will require monitoring of threatened populations and periodic reviews of red lists and protected species lists. As species with narrower geographic distribution ranges, altitudinal intervals and smaller population sizes presumably have lower levels of phenotypical plasticity and genetic diversity, they are also the main candidates for jeopardization or extinction in situations of abrupt environmental changes. Within the conservation framework, the detection of serious population decline in species that are not threatened at present will require special attention. The regional governments should take on the main contribution to this work.

Supporting *ex situ* conservation techniques could be crucial for the threatened flora. The conclusions of the Red Book (Moreno Saiz *et al*. 2003) revealed that barely 40% of the more endangered species are currently represented by at least one of their populations in germplasm collections.

Ecological restoration and climate change

Ecological restoration should incorporate criteria for the prevention of the effects of climate change. The use of exotic species in revegetation actions should be strictly regulated; indeed, among future restoration measures we can expect the more frequent use of those aimed at eradicating invasive species. Furthermore, restoration of plant cover should consider species tolerant to climate change tendencies and the prevention of generalised failing episodes in unfavourable years.

Review of forestry policy

As in the case of restoration, we should avoid reforestation practices involving species or seedling sources that are intolerant to climate change or to the associated risks of plagues and fires. Mixed plantations are recommendable, in order to prevent failures which are to be expected in climatologically adverse years, the frequency of which can be expected to increase. Criteria based on productivity should be abandoned in many parts of Spain, in favour of ones related to the fight against erosion, post-fire regeneration and the conservation of biodiversity. The use of shrub species in reforestation should become more frequent, for the same reasons and because of the role they play in facilitating the regeneration of Mediterranean woody species (Vallejo *et al*. 2003). Aridification will promote the abandonment of unproductive agricultural land, in which the natural dynamics of secondary succession might be very slow under the new climate conditions; reforestation of these areas with species tolerant to drought and elastic with regard to fires, will be necessary in certain situations in order to avoid degradation tendencies.

Forest management techniques ought to be adapted to a framework in which the risk of erosion, fires, loss of edaphic organic matter and deficient regeneration of forest cover will be critical. Forestry treatments should pay particular attention to enclaves that can play a role as refuges or migratory stages in the displacement of flora, as well as in general to the floristic diversity of the understorey. As we learn more about the processes of vegetation displacement, through techniques of selective extraction and even planned reforestation, we will be more capable of implementing the replacement of forests. A suitable selection of the species to be used in afforestations, in particular those carried out under the CAP frame of subsidies, could contribute to mitigate the migratory constraints of many woody plants, above all in territories affected by aridification and land abandonment. In many Mediterranean types of woodland, there will also be a need for forestry treatments aimed at improving stand structure in order to favour its hydric efficiency (see Chapter 9).

Regulation of livestock farming and hunting uses in forest systems

Excessive pressure by livestock in forests and shrublands subjected to intense hydric stress can accelerate the recession of these due to insufficient natural regeneration and to the damage inflicted upon adult individuals through browsing. The regulation of carrying capacities is important in the areas of Mediterranean woodlands which are still well-conserved and used for hunting, as these have shown recent tendencies towards intensification. In the territories most affected by aridification, however, decreased plant productivity will lead to the economic collapse of extensive livestock farming, which will give rise to the impoverishment of herbaceous flora. The possible increase in livestock farming in high mountain areas could delay the altitudinal progression of forest vegetation, although it would also accelerate the species turnover of the herbaceous flora.

Land use planning and environmental assessment

Several of the adaptational options mentioned need to be planned from the perspective of a type of land planning that takes into consideration the impacts of climate change, in order to minimise the synergic interactions between climate change and changes in uses. Within the new framework for sustainability to be established as the climate changes, environmental impact assessment will have to consider the interactions between the environmental impacts of projects and impacts deriving from climate change. With regard to plant diversity, and within this new framework, we must begin to evaluate impacts such as the fragmentation and reduction of habitats, the introduction of exotic flora, the intensification of uses, excessive water consumption associated with certain uses (see Chapter 3), or effects upon habitats that are unprotected but which could constitute important areas for the redistribution of vegetation. The development of strategic environmental assessments and the application of these to the issue of biodiversity conservation (Díaz *et al*. 2001) are tools that could contribute to the necessary change in scale and perspective.

5.6. REPERCUSSIONS FOR OTHER SECTORS OR AREAS

The ecological role of plants as primary producers means that changes in flora and vegetation will have direct or indirect effects on almost all the sectors considered in this Report. Loss of floristic diversity will affect the different goods and services provided by biodiversity. These losses are especially relevant in the case of Spain, because our country houses a high proportion of Europe's biodiversity as was pointed out in section 5.1. The following is a succinct enumeration of these connections.

Changes in the structure and composition of vegetation in turn have effects on climate, through modifications of the albedo, changes in carbon balances, emission of VOCs, and, through interactions with the fire regime, they also contribute to CO_2 emissions. The relationships between plant diversity and the functioning of terrestrial ecosystems, continental aquatic ecosystems, coastal areas and edaphic and hydric resources in general have been described in the corresponding Chapters (2, 4, 7 and 8) and in section 3. Interactions between plant and animal species are particularly important for both of them, are sensitive to changes in phenology and distribution caused by climate change, and they have been mentioned in Chapters 2 and 6 and in section 3. The relationship with natural risks is mainly established through fires and surface erosion processes (Chapter 12). Phenological modifications, changes in plant productivity and in the chemical composition of phytomass and flora displacement will have repercussions for the agriculture and livestock farming (Chapter 10) and forestry (Chapter 9) sectors; in turn, the interactions between changes in the uses associated with these sectors and climate change, constitute one of the most important determinant factors of the future dynamics of plant diversity. According to the World Health Organisation (WHO 2003), one of the consequences of climate change that is insufficiently known and which requires continuous follow-up refers to changes in air pollution and the associated aero-allergenic levels. With regard to the latter, changes in the flowering schedule of many species (García-Mozo *et al*. 2002a) would give rise to a prolongation of pollen in the air, with the consequent repercussions for health: an increase in the number of patients with allergy and in the duration of the allergic symptoms, the associated increased pharmaceutical expense, higher work and school absenteeism, increased hospital emergency admissions and even changes in the temporality of tourism. This health problem has implications at local, regional and even national level. Also related to tourism, it should be pointed out that certain degradation processes of plant cover that could be caused by climate change, such as deforestation caused by drought, plagues or fires, erosion, homogenisation of landscapes, etc., would involve losses of natural attraction for potential visitors, and this could locally affect tourism demand and the economy that depends on this sector.

5.7. MAIN UNCERTAINTIES AND KNOWLEDGE GAPS

Uncertainties involving the possible impacts of climate change are mainly based on our ignorance of how the different factors in play will come together in time. Thus, for example, we avail of reasonably reliable climate series that describe the situation of climatic elements within one century, but effects on plant diversity might be substantially different, depending on the temporal trajectories of the changes, especially in relation to rainfall. The incidence of extreme events and changes in climatic stochasticity are more difficult to accurately predict, but, from what we know, these direct and indirect effects could be more intense and irreversible than those produced by average climate tendencies (Walther *et al*. 2002).

Synergies between climate change and other components of global change (related to atmosphere and land uses) constitute another of the elements of uncertainty. The signs indicate that the latter will generally accentuate the impacts of climate change, especially in relation to aridification.

Responses by species also constitute another source of uncertainty, due to the variability to be expected, which has been verified in the sign and magnitude, and in the response times. The high number of species involved increases this uncertainty, and this is an aspect that therefore requires great effort with regard to research, which should foreseeably be based on the definition of the functional attributes of species conditioning their types of response. The uncertainty is greater in that species interaction will play a decisive role (see section 3.1), because the modification of these interactions can in many cases cancel out the individual responses to be expected.

Lastly, the temporal and spatial scales of the impacts represent another of the challenges in research of climate change. Most of the information on biological responses has been obtained at individual, population or community scales, but effects at landscape or regional scales are more difficult to model, and therefore, to predict. Furthermore, the spatial resolution of future climate projections have not yet reached the detail of the mesoclimatic variations induced by topography and relief, which could be vital for the subsistence of communities and species in refuge enclaves.

5.8. DETECTING CHANGE

The signal of climate change confirmed to date is still moderate (see section 5.3.1) and the effects are therefore beginning to be detected only partially. The most extensive and tangible evidence concerning plants in our latitudes refers to changes in the phenology of certain species and to the growth patterns detected in the annual rings of trees (Hughes 2000, Walther *et al*. 2002, Gitay *et al*. 2002). There is also evidence of changes in the composition of certain communities, of movements of flora, both autochthonous and invasive, and even of the virulence of certain plagues, which have been interpreted as consequences of climate change. This evidence, referred to in section 5.3.1.4, is often difficult to separate from other impacts of global change and, in any case, there is not yet a sufficient body of data to constitute an indicator. We also avail of studies that should how the impacts of climate change can occur in plants, and of evidence of the effects of other components of global change (chemical composition of the atmosphere, changes in land uses) on plant diversity, which are also referred to in section 5.3.1.

The magnitude and complexity of the foreseeable impacts therefore call for more detailed follow-up, in order to confirm responses expected and to predict the unexpected ones in time. The indicators of change assigned here are those that can provide information that is disassociated from the effects of other components of global change, for which previous series of data are available to establish references and projections, and that can provide in the short or medium term values indicating the magnitude of the effects of climate change upon plants. Furthermore, the interpretability of these data, the existence of international databases, the comparability of these and the validation methods are now sufficiently established or can be developed with little effort (Erhard *et al*. 2002). Other indicators the future development of which is considered to be feasible and recommendable are described in section 5.10.

Phenological databases

Different phases of the life cycle of plants depend on the annual course of the climate, especially in extratropical climates. This physiological triggering of these phases is controlled by temperature in many plants, and the response by these to climate change is one of the first ones to be recorded. It should be kept in mind, however, that, depending on the species and life cycle phases considered, this control could be exercised by other factors, such as photoperiod, water availability or even by combinations of factors (Larcher 1995).

But for sensitive species, flowering and foliation have been advanced in the last 3-5 decades at a rhythm of between 1.4-3.1 days/decade in Europe and somewhat less (1.2-2(3.8)) in North America (Ahas 1999, Chmielewski and Rötzer 2002, Menzel and Estrella 2001, Menzel *et al*. 2002, Walther *et al*. 2002). The changes are less consistent in the summer or autumn phases (Bradley *et al*. 1999, Menzel *et al*. 2001, 2002), but in Europe, the withering of the leaves of deciduous species is being retarded by between 0.3-1.6 days/decade. As a result, the period of vegetative activity has been increasing by an average of 3.6 days/decade over the last half century (Menzel and Fabian 1999, Peñuelas and Filella 2001). These values do not deviate greatly from those deduced from the analysis of vegetation growth periods through satellite images (Myneni *et al*. 1997) and of annual cycles of atmospheric CO_2 concentrations (Keeling *et al*. 1996), which also indicates that the lengthening of the period of vegetative activity has become more evident in the last two decades. Phenological modifications do not occur in a territorially homogeneous fashion, and allow areas to be detected in which warming does not occur at the same rhythm, as happens in parts if SE Europe (Menzel and Fabian 1999, Chmielewski and Rötzer 2002), or even cold spells recorded throughout the last century (Ahas 1999). The data available in NE Spain (Peñuelas *et al*. 2002) show advances of 1-5 weeks in foliation and of 1-10 weeks in flowering for the last half century, and delays in leaf shedding of 1-2 weeks, with mean values in the response that should be considered high in comparison with those of the European context. The responses are very variable among species; with regard to phenological advance, herbaceous and early flowering plants respond better than woody species and, in general, late flowering ones (Post and Stenseth 1999).

Thus, phenological studies are a powerful tool for the follow-up of the biotic response to climate change and, in spite of the fact that classical registers of phenological data have been criticised by different sectors of the scientific community, they have now been very much revitalised for this reason (Lieth 1997, Schwartz 1999, Wuetrich 2000, McCarty 2001, Menzel 2002). In the European framework, the main initiative in this context is the *European Phenology Network*, EPN, www.dow.wau.nl/msa/epn), created for the standardisation of procedures (Bruns and van Vliet 2003), for the co-ordination and integration of information and work groups (Bruns *et al*. 2003, Schwartz 2003) and for the development of education programmes. An attempt has been made to promote international phenological follow-up networks by GLOBE (*Global Learning to Benefit the Environment*) (Peñuelas and Filella 2001).

Aerobiological databases

Aerobiology is being used as a tool for the study of the male floral phenology of many plants pollinated by wind, due to the fact that the presence of grains of pollen in the air is a consequence of the phenological state of flowering. In this sense, aerobiological databases are proving to be useful for these studies, because temporal series of at least 20 years are available in several countries. Aerobiological data are collected on a daily basis with devices that suck constant volumes of air, which allows the evolution of floral phenology to be studied in detail. These apparatuses have a weekly autonomy and a cover of around a 50 km radius, depending on the topography and the capacity to disperse pollen grains of the species to be studied.

The Spanish Aerobiology Network (REA, Red Española de Aerobiología) manages about 50 sampling stations distributed throughout the country. Recently, it has been created the Red Portuguesa de Aerobiología (RPA, Portuguese Aerobiology Network) which, together with the REA, provides knowledge of the airborn pollen content throughout the whole Iberian Peninsula. Although the REA was formally set up in 1992, at certain sampling points it avails of continuous data from 1982. This database is highlighting the sensitivity of the different species to climate change.

Herbaceous species have a different behaviour pattern than woody plants. In the Mediterranean area, development and flowering depend more on rainfall regime, and the effects of temporal distribution are more important than the total quantity recorded (Emberlin *et al*. 2000). The influence of climate change has not been very marked up to the present, as the main changes have been seen in temperatures. A change in rainfall regime towards a greater torrentiality or a seasonal redistribution of rainfall would increase the effects.

With regard to woody species, temperature is the factor which most affects the development of the flower buds of trees that flower in early spring, such as the Holm oak, whereas the trees that flower later, like the olive or the cork oak, are more affected by the photoperiod (García-Mozo *et al*. 2002; Galán *et al*. 2001, 2004; see also Osborne *et al*. 2000). An advance in budding date of the trees and of flowering in general could make them more vulnerable to late frosts. An increase in temperature would accelerate the development of buds and flowers. Likewise, different responses have been observed in one same species depending on the geographic area, which is to be expected if we take into account the different magnitudes and directions of climate change according to territories.

At the European level, work is being done in this line using aerobiological data, through a European network, the *European Aeroallergen Network* (EAN), to which the REA belongs, and which is also co-ordinated with the EPN. Different studies have highlighted a generalised advance at European level of the pollen season of different species (D'Odorico et al. 2002, Emberlin et al. 1997, 2000). Most of the available pollen series in Spain are still too short, but they appear to indicate an advance in the flowering of many tree species. This advance is more consistent for the pollen season of *Quercus* at inland sites of the Peninsula (García-Mozo *et al*. 2002a, b). Using aerobiological and meteorological databases it has been possible to build models for the climatic determinants of flowering of some species. Applying these models to climate projections for the next century, it has been foreseen advances of up to 6 weeks in flowering time for early-flowering oaks in areas of central Spain like León or Madrid.

The increase in productivity induced by the elevation of CO_2 atmospheric concentration also implies a higher production of pollen by anther, as it has been experimentally proved (Rogers *et* *al*. 2004). Pollen series from different European stations show an increasing trend in the airborne pollen concentration. Such evidence is not clear in Spain still, probably due to the high interannual variability which will require the availability of longer series of pollen data.

The aerobiological data can be shown by means of tables and graphs that are easily interpreted by the people in charge of evaluating the results of the follow-up. The REA has much experience in the dissemination of aerobiological data, this being one of its main objectives since it was set up. Data must be statistically analyzed to prove that tendencies are not the result from interannual fluctuations of climate.

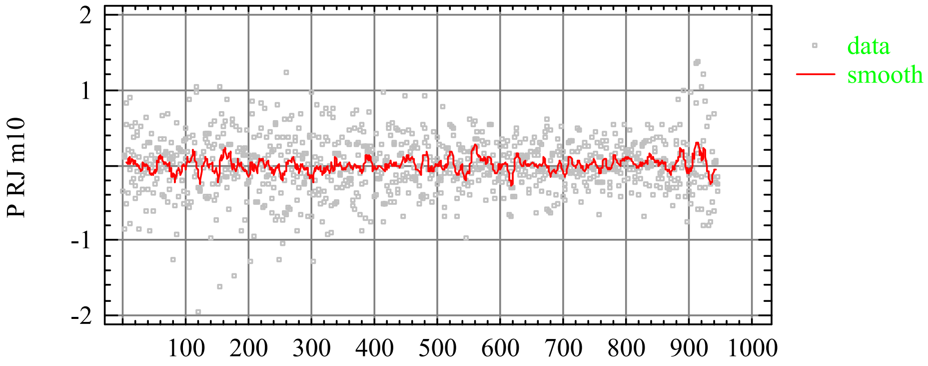
Dendrochronological database

Growth in thickness of trees, calculated using the annual growth rings, is related to climatic conditions, and can be used to confirm the effects of climate change. Dendrochronological and dendroclimatic studies are relatively recent in Spain, as they practically began with the studies carried out by Creus and Puigdefábregas in 1976 and 1983. Since 1984 dendroclimatology has been developed through studies of species of *Pinus* and of deciduous *Quercus,* apart from some other genera of conifers and of the oak family. Among the researchers who have most contributed to these activities, we can highlight J. Creus, A. Fernández-Cancio, R. Génova, M. Génova, E. Manrique, K. Ritcher, E. Trobajo, E. Gutiérrez, and A. Pérez-Antelo.

The dendrochronological database used as a reference for researching climate currently has 1,064 series, of which 940, corresponding to 574 trees, are of great interest due to their longevity and sensitivity (Table 5.6). The chronologies cover a time interval from 1050 to 1997. In dendroclimatic studies, the information supplied by the weather stations started to be of use in the year 1945, and the 1945-1997 interval was used to calibrate and verify the models. Spatially, the cover of the Peninsula is complete, as a new methodology has been developed (Fernández-Cancio and Manrique 1997) which, using the classical reconstruction systems of the Tree Ring Laboratory (Tucson, Arizona), allows for the reconstruction of any monthly meteorological variable in those stations that have at least 50 years of instrumental records.

This method is based on the selective reconstruction of each variable, correlating it with the total from the database. The model applied is calibrated and verified and its statistical reliability is analysed through comparison with random simulations from climate records, according to the distribution of the data on the real variable. Thus, 57% of the variables can be reconstructed in a reliable manner. In the year 1300, the variance started to become stable, and after 1500 the reconstructions are valid for the whole of the territory. The scope of the climatic signal for temperatures is very great, as it reaches the Canary Isles in the South and it has not been determined to what part of Europe in the North. For precipitation, however, the degree of reliability is much lower. The Little Ice Age (LIA) is completely defined, as is the evolution of the climatic variability (Manrique and Fernández-Cancio 1999, 2000). Spatially, the whole Iberian Peninsula can be reconstructed (Candelas 2000).

The signal (Fig. 5.7) tallies quantitatively with the tendencies detected in the climate evolution of the Northern Hemisphere (www.ngdc.noaa.gov/paleo/recons.html). Some of the chronologies obtained have been incorporated into the Geophysical Database of dendrochronological character of the NOAA *Paleoclimatology Program* through the *International Tree Ring Data Bank* (ITRDB). As can be seen in the standardised reconstructions of the millennium in 320 Spanish stations (Fig. 5.7), the precipitation and temperature episode in the last half century, essentially warm and alternatively humid and very dry, appears to show unique characteristics in Spain with regard to tendencies and variability of the last 1,000 years. From this point of view, we can speak of present climate change in relation to the tendencies of the last millennium.



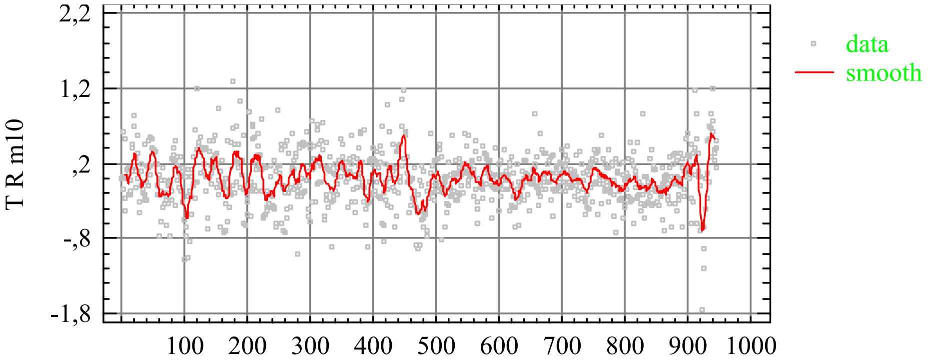


Fig. 5.7. *Representation of precipitation (above) and mean temperature (below) standardised and dendrochronologically reconstructed for 320 stations and their moving averages over 10 years. The temporal interval reconstructed (in abscissas) covers the period from 1050 to 1997. Patterns of high variability are observed from 0 to 400 and the anomalous behaviour of the last 50 years, which corresponds to real data.*

Table 5.6. *Zones and characteristics of the dendroclimatic database*

Zone	Number of series	Number of trees	Species	Maximum length of the series (years)
Guadarrama	259	134	*Pinus nigra*	505 (La Jarosa)
Cuenca	150	91	*Pinus nigra*	629 (Uña)
Galicia	59	40	*Quercus spp.*	432 (Invernadero)
Gredos	43	22	*Pinus nigra*	322 (Piedralaves)
Demanda-Urbión	129	76	*Pinus uncinata*	528 (Vinuesa)
Pyrenees	98	65	*Pinus uncinata*	690 (Aigüestortes)
Alcaraz-Segura	74	50	*Pinus nigra*	409 (Paterna)
Cazorla-Filabres-María	60	40	*Pinus nigra*	939 (Sierra de Cazorla)
Teruel	68	56	*Pinus nigra*	410 (Bellena)

It is possible to locally identify, in each weather station, the behaviour and evolution of the present climatic episode. The additional reconstruction of the climate enables us to detect the impact on vegetation, and, as a result, the changes tendencies therein can be analysed, simulating current and future displacements. An ARIMA analysis of the temporal series also enables us to get a qualitative idea of the predictions over the next ten years. The managers of the forestry, hydrology and population protection sectors could be the main beneficiaries of these predictions.

Remote sensing

The information provided by the remote sensors installed in the satellites has multiple environmental applications, among which are recent and important contributions regarding the impacts of climate change on vegetation. Analysis of the temporal series of the standardised difference vegetation index (NDVI) has confirmed the lengthening of the vegetative period in the northern hemisphere (Myneni *et al*. 1997, Chen and Pan 2002). Available since the 70s, the series of satellite images also enable us to quantify, with increasing resolution, changes in uses and plant cover in territories, and through these, changes in landscape structure, one of the indicators of biodiversity considered to be most promising (Fernández-González 2002). With the use of temporal series, biogeographic limits can be characterised that are independent from the different types of present vegetation (Lobo *et al*. 1997), and these could be used in the follow-up of vegetation displacements brought on by climate change. With modern sensors, it is becoming possible to evaluate with increasing resolution the water content, the physiological state or the growth of the vegetation. Technology based on remote sensing therefore constitutes and important tool for the follow-up of the impacts of climate change.

5.9. IMPLICATIONS FOR POLICIES

The objective of reaching sustainable forms of development has met with a big problem in climate change. The European Union recently established, in the Göteborg summit (2001), the commitment of halting losses of biological diversity within its territory by 2010, as an inherent implication of environmentally compatible development. Avoiding the added losses of biodiversity derived from the impacts of climate change, a global conflict requiring global measures, is a challenge difficult to deal with through local measures, the efficiency of which is always limited. The sectorial policies affected by the impacts of climate change on plant diversity were mentioned in section 5.5 and are fundamentally related to the responsibilities of the regional autonomies or local administrations. Thus, on one hand, the strategies designed in this respect require a broader geographic framework in which they can be co-ordinated and integrated. On the other hand, within the framework of the initiatives deployed by the EU, the large proportion of European biodiversity contained in Spain should be highlighted. The magnitude of the foreseeable impacts of climate change make advisable a thorough revision of the Spain's Strategy for conservation of biodiversity (MIMAM 1999).

Land use planning should incorporate consideration of the impacts of climate change, at least in an attempt to minimise the negative interactions between these and changes in land uses, in particular with regard to the effects of infrastructures and housing on the fragmentation and degradation of habitats and the geographic displacements of other uses brought about by climate change. The management of natural resources is faced with considerable changes in the productivity of forestry, agriculture and livestock farming, in the future location of these and in their modalities of intensification, in contexts that will be difficult to sustain in many cases. The environmental quality and evaluation area will also have to assume the new framework of interactions generated by climate change; the application of strategic environmental evaluations, instead of evaluations of the individual impact of each project, could contribute to the change in perspective.

The protection of nature, together with ecological restoration in the broad sense, is the sectorial policy most directly involved in the impacts of climate change on biodiversity, as has been indicated in section 5. These impacts are starting, and will do so to a much greater extent in the future, to throw light upon the value of ecosystems and the direct and indirect benefits the offer to humankind, which, unfortunately, are too often underrated, especially with regard to economic appraisal. It is therefore the time to call attention to this through environmental education and awareness programmes, in order to pave the way for sectorial policies capable of attenuating impacts and to involve administrations and citizens in their development and implementation.

5.10. MAIN RESEARCH NEEDS

Research into the impacts of climate change on vegetation can be structured into three main, interconnected lines: follow-up of the real changes occurring or that will occur, the responses by species and communities to foreseeable changes and the design of predictive models, based on the information provided by the previous ones and on climate projections, in order to anticipate the floristic and plant changes to be expected. The continuity of the follow-up will allow the degree of fit of the predictions by the models to be contrasted, and to introduce, where necessary, and with sufficient advance, measures for correction or attenuation of the impacts.
Follow-up activity currently comprises several unequally developed levels. We should promote the participation of research groups in the phenological follow-up networks, given that current participation is very low (for instance, there are only two groups in the EPN, apart from the REA). Furthermore, it is unlikely that research groups will become involved in long-term follow-up without any guarantee of financial backup. With regard to aerobiological follow-up, although there are a large number of aerobiological stations in Spain run by the REA, there are also big geographical areas that are insufficiently covered, such as Castilla y León and Castilla-La Mancha. This, along with the costs of maintaining the collector equipment and in particular of the interpretation of the samples, which require many hours of analysis at the optical microscope, constitute the funding needs of the network, which have been heretofore provided by the public administrations in exchange for the information generated for alergolology applications.

The possibilities of applying the dendrochronological database as an indicator of climate change still require methodological developments. Full comprehension of the scope of the present disturbance and of its characteristics can be attained by seeking more long-lived trees over 500 years, and by analysing fossil and sub-fossil wood, with which to strengthen the signal of the warm early-medieval period (1000-1300 AD) and the preceding big climatic episode (600-1000 AD).

Apart from the development of follow-up based on remote sensing, as was pointed out in section 8, follow-up in the field of floristic changes and changes in vegetation constitutes another vital element. The most interesting approach would consist of articulating a network of plots for permanent follow-up, representative of different habitats and landscapes with differing degrees of vulnerability to the effects of climate change, and suitably distributed according to Spain's biogeographic units. The network of protected areas could constitute a framework for the location of the follow-up points, because these provide an acceptable level of control of land uses and avail of backup cartographic and floristic information. The initiative could be co-ordinated with the follow-up plans required by the Natura 2000 network and could benefit from the possibilities of co-funding by the EU. Apart from periodic sampling of floristic diversity at different scales of spatial resolution, the incorporation could also be contemplated of the phenological and demographic follow-up of keystone species or of those selected and representative of different models of response to climate change. The follow-up procedures should be able to be linked to procedures of adaptative management in order to evaluate the effects on biodiversity of specific actions and the interactions of these with climate change. About the monitoring of high mountain habitats the EU research project GLORIA-Europe is functioning with a network of permanent plots in the summits of 17 European ranges, including Sierra Nevada and the Pyrenees (Pauli *et al*. 2004; www.gloria.ac.at). This initiative is been enlarged at the global scale through GLORIA-Worldwide. The network of plots for the follow-up of forest damage (Montoya and López Arias 1997), should also maintain its activities, which are necessary for evaluating the effects of droughts and plagues.

In the same sense, and in the frame of the Atlas of Threatened Flora (Moreno Saiz *et al*. 2003), a project of population monitoring has begun for 40 endangered plant species, which will allow to detect future trends in these species and draw conclusions and management recommendations. This demographic monitoring should be accompanied by genetic studies (breeding systems, levels of endogamy, genetic diversity within and among populations) to properly evaluate the vulnerability and the importance for conservation of the different populations (Hampe 2004). Another complementary network we could consider is the one dealing with the follow-up of post-fire regeneration after forest fires, because big changes are to be expected following these disturbances.

In dealing with the responses by species and communities to climate change, experimental and fieldwork approaches are needed. The latter, like the aforementioned follow-up systems, should have a long-term focus, due to the characteristics of climatic processes and to the fluctuating nature of Mediterranean climates. The lack of this type of long-term approach in our country has been pointed out by numerous researchers (Moreno and Fellous 1997, Herrera 2001, Zamora *et al*. 2001, Hampe 2004). Modification of inter-species interactions, the characterisation of functional groups of plants with similar responses to climate changes and the evaluation of indicators of the effects of climate change on biodiversity (Díaz 2002, Fernández-González 2002) are some of the priority themes in this respect.

Lastly, the development of predictive models of the dynamics of floristic diversity under the pressures of climate change, will constitute the most elaborate tool for generating projections, designing mitigation measures and evaluating their efficiency (Nualart 2003, Thomas *et al*. 2004). The models should progressively incorporate resolutions at landscape and regional scales, the effects of fragmentation, capacities of the species for dispersal and migration, indirect effects of climate change and interactions with other components of global change. All these respects need technical development. The reliability of the projections depends upon the quality of the information about floristic and vegetational diversity. Databases on the distribution of floristic and plant diversity in Spain is beginning to gain importance, although there are still shortcomings in relation to the spatial resolution of the data, and above all, in the compilation of the very dispersed documental sources coming from publications, cartography and scientific collections (herbaria). Support is needed for the initiatives aimed to computerize this kind of data, among which are at the national level the project Anthos (www.programanthos.org), carried out by the Botanical Garden of Madrid, that has compiled more than 700000 floristic records in Spain, and the Data Bank on Biodiversity of Catalonia (BDBC, http://biodiver.bio.ub.es/bioca), with similar aims but restricted to the Catalonian countries. At the international level are the Global Biodiversity Information Facility (GBIF, www.gbif.org) and the Biological Collection Access Service for Europe (BIOCASE, www.biocase.org). The Atlases of Threatened Flora and of Natural and Seminatural Habitats, promoted by the Direction General for Biodiversity Conservation (Ministery of the Environment), constitute other relevant milestones in this direction, which still requires the compilation of the huge phytosociological databases, only in part published, and the widening to other componentes of vegetal diversity. The usefulness of these data in studies about the impacts of climate change will require a good accuracy in both georeferentiation, with spatial resolution at 1 km^2 or more detailed, and in chronoreferentiation (date) of the records.

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