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Influence of fire on the stability of mediterranean forest ecosystems

GIENS (FRANCE) 23-26 /03/1987

Editors :
Ph. BOURDEAU, Ch. ROLANDO and A. TELLER

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FERN - EUROPEAN SCIENCE FOUNDATION
GIENS WORKSHOP (23 - 26/3/1987)

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FOREWORD

This Workshop was organized in the frame of the "Forest Ecosystems Research Network" (FERN) of the European Science Foundation.

In March 1986, when the FERN project was launched, it was decided that one Working Group should focus on the problems specific to the Mediterranean forest ecosystems, mainly fire and grazing. Professors P. Quézel (France), A. Escarre (Spain) and Ph. Bourdeau (CEC), who were appointed as co-ordinators of this Working Group, decided to organize a Workshop on "Influence of fire on the stability of Mediterranean forest ecosystems", which took place in souhtern France (Giens) in March 1987. Since the problems of grazing are complex and not really typical of European forests, the emphasis of the Workshop was to be on fire, whose role and implications are well known in the Mediterranean Basin, and on which important documentation exists.

Fire has always been a major problem in the Mediterranean Basin. Most plants of the Mediterranean vegetation possess strategies to withstand and survive fire which has shaped the Mediterranean landscape (e.g. matorral in Spain, garrigue in France, macchia in Italy, phrygane in Greece). This problem is all the more acute since the present abandonment of marginal land by farmers and shepherds as a result of the rapid increase in living standards and the ensuing rural depopulation.

The objective of the Workshop was to review European relevant research and to identify the gaps and priorities of future research to be undertaken.

The programme of the Workshop was sub-divided into four themes of discussion :
- Fire in the Mediterranean region : past and present evaluation ;
- Effects of fire on structure, dynamics and regeneration of forest ecosystems ;
- Effects of fires on soil and recolonization by animals of studied biotopes ;
- Comparative evaluation of ecological effects of prescribed and uncontrolled burning.

This sub-division corresponds to the different Chapters of these proceedings. For each area, a group of experts was appointed to review and to present relevant research potential and achievements in southern Europe. The contributions are printed in the main section of this book.

The Workshop was attended by over 30 scientists from Mediterranean countries, i.e. France, Italy, Spain and Portugal. On this occasion, the need to develop interdisciplinary research projects in which scientists, with different backgrounds and expertise, are working in close connection was underlined. Observations on data generated in an isolated field experiment are of little value if there is no possibility to relate these observations and data with others made in different conditions (e.g. wild fires, prescribed burning). Within a site, data to link post-fire observations with fire conditions are needed.

FERN could improve this situation by favouring contact between multidisciplinary teams, and supporting co-ordinated projects. Therefore, participants recommended the initiation of a European research project on experimental fires and their ecological implications. In order to set up a common protocol for future experimentation, Dr Rego (Portugal) is willing to organize a demonstration of a prescribed burning experiment in February 1989 with all interested research teams. It may be mentioned that one of the positive outcomes of this meeting has been the recent funding by the Environmental Research Programme of the Commission of the European Communities of two collaborative projects on forest fire prevention in Mediterranean countries (e.g. Italy, France, Greece and Portugal).

A great part of the success of this meeting was due to its careful organization by Professor Quézel and his collaborators in Marseille, whose help is gratefully acknowledged.

We hope that this publication will initiate further study and experimentation on Mediterranean forest ecosystems and will reinforce multidisciplinary co-operation among European research workers so that a better management of southern European forests can be achieved.

Ph. BOURDEAU
Chairman, Steering Committee
FERN (ESF)

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The role of fire from palaeoecological data.

A. PONS[*]
M. THINON[**]

SUMMARY - Pollenanalytical and pedoanthracological data show that burning has been a constant means for man to clear areas necessary to his activities for at least five millenia. The changes induced by this practice are so far-reaching that modern ecological analysis cannot determine the real vegetal potentialities in western Mediterranean regions.

KEYWORDS : Pollen analysis, archeoanthracology, pedoanthracology, palaeoecology, burning, charcoals, timberline, forest potentialities, pastoralism, Mediterranean region.

Besides written documents, two palaeoecological branches provide information on the role of fire in a distant past : anthracology and pollen analysis.

Anthracology covers two fields of research :

- archeoanthracology studies burnt woods from bivouacs or habitats as well as wood remains of tools, weapons, furniture or buildings from the prehistoric past;
- pedoanthracology studies more or less carbonized wood pieces buried in soils and not related to human settlements.

Archeoanthracology mainly informs us about wood species utilized by man at the different stages of his history either as a source of energy or as a material. Through the knowledge of the different species involved, a conjectural reconstruction can be made of the flora and - with more uncertainty - of the surrounding vegetation : but it does not provide any data on fires.

On the contrary, pedoanthracology deals with direct evidence of past fires. It informs us about burnt plants and sometimes local vegetal formations formerly destroyed by fire, and thereby also about the activities related to those fires.

Pollen analysis makes it possible to reconstruct the past regional vegetation through the study of the spores and pollen contents of sediment layers in suitable sites (humid places favourable to the deposition of organic material).

I - THE ROLE OF FIRE IN THE REMOTE PAST (A.P.)

Though pollen analysis can evidence and accurately describe changes in vegetation, it gives but indirect inferences concerning the precise role played by fire in clearings. There are two criteria which permit the presumption that clearings can be attributed to fire over more or less long periods : first the absence of selection in the species that are victims of deforestation - all the components of the vegetation are affected in proportion to their initial importance -, second, the more or less transient occurrence of interim species known for their great capability of germinating and growing on burnt soils (Pinus, Betula), added to a marked heliophily. Often, the presence of a few pollen grains from herbaceous plants more or less related to man's agricultural activity, suggests a human origin for this phenomenon.

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Sometimes, the levels involved are rich in microscopic carbonized wood remains, which confirms this assumption.

The most detailed pollenanalytical study of man's intervention on vegetation through fire in a precise region was carried out in Denmark - some 40 years ago - by Iversen (1941 and 1949) and his collaborators. The conclusions directly relevant to this Symposium are the following :

- the impact of man on vegetation was on the whole extremely localised and transient up to the time when he became a stock breeder and/or a farmer;
- in order to find space for agricultural or pastoral activities, man set fire to the natural forest, which burnt all the more easily as it contained a great amount of dead wood (on a scale on which no forest today can give an idea) ;
- there is a disproportion between the meager prehistoric populations and the large tracks of land that were burnt because the exploited areas were moved from one place to another every 3 or 4 decades, just the time necessary to take advantage of the illusory fertilisation resulting from the rapid mineralisation of the forest biomass.

Considering that the Mediterranean climate is more constraining for wood species than the Oceanic climate, it is _a priori_ obvious that if the first conclusion may well be questioned in our region, the two others are quite relevant and can even be amplified.

Therefore, from the discussion of these three points and in view of the set of palaeoecological and especially palynological data available, an attempt can be made at a chronology of fire and a global estimation of its role in the Mediterranean region.

A - The chronology

1) Practically no palynological or anthracological data can conclusively prove that fire played a decisive role in the botanical period well before the Neolithic development of stock-breeding and agriculture (as land uses). This excludes the possible occurrence of natural fires, even during periods with an incontestable summer drought, such as the first part of the Postglacial (between 10,000 and 6,000 Before Present) or the Lateglacial Interstadial (between 13,000 and 11,000 Before Present). In the same manner, it precludes the idea of an "accidental" incendiary action of man, unknown in the rest of Europe, but which could be related to the Mediterranean summer drought.

2) However, at relatively early periods, transitory clearings in deciduous forests are reported which present the dynamics of classical _landnams_ (a Scandinavian word for transitory clearings) through fire, but are not associated with any agricultural evidence (lower Rhône valley, about 8,300 B.P. : H. LAVAL-TRIAT 1979). There are some indirect arguments that these interventions were only aimed at creating pasture land.

But in the present state of research, it is not possible to detect clearings resulting from fires set in the Near East more than 10,000 years ago in order to gain areas for agriculture and stock breeding (MOORE 1979). In particular, the deforested aspect of vegetation known in several sites of the Near East during the Lateglacial interstadial - towards 12,000 B.P. - cannot be related to fire : it is accounted for by a rise in temperature not associated with a proportionate rise in precipitation (BOTTEMA & VAN ZEIST 1981).

3) In Italy and south-eastern France, much earlier than in the rest of Europe, classical _landnams_ occur, resulting from stock breeding, cereal cultivation and/or to favour wild edible herbaceous heliophytes (vetches and peas). At Courthezon, in the lower Rhône valley, a soil horizon dated slightly after 7,350 (+/- 170) B.P., with a coal level at its basis, reflects the transitory clearing of a deciduous _Quercus_ forest; moreover, the horizon contains a few pollen grains of cereal* as well as consistent percentages of pollen grains of _Labiaeae_ and _Papilionaceae_, the usual accompanying species in cereal cultures of prehistoric times or antiquity, and grains of _Plantago_ and other ruderosegetal herbs (LAVAL-TRIAT 1979). Similar facts have been reported from other sites between ca 7,000 and 5,000 B.P. - for example in the middle Durance valley (BEAULIEU 1977).

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* All cereals are autogamous (except rye) and produce very few large and poorly dispersed pollen grains.

These clearings always concern very small areas where the abandonment of cultivation causes the brief and sudden occurrence of heliophytes (Pinus and Corylus, Juniperus). In all the cases the re-establishment of the initial forest occurred rapidly and without any difficulty.

4) Things were different with the increase in clearings for pasture and culture land which started at about 4,500 - 4,700 B.P. This happened in numerous sites in plains and on low plateaus, and in some mountain areas (from the Rif to the western Taurus but also in the Southern Alps, Corsica and south-eastern Greece) (REILLE et al. 1980). The recovery is not always perfect : there is no substitution of formation but the vegetation that occurs after the clearing episode is not quite the same as before. New species -for example sclerophilous oaks - intrude among the dominant species.

The result of this transient neolithic deforestation was aggravated every time it was repeated in the course of the Older and Middle Bronze Age centuries.

5) The next stage corresponding approximately to the younger Bronze Age and Antiquity in the classical sense, is marked by a generalisation of clearings obviously by using fire (REILLE et al. 1980). It began at different times depending on places : slightly after 3,200 B.P. in north-western Greece, towards 2,800 B.P. in Corsica, 2,700 B.P. on the Dalmatian coast ... until the Roman apogee around the onset of the Christian era.

This stage, which probably corresponded everywhere to an increase in population, ended with the fall of the Roman empire. During the five to seven centuries preceding the medieval renewal (from the Arab invasions to the monastic impulse) incontestable forest dynamics played its role undoubtedly. This enables us to make a first statement of the action of fire which will become more and more severe and act in different ways over during the last millenia (THINON following paper).

B - Survey of the action of fire in the remote past

It is marked by three points :

1. The most general fact is the decline of deciduous Quercus forests in regions corresponding today to the subhumid mediterranean climate : the successive sudden decreases of the Quercus pubescens forest were generally to the advantage of the sclerophyllous oak forests (that is to say Quercus ilex + Quercus coccifera) in Provence (TRIAT-LAVAL 1979) and Istria (BEUG 1977). Quercus ilex and Buxus sempervirens in Languedoc (VERNET 1973), whereas in the Rif Quercus ilex (including Q. rotundifolia) and Q. suber forests (REILLE 1977) extended to the detriment of Q. canariensis and Q. toza. In Kroumiria, at lower elevation, the almost total extinction of a Q.canariensis forest favoured the expansion of Q. suber (BEN TIBA and REILLE 1982) and in Dalmatia the clearing of the deciduous forest, at first beneficial to Carpinus orientalis, then promoted the establishment of a vegetation with less and less deciduous taxa and more and more Pistacia, Phillyrea and Juniperus (BEUG 1967).

2. In Mediterranean mountain regions, other important formations receded more or less : Cedrus atlantica forest in the Rif (REILLE 1977), the High Atlas (REILLE 1976) and central Tunisia (region of Sakiet Sidi Youssef : THINON unpublished). Juniperus thurifera formations in the High Atlas (REILLE ibid.). Fagus sylvatica and Abies alba forest in Corsica (REILLE 1975) and in the moutainous fringe of Provence (TRIAT-LAVAL 1979), mixed forests with Cedrus libani in Anatolia (VAN ZEIST et al. 1975).

3. A first expansion of Pinus formations is evidenced. They are Pinus pinaster forests in the High Atlas (REILLE 1976) or Corsica, (REILLE 1975), Pinus halepensis in Languedoc (VERNET 1972), Provence (TRIAT-LAVAL 1978), Dalmatia (BEUG 1967), Pinus nigra in Anatolia (VAN ZIEST et al. 1975). Pinus sylvestris on the Massif Central Mediterranean border (REILLE & PONS 1982, BEAULIEU, PONS & REILLE 1984).

The role of Mediterranean shrublands is more difficult to apprehend in this first survey : most of the plants that constitute them produce a rare or a poorly characterized pollen. There are two arguments suggesting that they had a first notable anthropogenic expansion :

- they maintain fairly constant sociological relationships with Pines.
- archeoanthracology (VERNET 1973) as well as the study of plant remains from travertine (PONS 1969) show a significant increase in the proportion of shrub taxa remains.

Moreover, in regions with semi-arid and a fortiori arid Mediterranean climates, it is difficult as yet to outline the history of fire and hence to describe objectively its consequences : here the sites suitable for pollen analysis are very few and no definite results have been obtained from pedo-anthracology as yet. However there is strong evidence for the disappearance of a great part of the plant biomass favoured by climate. Under cold semi-arid conditions, such as which prevailed at Padul (near Granada) for example, the Quercus ilex forest settled very early at the end of the last glacial (PONS & REILLE 1986), long before any human interference; then, as a result of the latter, vegetation was reduced to a sparse scrubland or even mere herbaceous formations.

II - THE FIRE FACTOR IN PROVENCE OVER THE LAST 2000 YEARS (M.T.)

Many authors have shown how traditional cultivation practices, especially those using fire (grubbing, burn beating, pasture fires, etc.) led to a gradual regression of biological capital (KUHNHOLTZ-LORDAT 1938, 1957). This is particularly evident in the mediterranean climate where a dry period during the growing season intensifies the consequences of damage by slowing down the vegetation dynamism. The quantitatively poor vegetation cover and the soil characteristics (JORDA & VAUDOUR 1980) show the cumulative effect ot successive human interventions over a 8,000 year period. At the beginning of this period the first shepherds and farmers appeared in Provence on the North-Western Mediterranean shores (COURTIN 1974).

Old local customs using burning have been described by several authors (DARLUC 1782, DE RIBBE 1857, 1858, 1865, 1866), as well as their negative consequences as they often led to major fires on a scale of those of today, though they were generally ignored (AMOURIC 1985).

These essays, ethnological studies and indirect plant phyto-historical reports, such as those provided by palynology, indicate the existence of burning customs, though we cannot verify their reality, history or effect on vegetation on a local scale. A new scientific branch, pedoanthracology , sheds new light on these questions. This method, which has only recently been developed in Marseille (THINON 1978, 1979) is based on the botanical identification and dating of wood charcoals found in most soils, and which were left there by fires of mainly anthropic origin. It makes possible the recreation of the ligneous vegetation evolution over the past thousand years on an unequal spatial scale and often in relation to human activities.

A - Methodological summary

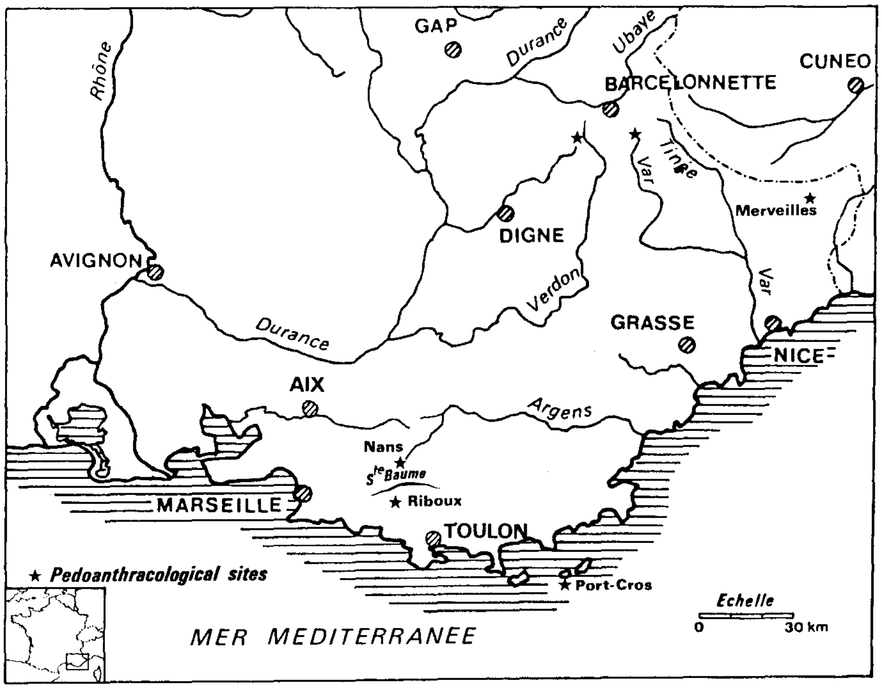
After digging a pedological trench, samples of many kilograms of sediment are taken at different levels. Wood charcoals which are generally no more than 4 to 5 millimeters in length are extracted by sifting in water down to a 400 micrometer-mesh screen, then cleaned and dried. Their botanical identification is made by observation of anatomical characteristics on the 3 wood planes (transverse, tangential and radial). The identification uses an incident light microscope with differential interference contrast or, if necessary, a scanning electron microscope. Anatomical characteristics have been codified from a study of reference samples and are checked by micro-computer.

The small amount of pedoanthracological material (a few milligrams for large charcoals) makes radiocarbon dating possible only by mass spectrometry. This type of chronological analysis has only been used very recently and only limited results have been obtained so far. These results indicate an age of a hundred years for charcoals found in the first 10 centimeters of a sample from the Sainte Baume mountains (Var, France) and 2,000 years of age for those at 60 centimeters, with consistent dates for intermediate levels. These data agree with the first series of dating made by counting carried out on Mont Ventoux charcoals (THINON 1978). We can therefore consider that, according to current pedoanthracological research, there seems to be a certain chronological stratification of charcoals in soils.

B - Application of pedoanthracology to the knowledge of past uses of fire in Provence

1) Demonstration of 2 types of environmental exploitation techniques in the Sainte Baume mountains (Var, France).

a) Riboux municipal land, southern side of the mountain, altitude 530 m. Current vegetation is of garrigue type : Quercus ilex, Phillyrea latifolia and Pinus halepensis. Shrub



layer : Rosmarinus officinalis, Juniperus oxycedrus, Rhus coriaria, Thymus vulgaris, Lavandula latifolia, Fumana ericoides.

The study of the pedoanthracological profile shows

0 - 10 cm. depth :	Juniperus sp. (most of the material)
	Quercus sp. deciduous (Q. pubescens ?)
	Pinus halepensis
	Sorbus domestica
	Prunus spinosa
20 - 30 cm.	Juniperus sp. (largely dominant)
	Several unidentified angiosperms
40 - 50 cm.	Quercus sp. deciduous (Q. pubescens ?)

The classic dynamic analysis of current vegetation includes this group in the Holly Oak and Phoenician Juniper series (OZENDA 1966). The last fire dates back to 1943 according to René Molinier (MOLINIER 1944) who mapped the area (site samples 1935-39) as juniper shrubland with Juniperus oxycedrus connected phytosociologically to the Rosmarinetalia order. This document and the blackened remains of junipers which can still frequently be found here explain perfectly the high frequence of junipers in the upper levels.

The Holly Oak (Quercus ilex L.) currently present and corresponding to the equilibrium stage of the formation described is not found in the charcoals where it is replaced by a deciduous oak, probably Quercus pubescens Willd. According to traditional ideas, this last taxon characterizes a higher altitudinal level or more humid regions.

The land was primarily used as grazing grounds as can be seen by the numerous ruins of sheep farms, the municipal archives and the reports of old shepherds. The clear dominance of junipers can be explained by selection, as grazing animals do not eat this species.

This kind of environmental usage has practically ceased to exist and natural conditions have developed for more than 40 years. A detailed study of the area shows that Quercus pubescens seedling growth prefigures the return of a vegetation type similar to that found in deeper pedoanthracological layers.

b) Nans Les Pins municipal land, north side of the mountain, altitude 424 m.

Current vegetation is represented by a preforest stage dominated by Quercus pubescens along with Quercus ilex and Pinus halepensis. The shrub layer consists of Phillyrea latifolia, Viburnum tinus, Arbutus unedo, Erica arborea, Cytisus sessilifolius and Coronilla emerus.

Pedoanthracological profile
10 - 15 cm. depth : *Quercus ilex*
Rhamnus alaternus
Arbutus unedo
Quercus sp. deciduous (*Q. pubescens* ?)
Fagus sylvatica
Prunus mahaleb
Cerasus avium
Taxus baccata
Juniperus sp.
15 - 25 cm. : *Quercus* sp. deciduous (*Q. pubescens* ?)
Fagus sylvatica
Quercus ilex
25 - 35 cm. : *Quercus* sp. deciduous (*Q. pubescens* ?)
Quercus ilex
35 - 50 cm. : *Quercus* sp. deciduous (*Q. pubescens* ?)
Quercus ilex
Acer opalus
60 - 75 cm. : *Quercus* sp. deciduous (*Q. pubescens* ?)
Quercus ilex
Prunus sp.

An analysis of current vegetation suggests the *Quercus pubescens* Mediterranean series (GAUSSEN 1948, OZENDA 1954, 1966) with the beginning of an evolution towards a *Quercus pubescens* Supramediterranean series (OZENDA 1966), a description which is ambiguous in relation to the series' definition.

In a study dating from 1926, LAURENT quotes the Toulon maritime archives which mention the visits to the "Vigueries de Provence" forests of commissioners of the Marine Royale from 1720 to 1725. This document indicates the presence of beech trees, *Fagus sylvatica* L., in a part of Nans Les Pins municipal land where they no longer exist. The pedoanthracological profile of this zone shows the presence of *Fagus sylvatica* in the upper levels where this usually mesophilous species coexists with Mediterranean taxa such as *Quercus ilex* and *Arbutus unedo*. We can still observe the coexistence of these species in the Chartreuse de Valbonne forest (Gard, France) at an elevation of 250 meters.

In 1926 LAURENT described a *Pinus halepensis* forest with *Quercus ilex* and *Quercus coccifera* on this site. MOLINIER in his 1944 study (*op. cit.*) includes a "*Quercus ilex* association". Since 1926 the site has thus undergone considerable changes and today we find many *Quercus pubescens* oaks and even mesophilous herb species existing in the neighbouring well-known forest. Three observations carried out over 60 years show different dynamic evolutions, the last one suggesting an increase in mesophilous species, closer to the pedoanthracological analysis which has shown the presence of a species absent for more than 2 centuries.

We should note the absence in the pedoanthracological profile of heliophilous species' charcoals such as pines and particularly (appearing only once close to the surface) of junipers which are characteristic of grazing lands. The relatively evolved stage of the current formation and the permanent presence of oaks confirm its forested nature, which is recalled in its toponymy : "Le Défens" (The protected area).

2) Recent pedogenesis on Port-Cros island (Var, France).

Current vegetation is dominated by *Quercus ilex* along with other species of the *Erica arborea* maquis. *Pinus halepensis* is also present in large quantities in the dominant layer.

The anthropic pressure has long been important on the island as deep charcoals show the generalized presence of *Erica arborea*, as well as *Arbutus unedo* and *Quercus ilex*. *Pinus halepensis* appears only in the surface levels. The easily alterable nature of the geological substrate (micaschists, phyllades) causes a rapid pedogenesis. Along with this, the abundance of charcoals throughout the soil, which corresponds to a degraded vegetation, indicates that the island's soils are relatively recent and were constituted during a period of human exploitation. The original soils were eliminated by erosion towards the sea, a phenomenon clearly visible in neighbouring Porquerolles island. Palynological studies in Corsica (REILLE 1977) showed the presence of an Erica arborea maquis probably preceding anthropic degradations. However, the current vegetation evolution on Port-Cros towards a forest stage after only a century of abandonment suggests that, despite the absence of datings, the development of the maquis corresponds to human activities.

3) Altitudinal decrease in the timberline

Several pedoanthracological profiles in the Southern Alps (upper valleys of the Verdon and the Var rivers, Vallée des Merveilles) under grasslands characterizing the Alpine belt, with Alopecurus gerardi, Trifolium alpinum, Nardus stricta, Ranunculus pyrenaeus, Carex sempervirens, etc., have revealed charcoals from Subalpine belt tree and shrub species. These charcoals show that these alpine pastures, from which trees and shrubs are completely absent today and which are located several hundred meters above the actual timberline, were colonized by Larix decidua, Pinus cembra, Juniperus and Laburnum. These last two shrub taxa are plants left by grazing animals and are good indicators of grazing. They are often found in charcoals from high altitude soils.

Pedoanthracology shows that human activities have in fact prevented the potential development of forests in certain areas previously considered as lacking tree species naturally.

C - Long-term effect of fire on vegetation

Wood charcoals and the observation of post-fire vegetation dynamics can give certain indications as to the modifications caused by fire over time.

The preceding examples show that in our region fire alone is incapable of effecting a new vegetation landscape. The Défens of Nans Les Pins has conserved over the centuries its characteristics of more or less closed deciduous forest. The management of burnt sites is a determining factor and grazing is one of the principal causes of modification. The study of wood charcoals suggests true grazing landscapes still visible today which are dominated by species left uneaten and which have good regenerative capacity after fire. Vegetation structures most susceptible to fire damage today are the result of the dual pressure of fire and grazing animals. The increased biomass caused by the development of pioneer trees (pines) after the abandon of traditional management techniques leads to the spectacular characteristics of today's fires.

It is interesting to note that plants which precede equilibrated forest stages (maples, rowans, oaks) resist damage and gradually gain importance even when their above-ground structures are partially or completely destroyed. Today's fires, with a local frequency of 3 to 5 per century, only slow down the growth of a resistant forest when seed bearers exist nearby. In the past, young shoots were systematically destroyed by grazing animals or shepherds and it was impossible for stumps to survive. Further, the coppice short revolution management did not favour seed production.

In the pedoanthracological profiles, deciduous charcoals in layers rich in sclerophyllous pyrophytes demonstrate the existence of periods of abandonment where the natural vegetation dynamism could re-affirm itself.

CONCLUSIONS

The set of palaeoecological data presented here on the past role of fire leads to four conclusions :

1) Fire was a constant, if not exclusive, means for man to clear areas necessary to his activities.

2) In the Mediterranean region, it was utilized at a much earlier date than in the rest of Europe. Fire began to be used sporadically towards 8,000 B.P. for yet uncertain reasons; then, between 7,000 and 5,000 B.P. it became more frequent for definite purposes and from 4,500 onwards it was widely used. It has been a general practice from dates that may vary but are well anterior to 3,000 B.P., in connection with agriculture and stock breeding. The conjunction of these impacts led to an already deeply modified vegetation at the end of the gallo-roman times.

3) Pedoanthracological analyses show that in Provence fire has been a universal and permanent tool during the past two thousand years. This practice, related to pastoralism and agriculture, often led to deep changes in the vegetation. The results of pedoanthracology, based on objective historical data often contradictory to the classical dynamic and speculative interpretations, are consistent with the present evolution of areas unexploited for several decades, and show that forest potentialities are largely underestimated in the Mediterranean region.

4) On the whole, the changes induced by this necessary and constant auxiliary of man are so great that an analysis of the present relationships between ecosystems and environmental factors can be significant only if it takes into prior consideration all the anthropogenic past of the ecosystems.

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ECOLOGIA MEDITERRANEA - Tome XIII (fascicule 4) - 1987

Vegetation wildfires in the mediterranean basin: evolution and trends.

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SUMMARY - Vegetation fires burnt an average 200,000 ha annually in the Mediterranean Basin between 1960 and 1970; the figure attained over 400,000 ha in the 1970's and over 600,000 ha in the 1980's. This three fold increase over 20 years is turning into an annual summer-catastrophe in the Basin, in spite of heavy investments and large budget expenditure spent to prevent and combat the scourge.

The paper gives detailed figures of number and acreage of fires in the various countries of the Basin, analyses the causes and evolutionary trends; some further regional details are provided for southern France.

The cost of preventing and combatting vegetation wild fires is evaluated for each country. The global cost for the Basin is in the vicinity of one billion U.S.$, as an annual average in the 1980's. But this figure does not take into account a number of havocs which are difficult to quantify such as : erosion, sedimentation, the silting up of reservoirs, floodings, loss of nutrients, hence decrease of fertility and productivity; loss of wildlife and game, of amenities and aesthetic value, etc.

The author advocates an expansion of research, demonstration and education of the public and politicians on the use of prescribed burning in conjunction with the systematic utilisation of browsing animals (cervidae, goats, camelidae) in order to reduce the fuel buildup, hence fire hazards and the ever increasing financial burden on rural communities, not to speak of losses in human life.

KEYWORDS : Wild fires, bush fires, Mediterranean Basin, native vegetation, forest, shrublands, fire ecology, landscape ecology, forest management, range management, wildlife and game management, land use.

VEGETATION WILD FIRES IN THE MEDITERRANEAN BASIN: EVOLUTION AND TRENDS

I - FIRE OCCURRENCE IN THE MEDITERRANEAN BASIN

Wild fires burnt an average of 660,000 hectares of forest and shrubland each year in the 16 countries of the Mediterranean Basin shown in Table 1, with an annual minimum of 382,000 ha and a maximum of 850,000 ha. For the same countries and the same period the number of wild fires declared averaged 35,600 per annum, with a low of 27,000 and a high of 45,500.

The number of wild fires and the areas affected have considerably increased over the past 25 years. The average annual surface burnt has evolved as follows:

Period	1960-71	1975-80	1981-85
Hectares burnt per annum	200,000	470,000	660,000

The average annual increase has thus been of 20,000 ha/year between 1965 and 1977 and 32,000 ha/year between 1978 and 1983.

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Table 1 -

Mediterranean Basin

Forest & Shrubland fires 1960-1985 evolution & trends
areas burnt in hectares
(sources : U.N. Economic Commission for Europe/FAO, 1986; Velez, 1986, 1987; Le Houérou, 1973)

	1		2			3			4			5					
Country	Forest & Shrubland 1984 Total area		1960-1971			1975-1984			1981-1985			1975-84 / 1960-71			1981-85 / 1960-71		
			Surface burnt area (S)	Number of fires (N)	S/N	S	N	S/N	S	N	S/N	S	N	S/N	S	N	S/N
	Surface 10³ Ha	Affor. rate %															
Albania	1038	38	-	-	-	-	-	-	-	-	-	-	-	-			
Algeria	4383	1.8	12000	300	40	-	-	-	55000	904	61				4.6	3	1.5
Cyprus	171	18.5	1100	56	19	2450	80	31	3870	78	50	2.23	1.43	1.61	3.5	1.4	2.6
Egypt	2	0.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
France	14604	26.8	38000	1260	30	42650	5400	7.9	42083	4908	8.6	1.12	4.29	0.26	0.99	0.90	1.1
Greece	2620	20.0	8300	522	16	32930	1015	32.4	53501	1168	45.8	3.97	1.94	2.02	6.45	2.2	42.9
Israel	116	5.7	3300	330	10	3280	920	3.6	2768	965	2.87	0.99	2.79	0.36	0.84	2.92	0.29
Italy	6410	21.8	19000	1800	11	127320	6210	20.5	167780	11832	14.2	6.70	3.45	1.86	8.8	6.61	0.3
Jordan	41	0.4	-	-	-	-	-	-	299	62	4.8	-	-	-	-	-	-
Lebanon	82	8.0	-	-	-	-	-	-	1200	-		-	-	-	-	-	-
Libya	640	0.4							16	3	5.3	-	-	-	-	-	-
Malta	1	3.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Morocco	5200	11.7	2500	60	42	-	-	-	4913	185	26.6	-	-	-	1.96	3.1	0.6
Portugal	3641	39.8	-	-	-	64185	3430	18.7	57482	5281	10.9	-	-	-	0.9	1.5	0.6
Spain	15625	31.3	31000	1200	26	209200	6110	34.2	240330	8453	28.4	6.75	5.09	1.32	7.7	7.0	0.05
Syria	498	2.7	-	-	-	-	-	-	1500	-	-	-	-	-	-	-	-
Tunisia	559	3.6	6000	50	120	-	-	-	1562	101	15.5	-	-	-	0.25	2	0.13
Turkey	20199	26.2	7300	518	14	14400	1140	12.6	9282	1225	7.6	1.97	2.20	0.90	1.3	2.3	0.5
Yugoslavia	9294	36.4	1200	300	4	11540	814	14	15708	9131	7.2	9.62	2.7	3.5	13	3	1.8
Total	85124	-	129700	6396	20	507955	25119	20	657294	36078	18	3.9	3.9	1.0	5.1	5.6	0.9

A few words of caution however would seem appropriate as the statistical base may vary from one country to another, and also from one period to another within a given country. The orders of magnitude nonetheless remain valid, all the more so as most of the areas burnt come from a limited number of large (over 100 ha) to very large (over 1000 ha) fires within any given country; these large fires are well documented, and therefore even if there are some discrepancies in the statistical definition of small fires, this does not significantly affect the overall areas concerned.

In recent years, the heaviest tolls were taken in:

- 1978 : 753,000 ha
- 1979 : 589,000 ha
- 1981 : 759,000 ha
- 1985 : 850,000 ha

The years of least fire occurrence were:

- 1975 : 341,000 ha
- 1976 : 325,000 ha
- 1977 : 284,000 ha
- 1984 : 310,000 ha

However, the statistics of fire occurrence given above are actually below what would constitute a realistic figure for several reasons:

- There are no data available for some countries such as Albania and Portugal before 1970.
- Other countries have scarce, irregular and partial data, such as : Algeria, Jordan, Lebanon, Libya, Morocco, Syria, Tunisia.
- Some Mediterranean countries - or even partly Mediterranean countries - have not been considered, viz: Iraq, Iran, Saudi Arabia, Oman, Afghanistan, Pakistan.
- In some countries such as France peri-urban and rural fires have been disregarded to take only into account forest and sclerophyllous shrubland fires.

An overall classification of burnt surface areas for the periods 1975-1984 and 1981-1985 gives the following figures in decreasing order of importance:

	1975-84		1981-85
Spain	210,000 ha	Spain	240,000 ha
Italy	127,000 ha	Italy	168,000 ha
Portugal	64,000 ha	Portugal	57,000 ha
Algeria	55,000 ha	Algeria	55,000 ha
France	43,000 ha	Greece	54,000 ha
Greece	33,000 ha	France	42,000 ha
Turkey	14,000 ha	Yugoslavia	16,000 ha
Yugoslavia	12,000 ha	Turkey	9,000 ha

When, for the same reference periods, the average area burnt annually is expressed as a percentage of the national forest and shrubland acreage or as a percentage of the Mediterranean natural vegetation (see Table 2) the following figures emerge.

	National		Mediterranean	
	1975-84	1981-85	1975-84	1981-85
Israel	2.7	2.4	2.7	2.4
Italy	2.1	2.6	2.7	3.2
Greece	1.3	2.0	1.6	2.5
Cyprus	1.4	1.4	2.3	2.3
France	0.3	0.3	2.3	2.3
Portugal	1.8	1.6	2.2	2.0
Spain	1.4	1.5	1.7	1.9

Yugoslavia	0.1	0.2	1.0	1.0
Algeria	1.0	1.0	1.0	1.0
Jordan	-	0.45	-	0.45
Syria	-	0.34	-	0.34
Tunisia	-	0.19	-	0.19
Morocco	-	0.09	-	0.09
Medit. Basin	0.6	0.8	0.9	1.2

Table 2 - Areas of total forest and shrubland, together with afforestation rates in 1000 Hectares.

Countries	Land Area x 10^3 ha	Forest & Shrubland x 10^3 ha	Affor. %	Medit.Forest x 10^3 ha	% of total	Estimate Afforestation Rate Percentage in Medit. zone
Albania	2740	1038	38.0	104	10.0	20.0
Algeria	238174	4383	1.8	4383	100.0	1.8
Cyprus	924	171	18.5	171	100.0	18.5
Egypt	100145	2	0.1	2	0.1	0.1
France	54563	14604	26.8	1800	12.0	25.0
Greece	13080	2620	20.0	2100	79.0	20.0
Israel	2033	116	5.7	116	100.0	5.7
Italy	29402	6410	21.8	4500	70.0	22.0
Jordan	9718	41	0.4	41	100.0	0.4
Lebanon	1023	82	8.0	82	100.0	8.0
Libya	175954	640	0.4	640	100.0	0.4
Malta	32	1	3.1	1	100.0	1.0
Morocco	44630	5200	12.0	5200	100.0	12.0
Portugal	9164	3641	40.0	2900	80.0	38.0
Spain	49940	15625	31.0	12500	80.0	30.0
Syria	18405	498	3.0	498	100.0	3.0
Tunisia	15336	559	4.0	559	100.0	4.0
Turkey	77076	20199	26.0	17000	84.0	23.0
Yugoslavia	25540	9294	36.0	1000	11.0	25.0
Total	867879	85124	10.0	54350	64.0	10.0

Table 1 shows that the evolution of burnt surfaces has been quite rapid over the past 25 years since the ratio of the surfaces burnt during the 1975-84 period was 4 times larger than that of the 1960-71 period, while the ratio of the 1981-85 period to the 1960-71 period reached a factor of 5 for the Mediterranean Basin as a whole, and a factor of 6 to 9 for three of the main countries concerned : Greece, Italy and Spain. North of the Mediterranean Sea the proportion of Mediterranean forest and shrubland burnt every year varied from 2 to 3.2% and these percentages are fairly consistent from one country to the next. To the South and the East of the Basin the situation is quite different for socio-cultural and socio-economic reasons. In these countries forest and shrubland are subject to heavy grazing, particularly by goats, so that there is little, if any, fuel accumulation and therefore less fire hazard. Devastating fires may however occur, such as in Algeria where 221,000 hectares burned in 1983 and 169,000 in 1881 !

There are, however, large differences from one country to another under conditions seemingly otherwise identical such as in Algeria, Tunisia and Morocco where the rates of burning are sharply contrasted : 1.0% of forest and shrubland are burned as an annual average in Algeria; 0.2% in Tunisia but less than 0.1% in Morocco ! The situation in these respects has remained similar for these countries for the past 100 years or so (LE HOUEROU 1977).

As far as the annual number of fires is concerned, the overall increase in the 1975-84 period over the 1960-71 period is similar to the increase in surface: four-fold; and the figure is almost six-fold for the period 1981-85 in respect to 1960-71. This increment factor, however, has been quite uneven; for the first mentioned reference periods of comparison the increment gave the following factors: Spain 5; France 4; Italy 3.5; Israel 3; Yugoslavia 3; Turkey 2; Cyprus 1.4. The mean surface per fire thus remained stable (20 ha) between 1960 and 1984 but decreased slightly (18 ha) in the 1980's. But here again, the situation varies a good deal from one country to another. The factor of increment in 1975-84 in respect to 1960-71 was the following: Yugoslavia 3.5; Greece 2; Italy 1.9 and Cyprus 1.6; it decreased to 0.9 in Turkey, 0.36 in Israel and 0.26 in France due to more efficient fire fighting management in the latter set of countries.

II - CAUSES OF WILD FIRES

The causes of wild fires may be divided into two groups, i.e. general causes and proximate causes.

General causes

Among the general causes are the meteorological factors of: summer drought, high summer temperatures, low air humidity, locally strong winds in the summer season: "mistral" in France, "tramontane" in Spain, Italy and France, "Foehn" in various mountainous regions and other local dry and hot winds such as "Ponant" in S-W Spain, "Levant" in the Gibraltar Straits, "Sharav" in Israel, "Khamsin" in the Near East and Egypt, "Ghibli" in Libya, "Sirocco" in Italy and N. Africa, "Chergui" in Morocco, etc.

Obviously, all other conditions being equal, fire hazards increase with the length and intensity of the annual drought period, up to a point where aridity becomes a limiting factor to biomass accumulation, whereupon fire hazards decrease due to fuel shortage. The worst situation climatically is found under semi-arid to sub-humid bioclimates with rainfalls between 400 and 1000 mm per annum. In more humid zones, the dry season is somewhat shorter and less intense and consequently less hazardous. In arid zones the fuel load is lower, vegetation is usually not continuous, and consequently fires can hardly extend to large areas.

Among general causes are also the nature and structure of Mediterranean forests and shrublands with their large areas of pines and sclerophyllous shrubland both of which are highly flammable. This particular point has been extensively studied by Trabaud (1971, 1976, 1979) and reviewed by Le Houérou (1973, 1977), Susmel (1973), Velez (1986, 1987) and others.

A further general, albeit indirect, cause of fire hazard is paradoxically a rapid increase in living standards in the countries of the northern shores of the Mediterranean over the 30 to 40 years and the ensuing rural depopulation, the abandonment of extensive animal husbandry and of forest and shrubland grazing etc. These socio-economic conditions have resulted in a general buildup of fuel in the more developed countries, a factor that has been often worsened by the increasing lack of economic interest in fuelwood and charcoal in these countries. Conversely, on the southern and eastern shores of the Basin the situation is exactly the opposite: grazing pressure and woodcutting have constantly and sharply increased since World War II. This is due to low standards of living, high and ever increasing demographic pressure in rural areas, ever increasing grazing pressure and stocking rates; all these causes have resulted in a general decrease in vegetation cover, in biomass and in wooded surfaces, hence in fire hazards.

Proximate causes

In the Basin as a whole the origin of about 50% of the wild fires remains unknown ! The main registered causes are : carelessness, arson, grazing fires, and, locally important, lightning.

As regards carelessness, the blame is often put on tourism in developed countries, although this may be doubtful : a survey in southern France in 1986 showed that almost 90% of wild fires were ignited by local people. It is true, however, that a very large proportion of fires, particularly those of unknown origin, are initiated within a distance of 20 m from roads and tracks, which suggests arson by pyromaniacs. Thus both tourists and natives, combined with careless picnickers and smokers, may indeed be responsible for a large proportion of the 50% fires of unknown origin.

Careless fires also include voluntary burning for grazing or for clearing by both farmers and graziers : fires that escape to shrubland and forest. This is particularly the case in the least developed parts of otherwise developed countries, such as some of the Mediterranean islands : Corsica, Sardinia, Crete, Cyprus, etc. but also in some continental provinces, to the north of the Basin. Another increasing cause in developed countries is the burning of unauthorized garbage dumps, particularly in highly touristic areas; these may locally represent up to 10% of wild fire ignitions. Careless fires overall represent some 20 to 25% of those of known origin.

Voluntary fires are those which have been lit on purpose; they may be criminal or not; they represent overall 30 to 40% in the Basin as a whole. Among the non-criminal fires, grazing fires come first in less developed regions where extensive shrubland grazing still prevails. Criminal fires represent 10 to 20% of those of known origin. The main causes of arson are private revenge between natives, hunting conflicts, disputes related to official forest policy, fires ignited in order to induce public investment and create employment and local income, conflicts related to land use and land tenure, ownership of forests and shrublands, fires lit to frighten away pests and predators (boars, wolves, jackals, etc.). Local politics, terrorism and warfare may be significant causes locally : for example, secessionism in Corsica, periods of general elections in Turkey, independence war in Algeria, war of secession in Cyprus, civil war in Lebanon.

Lightning may also be a significant cause locally, particularly in mountainous areas; in southern France, for instance less than 5% of the fires of known origin are due to lightning in lowlands, but this proportion reached 10 to 20% in highlands and mountains in 1986 (Prométhée Project 1987).

Among the causes one should also mention the local land tenure regimes. Most wild fires occur in private forests, woodlands and shrublands, at least to the north of the Basin; not only do private forest and shrubland represent 60 to 80% of the areas of natural vegetation but the number of fires per surface unit is also higher in private land or in communal land which is not submitted to the state forestry regime of management.

A final remark about the causes of forest and shrubland fires, is that there should be some sort of agreement between countries about the definition of wild fires and their causes. The statistics differ markedly from country to country as to the inventory of the causes of fire; in France, for instance, the categories are : unknown, carelessness, criminal, lightning, and various; in Spain the categories are : unknown, carelessness, voluntary, lightning, railways and various. There is no reference to arson or criminal fires, which in turn may be voluntary or due to carelessness. An international agreement would certainly help in interpreting the statistical data, and thereby an understanding of the facts; a clear understanding of the facts would, in turn, help the setting up of realistic prevention and fighting strategies.

III - ASSESSMENT OF THE ECONOMIC COST OF FOREST AND SHRUBLAND FIRES

The estimate of the overall economic cost of wild fires in the Mediterranean Basin includes five main items :

- Cost of prevention :
    - . Investment, labour, running costs
- Cost of fire fighting :
    - . Investment, labour, running expenses
- Monetary value of the biomass lost
- Cost of rehabilitation
- Ecological cost (erosion, sedimentation, nutrients, human life).
- Manpower.

A survey of 46 large fires on 100 ha or more and covering a total surface of 25,000 ha, in Southern France in 1986 (Projet Prométhée, 1987) came up with a figure of 10 man/hour per hectare (0.1 to 56.0), i.e. about 500 FF or USD 82 per hectare burnt, in terms of fire fighting costs.

The cost of prevention, i.e. opening tracks in forest areas, opening and maintaining firebreaks, establishing ponds, etc. cost an average of 600 FF or USD 100 per hectare over the same period.

The annual investments incurred in the following six countries for the 1981-1983 period were of the order of magnitude of USD 200 per hectare burned for Portugal, Spain, France, Italy, Greece & Turkey; this figure has been calculated from the data published by Velez (1986). If we estimate that the running costs for prevention and fire fighting are about equal to the cost of labour we have another USD 80 per hectare.

The cost of prevention and fire fighting is thus : USD 462 per hectare per year; or in round figures : <u>USD 500/ha/Year</u>.

The estimate of the biomass lost and of the cost of rehabilitation varies widely from one country to another and from one vegetation type to another. The following figures are put forward in Italy and Spain:

- Spain	Biomass	230	(USD per ha)
(Velez 1987)	Rehabilitation	700	"
	Total	930	"
- Italy	Biomass	575	"
(Istit. Centr.	Rehabilitation	738	"
Statist. 1984)	Total	1313	"

Taking the mean of these two overall figures we arrive at a value of USD 1013 per hectare, i.e. about USD 1000 in round figures.

The overall cost is thus 500 + 1000 = USD 1500/ha/Year.

If we multiply this cost by the average number of hectares burnt annually, we reach the following figures for the 1981-85 period :

Mediterranean Basin : 9.9×10^8 = 1 Billion USD/Year.

Algeria	8.25×10^7	USD/Year
Cyprus	5.8×10^6	
France	6.3×10^7	
Greece	8.1×10^7	
Israel	4.2×10^6	
Italy	2.5×10^8	
Jordan	4.5×10^5	
Lebanon	1.8×10^6	
Libya	2.4×10^4	
Morocco	7.35×10^6	
Portugal	8.6×10^7	
Spain	3.6×10^8	
Tunisia	2.7×10^6	
Turkey	1.4×10^7	
Yugoslavia	2.4×10^7	

Although these costs are huge, they are still far below the actual loss because a number of items have not been accounted for : these include loss of houses, properties, amenities, power lines, rural houses, crops, industrial properties, livestock, human life (152 in Spain from 1961 to 1985) etc.

In addition to this should be added the ecological cost : loss of nutrients, hence fertility and production; erosion and sedimentation, viz. silting of reservoirs; loss of wildlife and game; loss of recreational facilities and aesthetic value, etc. all of which are difficult to quantify.

IV - FOREST AND SHRUBLAND FIRES IN SOUTHERN FRANCE : AN EVALUATION

The area concerned covers 14 "départements" located in four economic regions :

"Département"	Postal	Area in 1000 hectares (1)			
	code	Total	Forest(F)	Shrublands(S)	F + S (%)
- Alpes de Haute Provence	04	694.4	293.4	13.9	44.2
- Hautes Alpes	05	552.0	161.4	-	29.2
- Alpes Maritimes	06	429.8	174.6	20.8	45.5
- Ardèche	07	555.6	224.5	52.3	49.8
- Aude	11	634.2	150.5	118.8	42.5
- Bouches du Rhône	13	524.8	80.9	81.4	30.9
- Corse du Sud	2 A	402.0	131.3	162.6	73.0
- Haute Corse	2 B	466.0	97.1	207.2	65.3
- Gard	30	584.8	171.5	78.6	42.8
- Hérault	34	622.4	162.3	53.9	34.7
- Lozère	48	517.0	206.1	6.1	41.0
- Pyrénées Orientales	66	414.3	114.2	67.3	43.8
- Var	83	602.3	280.0	94.0	62.1
- Vaucluse	84	357.8	109.0	28.6	38.5
Total		7357.4	2355.8	985.5	45.4
Afforestation rate		___	32%	13.4	45.4

The average number of forest and shrubland fires for the period 1977-86 was 3500 per annum; the average burned annually was 43000 ha over the same decade. The average size of each fire was thus 12.3 hectares for the same reference period. The highest number of fires in a given year during that period was 4300 in 1978, and the lowest 1800 in 1977.

The largest area burnt in a given year was 55000 ha in 1979 and the smallest 17000 ha in 1980. The highest occurrences both in terms of numbers and area are in the "départements" of Corsica and Var which together represent nearly a half of fire occurrence, with Corsica alone being responsible for one third of the total. Some 80 to 85% of the affected areas burn in July-August, while only some 50% of fires are ignited in that period. Most fires occur in day-time and 70% of them between 10.00 hrs to 18.00 hrs. There is a small peak of fire occurrence in March in the montainous areas of Lozère, partly due to lightning, but these do not cover large areas.

Large balance sheets in terms of surface were the result of a rather small number of large fires. In 1986 for instance, a year close to the average for the past decade, 64 fires of 100 hectares or more covered 38000 ha, i.e. 82% of the annual total; while 12 fires of 1000 ha or more burned 25000 ha, i.e. 53% of the total for that year.

Still for 1986, 3 periods totalling 12 days in July-August witnessed the burning of 68% of the annual total; on 23-24 August alone 16000 ha, that is 35% of the annual total, were burned !

Fires of 100-500 ha were responsible for 19% of the destruction, those of 500 to 1000 ha represented 10%, while fires of more than 1000 ha destroyed 53% of the total. The cause of forest and shrubland fires vary greatly from one "département" to another. In 1986, for instance, the average number of fires of unknown origin was 51%, broken down as follows :

- 85% in Bouches du Rhône
- 70 Corsica
- 69 Vaucluse
- 63 Hérault & Hautes Alpes

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(1) Source : Inventaire Forestier National, unpublished, data from 1976 to 1984.

- 62	Alpes Maritimes
- 33	Ardèche
- 22	Alpes de Haute Provence
- 0	Lozère

Lightning was a significant cause in montainous areas : 10-20% of all fires were ignited by lightning in Alpes de Haute Provence, Hautes Alpes, Ardèche, Aude, Lozère and Vaucluse; but less than 5% in Alpes Maritimes, Bouches du Rhône, Corse, Gard, Pyrénées Orientales and Var. Forestry and farming practices were responsible for 20 to 60% of the fires of known origin in Alpes de Haute Provence, Hautes Alpes, Alpes Maritimes, Lozère, Var and Vaucluse. Malice was the cause of some 15% of the fires of known origin, but the figure varied greatly : 0 in Lozère, 5% in Var, 10% in Alpes de Haute Provence, 15% in Pyrénées Orientales and Aude, 20% in Ardèche and Gard.

92% of criminal fires (arson) were lit by males, and 87% by adults and teenagers; 90% of the identified or suspected incendiaries were natives of Southern France. For fires ignited accidentally, 95% were lit by males, 91% by adults and 88% by natives, and 96% by farmers or graziers.

Of the 46000 ha burnt in 1986, 89% were privately owned forest and shrubland, while state-owned and communal land represented only 11%. However, as usual, there were large differences from one "département" to another. The following Table provides the details.

	Surface burned		Area of burnt state-owned land	
"Département"	(Ha)	% Forest & Shrub	(ha)	%
Alpes de Haute Provence	277.7	0.09	56	19
Hautes Alpes	50.0	0.03	6	6
Alpes Maritimes	10574.0	5.41	588	6
Ardèche	1554.0	0.56	5.4	3.5
Aude	4053.0	1.51	205	5
Bouches du Rhône	3205.0	1.98	458	14
Corse du Sud	2137.0	0.73	0	0
Haute Corse	6302.0	2.07	0	0
Gard	1935.0	0.77	462	24
Hérault	713.0	0.33	41	6
Lozère	831.0	0.39	41	5
Pyrénées Orientales	5628.0	3.10	1239	22
Var	8748.0	2.34	2019	23
Vaucluse	114.0	0.08	3	3
Total	46120	1.38	5104	11

V - CONCLUSIONS AND RECOMMENDATIONS

As mentioned above, it would seem desirable to reach some sort of agreement on the definition of the various categories of forest and shrubland fires and of their causes in the Mediterranean region. Such harmonization, at least within the European Economic Community, would greatly facilitate discussions between specialists, as well as the interpretation of fire statistics and would therefore assist in the exchange of information and experience, thus helping in the formulation of common and realistic strategies for fire prevention, suppression and management. This matter should be given high priority in the international coordinating bodies since, as we have shown, a minimum surface of 660000 hectares burns on average each year in the Basin at a cost of over one billion dollars (or ECU).

A more careful evaluation of the costs also seems desirable either in terms of manpower, running expenses, investment, biomass loss or cost of rehabilitation of burned areas. It

The difficult problem of the evaluation of the ecological cost of wild fires should also be addressed, at least at the research level. It constitutes an interesting challenge for forest and environment specialists to meet in collaboration with socio-economists. The quantification of such costs would in addition help to convince decision makers and politicians to provide further means for fire prevention and suppression.

It also seems that more resources should be allocated to research and demonstration on controlled and prescribed burning not only within firebreaks as is the case at present, but also as a silvicultural management tool in Mediterranean forests and shrublands. LIACOS in Greece, for instance, has long ago demonstrated that Brucia and Aleppo pine plantations under semi-arid and sub-humid Mediterranean climates can actually be managed with prescribed burning, and that this reduces the fuel load - hence wild fire hazard - by a factor of 6 (25 to 4 t/Ha in his experiments), with in addition greatly increased productivity and considerably shorter rotations.

Furthermore, the conversion of Mediterranean shrublands into sown pastures has been successfully achieved on a large scale in California and on a smaller scale in the Mediterranean, particularly in France (ETIENNE 1977), Greece (PAPANASTASIS & LIACOS 1980), Portugal and Spain. The technique of sowing herbaceous forage species in the ashes after a burn or a wild fire has been demonstrated to be cheap and effective under semi-arid to humid mediterranean climates, and well adapted to steep slopes and non-arable land. Perhaps 20 to 30% of Mediterranean shrublands could in principle be treated in this way with or without the presence of isolated trees; the latter alternative would create landscapes of the Dehesa and Montado type which are extremely effective in terms of fire prevention in addition to being aesthetic and productive in multipurpose production systems.

The ever increasing buildup of fuel in Mediterranean fores and shrublands as a result of rural depopulation and the abandonment of marginal lands, and the ever increasing surfaces burned annually by wild fires will sooner or later make new legislations necessary as well as the adoption of new methods of prevention such as those mentioned above, for the methods presently used obviously do not work. They only result in an ever increasing financial burden on nations and communities.

The combination of prescribed burning with the utilization of browsing animals such as goats and cervideae, could contribute to the solution. Wildlife management in particular could significantly reduce fire hazard as has been shown in the US and New Zealand with the utilization of appropriate fencing. Wildlife, in combination or not with livestock, may furthermore create significant income. In Texas, for example, ranchers do at present make more money from wildlife (deer, quail, turkey) than from livestock on the same land.

Prescribed burning took more than a quarter of a century to be accepted in the US. But nowadays it is largely used as a forestry management tool over more than 30 million hectares, particularly in pine forests and plantations, but also in range and wildlife management. Let us therefore take advantage of the American experience and train young scientists in fire ecology (for this is a science as well as an art).

Last, but not least, legal, sociological and economic studies seem badly needed to better apprehend the links between fire prevention and land use, hunting, recreation and various rural activities, in order to arrive at solutions, including prescribed burning and wildlife management, which would be acceptable to all the various parties concerned.

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Dynamics after fire of sclerophyllous plant communities in the mediterranean basin.

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SUMMARY - Fire has significantly influenced the evolution of ecosystems throughout the Mediterranean Basin. Old concepts considered fire as a destructive force bringing vegetation back to an initial starting stage. By reviewing recent European works, it appears that all authors agree that fire does not really change both flora and structure of plant communities. Ecosystems tend to reach a state similar to the one existing before fire. Most plants of the Mediterranean vegetation possess strategies to withstand and survive fire.

KEYWORDS : Dynamics, Mediterranean Basin.

INTRODUCTION

Fire is such an ancient, universal ecological force that it has played an important part in shaping many of the vegetative communities and much of the landscape in the Mediterranean Basin. The aridity of the climate has played an undeniable, but not always preponderant, role; combined with the nutrient poorness of soils, it contributed to create landscapes such as the matorral and tomillares in Spain, maquis and garrigues in southern France, macchia in Italy, xerovuni and phrygana in Greece, choresh and batha in Israel, gatha nabati in Syria and Lebanon. But these various landscapes have been modelled by human action, which has been strongly exerted in these areas.

Just how ancient fire is, cannot be known for certain but there can be no doubt that its first occurrence predates mankind by a considerable margin; it may be as old as terrestrial vegetation (HARRIS 1958, KOMAREK 1973). Lightning is one of the natural causes of vegetation fires, from the tundra to tropical forests (KOMAREK 1964, 1967, 1968, TAYLOR 1969). In the same way, other non-anthropogenic phenomena may also have caused fires, such as volcanic eruptions, spontaneous combustion (VIOSCA 1931) and sparks produced by falling rocks (HENNICKER-GOTLEY 1936).

At first, fire was a natural component which appeared more or less regularly in the natural cycle of vegetation succession. Its advent permitted the rejuvenation of some stands and created a mosaic of plant communities.

However, man's appearance on the scene disturbed this balance of nature substituting an artificial situation and upsetting the previous order. Man has used and misused fire; this, associated with forest felling, grazing by domestic animals and extensive but aggressive cultivation (uprooting of plants) contributed to the constitution of the vegetational landscapes encountered today.

In many ecosystems, fire controls the age, structure and species composition. Fire acts with different frequencies and intensities, depending upon the vegetation and the climatic situations. Thus, vegetation composition and structure depend on climate, fire frequency and intensity, while fire frequency and intensity in turn depend on vegetation structure and climatic regimes.

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OLD CONCEPTS

Research dealing with the problem of wild fires is characterized by two periods : a "pioneer" period during which there were very few works concerning this subject; and a more recent period where the fire problem was considered in a different way and tackled by rigorous methods and techniques.

The former period can be itself subdivided into two parts : a first one during which publications evoked fire without studying thoroughly its action on the ecosystem; a second one during which most of the authors, under the clementsian ideas, were anxious at all costs to show that the vegetation dynamics after fire followed a succession of stages or "associations" strictly in a straight linear way, without giving their attention to the responses of plants constituting the communities.

The first period is characterized by the works of the precursors. RIBBE (1865, 1869) recorded the importance of the understorey in the propagation of fire; he stated the dominance of Erica arborea and the rapid colonization of burnt sites by Cistus spp. He also recorded the action of pine cones in propagating wild fires. But he particularly discussed the causes of wild fires, their history and the legal problems linked to them.

Other authors then discussed and concluded partly in the same way as RIBBE about fire influence on the ecosystem. JACQUEMET (1907) attributed deforestation to wild fires; COTTE (1911) evoked the effect of fire on Quercus suber and Pinus pinaster. FLAHAULT (1924) studied more thoroughly the phenomenon, talking about the effects of fire on vegetation and soil; he recorded the first species getting established in burnt areas : Cytisus purgans, Erica arborea, Cistus monspeliensis and Quercus coccifera.

DUCAMP (1932), LAURENT (1937) tried to explain the effect of wild fires on vegetation. They particularly considered the problems of erosion and forest "degradation".

In practice, except for the forerunner RIBBE, it can be shown that of all these authors studying the wild fire phenomenon and the havoc suffered by vegetation, only a few obtained precise observations or carried out careful studies of causes and consequences in particular, for fire effects. More often than not they provided only general and fragmentary comments.

On the other hand during that period of time, more numerous (15) were the authors considering the wild fire problem from the juridical and financial points of view, or proposing prevention means, protection works and fire control techniques.

The development of the phytosociological method in the 1930's stimulated the activity for naturalists who then possessed a tool allowing them to study more precisely the impact of fire on vegetation.

Because of the North-american Clementsian theory, researchers have, for years, considered succession as a directional change of vegetation types, each successive type establishing itself because the preceding type has modified the site in a way favourable for its successor, this sequence finally ending in a climax type which is stable and self-maintaining under the current conditions of site and climate. This was seen as a deterministic process. If the climax was destroyed by some catastrophic event (e.g. fire), it was believed that its preceding stages could be repeated and the same climax eventually re-established, provided the climate was unchanged.

During this latter period, the phytosociological school dominated in Europe. BRAUN-BLANQUET (1935, 1936) considered wild fire as a factor as important as tree felling and overgrazing, in the regressive dynamics of vegetation. According to his opinion, he presented the degradation stages of the Quercus ilex forest as successive associations from the forest to the grass swards.

KORNAS (1958) studied the causes of the regressive succession in the Gardiole Mountains (near Montpellier). By comparing different types of plant associations, he tried to describe the regression stages in the same way as BRAUN-BLANQUET. Deepening the method and giving more details, his work is one of the best examples of this type of description of succession (fig. 1).

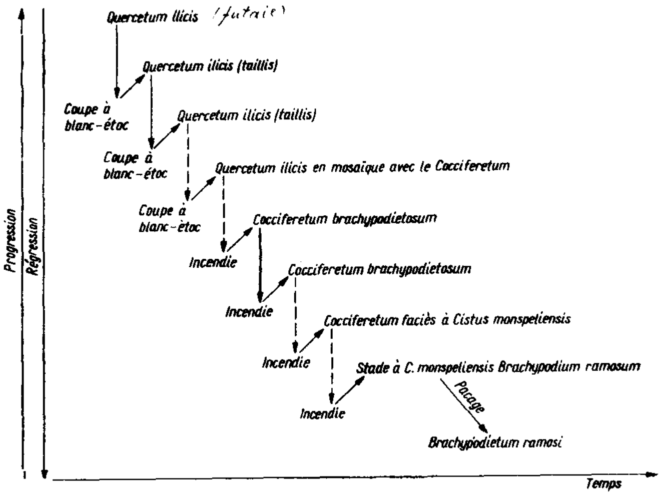

Fig. 1 Example of a schematic diagram of succession of plant communities according to the classical school (KORNAS 1958).

In Provence, MOLINIER (1953, 1968) and, MOLINIER & MOLINIER (1971), also approached this problem; but they did not try to analyse the process of vegetative dynamics after fire. They were only considering a wild fire as a factor contributing to the degradation of plant communities and gave diagrams similar to those of KORNAS.

At the end of that period, one author distinguished himself : KUHNHOLTZ-LORDAT, who did not follow the strict phytosociological method and who focused his research on fire action. KUHNHOLTZ-LORDAT (1938) was the first to give a definition of "pyrophytes" and a list of pyrophytic plant communities. His preoccupation was the resistance of species to fire and their ability to regenerate after it. In 1952, he claimed that the current vegetation of Provence was nearly totally pyrophytic, and cited four primary pyrophytes : Quercus coccifera, Erica multiflora, Cistus monspeliensis and Pinus halepensis. He associated soil degradation to fire, and established that the type of vegetation in the Mediterranean area depends on the frequency of fire : a short periodicity favours species with vegetative reproduction; a long periodicity is in favour of seeders.

Later, KUHNHOLTZ-LORDAT (1958) defined more precisely the term and classification of pyrophytes. He distinguished (1) pyrophytes with passive resistance because of their constitution, i.e. with high water content in their tissues (succulents, e.g. Agave) or thick bark (Quercus suber); (2) pyrophytes with vegetative reproduction which resprout after fire, either by epicormic shoots as Arbutus unedo (KUHNHOLTZ-LORDAT 1958) or by below-ground resprouts (Quercus coccifera, Pteridium aquilinum and several perennial grasses); (3) pyrophytes with an indirect resistance to fire which create an unfavourable environment for plants around them; (4) social pyrophytes which reproduce massively by seed (e.g. Mediterranean Pinus spp., Cistus spp.). However, KUHNHOLTZ-LORDAT still continued to give flow diagrams of succession without studying or furnishing real evidence of vegetation dynamics after fire. Following identical concepts, BARRY (1960) in his study near Nîmes (France) presented diagrams of successive stages of post-fire regeneration.

All the preceding authors considered that fire was creating series of more and more degraded successive stages. They only compared in a synchronic way different associations which they a priori thought were following a succession. The actual processes by which vegetation is recovering after fire were not precisely tackled.

Still today some authors present the same flow diagrams without providing any precise data from field observations, starting from "original" forests down to shrublands and open swards (LE HOUEROU 1974, MACCHIA et al. 1984).

As it can be seen, most of the authors speculated from scanty data. Fortunately, present studies have now started with less preconceived idea and their results will certainly give new perspectives in the plant dynamics after fire.

FLORISTIC COMPOSITION

In any study on plant community dynamics, floristic changes through time should be considered on one side, and structural changes which occur through plant growth or species composition on the other side.

Spain

In the SW of Spain (Gulf of Cadiz), GARCIA NOVO (1977) described the succession stages of vegetation recovery after fire of one type of matorral growing in the Donana National Park. He identified 5 stages.

First stage : 0-3 months. No seed germination, no annual plants. Resistant species recover after fire. Two species show an outstanding recovery : Chamaerops humilis and Daphne gnidium.

Second stage : 1 year. Seeds of both woody and herb species germinate. Fire resistant species start regrowing. Many annuals are present at this stage. Among woody perennials, Stauracanthus genistoidis grows very easily from its thick base. Erica australis, E. arborea, Ulex minor also regrow but they take much more time to start. Some species with subterranean organs are sprouting. Non-resprouting species such as Halimium halimifolium, Cistus libanotis, C. salvifolius, Rosmarinus officinalis profusely germinate after fire.

Third stage : second year. At this stage, species having germinated from seeds and species having resprouted from their base or subterranean organs grow regularly (Pteridium aquilinum, Chamaerops humilis, Stauracanthus genistoidis). There is no seed germination from woody species. Grasses reach their maximum development.

Fourth stage : third to fourth year; large increase in shrub species. Matorral composition evolves towards mature matorral types according to topography variations. Herb layer declines.

Fifth stage : from the fifth year onwards; the matorral recovers its original composition and structure. GARCIA NOVO (1977) specified : Erica matorral is indistinguishable from a mature stand 10 years after fire; Lavandula stoechas matorral, growing more slowly, has the same floristic composition 10 years after fire, but plant cover and size are much smaller.

GARCIA NOVO gives a good floristic description for the years after fire, but, unfortunately, he does not either give the number of species encountered, or the floristic richness compared to the burnt sites.

Northwards, in the area of Alicante, MANSANET TEROL (1982) has studied the development for 10 years of a matorral with Rosmarinus officinalis and Ulex parviflorus (Rosmarineto Lithospermetum) dominated by a tree layer of Pinus halepensis, and compared with different burnt sites. He found that the community was restored after 10 years, except for the Aleppo pine seedlings present in all the sites which did not reach their adult size at that time. Floristic richness did not seem to change through time, but this result can be due to the (very large) area of the plots he used (400 m^2). When he compared the life form ratios during the study years, he stated that the proportions remained constant through time, except for a greater abundance of therophytes during the 5 first years after fire (Table 1) which show an increase of annuals at that time.

Table 1 - Percentage of life forms through years in the MANSANET TEROL'S (1982) study

Years after fire Life forms	1	2	3	4	5	6	7	8	9	10
Phanerophytes	35	24	36	35	38	37	33	39,5	27	32
Chamaephytes	33	35	39	46	46	40	44	39,5	40	45
Hemicryptophytes	16	24	19	13	8	16	15	14	24	18
Geophytes	8	8	6	3	4	7	8	7	9	5
Therophytes	8	9	-	3	4	-	-	-	-	-

In the region of Valencia, SANROQUE *et al.* (1985) studied a dense *matorral* belonging to Rosmarino-Ericion, where *Erice multiflora*, *Rosmarinus officinalis* and *Ulex parviflorus* were the dominant species, overtopped by a tree layer of *Pinus halepensis* (therefore a community comparable with the one of MANSANET TEROL) on 3 sites from one to two years after fire. The number of observed species was higher in the burnt area in comparison with the control area because of the presence of perennial and annual herbaceous species. The same species did re-appear after fire. If these authors gave the floristic richness and composition of the burnt sites, unfortunately they did not record the changes for a long period of time (only for about 10 years).

In the mountain of Garraf (South of Barcelona) on compact limestone, PAPIO (1984, 1985), in a 3 year study, observed that the rapid recovery of vegetation after fire was through sprouts or seedlings, depending on species. Most of the species had appeared during the first 18 months. Unfortunately, the plots used are too small (1 m^2 quadrats) to be characteristic of the community (i.e. *Quercus coccifera* garrigue) and to show the floristic richness of the complete species composition. However, he gave a good description of the phenological stages.

In Majorca (Balearic Islands), MOREY (person. communic.) studied a shrub community on calcareous soil during the first three years. The tendency was to recover the previous community. Practically all the perennials would sprout; 7% only showed a seed strategy. Floristic composition was maximum during the first years after fire.

France

The dynamics of vegetation after fire was studied in the calcareous garrigues of Bas-Languedoc by TRABAUD (1970, 1980, 1983), TRABAUD & LEPART (1980, 1981). After fire, the composition of plant communities returns quickly to its initial state. Most often the species which are present 12 years after fire are the first ones to appear and become more and more numerous through time. Forty-seven plots were studied, located in 8 types of plant community representative of the area. The author did not give the floristic composition year after year but he compared his results with BRAUN-BLANQUET's associations which he referred to, so that comparison could be easily done. The different communities were studied by the diachronic method for 10 or 12 years.

One year after fire 70% of the study plots had more than 75% of the species which will be present 10 or 12 years later. Two years after fire, this percentage was over 80%, and after 5 years it reached 100%. The reversion towards a metastable state, at least for the time considered, was quickly accomplished.

This study of the development of floristic richness following fire shows that the different communities follow a highly general model (fig. 2). During the first months after fire there were relatively few species. Floristic richness reached a maximum between the first and the third year after fire, and finally tended to stabilize after the fifth year.

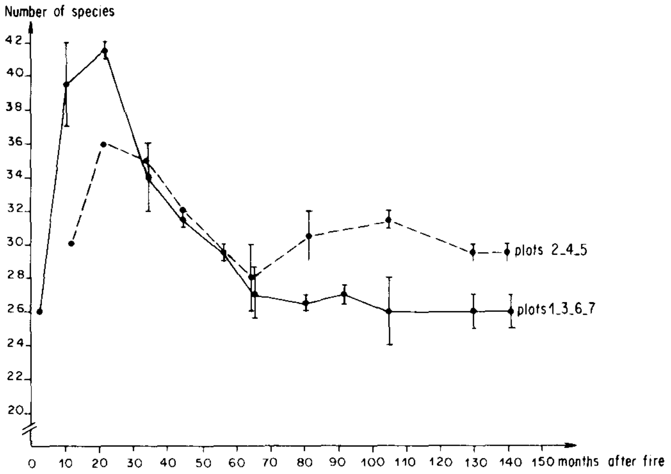

Fig. 2 Floristic richness of dense *Quercus ilex* coppices (TRABAUD 1980).

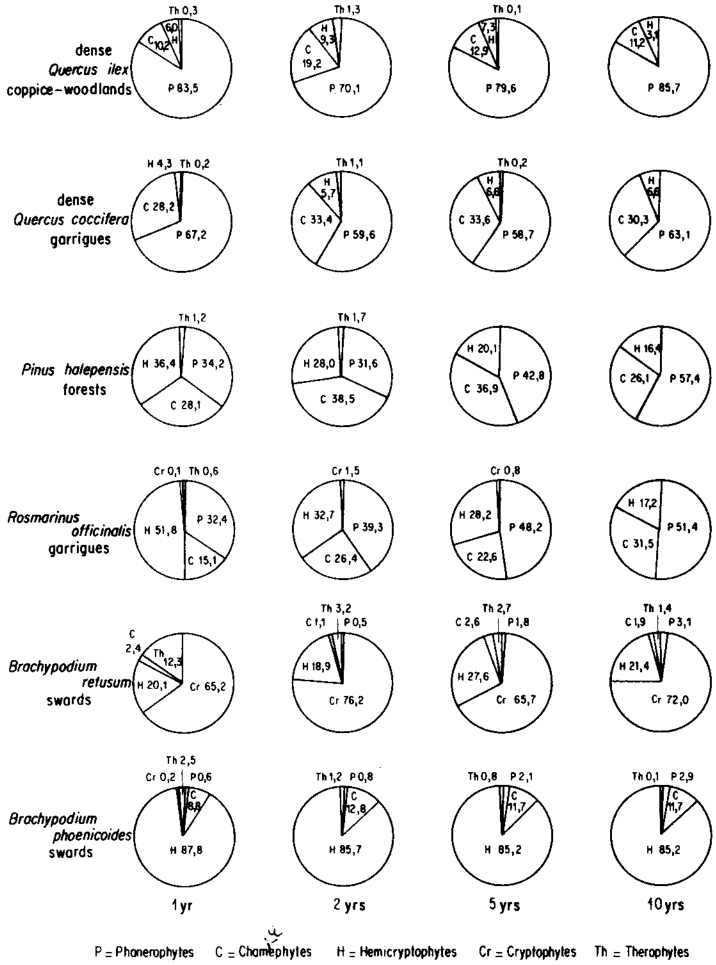

Fig. 3 Post-fire changes of life forms (according to the vegetation quantity and through time) of different plant communities (TRABAUD 1980).

The richness of the intermediate stages results from short-lived species which are progressively eliminated during the vegetation recovery. Most often these species are therophytes which do not belong to the communities. The generally higher number of species during the first three years can be attributed to the fire-induced opening of vegetation cover, the disappearance of litter and to the richness in nutrients of the upper soil. These conditions favour the establishment of alien species which will later disappear as plant cover closes up.

The reestablishment of plants after wild fire can be considered by studying the life forms (RAUNKIAER 1905) which characterize the species at the adult size. The quantitative method used by TRABAUD (1980) better reflects the actual quantity of space occupied by the plants, because the presence of a species alone, can represent either a small annual or a large tree. The proportion of the different life forms remains practically the same from the first to the 10th year after fire (fig. 3). Only the Pinus halepensis forests and the Rosmarinus garrigues deviate from this tendency : the reestablishment of phanerophytes concomitantly with the decrease of hemicryptophytes is progressive as communities get older. This is due to the fact that Pinus halepensis and Rosmarinus officinalis, the dominant species, reproduce only by seed after fire, and take a longer time to reach their adult size.

In the siliceous mountains of the Albères and the Aspres (eastern French Pyrénées), PRODON & FONS (1980), PETER (1981) and PRODON et al. (1984) by studying in a synchronic way 6 types of communities going from grass swards to oak forests (Quercus ilex or Q. suber) in plots one year to four years after fire found that reappearance of species was immediate after fire.
They recorded the names of invading species the first years after fire which include numerous

annuals. They observed that floristic richness was always higher in burnt sites in comparison to unburnt ones. They also recorded that in the sward-forest sequence, floristic richness was always the lowest in the forest. Moreover, during the first two years after fire, great modifications occur in the proportion of life forms : therophytes being extremely abundant. However, they concluded that the species which re-established after fire were the species present before fire.

In the same area (Albères) TRABAUD (not published) studied the recovery of a high maquis of Erica and Q. ilex forest, and in similar sites of Erica arborea and Calycotome spinosa matorrals of Cape de Creus (NE Spain), FRANQUESA (in publication) reached similar conclusions. Rapid recolonization is basically coming from sprouts of prolific species, with a rapid stabilization of the vegetation state; the species which are dominant appear during the first years of recolonization and are species present before fire; exogenous invading species (most often therophytes) appear and disappear quickly (they are present only during the first years); and there is no actual succession process.

Italy

In Italy, there is only the study in Sicily of DI BENEDETTO et al. (1963) in a pine forest of Pinus laricio which presents a sketch of vegetation dynamics after fire. Unfortunately, the phytosociological relevés of the burnt area are not dated so that the elapsed time since the burn is not known.

Greece

Post-fire regeneration of plant species in this part of the eastern Mediterranean Basin has been studied in phryganas (PAPANASTASIS 1977a, b, ARIANOUTSOU & MARGARIS 1981, ARIANOUTSOU 1984). This type of vegetation is mostly characterized by Sarcopoterium spinosum, Phlomis fruticosa, Euphorbia acanthothamnos, Cistus monspeliensis and C. salvifolius. Post-fire regeneration of plant species occurs through the resprouting of underground organs and the seed germination. The beginning of post-fire succession starts with herbaceous vegetation. Annual species dominate the burnt phrygana in the first year, and then decline rapidly. Legumes (Trifolium, Medicago, etc.) are the prevailing species. Also quite abundant are annual grasses (PAPANASTASIS 1977a, b). These species can strongly compete with the phrygana seedlings which appear after fire. However, this situation changes gradually so that, by the end of the seventh post-fire year, the percentages of woody and herbaceous species are almost the same as before. Unfortunately these authors do neither give the floristic richness of the plots during the succession process, nor study sites for a long period of time (e.g. 10 years).

Studying a period of 60 years by the synchronic method in Cistus stands (C. villosus, C. salvifolius), TROUMBIS (1985) found that floristic richness did not practically vary along the successive stages; the same was found for diversity. On the contrary, there were great floristic variations between years depending if two consecutive years were dry or humid, because of the possible resulting burst of annuals.

Other Countries

Despite his numerous publications dealing with the fire impact in Israel, NAVEH (1974a, b, 1975), NAVEH & DAN (1973) did not describe the dynamics of the vegetation after burning. From his work, the same trend as in the already quoted studies is occurring; i.e. the same preexisting species reinvade burnt areas. He emphasized the role and dominance of annual species during the two first years after fire.

Similar results were observed in a study in Tunisia by DEBAZAC (cited by LE HOUEROU 1974) in Quercus suber forest. In the first year, annuals covered 100% of the ground, with some shrubs resprouting. Ten years after, it seemed that the dominant trees and shrubs existing before fire had recovered so that the final community was similar to the original one.

Conclusion

In conclusion, all the authors agree and reach the same inferences which are :

a) The abundance of herbaceous species (mostly annuals) in the first years after fire is quite remarkable.

b) The majority of species which are dominant during the recovery phase of the mature vegetation, are already present during the first few years after fire.

c) the reestablishment of the original plant community is a rapid phenomenon.

VEGETATION STRUCTURE

Spain

In his description of the changes in vegetation in Donana National Park, GARCIA NOVO (1977) described a dominance of the herb layer during the two first years. During the 3rd and 4th year, the matorral evolves towards a mature matorral; the herb layer is declining. Finally, from the 5th year onwards, the matorral recovered its original structure. Unfortunately, neither the height of the layers nor their covers are given.

In his study on the recovery of the burnt matorrals near Alicante, MANSANET TEROL (1982) gave the average height of the community according to the year after fire : from 15 cm in the first year up to 85 cm 10 years after fire. He also gave indices of abundance-dominance and presented the heights through time of the most frequent woody plants.

In a similar way, FRANQUESA (in publication) in the matorrals of Cape de Creus, gave the average heights of some widespread species, but the number of replications for each species is not given. A tentative sketch is presented to explain the possible dynamics of vegetative regeneration which, unfortunately, does not give a precise picture of the structure.

Similarly, PAPIO (1984) gives the heights of the most frequent woody species. Besides, the structure of the Quercus coccifera shrub is added, showing the relative proportion of leaves and branches. The average height of this type of garrigue reaches 50 cm in one year and 1 m in 8 years. The phytomass of the above-ground parts was 2,8 t ha^{-1} one year after fire, and 16,5 t ha^{-1} after 8 years.

France

In the calcareous region of Garrigues, TRABAUD (1980, 1982, 1983) found that stand strata get more and more complex through time after fire, and vegetation tends to grow up from the lower layer to the upper layers (fig. 4). As communities get progressively older, the importance of

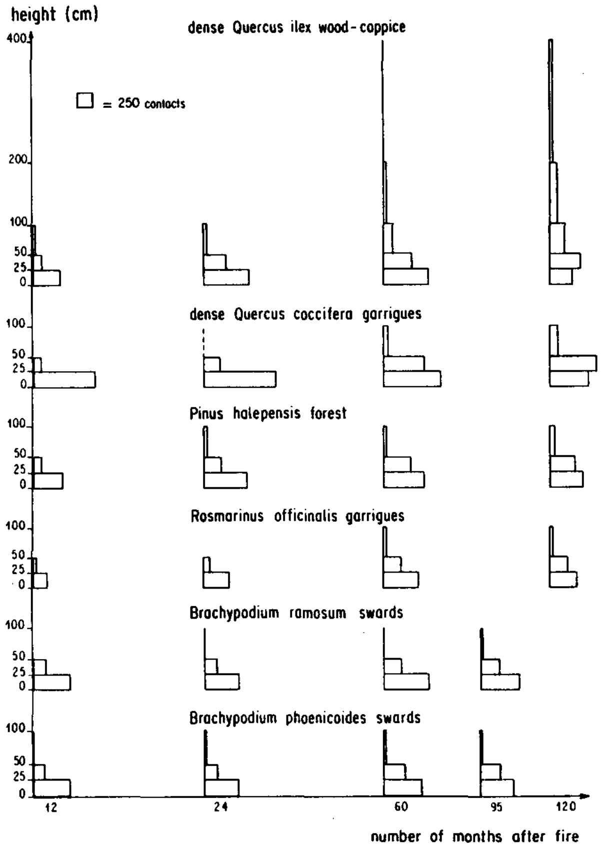

Fig. 4 Changes through time of the structure in burnt communities according to the number of contact points (TRABAUD 1980).

the lower layer (< 25 cm) decreases, whereas the higher layers increase. There is a progressive multiplicity of strata similar to the original ones : the higher strata appear later in the succession. These trends were measured quantitatively. During the 12 years of observation, Quercus ilex coppice only reaches the fifth layer (from 2 to 4 m). Pinus halepensis forests, though belonging to forest communities, did not generally grow beyond the third layer (1 m). This difference is due to the kind of survival strategy used by the dominant species of each community to regenerate after fire. Q. ilex sprouts vigorously from stumps and rapidly grows reaching 2 m height in 70 months, while P. halepensis can only regenerate by seed and thus shows a slower growth following fire (80 cm in 80 months).

The recovery of Pinus halepensis forests was more closely studied (TRABAUD et al. 1985 a, b). Whatever the type of pine forest being considered, three phases could be distinguished for the increase of the understorey phytomass : a first phase with a rapid increase of understorey lasting for two years after fire; a second phase, with a slower increase, where only shrubs grew; and a third phase, with no increase, where the understorey shrubs reached their adult size. Thirty years after fire, understorey phytomass ranged between 9 and 12 $t.ha^{-1}$. Litter, predominantly woody during the first years after fire, became foliar dominant as stands got older. Pine density increased to a maximum during the first fifteen years. It then decreased, probably because of mortality brought about by intra- and interspecific competition. The areal distribution of young pines was uniform in all the studied sites. This was apparently due to the superimposition of seed from various sources.

In the maquis growing on siliceous substrate of the Albères, PRODON (PRODON & FONS 1980, PRODON et al. 1984, PETER 1981) presented a structural schema of recovery but he only studied the first four years after fire. However, the foliar volume index allows to have an estimate of the vegetation quantity. This particularly shows that little-layered communities (swards, scrub) rapidly reached a state comparable to the initial one. In the same region, TRABAUD (data not published) is in agreement with these conclusions and ascertains that the structure of burnt communities has a tendency to return towards a state similar to the one existing before fire.

Greece

In this country, researchers have not particularly studied vegetation structure, except for the changes in phytomass. PAPANASTASIS (1977b) found a production of 3 $t.ha^{-1}$ in a Phlomis fruticosa community eight months after fire; whereas in a Sarcopoterium spinosum phrygana it was 1.2 $t.ha^{-1}$ in the first year, 1.6 $t.ha^{-1}$ in the second year, and 2.8 $t.ha^{-1}$ in the third year. In a community with the same dominant species, ARIANOUTSOU (1984) recorded a rapid growth (up to 5 $t.ha^{-1}$) during the first five years, which slows down after (fig. 5). The proportion of herb

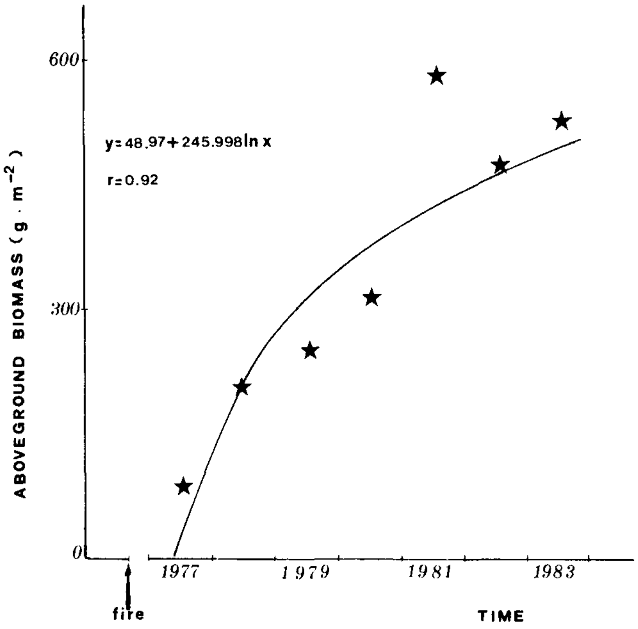

Fig. 5 Post-fire above-ground biomass for the four studied years. Fire occurred in July 1976 (ARIANOUTSOU 1984).

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phytomass was important during the first two years after fire, but then declined rapidly, shrubby species dominating from the third year onwards.

One of the most detailed studies on the changes in structure after fire is the one of TROUMBIS (1985) in Cistus stands. Cover recovery is rapid during the first years, then after 10 years it is relatively stabilized (indeed they are low shrublands). Cistus cover is practically unchanged from the sixth year onwards. Herbaceous species are very abundant during the first years and have a large cover; after 10 years this cover becomes constant, its importance varies according to sites. Vegetative spatial occupancy (recorded by the means of numbers of contacts from the species) shows a continuous increase of woody species through time, and important variations of herbaceous species. During recovery, the complexity of layers composition becomes higher and higher. At the beginning of recolonization the lower layers (< 40 cm) only constitute the main part of the vegetation, while afterwards higher layers (> 40 cm) appear and their importance increases more and more. In the same manner, the foliar organs are located in higher vegetation levels through time, whereas the woody organs are more and more numerous in lower layers.

Other Countries

In Israel, NAVEH (1974) has only given a few data on the growth of some woody species two years after fire.

Conclusions

After fire, plants appear rapidly and cover the ground surface. As burnt communities get old and return to a state similar to this of unburnt communities, structure becomes more and more complex with numerous layers. Herbaceous layers dominant during the beginning stages decrease, relayed by shrub and tree layers. In the same manner, the phytomass of herbaceous changes into woody dominant species.

GENERAL CONCLUSIONS

In the most recent studies all the authors agree that the development of vegetation after fire follows either the "initial floristic composition" model described by EGLER (1954), or the "inhibition" model of CONNELL & SLATYER (1977) :
all the pre-fire species are present immediately after fire, even if later on the relative abundance of individual ones changes. Thus, there is no real succession or floristic relays of different plant communities succeeding on the same sites as it is characteristic with secondary succession. In fact, Mediterranean vegetation does present a recovery phase by "direct" endogenous process, i.e. the species which existed before fire reoccupy the burnt sites, as opposed to an "indirect" or exogenous recovery characterized by a succession of stages as it is the case for old fields. The plants which persist are those which appear immediately after fire and which did already exist before fire.

The floristic composition and structure of burnt communities tend towards a metastable equilibrium similar to the one which existed without fire.

The plants of the Mediterranean region withstand fire by different survival vegetative and sexual traits. The present vegetation of the Mediterranean Basin results from many years of evolution during which plants have acquired mechanisms to overcome the effects of fire as well as climatic factors (such as summer drought). This evolutionary impact has been manifested by positive and vegetative feedback responses that enable direct fire tolerance or permit its avoidance, followed by vegetative and reproductive regeneration. Each type of species has developed different survival methods to survive disturbances and perpetuate themselves as well as the communities they belong to.

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Impact of forest fires on structure and architecture of mediterranean ecosystems.

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SUMMARY - The authors study the consequences of forest fires on potential forest (1) *ecosystems and on their varying stages of degradation. They evaluate the effects of disturbance on structure and architecture. Their dynamic approach takes into account criteria of specific and maximal diversity as well as the evolution of these criteria in different stages of the fire-related succession. They also show that the majority of Mediterranean ecosystems include numerous species called "pyrophytes" either because of their capacity to re-sprout from stumps or their ability to regenerate from seed after fires. Resilience and homeostasis were also studied in different types of potential ecosystems.*

KEYWORDS : Forest fires, structure, architecture, disturbance, diversity, resilience, homeostasis.

I - HISTORICAL BACKGROUND

Statistics on forest fires in France's Mediterranean regions, either by department or by region : Provence-Alpes-Côte d'Azur and Languedoc-Roussillon (BARBERO, LOISEL & QUEZEL 1986), show that during the last quarter of the 19th century, total burn area during large forest fires did not exceed 1000 hectares.

In the past, burn-beating was common practice in the Mediterranean, both for the regeneration of pastures (periodic bush burning) and for soil enrichment prior to cultivation (LE HOUEROU 1980).

This formerly common practice was abolished in France in 1841 in State and Communal Forests subject to the "Code Forestier" (Forest Laws). It continued to be used in private forests from which fires could reach state-owned forests where the shrub layer development was favoured by the absence or the irregularity of grazing after firewood cutting.

However, in private forests certain customs were respected. For example, the regeneration of the coppice was protected by forbidding access for 10 years after clearfelling and also by forestry techniques which aimed at facilitating regeneration by resprouts and runners (e.g. Coupe entre deux terres, Saut du piquet) (MIGLIORETTI, 1983).

At the end of the 19th century many marginal production zones were abandoned due to increased industrialisation and rural depopulation which further accentuated the abandon of these areas, particularly grazing lands. The areas, whether private forests, former grazing lands or farm lands on terraces and poor soil, gradually became covered with scrub. At the same time, anemochorous forest species increased on these shrublands : Pinus halepensis (35,000 hectares in 1900 - 180,000 today), Pinus pinea, Pinus sylvestris and Pinus pinaster. The last two species were widely used for carpentry and woodworking.

The development of pine forests and shrublands, the closing of bush communities formerly kept open by grazing, the resulting change in vegetation structure

(1) A "potential" forest is growing in optimal ecological conditions without almost any disturbance through human action.

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and architecture and the frequent lack of silvicultural treatment such as selective cutting in privately owned forests, all led to uncontrolled forest fires which slowly replaced burn-beating in many Mediterranean countries, especially those in the north.

Consequently, the evaluation of the extent of forest fire disturbance on ecosystems must be related to the types of communities affected by fire and their development stage. Disturbances do not affect the potential ecosystems in the same way either if fires are due to natural causes, such as lightening or volcanic eruption (cf. in taïga, tropical forest), or if they start in another type of ecosystem and then propagated in the stabilized forest(1) (Quercus ilex, Q. pubescens or Q. suber forest) or if they affect regularly burned forests which regenerate after fire (pre-forest Pinus halepensis communities, Pinus pinaster, P. pinea, P. sylvestris, etc.).

II - EVALUATION CRITERIA OF FOREST FIRE DISTURBANCES

In regard to their present characteristics, almost all of the natural Mediterranean ecosystems can be considered as disturbed. However, these natural ecosystems can be separated into two groups in relation to their present state and their spatial distribution : the more stable ones (i.e. potential ecosystems) and those subject to disturbances. In the latter case, the nature of the disturbance must be taken into account : e.g. cutting, grazing, windthrow, forest fire, etc.

The consequences of disturbance vary with the ecological situation and thus with the soil and climatic potential of the environment. Disturbances, particularly those linked with forest fires, also vary with the successional stage of a plant community, its nature, the spatial organisation of the succession and above all the frequency of fires.

Theoretical laws predicting the evolution of ecological systems have to be adjusted to the nature and intensity of disturbances affecting the ecosystem and to the dissemination and regeneration patterns of its constitutive species.

When no disturbance occurs, taking into account all of the animal and plant species characterising the last successional stages, the following trends can be observed, from the pioneer stage to the potential stage :

- increase in species diversity, increase in spatial heterogeneity
- increase of complexity in vegetation layers, increase in organisms' density, biomass and volume.
- development of community organisation and interdependant relationships between organisms
- optimum expression of the ecosystem's stabilization process by interspecific competition

Following these concepts, the potential forest stage should have the highest degree of diversity, both at the structural level, whether considered as a whole or by layer, and at the architectural level, where the interdependant relationships(2) between layers are taken into account. Generally, the more stable a potential ecosystem is, the more diverse it is, because, according to MARGALEFF (1979), it has the highest species diversity index. In fact, as we shall see later, this idea should be slightly modified if, instead of total specific diversity, floristic richness only is considered (fig. 1).

The concept of potential ecosystems' stability must be modulated as well; it is more a concept of permanence and persistence in relation to the environment, than of species diversity, which can sometimes be higher in natural potential ecosystems after non-recurrent disturbances, than in undisturbed ecosystems. Resilience (HOLLING 1973) and homeostasis (BREYMEYER 1979) in potential ecosystems vary with the type of environment and the nature of their constitutive species. Thus,

(1) a stabilized forest is a mature forest growing in an environment disturbed by man. The stability is only apparent and may change in the long-term.

(2) In potential forests, all strata and species are interacting so that competition occurs through interference.

ecosystems with different ecological potentials do not return to a stable state following disturbances after the same period of time. Also, similarly, resistance to non-recurrent disturbances is directly linked to the nature and to the biological characteristics of the ecosystem's species, although the rapidity to return to the equilibrium is linked to both the environment's potential and the reproductive and competitive capabilities of its constituting species.

On the other hand, with recurrent disturbances, the ecosystems finally reach a degradation threshold : new species appear and participate in the establishment of successional stages of populations adapted to the colonization of disturbed environments.

III - WILD FIRES AND ECOSYSTEMS' SPATIAL ORGANISATION

3-1 Impact on potential structures

All Mediterranean ecosystems are or have been disturbed by fires, but at different levels. It is thus difficult to describe and characterize the ecosystem stages by the disturbances they underwent. In this paper, we will first study the effect of fire on "potential forests" (i.e. *Quercus ilex*, *Quercus pubescens*, *Q. suber*) which are resistant to fire, then we will consider ecosystems which have undergone recurrent disturbances.

IMPACT OF DISTURBANCES BY FIRE ON ECOSYSTEMS

UNDISTURBED SYSTEMS :

* spatial-temporal directional development towards stability - permanence of potential formation related to station characteristics (climate - soil) - homeostasis

* augmentation of maximum diversity and of stational and interlayer heterogeneity - increased species density

* interspecific dependant relationships and interference competition

* forest diversity index maximized

DISTURBED SYSTEMS

repetitive ◄—— ——► occasional

- destruction of structures and architectures
- destruction of soil
- selection of certain stump sprouting species and species with numerous and resistant seeds
- quantitative increase in some species (dominant or codominant)
- decrease in tree layer diversity
- increase in floristic diversity of brush (first phase) followed by herb layer diversity (invading species, second phase)
- spatial independance of species and exploitative competition
- inapplicability of notion of forest index
- high spatial heterogeneity of single-layer or mosaic structures and systems

- low level of disturbance of average ecosystem equilibrium
- increase in overall floristic diversity
- slight decrease or increase in forest diversity index
- short resilience period
- rapid capability to restore potential homeostatic capacities

Fig. 1

- Systems with non-recurrent disturbances

Potential ecosystems are the least disturbed and are characterized by the most complex layers and architecture. Interdependant relationships are maximal (BARBERO and MIGLIORETTI 1984, 1986). For each layer and sub-layer, maximum diversity is reached because their constitutive species which compete with each other, have roughly the same abundance index and thus the same equitability index(1).

But, once again, these concepts must be used with caution. In fact, in their study on *Quercus ilex* potential forest ecosystems, BARBERO & LOISEL (1983) and MIGLIORETTI (1983) showed that the highest floristic richness was obtained in the pre-forest thickest and not in the potential structures of the *Epipactido-quercetum ilicis* (BARBERO & LOISEL 1983).

At the ecosystem level, there is a gradual process of disturbance. For example, in the forest, the effect of fire is more disturbing than windthrow. In the latter case, the resilience time is very short and often, particularly in sclerophyllous ecosystems, windthrow acts as an important factor for regeneration. Fire, on the other hand, has a direct destructive impact and immediately affects the soil by burning the organic matter, where it is abundant, as well as the rhizosphere. Fire has also an indirect effect on soil by increasing erosion, especially after the autumn rain.

For these reasons, fire exerts a selective pressure on ecosystem species (fig. 2).

- In the grass and shrub layers, non-resprouting species are eliminated and resprouting species develop because of their increased biological abilities. At the same time,

CHARACTERISTICS AND CONSEQUENCES OF DISTURBANCES BY FIRE

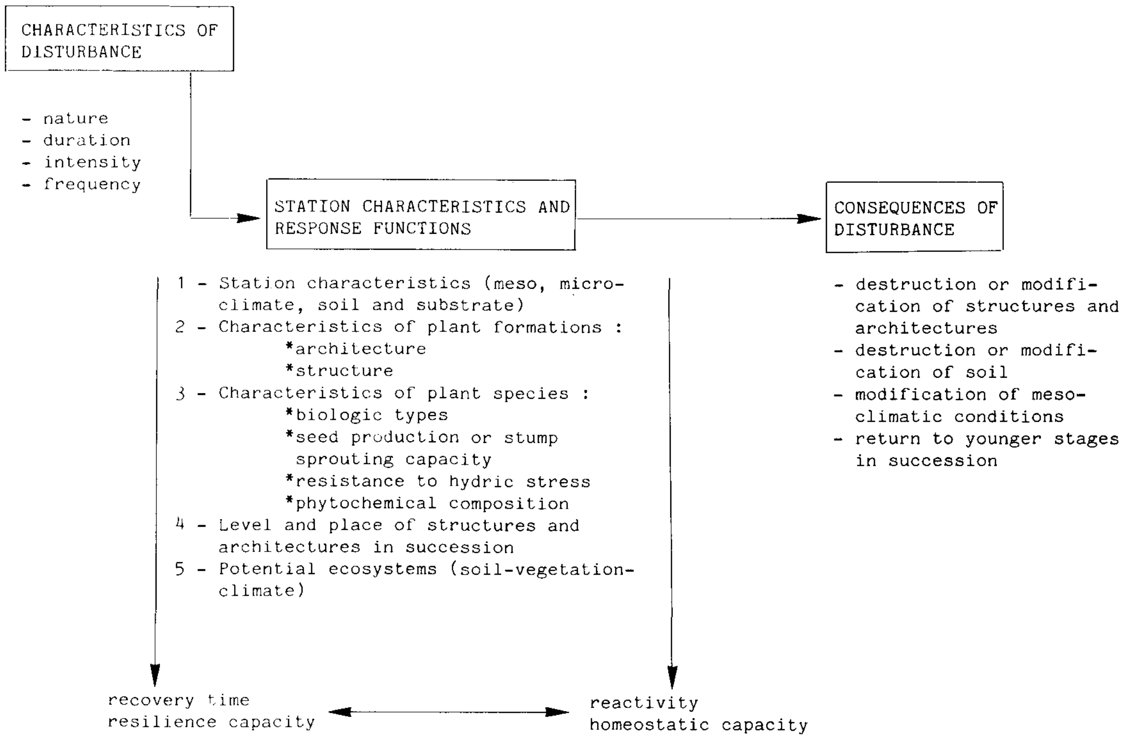

Fig. 2

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(1) The equitability index represents the ratio between floristic diversity and maximum diversity.

interdependant relationships are affected. Species not adapted to the new environmental pressures slow down or even stop their growth and reduce their seed production. Species richness is affected and, at the same time, the specific diversity index is modified because of a change resulting from modifications in the relative abundance-dominance index of fire-spread species. For this reason, it is important in such a disturbed ecosystem, to include in the calculation of the diversity index the ratio of the number of species characteristic to the forest ecosystem to the ecosystem's total number of species. The higher the ratio, the lower the level of the disturbance.

- In the tree layer, species are selected in the same way in the potential ecosystems and a loss in diversity occurs as poorly resprouting species are disadvantaged and even eliminated by disturbance. This is at least the case for a number of tree species such as Laurus nobilis and Celtis australis which used to be abundant in older Mediterranean forest ecosystems (PONS & VERNET 1971). However, among resprouting species, it is necessary to distinguish those which give off many shoots and thus are favoured by fire (e.g. Quercus ilex) from those (e.g. Quercus pubescens) with limited resprouting abilities (BARBERO & MIGLIORETTI 1986).

Fire causes by far the most important disturbances in ecosystems. It limits the ecosystem's homeostasis because of its effects on the soil and, more particularly, on the competitive equilibrium of the grass layer species, especially for the sylvatic- characteristic species. It is in these potential forest communities that the determination of ecological groups, based among others on humus nature, can best be applied (DUVIGNEAUD 1980).

For windthrow, as already indicated, the consequences are limited. On the other hand, when disturbance is linked to repeated cuttings, the differences in resprouting ability of the various species determine after a certain time the development of an ecosystem with a relatively stable composition where very demanding plants are eliminated, as it is the case for forest-mull hemisaprophytes : e.g. Epipactis microphylla, Epipactis latifolia, Cephalanthera ssp., Platanthera chlorantha in Quercus ilex forests (BARBERO & LOISEL 1983).

- Systems with recurrent disturbances

As indicated above, recurrent fire favours the development of resprouting species : i.e. Phillyrea media, P. angustifolia, Rhamnus alaternus, Quercus coccifera, Lonicera implexa, Jasminum fruticans, etc. in Quercus ilex zones; Erica arborea, Arbutus unedo, Erica scoparia, etc. in Quercus suber zones.

These plants can spread because of the regression of shrubs which are poorly adapted to forest fires : i.e. Viburnum tinus, Rhamnus infectoria, Rosa sempervirens, Cytisus triflorus, Cytisus candicans. The former species show considerable regeneration ability so that they rapidly occupy the initial community's volume preventing, at least in the beginning and if their density is sufficient, the latter species to grow or develop their young seedlings. The initial ecosystem obviously undergoes a loss in species richness ("drift"). On the other hand, when stump density is low, forest fires open the communities and lead to the development of species originating from shrub or pre-forest communities; floristic richness can then greatly increase. We can indicate the examples of the development of Pistacia terebinthus, P. lentiscus, Rhus coriaria and Osiris alba in Quercus ilex and Q. suber forests and of Rhus cotinus and Coriaria myrtifolia in Q. pubescens forests.

In these oak forests there is an increase in the total phanerogamous floristic richness which widely compensates the loss in sylvatic-characteristic species. On the contrary, the ratio of sylvatic-characteristic species to the total number of species in the community decreases. This trend is highly visible in sclerophyllous ecosystems undergoing recurrent disturbances. Formations originating from potential ecosystems behave like true ecotones in which the grass layer which characterized the initial ecosystem has almost disappeared, due to the impact of fire on the soil; the shrub layer shows a higher specific diversity.

When forest fires are frequent in structures where resprouting species have a low density, very aggressive plants rapidly settle. In contrast to the resprouting previous species, their population strategies are of the "R" type (1) because their biomass production per individual is low compared to a high investment in seed production. This leads to very high individual spatial densities.

In degradation zones of Quercus ilex forests, the following species can be found : Rosmarinus officinalis, Ulex parviflorus, Cistus albidus, C. monspeliensis; in those of Quercus suber forests : Calycotome spinosa, Cistus monspeliensis, C. salviaefolius; and last, in those of Quercus pubescens forests : Genista scorpius, G. cinerea, or Sarothamnus scoparius. Among these species, Calycotome spinosa, Genista cinerea or Sarothamnus scoparius, both resprout and regenerate from seed after fires.

This explains that with increased fire frequency, mosaic vegetation structures occur with very low floristic and specific diversity. A single dominant species or two co-dominant species with the same regeneration strategies then thrive, generally associated with a few others of very low density.

These poor communities, usually one-layered, often have their specific distribution determined by the site's pedology or geomorphology. Their constitutive species have low trophic requirements, high vegetative or sexual reproduction abilities and are generally sclerophyllous plants. In the Quercus ilex zone, we can mention Quercus coccifera on compact limestone, Rosmarinus officinalis on marly limestone or marl and Ulex parviflorus on clay colluvia.

In areas where resprouting species are rare, once again typical "R" strategy plants develop. They can be either from the Cistaceae family : Fumana thymifolia, F. loevipes, F. ericoïdes, Cistus albinus, C. monspeliensis, C. salviaefolius, C. ladaniferus; or from the Labiatae family : Lavandula vera, L. latifolia, Satureia montana, Thymus vulgaris. It should be pointed out that these substitution mechanisms are widespread in the Mediterranean region (NAVEH 1973, 1975, 1977, LE HOUEROU 1973, 1980; LIACOS 1973, MOONEY & CONRAD 1978, TRABAUD 1978, 1979, MARGARIS 1979).

Often, during the two or three years following a forest fire, before resprouting or seed-regenerating woody plants conquer all of the vegetation carpet, the environment is temporarily occupied by nitratophytes : Chenopodium ssp., Erigeron ssp., Atriplex ssp., Polygonum ssp., Urtica ssp., Psoralea ssp. or on degraded soils by therophytes, which act as temporary invaders, organising highly heterogeneous and shortlived stages of the post-fire succession. The latter species, revealing the extremely diversified seed richness of the soil, are called "anthracophytes" by LE HOUEROU (1980) for they are not characteristic of burnt sites but of soil enriched by fertilising elements (like ash).

True pyrophytes (KUHNHOLTZ-LORDAT 1939) are species whose propagation, multiplication or reproduction are stimulated by fire, or which show different fire-resisting mechanisms. Around the Mediterranean, not a single species has its ecology specifically linked to burnt environments, though this is not the case for other mediterranean zones, such as California (QUEZEL 1981).

On deep soil, more frequent fires even lead to the preceeding species' disappearance and to their replacement by herbaceous species such as Brachypodium ssp., Festuca ssp., Bromus ssp. Such a community shows a high floristic richness and specific diversity after fire disturbance. Grazing is necessary to avoid the growth of woody plants and to ensure the maintenance of herbaceous species.

As we have seen, fire has a large range of consequences on potential forest ecosystems. When disturbances do not modify the ecosystem's equilibrium in a very important way, specific richness can even increase, as well as the floristic diversity if all species, and not only the sylvatic-characteristic species, are considered. In sclerophyllous communities, where competition for light is high, the opening of such stands creates conditions favourable for a very rich grass layer, different from the potential ecosystem.

When the disturbance stops and the forest ecosystem's complex architecture is gradually re-organised, interspecific competition mechanisms become fully active and the number of sylvatic-characteristic species able to co-exist in each layer tends to stabilize; maximum

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(1) Species with a "R" survival strategy produce a lot of seeds to regenerate. They are therefore favoured in the first stages of plant succession.

diversity is then reached. On the contrary, when disturbances are recurrent and important, specific richness drops significantly. For a given formation type and in its corresponding ecosystems, only species with a short reproduction cycle, where "R" strategy prevails over vegetative biomass production, can survive.

3.2 - Impact on the structures of the forests originated from post-cultivation systems

Numerous forests have taken root on abandoned cultivated lands or open pastures especially since the main period of rural depopulation during the 19th and the beginning of the 20th century. Pinus halepensis, a typical pyrophyte, is found in these young ecosystems, which explains its recent and continuous extension, even if it is frequently subject to forest fire. In these forest ecosystems, the initial dynamics corresponds essentially to the successional stages defined by MOLINIER (1968) and LOISEL (1976) for Provence and by BRAUN-BLANQUET (1936) and TRABAUD (1976) for Languedoc :

The initial stage is characterized by Brachypodium ramosum or B. phoenicoïdes grasslands, theoretically used as pastures. Such grasslands are rich and diversified in Gramineae, Papilionoideae, Compositae, Umbelliferae and many geophytes (Gagea ssp., Muscari ssp., Ophrys ssp., Scilla ssp., Allium ssp., Gladiolus ssp., etc.) but must be regularly grazed by livestock to be maintained.

The next stage is characterized by the appearance of "Tomillares", with Thymus vulgaris, Dorycnium suffruticosum, Staehelina dubia, Lavandula latifolia, which are not very diversified even at their optimum because of allelopathic reactions.

On these slow growing communities, Juniperus oxycedrus formations appear with "R" strategy species with higher biomass production than before. They usually are Papilionoïdeae : Spartium junceum, Ulex parviflorus, Genista scorpius, Calycotome spinosa.

Pinus halepensis can settle and regenerate in such formations, either at the first stages or later on. Once developed, Pinus halepensis forests reach a maximum of production at the age of 70 (ABBAS, 1986), but tree fertility occurs much sooner from the age of 10 to 15. In such ecosystems, fire starts very easily and causes soil degradation. A rapid recolonization then occurs with "R" strategy species : Rosmarinus officinalis, Cistus ssp., or several resprouting Papilionoïdeae such as Ulex parviflorus or Calycotome spinosa.

Pinus halepensis itself shows high regeneration abilities in such communities. But recolonization always depends on the presence of seedbearing trees situated near the burnt site (ABBAS, BARBERO & LOISEL 1984).

Fire frequency has an effect on the complexity or the simplification of the structures of the forest. If the period between two fires is long (e.g. 15-30 years), the structure is complex and diversified, with trees, high and low shrubs, lianas, chamaephytes and grass. After recurrent fires, the structure tends to be very simple with only low shrubs and grass. Shrub communities are often characterized by a single dominant species or two co-dominant species.

In extremely degraded environments, if the soil enables the development of a grass layer, hemicryptophytes, geophytes and therophytes are co-dominant : the therophyte number is increasing noticeably with fire frequency and soil poorness.

On rich soils, with a continuous cover of Gramineae, the regeneration of seeds brought by birds, water or wind is often unsuccessful even if possible, because of interspecific competition. The species richness of these formations remains low because of the short colonization period between two consecutive fires.

When the colonization period is long and the pine forests do not burn, a change occurs in the underbrush. Heliophilous Papilionoïdeae regress and ornithochorous species which support competition in shade conditions develop : Phillyrea media, P. angustifolia, Viburnum tinus, Rhamnus alaternus, Asparagus acutifolius. Barochorous species such as Quercus pubescens and Q. ilex appear last and create a sub-layer in the pine forest.

The absence of fire disturbance leads to the development of an ecosystem whose floristic richness increases with time until the Quercus ilex canopy limits sunlight availability to the underbrush. Species which can grow well in this pre-potential ecosystem show "K" type demographic strategies whose aim is to constitute stabilized populations in saturated environments. They can be contrasted with the species constituting young pine forest understorey which expand after fire because of their fast colonizing cycle.

Mediterranean forest ecosystems subject to fires can thus be divided into two groups :

- Those originated from initial potential forest ecosystems where resprouting species prevail. When fire is frequent, they are characterized either by a single dominant species (for example Quercus coccifera) or two co-dominant species (for example Erica arborea and Arbutus unedo) depending on the substrate.
- Those originated from forest ecosystems on abandoned cultivation land, where anemochorous and "R" strategy species prevail. Once again when fire is frequent, only one or two species remain : Rosmarinus officinalis, Cistus ssp., Calycotome spinosa.

Fire frequency and intensity, plant resistance and regeneration abilities are the main characteristics leading to the installation of post-fire structures in which productivity decreases with increasing fire frequency. This can be explained by the fact that "R" strategy species produce less than "K" strategy species and that the environment loses its nutrient.

3.3 - Impact on architecture

Fire has a highly variable impact on forest architecture and the role of fire varies depending on the ecosystem's type and structure.

In pine forest systems, whether Pinus pinaster, P. pinea, P. sylvestris, or P. halepensis, fire affects the entire structure, but post-fire regeneration strategies are very different depending on the initial structure.

Pine forest with a shrub underbrush is usually a two-layered community because shrub density generally prevents the development of grass layer. Regeneration is rapid : if resprouting species are present, a replacement shrub develops; if absent, a one-layered grass ecosystem develops.

The rapidity of pine regeneration depends on the proximity of neighbouring seed-bearing trees from the site. If close enough, recolonization is fast and seedlings are numerous on the burnt site. However, pine seedling development is often staggered which allows the installation of shrub species on sites already colonized by pine seedlings of different ages. From there, arises a multi-layered highly heterogeneous architectural system.

These spatial recolonizing models can explain the large variations in height observed in Mediterranean pine forests, and particularly in those frequently disturbed by fire.

In potential forest systems, fire frequency has also an impact on architecture. Generally, the number of resprouting species increases in each layer. For this reason, in many zones of Quercus ilex mixed with Q. pubescens, recurrent fires induced the dominance of Q. ilex to the detriment of Q. pubescens because of its better resprouting ability (BARBERO & MIGLIORETTI 1986). In zones where mixed Q. ilex-Q. coccifera communities occur, the latter has such a strong regeneration ability that its stands were always widely favoured.

In any case, forest fires destroy the preexisting architecture. Its reconstruction is directly related to the nature of the initial species community to their density and resprouting abilities :

-- in densely populated Quercus ilex stands, a one-layered system will prevail for many years after fires;

-- in a resprouting shrub system including Q. ilex stands, a two-layered system will rapidly develop after fire;

-- a two-layered system (herb and tree layers) will also replace Q. ilex and seed regenerating shrubs (Cistus ssp., Genista ssp., ...) after fire.

As a general rule, forest fires lead to a simplification of forest architecture.

There is a case, however, where the tree layer is not destroyed after burning : the Quercus suber community. The ecosystem general layering remains, even after frequent fires. Nonetheless, a selection takes place in the shrub layer in favour of resprouting species : Erica arborea, E. scoparia, Arbutus unedo. Because of competition in a system where shrub cover is high, the grass layer is either very scarce or even nonexistent. Recurrent forest fires do contribute to soil impoverishment there and thus to the creation of a two-layered system.

In Quercus suber-Quercus ilex or Quercus suber-Quercus pubescens mixed communities, high fire frequency can eventually lead to Quercus suber as the only remaining species. In thermophilous littoral communities of Quercus suber recurrent fires lead to an important drop in species richness and diversity. Shrubs such as Pistacia lentiscus and Calycotome spinosa expand while others, such as Myrtus communis and Genista linifolia which resprout and regenerate less easily, regress.

IV - PERIODIC BUSH BURNING AND ECOSYSTEMS' SPATIAL ORGANISATION

For centuries burn-beating and mainly periodic bush burning have been the main cause of disturbance in forest ecosystems whose structure and architecture were considerably simplified. Fires are still widely used for different purposes nowadays, at both low and high altitudes. In the Mediterranean region, however, due to rural depopulation and multiplication of scrub zones, they have been progressively replaced by uncontrolled fires.

This centuries-old technique of periodic bush burning has left many scars on pre-forest and forest ecosystems. In forest ecosystems the vegetation carpet is characterized by a single dominant chamaephytous species, either Gramineae or Papilionoïdeae. It offers low heterogeneity and low species richness and diversity.

Maximum diversity is never achieved and species diversity of the grass layer is always low compared to open grasslands. In general in the grass layer, there are only one or two dominant or co-dominant species associated with a few others, which always show very low abundance-dominance indices.

In this group are also Quercus ilex forests used as pastures. Their stands have a low stocking density, and a grass layer develops in which Brachypodium ramosum is dominant due to periodic bush burning. When grazing is abandoned, such structures are invaded by Quercus coccifera where available, or by other very aggressive species : Cistus ssp., Osyris alba, Rosmarinus officinalis, Ulex parviflorus. In the Quercus pubescens Mediterranean zone, there are forests where Brachypodium phoenicoides and B. pinnatum are dominant and in the Quercus pubescens supra-Mediterranean zone, there are forests where Brachypodium pinnatum is dominant.

All of these structures have their main origin in periodic bush burning, which highly favours species with vegetative reproduction ability. Some seed reproducing species are also selected : Genista hispanica, G. pilosa, G. sagitallis.

In zones where grazing lands were rationally maintained, without the systematic use of periodic bush burning, as in the chestnut forests of the Maures (Var, France) the grass layer showed a high floristic diversity (LOISEL & MERCURIN 1972). Fire frequency again is important. In areas where periodic bush burning is not frequent and soil is appropriate, the impact was to reconstitute a certain floristic diversity in relatively poor zones.

Historical surveys showed that in Ulex parviflorus and Genista cinerea stands, burning prior to cultivation brought a rapid increase in species richness and diversity in the first few years through the development of herbaceous species.

Once more, it is the high frequency of the disturbance which leads to an important drop in diversity through the replacement of numerous species either by a single dominant species with high vegetative reproduction abilities, or by a single species with high seed production and reproduction potential. This could explain the expansion of chamaephytous Papilionoideae in every Quercus pubescens community previously or still subject to periodic bush burning : Coronilla australis, Dorycnium suffruticosum, Genista pilosa, G. hispanica.

V - CONCLUSION

Recurrent forest fires have a very important impact on the structure and architecture of Mediterranean ecosystems.

- They greatly modify stabilization mechanisms in forest communities because they reduce the effects of interspecific competition through interference. If fires become more frequent in ecosystems containing resprouting plants, species with shorter colonizing cycles will develop.

- When fires are frequent, they modify the community organisation and lead to a decrease in species diversity. On the other hand, occasional fire can be responsible for an increase in species richness and diversity.

- They disturb and simplify vegetation layers and thus architecture. Wild fires usually eradicate every layer, except in Quercus suber forests where the tree layer remains after fire and where buds and branchlets start growing almost right after burning. Fires have a tendency to favour the layer in the progressive succession which corresponds to the previous dynamic stage :

- shrub layer in forest ecosystems
- grass layer in shrub ecosystems

We can conclude that the loss of diversity in numerous Mediterranean ecosystems, as well as the low index of evolution and of architectural differentiation, result from recurrent disturbances, whether from cutting or forest fires.

In the Mediterranean region, considering the importance of different types of disturbances, forest ecosystems with a well-organised architecture are extremely rare and their species richness is considerably lower on average than in deciduous temperate forests. Temperate forest ecosystems, whether Mediterranean or Middle-European, have a lower richness and diversity when compared to tropical forests which are less subject to recurrent disturbances.

In Mediterranean forest ecosystems, Quercus pubescens forests show a higher architectural heterogeneity than sclerophyllous forests characterized by simpler architectural models.

Forest fires, if they decrease vertical organisation, i.e. the architecture of the forest, greatly increase spatial heterogeneity. Far from being identical, successional stages in grass or shrub layers vary significantly from one site to another. Such a diversity depends on the nature of the biological potential existing in the sites neighbouring the burnt sites which participate in the recolonization process, and on soil capacity. The spatial distribution of dominant or co-dominant species depends on climatic, geomorphologic and edaphic conditions. One of the main characteristics of the landscape in the studied areas which are shaped and constantly modified by fire, is to have very low species richness and diversity and at the same time a high spatial heterogeneity. The fire-shaped landscape perfectly corresponds to the definition of the mosaic community.

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Incidence of exogenous factors on the regeneration of Pinus halepensis after fires.

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SUMMARY - Is the expansion of the Aleppo pine (Pinus halepensis) in Mediterranean region directly linked to fire ?

To answer this question, the authors have analysed the biological characteristics of the species, the regeneration conditions after fire and, more particularly, the influence of exogenous factors on regeneration (presence of seed bearers, maintenance of burnt trees,...).

The investigation on Aleppo pine expansion on farmland leads to the conclusion that fire plays an indirect role only in the structural development of this species.

KEYWORDS : Pinus halepensis, structure, regeneration.

The recent history of the Aleppo pine is most closely connected to the phenomenon of rural depopulation and this remains true despite the incidence of fires which have destroyed and continue to destroy tens of thousands of hectares each year in Provence-Alpes-Côte d'Azur (PACA) and in Languedoc-Rousillon (LR).

Forestry statistics alone clearly demonstrates this fact :

- Prior to 1870, forestry engineers estimated that pine trees covered approximately 30,000 ha., mostly on grazing lands: traditionally, garrigues and maquis have always been important pasture lands.
- In 1878 the total surface area covered by Pinus halepensis was estimated at 36,000 ha.
- At the beginning of the century, the total surface area was 105,000 ha and reached 130,000 ha in 1950.

ACHERAR (1980) and ABBAS (1983) estimate that today Pinus halepensis covers 180,000 ha.

Pine forest area has thus increased by five, since the beginning of the century and, as ACHERAR (1980) has shown, pines increased by 157% in LR and 57% in Provence.

What are the reasons for this expansion ?

- Massive reforestation was undertaken at the end of last century, around 1880, when, according to CARMENTRAND (1940), pines were planted for crates and railway sleepers.
- the abandonment of cultivated areas over a period of time in Provence and Languedoc.
    - in the first phase, from 1878 to 1908, pine forests increased dramatically in Provence
    - in the second phase, from 1908 to today, pine forest colonization increased in Languedoc
- the abandonment of grazing land has favoured pine expansion
- the reduction of holly oak thickets which, according to ABBAS (1983), have decreased by a quarter since the beginning of the century in PACA and LR, has favoured the extension of pine.

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These factors point out the profound transformation of forested surfaces and the modifications in the landscape which have appeared since the beginning of the century.

The extension of pine shows its remarkable success in spite of fire which have destroyed on average 26,000 ha per year of Mediterranean forest during the past thirty years (BLAIS 1979). According to GADANT (1979), pine forests represent the most heavily damaged areas. LOISEL (1976) pointed out that the expansion of Aleppo pine has occurred at the expense of holly oak, particularly in the lower level of the meso-Mediterranean zone.

In the following section we will first examine the biological characteristics of this species which belongs to the group of colonizing conifers. We will then examine the conditions necessary for the success of seeds as related to the environment where they develop.

In the second part, we will look at the structural features of pine forests in relation to the plant communities in which they develop.

BIOLOGICAL CHARACTERISTICS OF ALEPPO PINE, SEED SPREADING AND GERMINATION

The Aleppo pine has an "R" type strategy with abundant anemochorous seeds. This gives the advantage of pine over other species with heavier diaspores whose spatial distribution is considerably slower.

This fact does not mean, however, that the Aleppo pine is systematically favoured by fire when it invades the soil and then the whole ecosystem. Pine expansion occurs after rural depopulation, which shows that fire is not indispensable for its dispersal.

The data on seed propagation in normal conditions show an early fertility at 15 to 20 years when a large quantity of seed is released. Seed spreading generally takes place from the end of July until August, September or October depending on the year for three-year-old cones and, occasionally, for mature two-year-old cone (LOISEL 1967, FRANCELET 1970, ACHERAR 1981, ABBAS 1983).

There is an abundant seed fall, but, to start its opening, the cone must be in contact with heat. Cones alternately open in the middle of the day and close at night with increased humidity. Certain cones, however, do not open before 4 or 5 years. They are very lignified by this time and are not totally destroyed during fire. This is one of the seed defense mechanisms against fire which is important in post-fire recolonization. Under natural conditions where low temperatures prevent cone opening, the cone remains closed even if it is dry, because its resin prevents internal drying out. Heat on a limited area of the cone causes a crack in the resin joining the scales. The cone then begins to dry out and the scales open (FRANCELET 1970).

Cones remain on the trees for a long time (DEBAZAC 1960) and their presence can be dangerous during fire because they may explode and spread the fire. However, old unopened cones will produce an abundant seed fall during fire. ABBAS, BARBERO & LOISEL (1984) have shown that 4 to 5-year-old unopened pine cones have a large quantity of fertile seeds, which is not the case for all conifers.

The Aleppo pine natural regeneration is favoured after fire because a large number of seeds remained in older cones, now fall to the ground. Of course, this is not the case if cones have been completely destroyed by fire.

Moreover, the natural regeneration of Aleppo pine is favoured by the increase un nutrients released by wood ash after fire (NATRAL 1962, KARSHAN 1973, TRABAUD 1980).

Seed dehydration through heat increases both the capacity and percentage of germination and its success depends, as we shall see later on, on the ecological characteristics of the station.

Time and period of the fire occurrence do not appear to influence the natural regeneration of Aleppo pine. When the population is destroyed over a large area and seed bearers are far from the burnt site, regeneration can only begin after a certain number of years following the fire and will depend on the relative proximity of seed bearers.

Seed sowing and germination experiments have been carried out in situ by LOISEL (1976) which showed that germination begins the fall following sowing, due to the favourable soil moisture. These seedlings, however, are highly sensitive to strong sunlight, particularly during the following summer, which can even provoke the death of the young plants.

Protection of the seedlings is necessary for their maintenance as long as they are not completely covered. In fact, in thick grassland, despite abundant seed fall, regeneration is characterized by a high rate of failure the following summer.

In young populations of 15 to 20-year-olds, the survival of the population is guaranteed as long as there is no fire because the 20-year-old trees can provide abundant fertile seeds out of the surviving ones.

But, if a fire occurs before this period of 20 years, the Aleppo pine population will be eradicated and regeneration will be much slower and will be due to allogenous seeds often coming from far away.

The Aleppo pine natural regeneration is irregular and depends on the particular situation. It is thus important to refrain from considering that fires automatically favour it.

The irregularity in regeneration is related to several factors and, despite abundant seed fall, all seeds are far from being fertile :

- insects and small rodents are frequent predators on seeds;
- bacterial and fungal parasites have a damaging effect on fertility;
- once out of the cone seeds rapidly lose their germinating capacity due to an accelerated depletion of their reserves;

- atmospheric humidity levels and strong climate variations are a barrier to the survival of young seedlings;
- young plants are often destroyed by rodents.

Thus fire has differing effects on the Aleppo pine's regeneration. This brings us to reconsider some of the reasons why it was thought that fire automatically favoured this species. In the case it is true, it is always indirectly linked to increased seed fall from older cones.

To check these hypotheses, we studied more than 5 ha of fire-burned areas in the Bouches-du-Rhône between 1973 and 1979.

In these zones, countings were made of seedlings obtained from surviving trees after fire from both felled and unfelled burnt trees. The area of the sampled plots was 100 m^2 and correlations were tested :

- between time of year of fire and the amount of seedlings observed per hectare
- between seedlings and surviving trees
- between seedlings and uncut burnt trees
- between seedlings and cut burnt trees

Sampling was stratified on the basis of simple ecological factors; elevation, exposition, substrate and soil nature. Many different sample plots were chosen within the burnt site in order to take site heterogeneity into consideration (ABBAS, BARBERO, LOISEL & QUEZEL 1984).

RELATION BETWEEN SLIGHTLY BURNT LIVING TREES AND POST-FIRE REGENERATION

The Aleppo pine post-fire regeneration is directly influenced by the number of trees left untouched after fire. The greater their density, the greater the chances of regeneration success. We see here the concept of dissemination pattern of seed bearers in relation to ecological conditions necessary for regeneration.

RELATION BETWEEN TIME OF YEAR OF FIRE AND NATURAL REGENERATION

We noted that a high degree of regeneration occurred after three August fires (more than 1000 seeds/ha). However, this does not necessarily mean that the month of August had a significative influence on regeneration. Fires in August, September and October can have poor regeneration. In fact, climate conditions of the following year, as well as the importance of predators and parasites could explain the significant spatial and temporal heterogeneity observed in regeneration (ABBAS, BARBERO & LOISEL 1984, LOISEL 1976).

RELATION BETWEEN POST-FIRE SEEDLINGS AND UNCUT BURNT TREES

On different experimental sites, both abundance and growth of seedlings vary considerably due to root competition and light conditions imposed by other tall resprouting plants and to crowding by Graminae (Brachypodium div. sp.). The interspecific competition seen indirectly through the species density and cover, is a determining factor in the success or failure of the Aleppo pine.

In forests where burnt trees are not all removed, seedling development is generally low and shows reduced growth. These conclusions concur with those of KARSHAN (1973) and ACHERAR (1981), which explain observations in undisturbed zones where Aleppo pine seed fall is high, but where seedling development is relatively rare.

This explains the low regeneration of pine under pine forest.

RELATION BETWEEN POST-FIRE SEEDLINGS AND BURNT TREES EITHER PARTIALLY OR ENTIRELY CUT

High regeneration and growth rates are evident in cleared areas, as well as in clearcut areas. The Aleppo pine is thus favoured by the regression of inhibitory factors, such as competition for light and moisture, but also by the removal of trees in the first year following a fire (KARSHAN 1973) and by the absence of a competitive herbaceous cover (LOISEL 1976).

FUTURE OF SEEDLINGS

Certain areas of the St. Mitre les Ramparts State forest were studied diachronically. Regeneration after the 1976 fire was high and the number of seedlings observed was 170 plants/m^2, most of them were of about one metre high. We can conclude that thinning out is high and that interspecific competition and difficult summer conditions account for this strategy (LOISEL 1976).

Our results are similar to those found by KARSHAN (1973) who showed in a similar study that the maximum number of Aleppo pine seedlings was 155/m^2 with a maximum height of 115cm.

It is obvious that these examples represent the optimum regeneration and that large variations exist in nature, depending on climate conditions, vegetation cover type and degree of predation.

PINE FOREST STRUCTURAL CHARACTERISTICS

The future of seedlings determines the spatial and temporal heterogeneity of Aleppo pine structure. There are many different types of structures and it is difficult to establish a synthetic typology between them.

The Aleppo pine occurs naturally in two major systems :

1) the first system is constituted by groups belonging to the pre-forest (maquis, garrigue) or the forest (coppice, high forest) and to their degradation stages which have been maintained for centuries by different uses. In these landscapes, the Aleppo pine is associated with resprouting species:

- Quercus ilex coppice under Aleppo pine forest
- Quercus coccifera shrubland under Aleppo pine forest
- Ericaceae shrubland (E. scoparia, E. multiflora) under Aleppo pine forest

2) the second system corresponds to pine forests developed on farming land and has the following sequence :

- perennial Gramineous grasslands (Brachypodium pinnatum, B. phenicoïdes)
- Tomillares with low chamaephytes followed by Papilionoideae shrublands

These successive stages are often colonized by pines at different time periods, which is quite different from what occurs in burnt areas. Heterogeneous populations are organised in terms of architecture.

The progressive evolution of this system is led by ornithochory. After 40 to 50 years, species such as Phillyrea media, Rhamnus alaternus, Lonicera implexa, Jasminum fruticans, Pistacia div. sp., Cormus, etc. develop or are able to resprout after fire.

After 70 to 80 years, barochorous species, such as oak, appear. Some pine forests from post-farming sequences have been studied by ABBAS (1986) in the Bargemon region. They show high biological potential.

Both major systems have been affected by fires of different periodicities. The recolonization of burnt areas is the following :

- a regeneration model of resprouting species characterizing highly-developed post-farming pine forests and traditional pre-forested systems where coppice regeneration by shoot is dominant.

- a regeneration model of species with a high germinative capacity, linked to high seed fall for "R" strategy species : Rosemarinus, Calycotome, Cistus, etc. (TRABAUD 1980).

In these different groups the Aleppo pine plays the role of colonizing conifer which rapidly occupies an area. Seed fall is related to proximity of seed bearers.

- seed fall can be very important and the spatial occupation of a site leads to a dense population which is thinned naturally through intraspecific competition due to limited trophic resources on poor soils (competition through exploitation). The resulting population contains even-aged trees and regeneration of the Aleppo pine underneath is very low.

- in areas further away from seed bearers, pine colonization will be at irregular time periods. It depends directly on interspecific competition and on the capability of aggressive species to occupy the vegetation carpet. If the initial cover is sparse, colonization can continue without interruption and the resulting forest community will be very heterogeneous (in terms of stocking density, age, diameter, height).

Older unburnt pine forests do not regenerate. They can be classified into two groups according to their dynamics : one group is characterized by the dominance of the Aleppo pine forest on coppice or green oak forest (i.e. pre-potential group); the other group is characterized by, for instance, pine forest on garrigue with Quercus coccifera or Erica shrublands. This less evolved group is blocked until windblow occurs due to Eastern or Mistral winds and creates gaps for regeneration.

Because pine silvo-pastoral system has been abandoned for many years, the directional evolution of different pine forests has led to a pre-potential forest represented, depending on the time periods and site potential, by :

- a multi-layered system with a complex upper-heterogeneous layer of Quercus ilex or Q. pubescens

- an increase in shrub layer diversity in favourable edaphic zones with the presence of sylvatic species in the herb layer.

Within a time-frame of 100 to 150 years, pre-potential pine forests will gradually show the disappearance of pine trees and the development of the potential forest populations on the former sites, i.e. Quercus ilex, Q. pubescens.

- the main factors which intervene in the structure composition and species richness are :

1) The time of the disturbance : the overall species richness is the greatest in pre-potential systems and the lowest in young pine forests.
2) The frequency of disturbance : the more frequent the disturbance is, the lower is the specific richness. A high disturbance frequency causes the survival of a few dominant of co-dominant species only.
3) Climate and especially the edaphic characteristics of the site : the rapidity of recovery of burnt pine forests is greater when the soil is deep and rich and when its moisture content guarantees seedling survival during the summer.
4) The diversity and floristic richness of plant communities near the burnt pine forest : species dissemination pattern in neighbouring burnt sites and type of diaspore dissemination, directly influence the structural models.

CONCLUSION

Fire is obviously an important factor in the spatial and temporal heterogeneity of Aleppo pine stands structure and architecture in the Mediterranean region.

As all pioneer species which are favoured by exploitation (i.e. gaps) over interference competition for which they are poorly adapted, the Aleppo pine will colonize open environments, whether disturbed by fire or abandoned by agriculture. In this sense the Aleppo pine is only indirectly favoured by fires.

The concept of being indirectly favoured by fire should be carefully used since we have shown that regeneration is not directly linked to fire. In fact, regeneration is far from being spontaneous after fire but depends on the presence or absence of seed-bearers near the burnt site, on the vegetation structure and on the climatic conditions of the following years.

Pine colonization strategy (with or without fire) in the initial successional stages leads to a high structural and architectural heterogeneity both in space and time.

These forests are true "unbalanced" systems developing colonization mechanisms in disturbed or abandoned areas where the edaphic resources can either be low or high, which favours the expression of many different types of spatial heterogeneity.

CHARACTERISTICS OF POST-FIRE AND POST FARMING COLONIZATION SYSTEMS

- low species diversity
- heterogeneous (diameter, height) monospecific tree layer ("unbalanced" pine forest system)
- high quantitative distribution of a few species
- high spatial heterogeneity, but low site heterogeneity
- predominance of "R" strategy species with high dissemination and germination abilities
- very low sylvatic index
- shorter or longer successional time-frame depending on the species in successional stages
- predominance of competition by exploitation.

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Ecological factors restricting the regeneration of Pinus brutia in Turkey.

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SUMMARY - Pinus brutia forest ecosystem forms a special part in Mediterranean forest ecosystems. This species covers about 3,096,064 hectares in pure and mixed stands in its natural distribution in Turkey. It is a contented (less-demanding) tree species, and has no special soil requirements.

Pinus brutia is the most important fire-adapted tree species of Turkey. The fire adaptability of this species arises from several characteristics. Finally, fire becomes one of the most decisive constituents of Pinus brutia forest ecosystems in the Mediterranean region.

In order to obtain proper and adequate Pinus brutia natural regeneration, the seed production capacity of the stand, the amount of seed produced, the dispersal period of the seed, and the germination properties of the seed must be well known. Following germination, the growth characteristics of the seedlings as well as their ecological demands are also important for the establishment of natural regeneration.

KEYWORDS : Mediterranean region, Pinus brutia, forest fires, natural regeneration.

I - INTRODUCTION

Turkey is located in the northern hemisphere between 36-42° latitude North and 26-45° longitude East. The country is surrounded by the Black Sea in the north, the Aegean Sea in the west and the Mediterranean Sea in the south. On the east, it is separated from Russia by the Trans-Caucasian mountains. The total area of the country is 780,000 km^2. Forest covers approximately 20,2 million hectares or 26 percent of the total land area.

The distribution of this forest land between the various forest categories is as follows :

Condition of the forest	High forest (ha)	Coppice (ha)	Total (ha)	% of forest area	% of total area
Normal-Productive	6,176,899	2,679,558	8,856,457	44	11.4
Degraded-Unproductive	4,757,708	6,585,131	11,342,839	56	14.5
TOTAL	10,934,607	9,264,689	20,199,296	100	25.9

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There are about 19 coniferous and 74 non-coniferous species in Turkey's forests, but only few of them form stands of economic importance. 54.4% of Turkey's forests are softwood and 45.6% are hard-wood species, in the following proportions :

Softwood Species	Percentages	Hardwood Species	Percentages
Pine (P.sylvestris, P. nigra, P.brutia)	38.5	Oak (Q.pedunculata, Q.sessiliflora, Q. cerris, Q. pubescens, Q. ilex, Q. infectoria, Q. coccifera, Q. aegilops)	25.9
Fir (A. nordmanniana, A. bornmülleriana, A. equi-trojani, A. cilicica)	6.8	Beech (F. orientalis)	8.5
Spruce (P. orientalis)	2.0	Hornbeam (C. betulus)	2.7
Cedar (C. libani)	3.5	Chestnut (C. vesca)	1.4
Juniper (various)	3.5	Alder (A. glutinosa)	0.9
Other conifers	0.1	Lime + Ash	0.9
TOTAL	54.4	Poplar (P.X. euramericana "I-214", P. nigra	2.5
		Other Hardwoods	2.8
		TOTAL	45.6

All the natural forests belong to the State, except for a small proportion (less than 1%) which is privately owned. The major part of Turkey's forests is badly deteriorated through devastation, bad management, forest fires, illegal cuttings, and the grazing of animals. Good, productive, high forests represent less than half of the total forest area, and the rest consists of degraded forests with little economic value, at least for the time being.

The climate and topography of Turkey have an adverse influence on the spread and distribution of forests; thus the strictest protection of these forests is necessary. On the whole, the climatic zone in which the country lies is not altogether favourable for forest growth, particularly with regard to humidity and rainfall that are so important to the existence and survival of forest trees.

Apart from these adverse conditions, the mountain ranges generally run parrallel to the coast in the north and south of the country, while the hinterland - the high Anatolian plateau - is shut off from the sea. The fact that the mountains rise abruptly from the sea and reach high altitudes over relatively short distances sometimes causes widely differing climates, as far as temperature and rainfall are concerned. In the west, however, mountain ranges run at right angles (perpendicular) to the Aegean coast, and there are many valleys formed by depressions which stretch eastwards about 160 km inland from the coast. Therefore the influence of the Aegean sea, with the moisture-bearing winds, can penetrate much farther inland than in places where the mountains form a barrier.

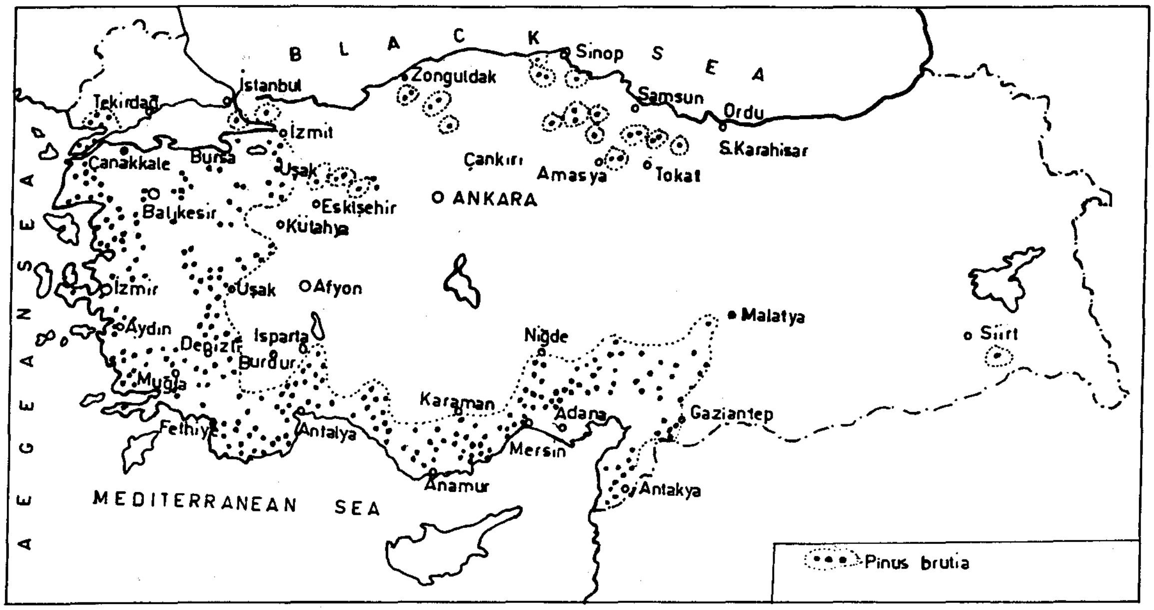
In general, Turkey is a country where wide dissimilarities in climate are to be found side by side, often without any transitional areas. For example, the rainfall in Turkey follows a pattern. The coastal strip is Turkey's most humid area, because the mountain ranges bring about the condensation of humid air masses. The further the distance from the coast the less rain there is, and the more arid the land. This fundamental trend, together with the variations in the seasonal distribution of the rain, causes important differences in vegetation.

All of these adverse climatic and topographic factors, along with human destruction, have been at work over the centuries to destroy the forests, both by reducing their extent and degrading their structure. These factors have contributed much towards transforming forest lands into the present steppe.

II - DISTRIBUTION OF PINUS BRUTIA FORESTS IN TURKEY

The Pinus brutia forest ecosystem constitutes a special part of the Mediterranean forest ecosystems. One of its main characteristics is the floristic and phytosociological heterogeneity arising from the biogeographic, climatic, historic, physiologic, bioclimatic, geomorphologic, and pedologic differences. The other is the sensitivity of this system to the influences on its ecological balance because of the lack of tolerance of this ecosystem (QUEZEL 1977). In other words, reaccomplishment of the ecological balance in this ecosystem becomes more difficult once the ecological balance is damaged by fires, grazing animals, illegal cuttings, early frosts, erosion, drought, or bad management. Therefore, it is very important to take into consideration the restricting ecological factors before making any adjustment or modification to this ecosystem.

Pinus brutia is well adapted to the Mediterranean climate, and covering about 3,096,064 hectares in pure and mixed stands in its natural distribution, it is one of the largest representatives of the pine species in Turkey. 47 percent (1,469,209 ha) of its distribution is found in the Mediterranean region, 40 percent (1,211,743 ha) in the Aegean region, 10 percent in the Marmara region, with the remaining 3 percent in the form of small stands on the inland slopes of Black Sea mountain ranges which also in some parts have a Mediterranean-type climate locally (Map 1, Table 1).



Pinus brutia forests are generally located on the sea-facing slopes of the Taurus mountains in the Mediterranean region. Mountain ranges reaching high altitudes in short distances prevent the distribution of this species further inland. Although the mean annual rainfall is adequate, for 3-5 months in the summer this region receives practically no rain at all. A summer drought generally characterizes the region where rainfall is received in heavy rainstorms in spring and winter.

In this region, Pinus brutia forms stands on the Taurus mountains, starting from sea level and rising up to 1,300m altitude (KANTARCI 1982). The wind generally blows in a west-southwesterly direction. The wind direction and the steepness of the slopes are the 2 principal determinants of forest fire behaviour in this region.

Table 1. Distribution of Pinus brutia according to Regional Forest Offices of Turkey (Turkish Forest Inventory 1980).

Regional Forest Office	Productive (ha)	Degraded (ha)	Total (ha)	%
Adana	165161	128537	293698	9
Adapazari	5121	2890	8011	-
Amasya	33129	33413	66542	2
Ankara	4611	6568	11179	-
Antalya	348433	130104	478437	16
Balikesir	48538	41662	90200	3
Bolu	8549	3603	12152	-
Bursa	48473	44962	93435	3
Canakkale	96521	65448	161969	5
Denizli	120090	82553	202643	7
Elazig	690	1322	2012	-
Eskisehir	25645	19273	44918	1
Kahraman Maras	87595	104926	192521	6
Kastamonu	747	2862	3609	-
Konya (Mersin)	23964	20906	44870	1
Istanbul	337	30	367	-
Izmir	180024	167624	347648	11
Isparta	51116	19551	70667	2
Mersin	216395	145016	361411	13
Mugla	305173	266070	571243	19
Sinop (Kastamonu)	11105	23130	34253	1
Zonguldak	2751	1546	4297	-
TOTAL	1,784,068	1,311,996	3,096,064	

The Aegean region, in contrast to the Mediterranean region, has mountain ranges running perpendicular to the coast, and therefore more humid air masses penetrate into western Anatolia. This region has the second largest Pinus brutia forest distribution in the country. The Regional Forest Office in Muğla alone has about 571,243 hectares of Pinus brutia forest resources, making it the largest among Forest Regional Offices in Turkey (Turkish Forest Inventory 1980). Mountains are relatively less steep, and annual rainfall is hardly adequate for forest growing. However, the summer drought season is shorter in this region. Since the mountain ranges run perpendicular to the coast, Pinus brutia composes forests starting from the coast and penetrating up to 300 km inland (SAATCIOĞLU 1969). Pinus brutia can also form stands at altitudes up to 800-900m in this region.

The Marmara region has the most orderly rainfall regime of the three areas considered and its drought season in the summer is also the shortest. Therefore, this region can be accepted as the continuation of the Aegean region as far as the climatic conditions are concerned. The Marmara region has about 10 percent of the total distribution;

Pinus brutia forests are mainly found on the Kom Mountains facing the Gulf of Saroz on the Thrace. Forests in these areas can reach up to 600-700m altitude.

Since the Black Sea region makes only a very limited contribution to the distribution of Pinus brutia forests, it is not taken into consideration here.

Distribution of Pinus brutia forests according to Regional Forest Offices of Turkey is given in Table 1.

Some climatic data for these regions is given in Table 2.

Table 2. Climatic data of some areas of Pinus brutia distributed in Turkey

Region	Place	Altitude	Mean annual Temp. (°C)	Minimum Temp.(°C)	Mean annual Rain (mm)	Rainy Days	Mean Rain of Driest month	Dry Months	Rainy Months
Black Sea	Samsun	44	14.3	-9,8	719	-	29,5	6,7,8	1,2,3 - 10,11,12
Region	Bafra	20	-	-	686	109	25	7,8	1,2,3 - 11,12
Marmara	Bursa	100	14.4	-25,7	709	-	15,6	6,7,8	1,2,3 - 11,12
Region	Orhaneli	525	-	-	582	85	2,9	7,8,9,10	1,2,3 - 12
	M. Kemalpasa	45	-	-	660	102	8,3	7,8	
	Canakkale	3	14.8	-11,5	627	-	7,8	6,7,8,9	1,2,3 - 10,11,12
	Ayvacik	275	-	-	936	70	7,5	6,7,8,9	1,2,3 - 11,12
Aegean	Balikesir	147	14.5	-21,8	603	96	6,5	6,7,8,9	1,2 - 11,12
Region	Bigadic	260	-	-	687	94	0,5	6,7,8,9,10	1,2 - 12
	Gölcük	6	-	-	724	108	24	6,7,8	1,2,3 - 10,11,12
	Izmir	25	17.5	-8,4	704	80	2,4	6,7,8,9	1,2,3 - 11,12
	Usak	919	12.3	-24,0	535	98	3,5	6,7,8,9	1,2,3,4,5 - 11,12
	Milas	45	18.0	-9,6	751	58	0,7	6,7,8,9	1,2,3 - 11,12
	Yatagan	950	-	-	643	65	2,4	6,7,8,9,10	1,2 - 11,12
Mediterranean	Marmaris	3	18.5	-4,0	1187	77	0	6,7,8,9	1,2,3 - 11,12
Region	Mugla	646	15.0	-12,6	1202	98	7,2	6,7,8,9	1,2,3 - 11,12
	Fethiye	3	19.0	-5,8	989	78	1,0	4,5,6,7,8,9	1,2,3 - 11,12
	Acipayam	946	-	-	494	76	1,4	6,7,8,9	1,2,3 - 11,12
	Kas	25	-	-	811	73	0	4,5,6,7,8,9	1,2,3 - 12
	Elmali	1113	13.1	-16,0	514	78	4,1	6,7,8,9,10	1,2 - 12
	Antalya	42	18.7	-7,1	1031	74	2,0	6,7,8,9	1,2,3 - 11,12
	Isparta	1043	12.1	-17,8	614	103	11,1	7,8,9	1,2 - 12
	Bucak	850	-	-	758	80	16,0	7,8,9	1,2 - 12
	Sütcüler	1000	-	-	892	63	9,4	7,8,9	1,2,3 - 12
	Alanya	10	18.8	-3,1	1042	72	0,2	6,7,8,9	1,2,3 - 11,12
	Gündogmus	930	-	-	1065	75	5,4	6,7,8,9	1,2,3 - 11,12
	Anamur	3	19.7	-4,7	1015	75	0,4	4,5,6,7,8,9	1,2,3 - 12
	Silifke	30	19.1	-6,3	629	54	1,2	4,5,6,7,8,9	1,2 - 12
	Gülnar	925	-	-	830	61	0	6,7,8,9	1,2 - 12
	Mersin	5	18.6	-6,6	601	46	5,2	5,6,7,8,9	1,2 - 12
	Gülek	950	-	-	903	72	2,1	7,8,9,10	1,2,3 - 12
	Adana	20	18.7	-7,1	625	74	4,2	6,7,8,9	1,2 - 12
	Pos (Pozanti)	860	-	-	1031	80	3,5	6,7,8,9,10	1,2,3 - 12
	Arsuz (Ulucinar)	10	-	-	680	73	2,0	6,7,8,9	1,2,3 - 12
	Antakya	100	18.2	-14,6	1157	93	3,0	7,8,9	1,2,3,4 - 12
	Yatagan	450	-	-	1034	91	2,3	6,7,8,9	1,2,3,4 - 11,12
	K. Maras	700	16.7	-13,9	694	71	0,7	6,7,8,9	1,2,3 - 12
	Sucati	800	-	-	412	64	1,7	5,6,7,8,9	1,2,3,4 - 10,11,12

Source : ARBEZ, N. (1971). Répartition, Ecologie et Variabilité de Pinus brutia en Turquie. FAO Report 3555.

III - SOME ECOLOGICAL FEATURES OF PINUS BRUTIA FORESTS

Pinus brutia is an easily contented and undemanding tree species, and has no special soil requirements. It can grow on many soil types, and even on the rocks whose cracks are filled with soil. Calcareous bedrock having many cracks filled with loamy soil creates one of the best growing sites for *Pinus brutia*. According to QUEZEL (1977), this species grows best over the calcareous and conglomerate bedrock.

Humus and dead organic material lying over the soil have an ecological influence on the soil, such as preventing soil erosion, maintaining soil humidity, and reducing evaporation from the soil. In addition, this organic material also protects the mineral soil surface from stroking effects of raindrops. Finally, it reduces surface runoff, and results in slow infiltration of rainfall into the soil. These functions of dead organic material regulate (and increase) the soil water conservation.

ODABAŞI (1983) stated that a layer of organic matter which does not restrict seedling roots in reaching to mineral soil is necessary for the establishment of *Pinus brutia* seedlings. ÖZDEMIR (1977) observed that burnt areas covered with needles, small branches, and cones following the fire offer more suitable conditions for seedling growing.

Natural and man-made fires have played a decisive role in the evolution of Mediterranean landscapes and its floristic composition. WALTER (1968) recognized fire as one of the major ecological factors which shaped the Mediterranean landscapes and fashioned its present mosaic-like pattern of regeneration and degradation. QUEZEL (1977) pointed out that *Pinus brutia* forest in lower altitudes have been experiencing many fires and many other factors detrimental to the health and integrity of the forests. These factors finally turned forest lands into maquis, grazing lands, and open areas.

Some of the important maquis flora found in the Mediterranean region are : *Arbutus andrachne, Calycotome villosa, Celtis australis, Ceratonia siliqua, Cistus creticus, Cistus salviifolius, Cotinus coggygria, Daphne sericea, Erica verticillata, Fontanesia phillyreoides, Juniperus oxycedrus, Laurus nobilis, Myrtus communis, Olea europea var. oleaster, Phillyrea media, Pistacia lentiscus, Pistacia terebinthus, Rhamnus palaestinus, Quercus coccifera, Quercus aegilops, Quercus haas, Quercus libani, Quercus infectoria subsp. boissieri, Smilax aspera, Styrax officinalis*.

Among these, some species catch fire rather easily turning ground fire to crown fire in *Pinus brutia* forests. Therefore, the management of these species requires special attention (ATALAY 1983).

Pinus brutia forests in Turkey also have been under the influence of many forest fires so far. Forest and fire relationship has resulted in a natural selection in which some trees and shrub species are left out of the system while others have adapted themselves to the ongoing conditions. This means that fire has become one of the decisive constituents of the Mediterranean forest ecosystems of Turkey.

Pinus brutia is the most important fire-adapted forest tree species of Turkey. Fire adaptibility of this species arises from its following characteristics :

1. *Pinus brutia* has a very thick bark serving as an effective insulator, which protects the cambium from burning and the resultant tree death.

2. *Pinus brutia* produces and holds adequate amounts of cones and seeds every year during its life-time, beginning at very young ages (4-7 years) and continuing to mature and overmature stages (ŞEFIK 1965, SAATÇIOĞLU 1967, ÖZDEMIR 1977).

3. *Pinus brutia* also holds some of its mature cones unopened, thus preventing seed dispersion for many years (something like 8-9 years) (ŞEFIK 1965; SAATÇIOĞLU 1967). These cones are called "serotinous cones"; the bracts of these cones are held by a high-temperature resin which melts during the passage of fire, thus enabling the seeds to fall (LOTAN 1975). These cones in a sense serve as an additional seed source, and as the life-insurance of the regeneration and of the tree.

4. *Pinus brutia* seeds falling to the soil can remain one or more years in the soil without losing germination capacity. This means that there are seeds in the soil for germination

every year. Whenever the soil conditions become favourable for germination, these seeds germinate quite easily.

5. Finally, Pinus brutia seeds reach the maturity stage in March, before the fire-season starts (ŞEFIK 1965, ALPACAR 1981). On the other hand, forest fires generally occur during hot and dry summer months - May through October in Turkey (BAŞ 1965). It is also during this period that approximately 92.5 percent of the seed dispersal takes place (OZDEMIR 1977). In other words, fire burnt areas will have a good chance of receiving mature seeds for germination.

IV - NATURAL REGENERATION OF PINUS BRUTIA

The first requirement for natural regeneration is the presence of seeds. In order to obtain adequate regeneration, the seed production capacity of the stand, the amount of seed produced, the dispersal period of the seed, and the germination properties of the seed must be well known (ODABAŞI 1983). Following germination, the growth characteristics of the seedlings as well as their ecological demands are also important for the establishment of the seedlings.

4.1. - Seed Dispersal Period of Pinus Brutia Stands

Pinus brutia stands generally produce an abundant seed crop in every 2-3 year period. In lower elevations, however, Pinus brutia stands produce a plentiful amount of seed crop every year. Even though seed fall seems to occur every month, the most abundant seed fall in fact takes place in July, August, September and October. Depending upon the weather conditions and elevations, this period can begin or end a little bit earlier or later. The seed fall of Pinus brutia at higher elevations (700-800m altitude) starts earlier (in July-September), than at lower elevations (250-300m altitude in August - October) (ODABAŞI 1983)

In general, the most abundant seed fall is observed in August, September and October. The amount of seed produced during this period is greater than half of the total seed fall. Since most of these fallen seeds retain their germination energy for a few years, and germinate when the conditions become suitable for germination, this creates another important seed source for natural regeneration in addition to the annual seed fall.

According to OZDEMIR (1977), the seed fall of Pinus brutia at lower elevations shows a linear increase in the first few years following the felling. At middle and high elevations, the abundant seed crop occurs every third year. Seed fall in Pinus brutia stands starts in early June and ends in January. The most abundant seed fall for these elevations takes place in August, September, and October (on average 71.9%). July and November are medium seed fall months (on an avergae 21.2%). The poorest seed fall months are June and December (on average 6.9%).

4.2. - Germination Time of Pinus Brutia Seeds

Germination time in natural regeneration is a significant value in the observation of regeneration and in the limitation of works in the regeneration area, as well as for protection measures.

Germination generally starts in November in lower elevations (250-300m), in February in middle elevations (350-650m), and in April in higher elevations (700-800m). Germination lasts until the end of June (ODABAŞI 1983). OZDEMIR (1977) found that germination in the Antalya region starts in the third week of January at the lower elevations and on the 18th of March at the higher elevations. The maximum germination dates are 15th of March in the lower elevations, 10th of April in the middle elevations, and 1st of May in higher elevations. The total germination period at lower elevations is 8.5 months, of which 5.5 months are in the spring and 3 months in the fall. At higher elevations it lasts a total of 8 months, of which 5 months are in the spring and 3 months in the fall. The germination on the middle elevations shows a continuity and lasts 8 months.

4.3. - Light, Temperature, and Moisture Requirements of Pinus Brutia Seedlings

Pinus brutia is a typical light-tolerant (light-demanding) tree species and needs a high light intensity at the beginning of its establishment as well as to enable it to grow sufficiently in its early seedling stage. OZDEMIR (1977) stated that the light intensity at the seed cutting stage must be around 65-70%. PAMAY (1965) found that if the light intensity is lower that 55-60% at the early seedling stage, seedlings either disappear or degenerate in 1-2 years. By losing their stem and crown characteristics seedlings develop like a shrub in a few years. Depending upon the site quality, seedlings need to be released from the sheltering effect of the stand in 1-3 years following the establishment.

Parallel to their high light requirement, Pinus brutia seedlings also require high temperatures to grow adequately. This species is very sensitive to frost, resistant to drought, and very tolerant in terms of moisture requirements.

Pinus brutia seedlings by developing long top roots can rapidly penetrate into the deeper soil horizons and thus make use of the soil moisture there. This enables the seedlings to survive under drought conditions. For Pinus brutia seedlings, top root growth is sometimes found to be 8-10 times greater than the stem growth and up to 65cm top root growth can be measured on 5-6 months old seedlings developed from the seeds (SAATÇIOĞLU 1969). ÖZDEMIR (1977) pointed out that particularly at lower and higher elevations, in spite of the soil moisture being rather insufficient throughout the year, the survival of the seedlings either on seeding or clearcut areas can be explained by the early germination and rooting. As a result of this, the roots can attain deeper parts of the soil.

4.4. - Survival and Death Rate of Pinus Brutia Seedlings in Natural Regeneration Areas

The number of seedlings and their survival and death rate over time in natural regeneration areas are affected by many factors. These factors can be summarized as "the amount of annual seed fall of the stand", "seed fall in previous years", "seed fall afterwards", "site quality", "extreme weather conditions", and "regeneration methods" (ODABAŞI 1983). Although most of these factors are interrelated, no specific research has yet been conducted to investigate the effects of any one of these specific factors. However, ÖZDEMIR (1977) found strong indications in his study that the number of seedlings in the first year is very much controlled by the seed fall quantity in the same season. There is also germination from the seeds fallen in previous years as well as from the seed dispersal afterwards. Elevation of the regeneration area and its climate also have strong influences on seedling survival. In lower elevations, both the germination rate and the following death rate of seedlings is higher than in the higher elevations. For successful regeneration, 2-3 biologically self-sufficient seedlings in one square metre is accepted as an adequate number (ODABAŞI 1983). ÖZDEMIR (1977) stated that the higher elevations are the best for formation and survival of the natural regeneration of Pinus brutia.

4.5. - Soil Preparation and Prescribed Burning

4.5.1. Soil Preparations

Soil preparation techniques have been one of the main constituents of natural regeneration for years. As an extremely variable characteristic of site quality, soil properties necessitate special attention in natural regeneration studies. Soil preparation has therefore been considered as an inseparable part of natural regeneration studies. SAATÇIOĞLU (1967) observed that Pinus brutia forest soils are generally covered with an undecomposed needle layer of various thicknesses. He also stated that breaking out of this layer, either the total area or strips of it, thereby opening up the mineral soil surface, constitutes the first and most important step in the natural regeneration of Pinus brutia in the Shelterwood system. On the other hand, ÖZDEMIR (1977) has stated that soil covered with a thin layer of cone-bearing branches after clear-cutting creates the most suitable conditions for natural regeneration of Pinus brutia.

In natural regeneration of Pinus brutia, especially in lower elevations, there is no need for soil preparation unless the thickness of the dead organic matter lying over the mineral soil exceeds 3cm, and/or a thick humus formation is found. On the other hand, soil preparation generally increases the success of regeneration in areas when there are heavy accumulations of dead organic material or where a thick humus layer has formed over the mineral soil, as well as in areas covered with heavy grasses, shrubs and other understory vegetation.

YAKA (1985) who is a forester in the Muğla Regional Forest Office, observed that soil covered with 1-3cm thick needle layer creates the best seedbed conditions. Where this layer is broken and the collected material is piled up, it also causes the seeds to be collected in the piles. However, he recommended soil preparation for areas covered with heavy grasses or having thick organic matter and humus accumulations over the mineral soil. Even in such areas, following soil preparation, cone-bearing branches must be laid out, and suitable conditions reestablished. Living material in such areas can be cut down to prevent the effects of shade on regeneration.

ERON (1981) suggested the laying out of cone-bearing branches, no larger than 2.5cm in diameter, on fire-burned productive areas following the clearcutting of fire-affected trees. After removal of the logs and the firewood, small cone-bearing branches from the leftover material which cannot be used for other purposes must be laid out homogeneously on the fire-burned areas. He also pointed out the following advantages of this practice :

1 - Heat generated from the fire can readily open up most of the mature cones as well as "serotinous cones" on the branches, and thereby seeds the area where the cone-bearing branches are laid out.

2 - Cone-bearing branches serve as an organic matter layer that is consumed by the fire, and prevent the erosive effect of raindrops on the surface soil, protects soil porosity, and therefore reduces surface runoff and soil erosion, especially on steep slopes.

3 - Small cone-bearing branches also keep the seeds from slipping down the slopes in the wake of soil erosion.

4 - Cone-bearing branches create better temperature-moisture conditions over the mineral soil and improve the seed germination and seedling growth at different exposures.

5 - Cone-bearing branches, by decomposing rather slowly, become the green manure for fire-burned areas in the long term.

6 - Cone-bearing branches also hide and protect the fallen seeds from the birds and other seed-eating animals like squirrels, rabbits, rats, etc.

4.5.2. Prescribed burning

Prescribed burning is regarded as a useful tool, if used carefully, to eliminate undesired vegetation, but site characteristics must be taken into consideration before this method of soil preparation is used. According to OZDEMIR (1977), the cheapest soil cultivation method, which includes the clearing of the litter and the living cover, is burning.

However, as regards soil cultivation, raking is the best procedure for natural regeneration. Ploughing can also be successfully carried out, particularly at lower and middle elevations, when there is no possibility for soil cultivation. OZDEMIR also stated that although rooting out gives good results, it is an expensive procedure. Areas where rooting out of the living cover is carried out give better results of regeneration by planting.

Prescribed burning for seedbed preparation and for the regeneration of Pinus brutia has been considered as a useful tool since the beginning of natural regeneration studies in Turkey (ORPAK 1968, OZDEMIR 1968). Both the abundance of Pinus brutia seedlings in burnt areas and the mosaic-like, even-aged stands of Pinus brutia forests confirm the fire-adaptability characteristics of these forests. Even today, there are disagreements among scientists about the beneficial and detrimental effects of fire on Pinus brutia regeneration. For example, SAATÇIOĞLU (1971) claims that fire burns out undesired dead and living material over the soil, and releases organically bound mineral nutrients to the soil as readily soluble salts; mineral nutrients thus become available for the seedlings. He also mentions that the total nitrogen lost during burning is not a large amount. In contrast, ZECH and ÇEPEL (1972) stated that water deficiency as well as the deficiency of nitrogen and phosphorus in the soil, along with the high pH values of surface soil, are the principal limiting factors of Pinus brutia forest development. Since humus has a high water-holding capacity and is a source of nitrogen, burning out of this material seems to be detrimental to forest health.

In spite of this, some of the successful examples of natural regeneration studies in burnt areas, being a cheap method for natural regeneration, encourage the foresters to use this method, especially where the dead and living organic material create problems for natural regeneration.

V - NATURAL REGENERATION METHODS OF PINUS BRUTIA

Selecting the most suitable method adapted to the biological properties of Pinus brutia is the first step towards natural regeneration. Considering the site characteristics before the application of the selected method is also important. Different site characteristics can result in the use of different regeneration methods, even for the same tree species.

There are two kinds of natural regeneration methods used for Pinus brutia in Turkey :

1 - Shelterwood,

2 - Strip clearcutting method.

5.1 - Shelterwood method

There are three phases of this method :

- Preparation cutting
- Seed cutting
- Clear cutting (or, last cutting)

These phases follow in sequence, and are used in accordance with the biological properties of Pinus brutia.

5.1.1 Preparation Cutting

According to ODABAŞI (1983), there is no need for preparation cutting in old (overmature) Pinus brutia stands. However, in mature stands which have normal or heavy closures, 2-3 preparation cuttings at 2-3 year intervals can be carried out. OZDEMIR's study (1977) revealed that preparation cuttings are necessary for Pinus brutia and that they have to be made 3 years before the seeding stage. This is because there is a great increase in the seed production in the fourth year following the preparation cuttings.

5.1.2 Seed Cutting

As a general rule, cutting must be carried out in abundant seed fall years. This also applies to Pinus brutia. However, since Pinus brutia forests produce (and hold) seeds every year, even if not in great amounts, seed cutting can be carried out outside the abundant seed fall years, accompanied by the laying out of cone-bearing branches over the regeneration area.

To determine the seed cutting period, seed dispersal and germination time must be taken into account. SAATÇIOĞLU (1971) suggested that the seed cutting and soil preparation be completed before the end of the fall, assuming that the main seed dispersal takes place in the fall. Many studies carried out in this field have shown that the main seed dispersal period starts in July, and therefore seed cutting and soil preparation need to be completed before the end of that month. Since the laying out of cone bearing branches on the regeneration area increases the success of regeneration, this practice is not to be forgotten following the seed cutting as these materials are obtained from the very trees that are felled during the seed cutting operation. Therefore, the cutting operation must be started after March. OZDEMIR (1977) found that the best time for seed cutting followed by the seedbed preparation is March-April or May.

5.1.3 Last (Clear) Cutting

Once Pinus brutia seedlings are established on the regeneration area, it is generally agreed that last cuttings must be carried out not later than 2 years afterwards to provide high light intensity for the seedlings. OZDEMIR (1977) suggested that the removal of the remaining trees with clear cuttings must be carried out within 2 years of germination. In other words, seedlings must not be left under the shading effect of the remaining trees for more than 2 years (or 2 growing seasons) because otherwise degeneration may set in among seedlings.

5.2 Strip Clearcutting

This method is also successful in developing pure and even-aged stands, and is being used successfully in regeneration studies of Pinus brutia. However, the inadequacy of the road network system for removal of the logs and fuelwood, the possibility of the soil erosion on steep slopes, and the danger of coverage of undesirable living vegetation on regeneration areas constitute the shortcomings of this method.

Cutting times (preparation, seeding, and last cuttings) and the soil preparation techniques for this method are the same as for the Shelter-wood method. Laying out of cone-bearing branches on the strips is an important element for increasing the success of regeneration in this method as well.

Although PAMAY (1966) and SAATÇIOĞLU (1971) suggested 20m as the width for the clearcutting strips at the beginning of such an operation, it was later found that wider strips are better for natural regeneration. SAATÇIOĞLU (1979) stated that 30-40m wide strips for Pinus brutia regeneration are more acceptable. OZDEMIR (1977) also found in his study that 30m wide strips and 40m wide strips covered with cone-bearing branches showed better regeneration properties than 25m wide strips.

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Ecological fire influences on Quercus suber forest ecosystems.

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SUMMARY - The effects of fire on the vegetation is one of the main factors in the present structure of Portugal's landscape.

Although caused exclusively through anthropogenic activities, the effects of fire have influenced the territory for so long that it can be considered as part of the natural evolutionary process.

In the north, the highly fire-resistent deciduous oak forests were almost totally destroyed by man; they were consequently substituted by pine forests which in the last decades, due to increment of the understorey, have become a high-risk fire system.

The effects of fire in the south of Portugal have produced a tree-steppe system, locally known as "montado", a complex, multiple land-use system offering such activities as cereal production, game species habitats, and even recreation and tourism.

Apart from substituting cork oak forest, the cork oak "montado" has also extended its distribution areas to formerly semi-evergreen forests. This may be attributed to the cork oak's strong fire resistence.

The cork oak "montado" occupies approximately one quarter of Portugal's forest area where it presently constitutes one of the most productive multiple land use systems.

KEYWORDS : Portugal, Quercus suber, fire-ecosystems, fire-regeneration

PRELIMINARY CONSIDERATIONS

The Quercus suber is the principal Fagaceae in Portugal, not only because it occurs over a wide area (1) but also because of the products obtained from it when exploited in "montados" (2).

The Cork oak "montado" constitutes a complex system of agriculture, forestry and grazing, with the technical possibilities of becoming a true multiple land use forest area. Apart from timber and cork materials, the cork oak is also exploited for its acorns which are used for fattening livestock (pigs, goat and sheep). At the same time cereal crops or pasture may be grown, either simultaneously or alternately. The "montado" also provides habitats for game species (associated to this tree steppe biotype), while recreation and touristic activities (at present largely unexploited in Portugal) could also be developed.

Quercus suber is presently spread over practically the whole of the Portuguese territory (Fig. 1) in spite of the country's wide climatic variability. For this reason, Quercus suber ecosystems require a more extensive framework than the other natural ecosystems as far as fire problems are concerned.

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(1) 667,000 ha - representing 7.5% of the area of Portugal and 22.5% of its forest area.

(2) In the last ten years, cork production alone reached an average annual total of 138,700 tonnes.

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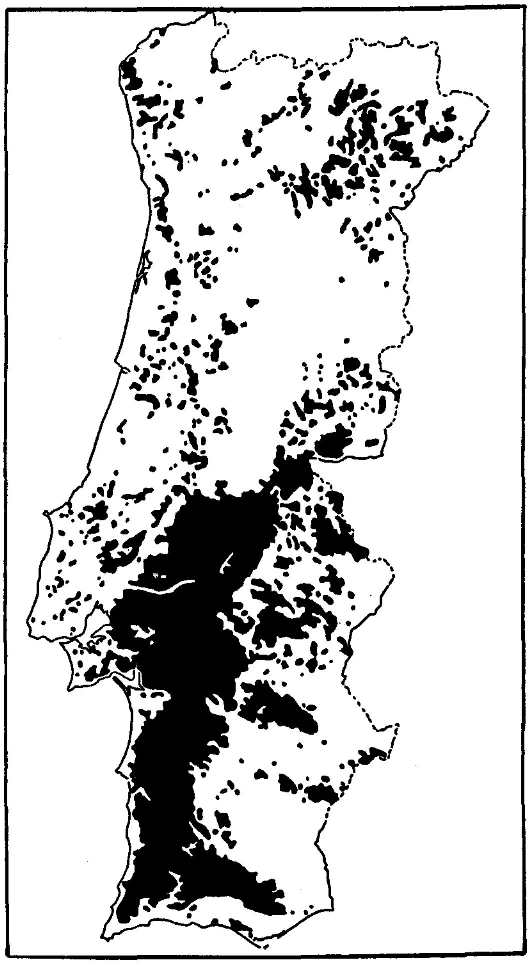

FIG. 1 - QUERCUS SUBER (CORK OAK) - ACTUAL OCCURRENCE IN PORTUGAL

NOTES ON PORTUGAL'S CLIMAX VEGETATION

In Portugal, the climax vegetation communities presents accentuated variations (Fig.2), considering the small area of the territory (89,000 km).

In this paper, we do not consider the sites whose edaphic characteristics have a minor influence on climatic factors. Rather we will concentrate our comments on the climax vegetation distribution (1), that is on structures that depend basically (at least in their more developed stages) on the climatic conditions prevailing in the ecotope.

For the major part of the Portuguese territory the climax communities present are almost exclusively Quercus species which are dominant and which show different ecological preferences as well as distinct physiological behaviours.

An easily observed characteristic is the phenological aspect of the variations in leaf persistence, as follows :

Quercus robur - a deciduous tree
Quercus pyrenaica - a deciduous tree though it may be considered sometimes as semi-evergreen (VILLAR 1958).
Quercus faginea (2) - a semi-evergreen tree

(1) to be used only as a vegetation model from a "climax pattern" of communities

(2) the taxon Quercus faginea is considered here sensu lato, in which is included Quercus canariensis

Quercus suber - an evergreen taxon though frequently observed as semi-evergreen (1)

Quercus rotundifolia and Quercus coccifera - evergreen trees.

From a still incomplete analysis, due to differences arising from non-climatic factors, it may be concluded that the following vegetation belt sequence : Quercus robur - Quercus pyrenaica - Quercus faginea. Quercus suber - Quercus rotundifolia - Quercus coccifera, accompanies the climatic variation occurring between the N.W. and S.E. of Portugal (2), particularly with respect to the value of hydric resources.

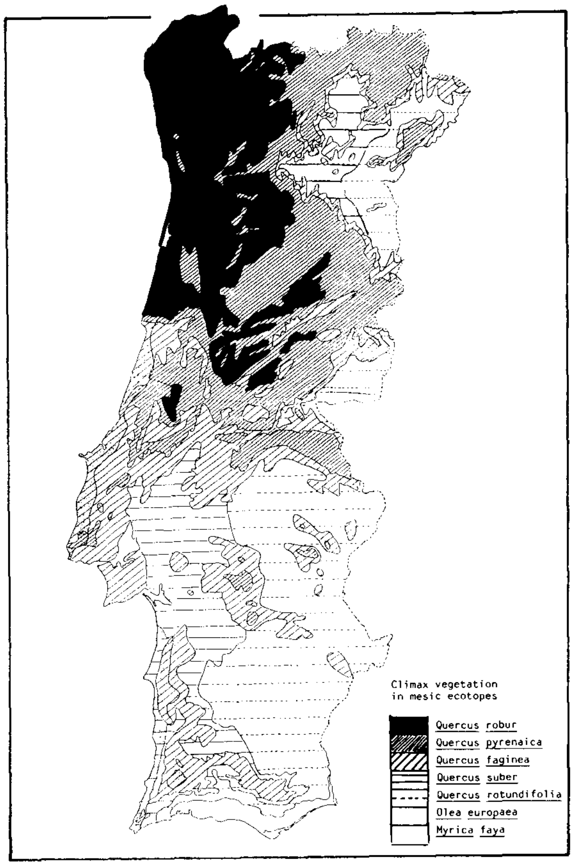

FIG. 2 - VEGETATION BELTS ZONATION IN PORTUGAL (MODEL)

However, the sequence occurring along the littoral bears marked differences to that occurring in the interior zone.

While the interior zone sequence is dominated by the Quercus pyrenaica and Quercus rotundifolia vegetation belts to the detriment of other Quercus species, the belts of which occur less frequently and sometimes not at all, the littoral sequence is dominated by the vegetation belts of Quercus robur, Quercus faginea, Quercus suber, and locally of Quercus coccifera and probably also of Myrica faya.

So the continental Portuguese sequence can be compared with the typical ones of the eastern Iberic Peninsula and South of France, particularly in relation to the floristic composition. However, in the case of the dominant taxa, Quercus pubescens is replaced in Portugal by Quercus pyrenaica and Quercus ilex by Quercus rotundifolia. On the other hand, the Portuguese littoral sequence is relatively similar to that occurring along the North African littoral, between Tunisia and Algeria.

(1) although the leaves can stay 2 or 3 years, the tree can lose all the leaves in very dry periods. Several trees can lose the leaves annually but only after the appearance of the new ones (PALHINHA 1939).

(2) also found in the zone running from west to east along the Douro valley.

1 Fraxinus angustifolia ssp a.
2 Laurus nobilis
3 Prunus lusitanica ssp l.
4 Persea Indica
5 Myrica faya
6 Phillyrea latifolia
7 Quercus canariensis

elements of degraded stages :
(a) Rhododendron ponticum
ssp baeticum
(b) Euphorbia obtusfolia

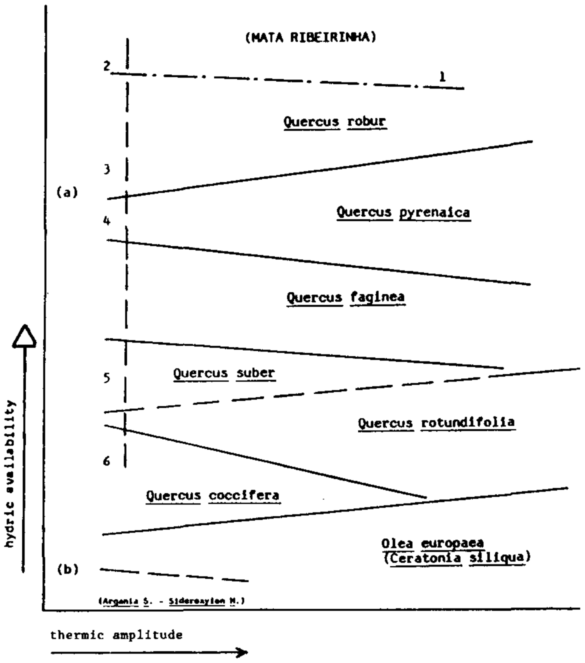

FIG 3 - MODEL OF THE VEGETATION BELTS IN PORTUGAL

The main reason for this variation is still not totally confirmed. However, several studies made so far suggest that thermic amplitude could be a significant factor in the competitive process between the mentioned taxa without modifying the basic vegetation belt sequence as shown in the variation pattern of Figure 3.

Quercus coccifera which, in most of its actual area of occurrence, is considered as a pyrophyte in climax structures behaves as an understorey element in semi-evergreen oak woodlands, although in several xeric Atlantic sites (as mentioned) it could form a climax together with Phillyrea latifolia.

Figure 2 shows the zones of the different climax vegetation in mesic sites in Portugal. In xeric or hydric sites the potential vegetation may not correspond to the one indicated.

An example of this can be given in the case of Quercus suber which occurs over practically the whole of the Portuguese territory, except for limestone regions and the highest mountain areas. In areas of potential belts of Quercus rotundifolia and Olea europaea it tends to behave as a riparian tree associated or marginally so, with the water courses. On the other hand, in potential belts of Quercus faginea, Quercus pyrenaica and even Quercus robur it behaves as a xerophyte or as an integrated secondary component of the climax structure.

INFLUENCE OF FIRE ON THE NATURAL VEGETATION

In general, the hardwood forests in Portugal associated with an environment of high water availability during the dry period, have a low fire risk. This is especially the case in pluristratified climax formations with high moisture levels under the vegetation cover. These formations constitute real barriers against the spread of fires which originate in high risk formations such as matorral associated or not with Pinus pinaster, a tree of great silvicultural value which in the last 700 years has gradually replaced the formerly deciduous and semi-evergreen species.

For centuries pine forests tolerated relatively well the occurrence of fire because of the traditional clearing of the understorey which was used for manure and cattle beds in past decades. However, rural zones have been abandoned and the traditional clearings are no longer practised. This has led to an excessive increment of dense matorral of Erica, Ulex, Genista and Cytisus species, transforming the Pine forests into high-risk systems (matorral).

In fact in the riparian deciduous woodlands of Alnus glutinosa, Ulmus minor, Salix sp., etc semi-riparian forests of Fraxinus angustifolia ssp. angustifolia, Celtis australis and Acer monspessulanum and oak forests of Quercus robur, the occurrence of fire is rare and when it does occur it is only marginal.

In the remaining oak fores, either deciduous or semi-evergreen, the effect of fire depends basically on what may be called the "strategy" of their dominant taxa.

The effect of fire on Quercus pyrenaica stimulates the sprouting of relatively vigorous shoots from underground stumps giving a tall bushy form to this Quercus which, associated with communities of Ericaceae and Leguminoseae species, presents a moderate to high fire risk resulting from the community of the strata.

As regards the semi-evergreen oak woodland dominated by Quercus faginea, little is known of its "strategy" in relation to burning. It is a well-known fact that Quercus faginea and its related taxa (1) disappear after the occurrence of fire (LE HOUEROU, 1977). It has been observed that in sandy soils (2) the after-effect of fire in woodlands, dominated by Quercus faginea, tends to lead to the appearance of a dwarf oak with a root-sprouting aspect which can attain a maximum height of 2 metres and can be classified taxonomically as Quercus lusitanica (3). This taxon may possibly be a result of a very particular "strategy" of the Quercus faginea in response to frequent fires.

The effects of fire on oak groves of this kind determine the colonization of other taxa with better strategy processes, such as Quercus suber on silicious soils, and Quercus rotundifolia on limestone soils where very frequent fires, associated with tree-cutting practices, particularly of cork oak trees could lead to the disappearance of these oaks and the appearance of "carrascal" type formations (4) with dominants of Quercus coccifera and Rhamnus alaternus (and Pistacia lentiscus, Rhamnus lycioides ssp. oleoides, etc.).

In woodlands which are almost or completely evergreen, fire implies a different process. In the case of Quercus suber dominated zones, the fires change the woodlands, transforming them into tree steppes known locally as "montado de sobro". This type of formation presents two well-defined strata : the canopy, formed by the cork oaks, and the low layer, formed by herbaceous or shrubby species, whose floristic structure depends on the frequency of fires (5). The cork oak has a strong resistance to burning when not stripped of its bark, and has a strong sprouting of shoots in the canopy.

(1) Such as Quercus canariensis - probably an ecotype of Quercus faginea found in the Quercus pyrenaica belt zone.

(2) Originated mainly from granite, schistose and sandstone.

(3) In the Flora Europaea (1964), it is referred to as Quercus fruticosa Brot.

(4) Dense, almost impenetrable thickets of 0.5 m to 1 m high, dominated by species with thick and thorny foliage and strong imbricated branches. Similar to Quercus coccifera garrigue in South of France.

(5) with a predominance of leguminoseae species in low-frequency fire zones and Cistaceae species in high-frequency fire zones.

In the case of Quercus rotundifolia dominated zones, the climax oak woodlands have indeed a low taxonomic diversity and the less intense burnings gradually transformed such woodlands into steppes with isolated Quercus rotundifolia trees ("montado de azinho"). More intense burnings eliminated the evergreen oak giving place to Cistaceae (Cistus ladanifer) or brooms (Lygos sphaerocarpa).

The "montados" constitute short-cycled fire ecosystems (structure is maintained despite annual or frequent burnings), whereas the formations of Cistus ladanifer heath ("esteval") occur within long-cycled fire ecosystems. Since they are not burned in the first few years, they pass into a low fire-risk phase (discontinuity of strata) which only increases with the appearance in the understorey of Ulex sp., Genista sp., Lavandula sp., etc. After fire, Cistus ladanifer regenerates quite easily from seeds.

NATURAL FIRE-HAZARD IN PORTUGAL

As regards the probability of the occurrence of fire in a determined zone, fire hazard may be considered in terms of ignition or propagation. The global hazard, defined as the probability for an area to burn is shown for the Portuguese territory in the gradients in Fig. 4 "Actual Fire Risk".

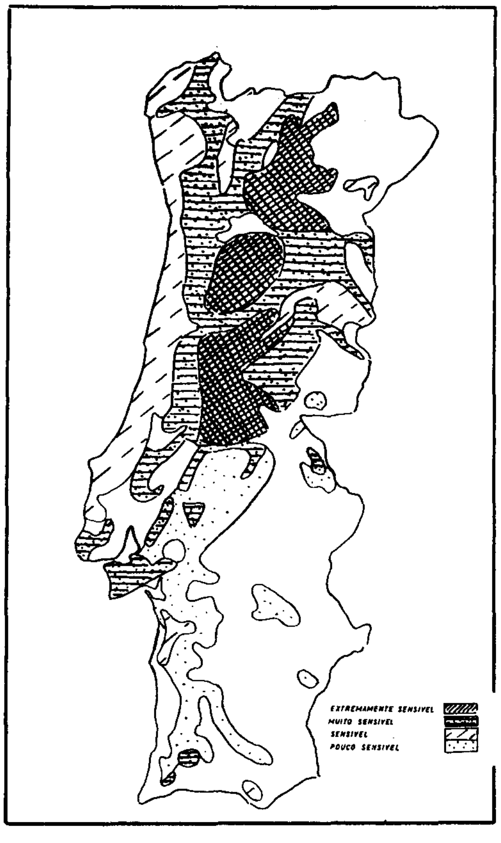

FIG 4 - ACTUAL FIRE RISK (Silva 1975)

With regard to potential hazard of fire (which covers only permanent factors) the zoning is clearly different, as can be seen from Fig. 5. This zoning was established through a qualitative integration of the determinant factors, namely the duration of the dry season (ALBUQUERQUE 1971), and maximum temperatures (DAVEAU 1980). Neither pluviosity factors nor atmospheric and soil moisture factors were considered, due to the fact that they exercise opposite influences on fire hazard : - low influence in the case of increasing humidity in the soil and understorey and high influence in the case of increasing vegetal fuel. In the cartography of Potential Fire hazard, the majority of high hazard zones coincide with areas in danger of desertification. Simultaneously, there is no link with the cartography of Actual Fire risk. The reason for this (as can be seen in Fig. 6) is the existence of fire-tolerant or fire-adapted forest structures in the south of Portugal (the "montados").

UNDERSTOREY REGENERATION OF *QUERCUS SUBER* FOREST

The *Quercus suber* is well protected against the high temperatures of burnings by its thick cortical tissue, which is why it is the only tree to remain after a fire.

All other tree species regenerate mainly by root or stump sprouting, so that after a fire occurrence there is a dominance of *Quercus suber* whenever this species is present in the oak forest.

This last situation is particularly important in *Quercus faginea* forests since after fire this species does not easily regenerate by sprouting new shoots. In its ecological area, *Quercus faginea* is thus replaced by the *Quercus lusitanica* after fire.

Apart from *Quercus lusitanica* which arises as the only probable indicator of cork oak forests and "montados" which have originally been semi-evergreen oak forests, there is a relatively homogeneous understorey although this can vary according to the different types of soils, water availability and the sequence in which vegetation is found.

In their more evolved stages cork oak forests may even contain deciduous species such as *Pyrus bourgaeana* and *Crataegus monogyna* ssp. *brevispina*. However, the majority of species occurring are evergreen, such as *Myrtus communis* (associated, as are the deciduous species previously mentioned, with the water courses) : *Myrica faya* (only in littoral cork oak forests associated with thermo-atlantic climatic characteristics) *Arbutus unedo*; *Quercus coccifera*, *Phillyrea latifolia*, *Rhamnus alaternus*, *Erica arborea*, *Viburnum tinus* and also xerophytes such as *Pistacia lentiscus*, *Phillyrea angustifolia*, *Olea europaea* and *Quercus rotundifolia*.

In these more evolved stages are frequently to be found scandent species such as *Smilax aspera*, *Hedera helix* ssp. *canariensis*, *Lonicera* sp., *Rubia peregrina*, etc; and also short woody species such as *Ruscus aculeatus*.

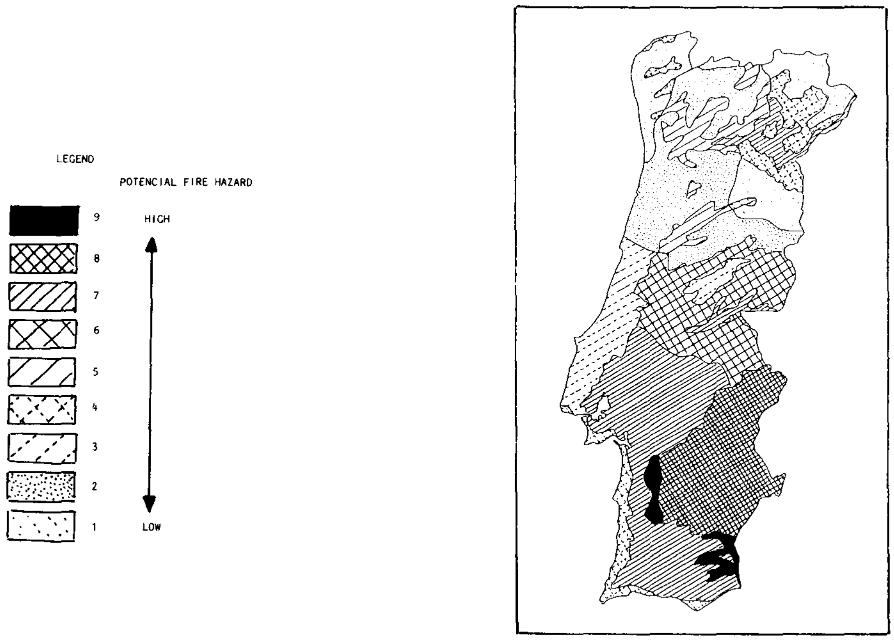

FIG. 5 - POTENCIAL FIRE HAZARD

The continued action of diverse anthropic-induced fires constitutes endangerment the majority of the species mentioned, which will give rise to very diversified *matorrals* with *Leguminoseae* and *Labiateae* dominants in the case of irregular fires, and *Cistaceae* dominants where there are very frequent fires. This latter group does not include *Cistus ladanifer* which, if it survives for 3 or 4 years without burning, develops in height and, due to the discontinuity of strata, acquires a strong resistance to fire. This characteristic is lost in phytosuccession due to diverse species such as *Ulex* sp., *Genista* sp., *Lavandula* sp. and *Thymus* sp. developing in the "esteval" understorey.

The fire-soil-water complex thus creates a notable diversity in the understorey of cork oak forests, so much so that *Quercus suber*, practically without competitors and often helped by man, continues to expand its area of natural occurrence - defined here as the area in which the species is capable of assuming a dominant role in the formation, directly competing with other species without the influence of fire or man.

With a very reduced number of studies to go by (NATIVIDADE 1950, GOES 1953 & CRUZ 1985), a concise picture of how the more characteristic species of cork oak forest are formed, may nevertheless be presented :

a) in deep but heavy soils the most characteristic element to be found is *Cistus monspeliensis* and to a certain extent *Cistus crispus*. Where there is an increase of water availability *Erica scoparia* ssp. *scoparia* occurs as the main dominant species.

b) in poorly developed soils from schist or "arenito" (sandstone with poor permeability due to its clay-cement) several characteristic species are found :

ba) in xeric sites : - *Cistus ladanifer, Erica australis* and *Pterospartum tridentatum*

bb) in mesic sites : - *Pterospartum tridentatum*, *Calluna vulgaris*, *Erica umbellata*, *Halimium ocymoides* and to a certain extent *Cistus crispus*

bc) in hygrophilous sites : - *Halimium lasianthum* spp. *lasianthum* and *Erica ciliaris*.

c) in sandy soils with impermeable substrata 0.5 to 1 m deep, sandstone soil with little clay, and sandy soils with a podzol horizon there is an abundance (though not necessarily characteristic) of : *Ulex parviflorus* ssp. *parviflorus* and *Cistus salvifolius*. As hydric availabilities increase there is a successive occurrence of *Genista triacanthos*, *Cistus psilosepalus* and *Ulex minor*.

d) in deep sandy soils, with strong xeric characteristics there is an abundance of the following characteristic species : - *Juniperus oxycedrus* ssp. *transtagana*, *Stauracanthus genistoides* ssp. *genistoides*, *Halimium commutatum* and *Halimium halimifolium*. As hydric availabilities increase there is a successive occurrence of *Erica erigena* and *Erica lusitanica*.

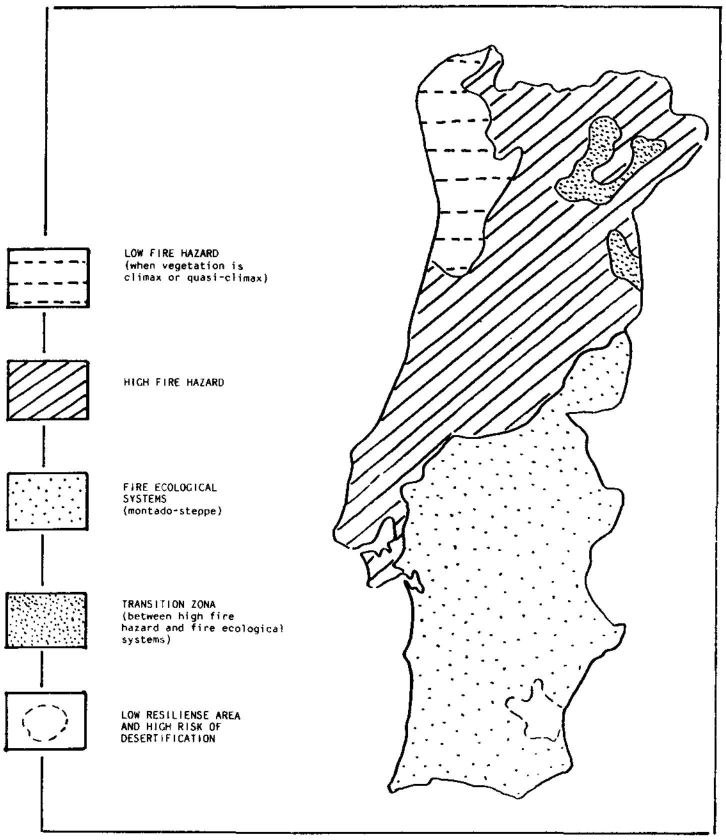

FIG. 6 - FIRE ECOSYSTEMS MACRO-ZONATION

Finally, as a result of cultural activities and of intensive grazing associated with very frequent fires, there is an almost complete elimination of woody components; the last ones to resist are usually the *Cistus salvifolius*, *Daphne gnidium* and, already with ruderal characteristics, *Dittrichia viscosa* (ssp. *viscosa* in the centre of Portugal and ssp. *revoluta* in the South). Hence an anthropogenic structure of the cork oak forest similar to a tree steppe formation appears, dominated by a single type of tree (*Quercus suber*), and with an understorey of grass and diverse herbaceous species.

It should further be mentioned that despite its resistance to fire, *Quercus suber* can be destroyed in certain conditions; it happens mainly along the littoral zone of South-West Portugal between Sines and Lagos, which is probably largely a potential cork oak climax area, with *Myrica faya* and *Arbutus unedo* as the main codominants where there are now only a few vestiges of *Quercus suber*.

Although there is no concrete evidence of it, it is supposed that this was due to the prevailing favourable environmental conditions (low thermic amplitude, elevated humidity in the dry season caused by fog-drip) which permitted the growth of abundant and continuous biomass which in conditions favourable to the occurrence of fire, would have been totally destroyed.

It can be concluded that these tree steppe formations dominated by *Quercus suber* constitute one of the best types of fire ecosystems in Portugal. Consequently, the need of further, more detailed, studies on this system becomes greater, especially if maximum effectiveness is to be obtained in land use planning (including conservation and increased natural patrimony) through multiple use of this type of forest.

One of the most interesting solutions is the expansion of cork oak "montado" to the centre of Portugal, which is presently occupied by extensive areas of high fire risk pine forests. This would imply not only an increase in land use diversity but also in fire prevention measures providing the "montados" were effectively managed at the field level, through grazing and controlled burning.

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ECOLOGIA MEDITERRANEA - Tome XIII (fascicule 4) - 1987

Effects of fire on structure, dynamics and regeneration of <u>Quercus pyrenaica</u> ecosystems.

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SUMMARY - Post-fire regeneration is studied in forty-three communities of Quercus pyrenaica in the province of Leon (Spain) burned by surface fires from a few weeks up to 60 years before the sampling, and is compared with four zones which have not been burned. A greater similarity in floristic composition is clearly to be seen in the different phases of one same community than between different communities with the same age of regeneration. The vegetation prior to the fire conditions, therefore, is to a great extent the one which will arise after it, the process constituting a compensatory autosuccession of the regression suffered.

Specific diversity increases in the first years, due mainly to an increase in species richness, showing later values more in accordance with each specific zone than to the time which has elapsed since the fire. All the communities studied are very heterogeneous, but the action of the fire can be seen as an additional generating factor of heterogeneity a few weeks after the burning.

From all this, it can be deduced that these communities have been adapted to fire for a very long time and that most of the species have developed selective mechanisms to regenerate themselves quite effectively after the fires.

KEYWORDS : post-fire regeneration, Quercus pyrenaica, autosuccession, diversity, heterogeneity.

INTRODUCTION

Due to the increment of the great number of fires detected in the last few years, above all in the Mediterranean zones, the theme of forest fires has acquired a considerable importance nowadays. Nevertheless, this phenomenon, although it has increased in the last few decades, does not constitute something exceptional, rather it is a normal ecological factor, which has influenced the dynamics of the ecosystems of these zones for a very long time. Even before the appearance of man, fires were produced by lightning. Fire frequency increased later on by the action of farmers and shepherds. This has conditioned the development of a vegetation adapted to resist fire, and which usually resprouts immediately after the burning. The shrubby formations proper to the Mediterranean climates which are adapted to more or less periodical fires, are numerous, e.g. the Californian chaparral (BISWELL 1974, PARSONS 1976, KEELEY & KEELEY 1981) and the maquia and garrigue in France (TRABAUD 1980, TRABAUD & LEPART 1980), Israel (NAVEH 1974) and Greece (PAPANASTASIS 1977, 1978).

The communities of <u>Quercus pyrenaica</u> discussed here cannot really be considered as bushes, but in most cases they cannot be considered as real woods either, because they are usually very much altered and the shrubby forms are predominant. In the zone studied, they are situated between the phytogeographic Mediterranean region and the Eurosiberian region (RIVAS <u>et al</u>. 1984). Their present state of degradation is due mostly to human activity, which includes the frequent fires, some of which must have been accidental, which would have extended from the neighbouring pasture grounds, and others ignited on purpose to make the area more practicable for cattle.

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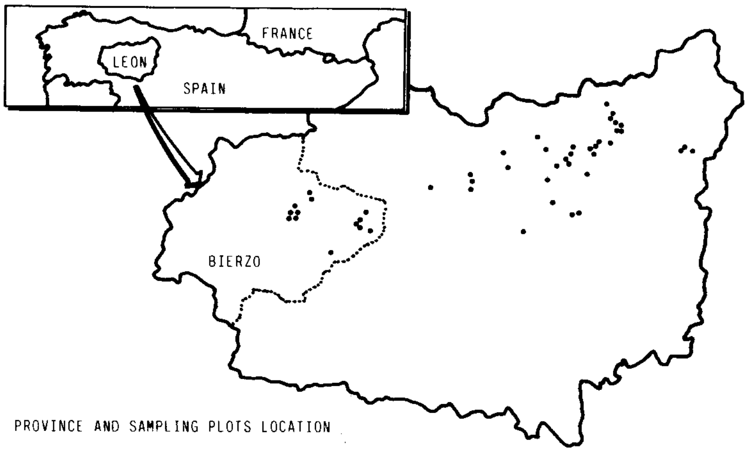


PROVINCE AND SAMPLING PLOTS LOCATION

The main aim of the present study is centred, therefore, on the effects of fire on the structure, dynamics and regeneration of these ecosystems with the view of determining their degree of adaptation to the fires. The ideal in this type of study is to follow the evolution of the same communities through time, but as this is often impossible, it is necessary to revert to space-temporal extrapolations. Nevertheless, we have at our disposal the real temporal evolution of the vegetation, for three consecutive years for some zones, and also including amongst them a spectrum ranging from a month to eight years.

MATERIAL AND METHOD

Description of the study zone

A total of 47 communities were studied, all of them located in the province of Leon (Spain). Thirteen of them were situated in the region of El Bierzo, with a microclimate of its own, defined by a higher level of humidity and more temperate temperatures; they have been designated as BA, BB, BC, BD, BE, BF, BG, BH, BI, BJ, BK, BL and BM respectively. The other 28, in the North-west of the province, are coded with the letters of the alphabet from A to Z; 18 of these, the most recently burned or of a larger extension, were sampled for three consecutive years, placing before their code letter the numbers 1, 2 or 3, corresponding to the first, second or third sampling. In 1A' it was impossible to carry out inventories because no vegetation had sprouted. Those named as PA, PB and PC are those burned at an earlier date, though for the last two, part of them was burnt again recently, and both phases of each community have been sampled. The remaining three, NQ1, NQ2 and NQ3 belong to zones which, according to the information available, had never suffered a fire. The location of these zones can be seen in Fig. 1, and the UTM coordinates, the date of the fire and the sampling, as well as some general characteristics, in Table I. Zones BI, BJ and BK suffered two fires within the stated period.

All of them were surface fires allowing the survival of a great part of the trees.

Sampling method

In each zone, an average of ten inventories carried out at random were performed, with a square sampling unit of 50 cm a side. In each unit the data of the species present were collected, and their degree of importance was expressed in terms of cover average. The main aim of these inventories was the analysis of the herbaceous vegetation, theoretically the most sensitive to the perturbations and the post-fire dynamics unfortunately well known in these ecosystems. Nevertheless, the bushes of small size, even the sprouts of shrubby or arboreous species included in the sampling units, were also registered.

Table I - Definition of geographical characteristics and date of burning and sampling of the studied zones.

ZONES	SITE	UTM COORDINATES	YEAR OF BURNING	YEAR OF SAMPLING	EXPOSURE	ALTITUDE	SLOPE
BA	Sta. Ma. del Sil	29TQH057253	1970-1982	1982	NW	700-800	Quite pronounced
BB	Toreno	29TQH059294	1972	1982	W	700-800	Very pronounced
BC	Toreno	29TQH019352	1972	1982	NE	750-900	Very pronounced
BD	Pardamaza	29TQH069389	1973	1982	SE	1300	Light
BE	Pardamaza	29TQH086370	1970	1982	SW	1300-1400	Variable
BL	Valdelaloba	29TQH021277	-	1982	-	700	Imperceptible
BM	Toreno	29TQH028335	1970	1982	N-NE	700-800	Very light
BF	Almagarinos	29TQH254300	1966	1982	N-NE	900-1000	Very pronounced
BG	Almagarinos	29TQH251305	1975	1982	N-NW	950-1000	Quite pronounced
BH	Almagarinos	29TQH246306	1966	1982	E	1000	Variable
BI	Sta. Marina	29TQH156187	1973-1979	1982	N	700-800	Quite pronounced
BJ	Pobladura	29TQH280318	1958-1980	1982	NW	1000	Quite pronounced
BK	Tremor de Abajo	29TQH252233	1963-1978	1982	NE	900-1000	Very pronounced
V	La Omanuela	30TTN553404	1979?	1982	E	1100	Quite pronounced
NQ1	Quintanilla	30TTN682401	-	1982	N	1000-1100	Quite pronounced
X	Garano	30TTN699425	1978	1981-83	S-SW	1100	Very light
Y	Vega Caballeros	30TTN686445	1980?	1981-83	W-SW	1000-1100	Variable
Z	Carbajal	30TTN871240	1981	1981-83	W	900	Very light
A'	Sta. Ma.del Monte	30TTN996292	1981	1981-83	SE	960	Very light
A	Sta. Ma.del Monte	30TUN026306	1980?	1981-83	-	960	Imperceptible
P	Ruiforco	30TTN953347	1978	1981-83	W y N	950-1050	Variable
O	Pedun y Pardavé	30TTN947413	1978?	1981-83	E	1000	Variable
N	La Valcueva	30TTN977467	1978	1981-83	NW	1100-1200	Quite pronounced
NQ2	Vegaquemada	30TUN083421	-	1982	S	950	Very light
LL	Campohermoso	30TUN018469	1979	1981-83	N y NW	1100	Quite pronounced
M	Campohermoso	30TUN017846	1978 o 79?	1981	S	1200	Quite pronounced
B	Bonar	30TUN115520	1980	1981-83	E-NE y N	1000-1100	Very pronounced
NQ3	Oville	30TUN095520	-	1982	N	1100-1200	Very pronounced
C	Bonar	30TUN110525	1978?	1981	SE	1000-1100	Very pronounced
D	Valdecastillo	30TUN113540	1978?	1981	SE	1100	Quite pronounced
E	Vegamian	30TUN155557	1981	1981-83	NW	1300-1400	Very pronounced
F	Lodares	30TUN167576	1978?	1981-83	S	1200	Very pronounced
G	Pallide	30TUN176580	1978?	1981-83	SW	1100-1200	Quite pronounced
H	Pallide	30TUN179576	1978?	1981	NW y W	1150-1250	Quite pronounced
I	Orones	30TUN175588	1980 o 81?	1981	W	1250	Quite pronounced
J'	Orones	30TUN173595	1980?	1981	SW	1200-1300	Quite pronounced
J	Orones	30TUN173603	1979 o 80?	1981-83	W y W-SW	1400	Quite pronounced
Q	Cofinal	30TUN127666	1978 o 79?	1981	W y SW	1300-1400	Variable
K	Valdepiélago	30TUN038505	1979	1981-83	E-SE	1100-1250	Quite pronounced
L	Nocedo	30TUN036535	1980?	1981-83	S	1300-1450	Quite pronounced
R	Vegacervera	30TTN937524	1979?	1981	S	1150-1200	Light
S	Felmin	30TTN925552	1979?	1981	S-SE	1100-1200	Quite pronounced
T	Puente de Alba	30TTN244442	1979?	1981-83	N y E	1000-1100	Variable
U	Buiza	30TTN805521	1975	1981-83	E-NE	1200-1300	Variable
PA	Prioro	30TUN386539	1920?	1982	S	1200-1300	Very light
PB	Prioro	30TUN358501	1962-1978	1982	S y NE	1200	Variable
PC	Prioro	30TUN417514	1965-1978	1982	NW	1250-1400	Very pronounced

Statistic treatment

Due to the enormous amount of data collected, a statistic analysis to quantify and thus clarify the information was considered necessary. For this, an average cover of the species in the total of the inventories belonging to each community and sampling is employed, except for those communities where we have no certainty about the year of the burning.

Three affinity analyses between the different groups of samplings are carried out: a temporal analysis, considering the samplings in order to their ages distinguishing between samplings with ages of post-fire regeneration ranging from one to three years; between samplings over three years after the fire; and a third analysis among communities located in El Bierzo, because of the peculiar characteristics of this region. For this, an index attributed to Steinhaus by MOTYKA et al. (1950), which is applied with quantitative variables, is used, and it is expressed with the formula:

$$S_{(x_1, x_2)} = \frac{2\,w}{A + B}\,100$$

where w = summation of the lowest cover value of the species present in the two samplings x_1 and x_2.

A = summation of all the cover values of all the species present in sampling x_1.
B = summation of all the cover values of all the species present in sampling x_2.

The results obtained are grouped together by means of the UPGMA method (SOKAL & MICHENER 1958) and are represented graphically in form of dendrograms.

The specific diversity is also calculated by the information index of SHANNON-WEAVER (1949):

$$H' = \sum_{i=1}^{s} p_i \log_2 p_i$$

where p_i = probability of finding species i.
s = n° of species.

Their evolution over the course of time as well as their components, richness and uniformity are studied.

The heterogeneity of each sampling is also studied by means of a modified expression of the formula of MARGALEF (1972):

$$\text{Het.} = H'_T - \frac{\sum_{i=1}^{n} H'_{\alpha i}}{n}$$

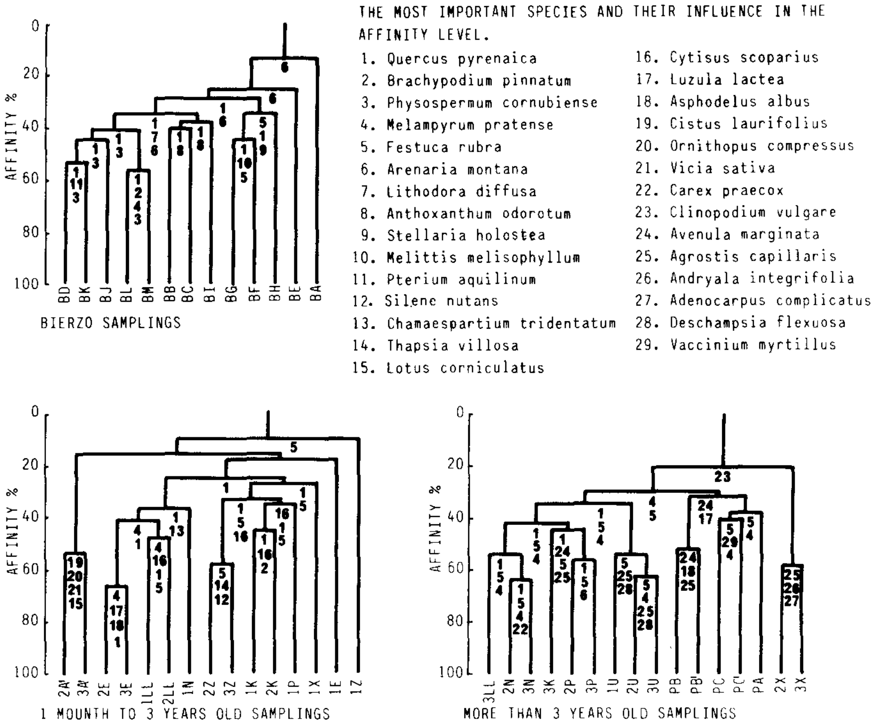
where H'_T = diversity of the sampling according to the Shannon-Weaver index.
$H'_{\alpha}i$ = diversity of each inventory of that sampling.
n = number of inventories performed in that sampling.

RESULTS AND DISCUSSION

In the affinity analysis of the samplings out from one month to three years after fire, we can see the biggest difference in the floristic composition of the other samplings compared with 1E and 1Z, where there were only a few weeks of regeneration, where the soil without vegetation is predominant, and with only a few perennial species, which differ in both cases as these 2 samplings do not associate with each other. The predominance of the vivacious species (TARREGA 1986), with only three annual species out of the 16 registered, suggests, in both cases, that most of them resisted the fire fundamentally by means of underground organs, from which plants rapidly sprouted. This is a phenomenon quoted by numerous authors (DEBUSSCHE *et al.* 1980, TRABAUD 1980, CASAL *et al.* 1984).

The separation of the samplings belonging to zone A' can also be appreciated from the dendrogram, which can be explained by the great abundance of *Cistus laurifolius* and *Ornithopus compressus* in it, which practically do not appear in the others, as well as by the large predominance of annual species which penetrate from the neighbouring pasture grounds, due to the lesser density of trees in this community, and which conditions a quite different floristic composition (Fig. 1).

Taking into account the remaining areas, which have in common that small sprouts of oak trees are found in them, we can distinguish two sub-groups. The first one is characterized by the presence of *Chamaespartium tridentatum*, which does not appear in the areas which are part of the other sub-group. The great abundance of *Melampyrum pratense* also stands out, which conditions the union of 2E, 3E, 1LL and 2LL. In the second group the biggest similitude appears between group 2Z and 3Z, which are the only samplings in which *Thapsia villosa* is present.



THE MOST IMPORTANT SPECIES AND THEIR INFLUENCE IN THE AFFINITY LEVEL.

1. Quercus pyrenaica
2. Brachypodium pinnatum
3. Physospermum cornubiense
4. Melampyrum pratense
5. Festuca rubra
6. Arenaria montana
7. Lithodora diffusa
8. Anthoxanthum odorotum
9. Stellaria holostea
10. Melittis melisophyllum
11. Pterium aquilinum
12. Silene nutans
13. Chamaespartium tridentatum
14. Thapsia villosa
15. Lotus corniculatus
16. Cytisus scoparius
17. Luzula lactea
18. Asphodelus albus
19. Cistus laurifolius
20. Ornithopus compressus
21. Vicia sativa
22. Carex praecox
23. Clinopodium vulgare
24. Avenula marginata
25. Agrostis capillaris
26. Andryala integrifolia
27. Adenocarpus complicatus
28. Deschampsia flexuosa
29. Vaccinium myrtillus

Fig. 1.- Affinity relations for three different analysis.

It can be appreciated that the association between the zones is not determined by the time that has elapsed since they suffered the fire, with the exception of the communities with only a few weeks of recovery which, for this reason, do not follow the same trend as the others. A preferential grouping can be detected in the other cases among the different samplings of a same area, which shows a greater resemblance among the different phases of a same community than among the samplings of the same age belonging to different zones.

In the dendrogram pertaining to the communities burnt at an earlier date (more than a three year post fire period) the connection between the samplings from a same zone is also apparent even where there are great differences in the age of regeneration, as in the case of PB and PC. The pair 2Z-3Z, differs from the others because it lacks Melampyrum pratense, a species highly abundant in the others. The influence of the geographic proximity in the associations can be observed, as in the group of areas located near the village of Prioro (PA, PB and PC) though this is not a general rule. The influence of many other factors which condition the floristic composition cannot be denied, the most important being connected to each specific area.

Similar results are obtained in the affinity analysis of the zones located in El Bierzo. The largest affinity group (BD, BK, BJ, BL and BM) is characterized by the presence of Physospermum cornubiense, a species which does not occur in the others. In the remaining groups the union is determined by the total of the species rather than by the exclusive apparition of one which makes them differ from the others. The association of the different zones in some cases is caused by the geographic proximity which conditions similar microclimatic and edaphic characteristics for vegetation, but in most cases it depends on the complex interaction of multiple factors, such as the orientation of the slope, the density and size of the trees, etc. It seems clear that the time elapsed since the attack of the fire is just one of the factors of this group. That it is not one of the most important factors is illustrated by the fact that no connection can be detected between the three areas which suffered a fire again more recently, (BI, BJ and BK) and the area which did not suffer the fire (BL) is not to be distinguished from the remaining ones (Fig. 1).

Global analysis of all zones (those with an approximate date of burning as well) were carried out, determining the similitude in relation to the Euclidean distance and using a computer for that purpose. Because of the great magnitude of the species found in all zones (339) which exceeded the capacity of the computer, it was necessary to leave aside the species which were found in less than five samplings, out of the total of 80 taken into account. A factor analysis in the principal components was also performed, starting from the same matrix of data. The results obtained (TARREGA 1986) confirm the lack of association between the communities of the same age of regeneration, and a bigger resemblance between samplings of the same zone, although the latter is masked by the suppression of species.

The fact that there is no discrimination between groups of areas burned at similar dates shows clearly the lack of successional stages, characterized by a clear and sequential floristic relay such as the one described in the classical theories of succession. Most of the authors who study the post-fire regeneration of plant communities obtain similar results (METHVEN et al. 1975, PURDIE & SLATYER 1976, SAUER 1977, OHMANN & GRIGAL 1979, TRABAUD 1980, 1986, NOBLE 1981, etc.). The development of the vegetation after the fire is conditioned to a great extent by the floristic composition of the community prior to it. A phase of invasion of pioneering species, mostly annual, detected by other authors, mainly in studies about the bush (DYRNESS 1973, ANDERSON & BAILEY 1979, CASAL 1984), cannot be detected. This is probably due to the survival of most of the oak trees which, although more or less damaged, partly preserve the structure previous to the fire, and usually give enough shade to slow down the massive invasion of pioneering species, normally of the heliophilous type.

The regeneration of Quercus pyrenaica ecosystems consists, therefore, in a process of autosuccession, compensatory to the regression suffered. Nevertheless, there do not exist cyclical changes of species as those described in the climax or subclimax of fire (MYTINGER 1979). To the surviving species, mostly oak trees, are added to those which resprout immediately after the burning, starting from persistant underground organs, and those which sprout shortly after springing from seeds which were already in the zone and resisted thanks to the bad diffusion of the heat in the soil, or due to seeds coming from the neighbouring communities.

This progressive incorporation of species conditions an increment of the diversity during the first two years (Fig. 2), although later on this tendency is not general in all the

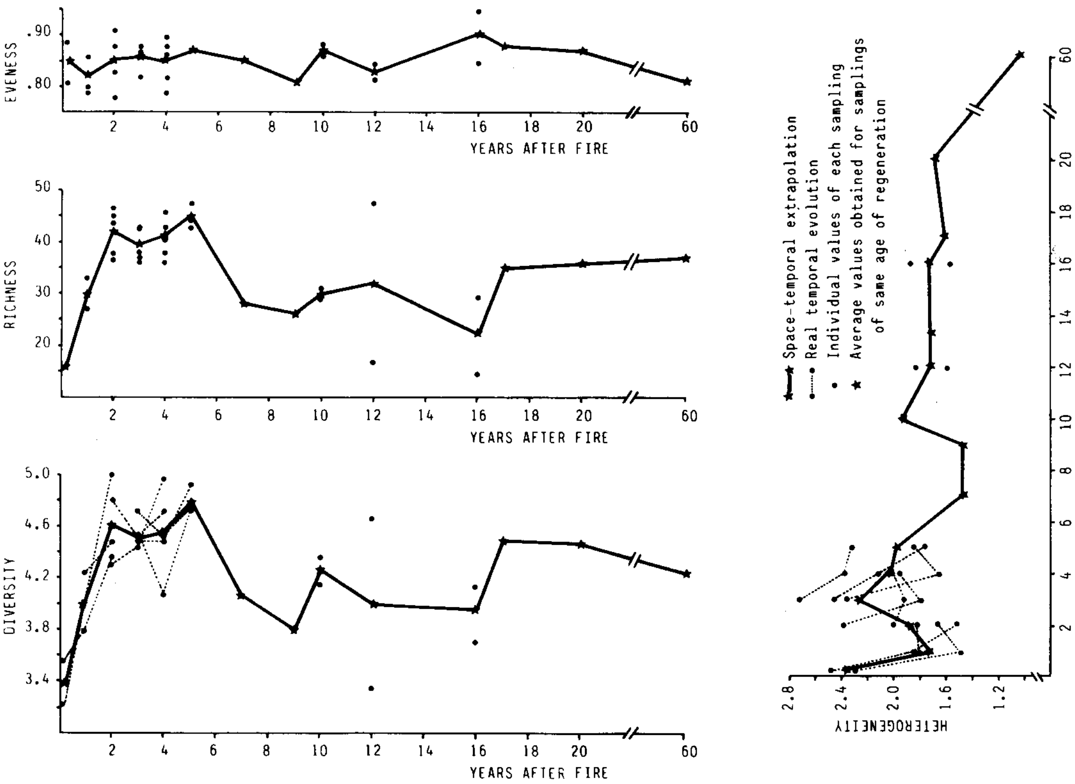


Fig. 2.- Structural parameters evolutions after fire.

zones. If the average value of the diversities of the zones of the same age is taken into account, a stabilization during the period up to five years can be noted, showing thereafter an irregular evolution, partly due to the fact that there are less data available. The progressive increment of the diversity in the first years after the fire followed by a posterior descent or stabilization is evident in most of the works on this subject (BELL & KOCH 1980, DEBUSSCHE *et al*. 1980, TRABAUD & LEPART 1980, CASAL 1982).

The increment of diversity is mostly due to the increment in the number of species (richness) as the great similarity between the 2 graphs shows, with a similar evolution. SHAFY & YARRANTON (1973) agree on this point. Uniformity has a more irregular behaviour, although with very small irregular oscillations in the course of time.

As far as the evolution of heterogeneity is concerned, if the average values are taken into account, it can be observed that the highest heterogeneity belongs to the communities just burned, the graph being afterwards irregular, although with an overall tendency to decrease (Fig. 2). It is probable that this tendency cannot be generalized because the oscillations can be better understood if they are considered linked to the heterogeneity values of each specific area, independently of the time elapsed since it burned. What really seems to be meaningful is the high heterogeneity of the youngest communities.

Fire can act as a perturbing element, exercising a more intense effect in some plots than in others depending to a very small extent on the microclimatic and topographic conditions. This determines if the vegetation is influenced to a greater or lesser extent and enables small, almost intact, mosaics to be observed in the same zone, surrounded by areas completely burned.

All the ecosystems studied show a great heterogeneity independently of their age of post-fire regeneration. Besides, they offer remarkable differences between each other, as can be verified by observing the deviation of some values from the average (for example, zones BE and BM, burnt twelve years before the study was carried out). Because of this, space-temporal extrapolations are not commendable in this type of community, and they must be only given an approximate value, backed by the temporal dynamics of the same communities.

CONCLUSION

The ecosystems of *Quercus pyrenaica* do generally recover well after the action of surface fires in a process of autosuccession which shows the presence of species adapted for a very long time to these perturbations. A progressive increment of the specific diversity of the herbaceous stratum up to two years after the burning can be observed. From that moment onwards it is difficult to detect structural differences caused by fire, and due to the survival of most of the oak trees, the passage of fire can only be recognized because of some burnt remains.

Nevertheless, it seems clear that the increment in the frequency of fires registered in the last few years constitutes an enormous factor of risk (TARREGA 1986). The fact that a system in a stage of recovery is again damaged after a very short time by a new fire, results in an increasingly greater degradation of the environment, which can lead to an irreversible alteration of the ecosystem.

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Post-fire dynamics of shrubland dominated by Papilionaceae plants.

M. CASAL★

SUMMARY - Shrublands dominated by aphyllous Papilionaceae are commonly seral communities in the Mediterranean Region derived from degradation of climatic forest, developed on poor soils and maintained by human action. Because of their morphology, these shrublands are very susceptible to fire.

In this paper, changes after fire in different Papilionaceae (Ulex, Cytisus, Genista, Calicotome, Chamaespartium) communities are analysed, specially in France and Spain. All these communities and species show similar trends of change.

The recovery from resprout is quantitatively the most important. A few woody species became dominant in the community thanks to this recolonization system; other species which recolonize only by seeds (Cistus or Ericaceae shrubs) became less dominant with fire. The same occurs for herbs and grasses.

The horizontal and vertical structures of the community change strongly : shrubs' height, cover and biomass increase with time. The herbaceous community shows an abundant displacement, the annual species being replaced by perennial species.

Species richness and diversity vary in a different way depending on the component : the number of annual species is unstable, whereas the diversity of perennial species progressively increases.

The maintenance of Papilionaceae shrublands is due to repeated, but not very recurrent, fires. The post-fire dynamics seems to fit Connell & Slatyer's succession model.

KEYWORDS : European shrublands, woody Papilionaceae, fire, dynamics, structure, diversity, regeneration strategies.

INTRODUCTION

In spite of their scant occurrence in the Mediterranean region and the frequent presence of sclerophyllous or malacophyllous species, the term "non-sclerophyllous Mediterranean vegetation" is often applied to a variety of shrubland communities dominated by aphyllous Papilionaceae species. These seral communities derived from the degradation of Quercus or Fagus sylvatica woods are in fact more common in peri-, supra- and oro-Mediterranean regions with more Continental or more Atlantic climates (TRABAUD & ALLAG 1976, BELLOT 1978). Most species are evergreen shrubs sharing certain morphological adaptations such as photosynthetic stems and/or spines, or leaves which are few, or transformed into spines, or only present in the early stages of the plant development. In climates with a pronounced summer period, Papilionaceae are the main species accompanied by Cistaceae and (in moister Atlantic climates) Ericaceae. Such communities generally develop on poor, acid, little-evolved soils (lithosols, rankers or woodland soils) at sites with relatively moderate summer drought. The Papilionaceae withstand these conditions due to their symbiosis with Rhyzobium, and the Ericaceae with mycorrhizae (MALAJCZUK & LAMONT 1981), and due to their adaptation to low-nutrient soils (AUBERT 1978). The aphyllous Papilionaceae communities are maintained by the repeated disorganisation through wild fires, stubble burning, or grazing, all related to human intervention.

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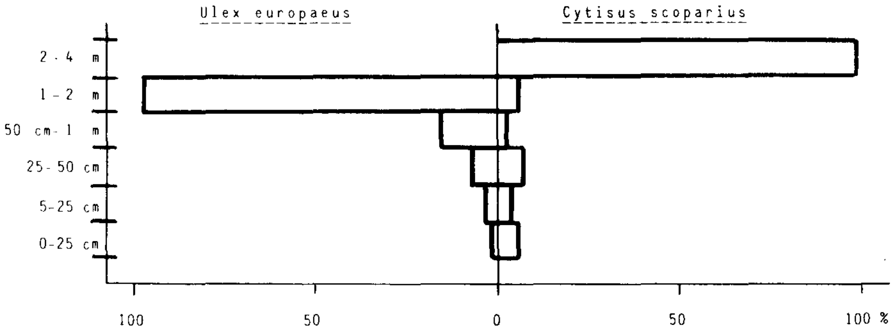

FIGURE 1.- Vertical profiles in U. europaeus communities and in C. scoparius communities, from % cover data (from BASANTA, 1984)

The morphology of their dominant species presents a finely divided biomass, a large unshed necromass and a structure with large gaps near the ground facilitating aeration during fire (Figure 1). These features make the communities highly susceptible to fire, though species containing aromatic resins or essential oils are relatively scarce.

Little research has been done on their post-fire dynamics, and the studies which have been carried out, whether diachronic or synchronic, have only covered short intervals and often failed to quantify the observed changes in structure and species composition. Communities of Ulex europaeus L. were studied by CASAL et al. (1984, 1986); the same shrublands as well as others dominated by Chamaespartium tridentatum (L.) P. Gibbs were examined by DALDA (1978) in NW Spain whereas communities of Calicotome spinosa (L.) Link were studied by FRANQUESA (1985) in NE Spain; DEBUSSCHE et al. (1980) focused on the Cytisus purgans (L.) Boiss and the Genista scorpius (L.) DC. communities in SE France, and TRABAUD & ALLAG (1976) on several shrublands in the same area, including Cytisus scoparius (L.) Link communities; finally, GONZALEZ BARTOLOME et al. (1986) studied the Cytisus balansae (syn. purgans) shrublands in Central Spain. All these studies showed that the post-fire development is strongly influenced by the pre-fire species composition, but the succession process is more complex because of the drop in density of some populations, the rapid increase of others and the brief appearance of transient species. Reforestation with conifers can increase the intensity and temperature of wild fires, and the introduction of Eucalyptus can promote the inflammability of the vegetation. In both cases, the risk of fire is increased, since the effect of any shading of the undergrowth is more than offset by the desiccation of the soil.

REGENERATION STRATEGIES

The dominant species of communities subject to periodic burning possess survival mechanisms ensuring the rapid recolonization of the environment after fire. Most are capable of both vegetative regrowth and regeneration from seed, but the former mechanism is usually the more important. Thus the after-fire resprouting guarantees the continued predominance of C. spinosa, C. scoparius, C. purgans, Ch. tridentatum, G. scorpius or U. europaeus over secondary or opportunistic species relying solely on the germination of seed. C. purgans, for example, produces abundant resprouts within a few days after burning, and its population is later reinforced by the presence of many seedlings in the wake of passing flocks of sheep; 8-10 years after the fire, dense bushy cover has developed to a height of 1 m (DEBUSSCHE et al. 1980).

Ulex, Cytisus and Chamaespartium generally owe their fire-resistance to their deep root systems. In the case of C. balansae, resprouts also emerge from surviving bases of branches, since before burning mature branches bow down to the ground and encourage the formation of a mantle of soil and litter. In some locations, such as the Sierra de Béjar (above 2100 m), the low branches accumulate a large and undecomposed necromass that builds up on the plant. In this case, fire kills all existing plants and regeneration relies exclusively on the soil seed bank (GONZALEZ BARTOLOME et al. 1986).

The production of post-fire resprouts fluctuates over the life cycle of the plant. Very young or very small plants fail to survive because their root systems are either not deep enough to escape high soil surface temperature, or have not yet accumulated sufficient reserves to enable resprouting. At the other extreme, very large, senescent plants lose their capacity for vegetative regeneration; C. scoparius, for example, only throws out resprouts when less than 10 years old (TRABAUD & ALLAG 1976). Loss of regenerative capacity may also occur to whole populations of resprouting species if they are subjected to over-frequent burning, which cause excessive depletion of their reserves (TRABAUD 1980).

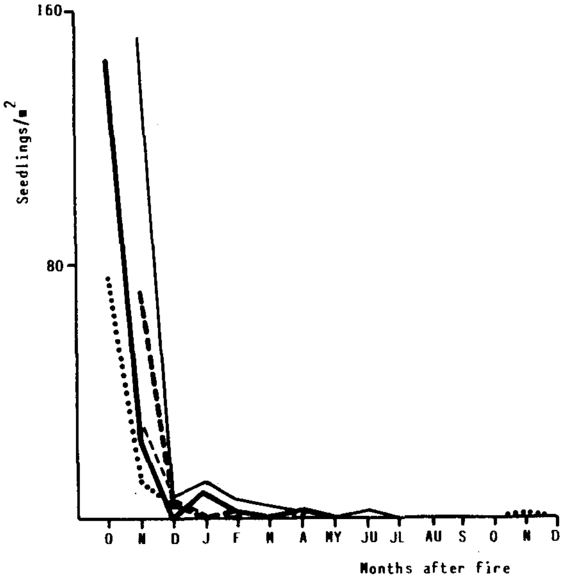

FIGURE 2.- Average number of U. europaeus seedlings emerging monthly. Each line corresponds to a sample pair (from PUENTES et al., 1985)

However, the seeds of many genera of Papilionaceae are very hard, impermeable and germinate with difficulty. Their prolonged viability leads to the accumulation of large seed banks (CROKER & BARTON 1957, MARTIN et al. 1975). U. europaeus seeds, for instance, may remain buried and viable for up to 30 years (MOSS 1959, ZABKIEWICZ 1979), and in stands in Galicia (NW Spain), PUENTES (1984) found seed banks of 645 seeds/m^2 (with 89% in the top 5 cm) and 1045 seeds/m^2 (with 86.5% in the top 10 cm). The germination of U. europaeus seeds is stimulated by high temperature shocks during a short time. After a shock at 80°C, the maximum stimulation is achieved after 20-30 minutes exposition and the germination rate decreases after 2 hours of exposition. However, the germination rate decreases after an exposure to higher temperatures even for a few minutes, e.g. at 120-150°C for 5 minutes only (PEREIRAS et al. 1985). The actual appearance of seedlings in a burnt stand also depends on the time of year; PUENTES et al. (1985) found that the sprouting of U. europaeus seedlings was intense in October and November but practically nil after May (Figure 2), and CASAL (1982) found that seedling emergence invariably exhibits a single peak during the first autumn after burning, regardless of when the fire took place. In the 14 months following the massive appearance of seedlings in autumn, 40% are lost, 65% of the losses occurring during the winter months (PUENTES et al. 1985). The size and evolution of a population of gorse seedlings is also strongly influenced by microtopography (Figure 3), both initial numbers and mortality decrease with increasing slope (PUENTES et al. 1985). In the case of Ch. tridentatum, mortality among young seedlings is likewise heavy (40-50% among seedlings < 1 cm), and in summer exceeds the rate of emergence, which is minimum at this time of year (BASANTA ALVES et al. 1986). Even when massive germination of seed takes place after fire, the plants so produced may little contribute to the final population : PUENTES (1984) found that plants produced from seed made up only 19% of the total biomass of a gorse stand 14 months after burning.

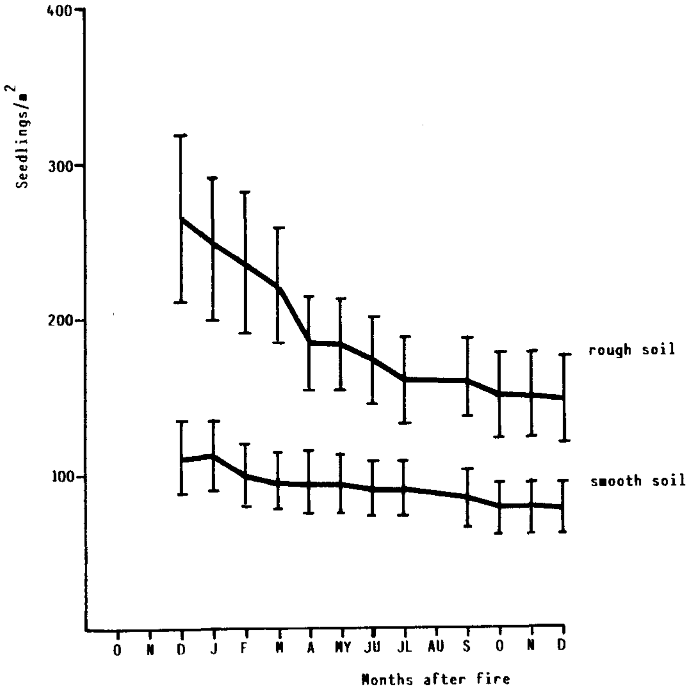

FIGURE 3.- Change along time in number of gorse seedlings according to microtopography (From PUENTES et al., 1985)

An apparent exception is G. scorpius, whose resprouts are less vigorous than those of Ulex or Cytisus; in G. scorpius communities, seed may be provided not only by seed banks but also by dispersal from adjacent stands via sheep, the regeneration from seed is more important and the vegetation developed is less dense than in Cytisus stands (DEBUSSCHE et al. 1980).

The internal distribution of the dominant populations of these communities depends on the abundance and size of the plants before burning and the density of resprouts. Pre-fire distribution is retained virtually intact for perennials regenerating with resprouts, such as Ulex spp., Ch. tridentatum, Halimium alyssoides (Lam.) C. Koch or Agrostis setacea Curtis (DALDA 1978). Populations of C. balansae, on the other hand, show a marked heterogeneity in size due to differential growth of new individuals from seed and the effects of competition. This effect tends to modify the pattern of the pre-fire distribution turning it to a random distribution (GONZALEZ BARTOLOME et al. 1986).

Those species with no resprouting ability are disadvantaged with regard to resprouting species. Even when germination is abundant (as is the case for the Cistaceae and Ericaceae), seedlings can only occupy the space left free by resprouts, and their development is very slow. For example, in Calicotome stands, Cistus albidus L. and Cistus monspeliensis L. have been found to produce no more than 150 and 20 seedlings/m^2 respectively. Erica cinerea L., Erica umbellata L. and Daboecia cantabrica (Hudson) C. Koch are in a similar situation in communities dominated by U. europaeus or Ch. tridentatum. If germination does not occur before the spring following burning, as in the cases of E. cinerea, E. umbellata and Calluna vulgaris (L.) Hull., the young seedlings may then have to compete against 6-9 month-old woody and herbaceous resprouts for a nutrient pool that has been depleted by winter rain (CASAL et al. 1984).

Over-frequent burning, or burning during the ripening of seeds may prevent post-fire regeneration from seed if the seed bank stored in the soil is not large or if soil seed survival is short. It is the case of certain Cistaceae, such as C. albidus and C. monspeliensis (FRANQUESA 1985).

CHANGES IN SPECIES RICHNESS

During the months following burning, the species richness of the burnt stand generally increases rapidly to a peak before declining more slowly to become stable at "normal" pre-fire community levels. An exception is provided by the C. spinosa shrublands studied by FRANQUESA (1985), which in the first 40 months after burning exhibited peaks of species richness each spring (an average of 28 species in ten 25 m^2 quadrats) and minima at the end of each summer (14 or so species). Figure 4A shows that in mesoxerophilous shrublands in Galicia (NW Spain), the single peak is attained on average 18 months after burning (in the first or second spring) and stability of species composition is reached after 4 years (CASAL et al. 1986).

Inventories taken in mesophilous and hydrophilous communities of Ulex spp. in this region, and in the Ch. tridentatum communities studied by DALDA (1978), show an increase in species richness after fire during the second and even the third year. In C. purgans shrubland, DEBUSSCHE et al. (1980) found that species richness was maximum when C. purgans provided 20% cover (in the second or third year). Species richness remained high during the fourth year, but then fell until there was approximately 60% of C. purgans cover; during the remainder of the 18-year study, the decline continued but much more slowly. The fact that the number of appearing and disappearing species decreases gradually after the first or second year means that the turnover of the community decreases. Similarly DEBUSSCHE et al. (1980) found in G. scorpius stands which they began to study seven years after burning, that the fall in species richness continued until at least the 18th year after fire.

Like the overall species richness, the spectrum of biological types in Galician shrubland reveals the selective action of repeated fires (DALDA 1978), but the populations of the various life forms evolve in different ways in the years immediately following burning. The number of woody species and herbaceous perennials increases rapidly during the first two years and continues to rise slowly thereafter, whereas the number of therophytes peaks after 16-20 months and then markedly falls, with minimal levels after 4 years (CASAL et al. 1986). According to DEBUSSCHE (1978), the peak in species richness is the result of an invasion of the area by annuals, formerly absent in the pre-fire vegetation. This general trend is modulated by seasonal fluctuations in the number of therophytes in accordance with the life cycles of the various species, the magnitude of the fluctuations being greater in stands with lesser shrub cover (FRANQUESA 1985). DEBUSSCHE (1978) and DEBUSSCHE et al. (1980) also reported that the number and proportion of therophyte species was greater in drier than in wetter environments, and greater at lower than at higher altitudes. These findings were supported by DALDA (1978) and DAGET (1980). In the G. scorpius stands studied by DEBUSSCHE et al. (1980) between 7 and 18 years after burning, the number of therophyte species fell until they totally disappeared 11 years after fire, and the number of hemicryptophyte species likewise dwindled, though some still remained after 18 years. In the final years of the succession, the proportion of phanerophytes and geophytes tends to increase at the expense of chamaephytes (ESCARRE 1979).

There is a considerable number of species whose persistence is assured in the middle term. These are all perennial species that were present in the pre-fire community, and which have survived the fire thanks to the resistance of roots, rhizomes or bulbs or to their seed bank. All these species reappear during the first year after burning, 87.5% of them in the first 5 months in shrub stands of NE Spain (FRANQUESA 1985).

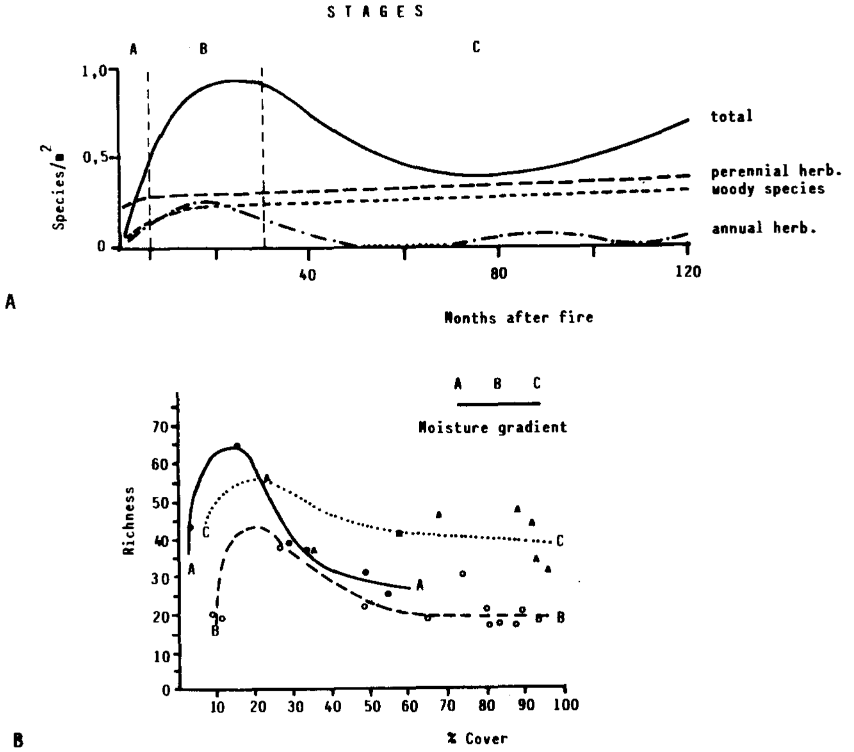

FIGURE 4.- A: Changes in specific richness after fire in U. europaeus shrubland (from CASAL et al., 1986)

B: Changes in richness along three post-fire successions in the C. purgans communities (In DEBUSSCHE et al., 1980)

The transient species colonizing open ground, on the other hand, may not have been present before the fire. Some of these opportunistic species, such as Aira praecox L., are typical of the first spring after burning. Others, such as Tuberaria guttata (L.) Fourr., will peak during the second spring. Others exhibit again population peaks during the first two post-fire springs. Furthermore, each group contains true vernal species and pre-vernal species (CASAL et al. 1986).

STRUCTURAL CHANGES

During the first stages of the post-fire regeneration of the plant community, the recolonization of the open ground by surviving and imported propagules, is hampered by the instability and degradation of the ecosystem. After a brief latency period, the area of open ground in burnt U. europaeus decreases exponentially, with an average 74% of the ground lacking woody cover at the end of the first year after burning, 30% after 3 years, and just 2% after 10 years (CASAL 1985). Recolonization is more rapid after a spring fire than after a fire occurring late in the year (Figure 5). The regeneration of shrub biomass follows a similar pattern : a rapid increase during the first three years after burning is followed by a gradual decrease in growth rate after 10 years as the community tends to stabilize, with biomass levels of 10-25 t/ha. Again, the recovery of biomass is more rapid after spring fires than after autumnal fires.

Figure 6 illustrates the quantitative differences between Ulex spp. and coexisting Ericaceae as regards the regeneration of biomass. Since U. europaeus alone makes up 70-75% of the total biomass of the stand 10 years after burning, Ulex biomass follows virtually the same trend as the total biomass, with very fast recovery during the first three years after the fire. The Ericaceae, on the other hand, exhibit a sigmoid trend, low regeneration rates during the first 1.5 years being followed by a rapid rise during the next 3.5 years and a subsequent deceleration. After 10 years, the biomass of the Ericaceae ranges from 0.5 to 4 t/ha. The lag between these two components of the ecosystem, which is due to the difference between their recolonization strategies, makes *U. europaeus* the more competitive of the two, and the heathers are gradually displaced under a frequent fire-regime. Thus the enormous areas occupied by gorse stands in Galicia today, and the marked predominance of *U. europaeus* over the various kinds of heather, may be explained by the recurrence of fires and other human interventions such as clearance in the semi natural vegetation, as it has been documented by BOUHIER (1979).

As time proceeds, the spatial structure of the post-fire community becomes more complex. DEBUSSCHE *et al*. (1980) found that the vertical distribution of green biomass in *C. purgans* stands invariably exhibited a single peak at a progressively greater height above ground level.

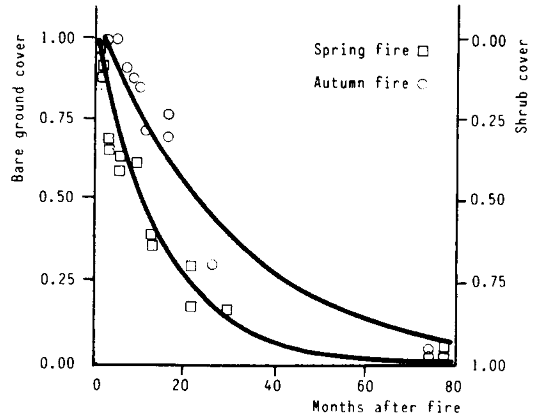

FIGURE 5.- Changes in shrub cover along time; the intensity and rate of recovery is influenced by the season of fire occurrance (From CASAL & GARCIA NOVO, 1985)

In the first few years after the disturbance, the horizontal cover provided by green biomass also rose, only to fall when the post-fire plants became senescent (Figure 7A). Unfallen necromass was located largely in the lower strata, with a single peak at a lower height than that of the green biomass. *G. scorpius* exhibited similar structural progress. *C. purgans* stems grew almost exclusively in a vertical direction, and herbaceous vegetation declined as shrub cover spread. OJEA (1986) found that the post-fire structural evolution of *U. europaeus* was slightly different (Figure 7B). In plants of medium height, unfallen necromass provided much more cover than green biomass, and the horizontal frequency of stems was much greater than in the case of *C. purgans*. The distribution of biomass was similar to the diagram of frequency : supporting organs increased progressively in the lower strata and gain in biomass levelled off above 90 cm except for stems, which kept increasing up to 1.5 m. As a result, the ratio of photosynthetic biomass to total biomass fell appreciably with increasing height (0.65-0.08).

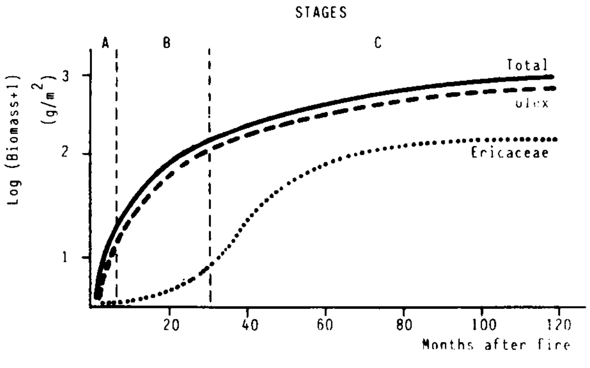

FIGURE 6.- Shrub biomass evolution after fire (Redrown from CASAL *et al*., 1984)

DIVERSITY

Diversity, as measured by the Shannon index, exhibits a trend similar to that of species richness (CASAL 1982), with a peak after 20-24 months being followed by a fall to normal values. Similar results were reported by DEBUSSCHE (1978). Following MARGALEF (1980), the peak can be interpreted as indicator of a chaotic situation which gives way to a tendency towards a greater degree of "organisation". In the present case, the "chaos" is due to the proliferation of populations of opportunistic species which take advantage of the new ecological conditions in the wake of the fire : open ground, lack of competition and greater availability of nutrients and light.

The contribution of specific populations changes with time. Some species which are initially abundant are later displaced by the progressive expansion of the more persistent species.

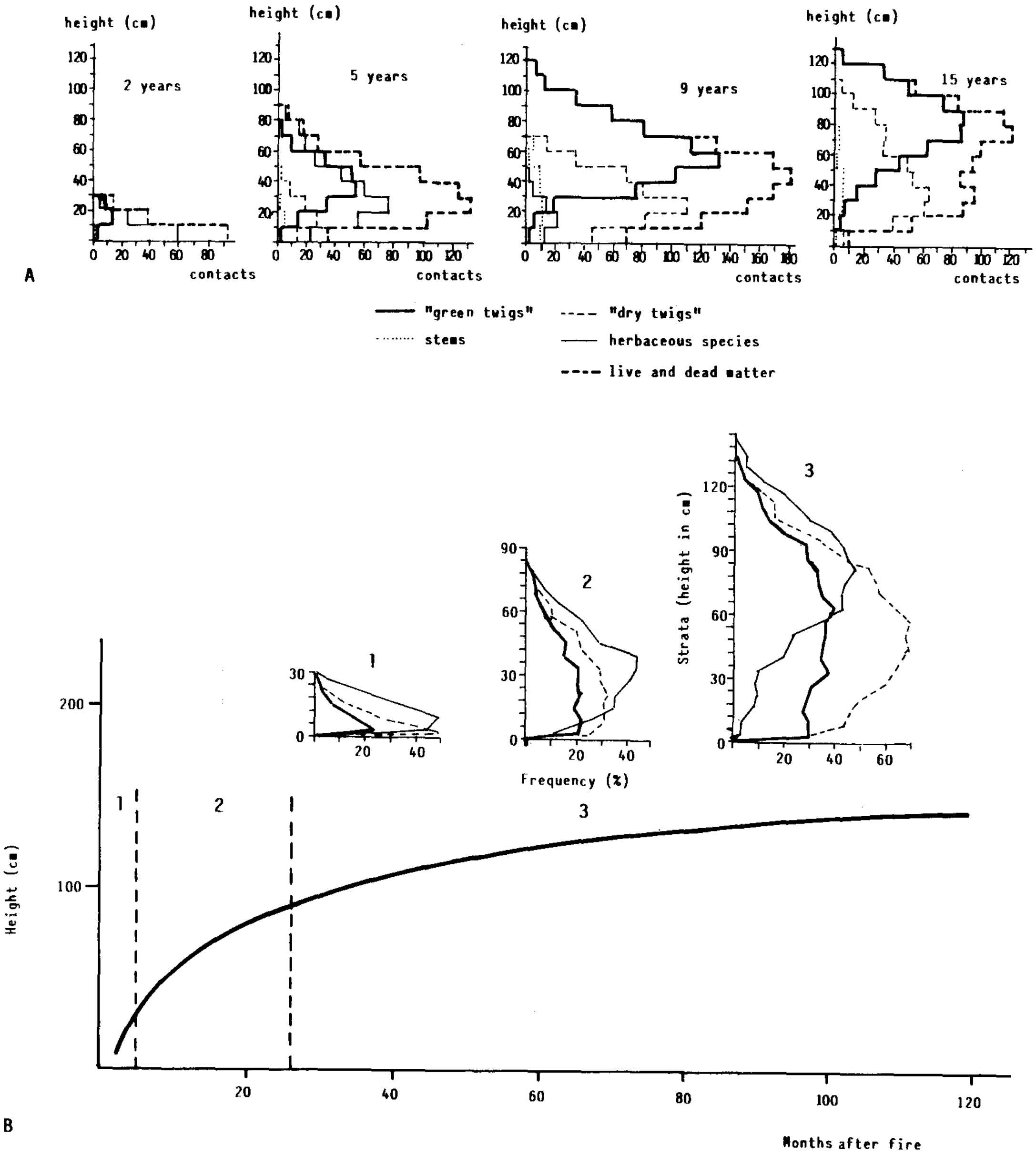

FIGURE 7.- A: Stratification of *C. purgans* communities at different stages (From DEBUSSCHE (1978) in DEBUSSCHE *et al.*, 1980)

B: Vertical structure of *U. europaeus* for green (——), dead (----) and woody (━━) structures (strata of 6 cm in height) (From OJEA, 1986)

Figure 8A shows dominance-diversity graphs based on frequency data for herbaceous species in a burnt U. europaeus stand (CASAL et al. 1986b); it can be seen that after an initial boom, annuals tend to disappear from the area. The dominant herbaceous species is always the perennial A. setacea. Dominance-diversity graphs for woody species based on linear cover data are shown in Figure 8B. Though the dominant species is always U. europaeus, the Ericaceae later acquire considerable importance too, as does the "other species" group. The marked increase in diversity is not due to an increase in the number of species, but rather to the reduction of the numerical imbalance among the populations of the species present.

The evolution of the diversity of the community is heavily dependent on the time of year at which burning occurred (CASAL & GARCIA NOVO 1986). Comparison of the development of woody species after spring and autumn fires in nearby U. europaeus stands showed similar trends as regards the species richness of the shrub vegetation, but differing trends as regards dominance and diversity.

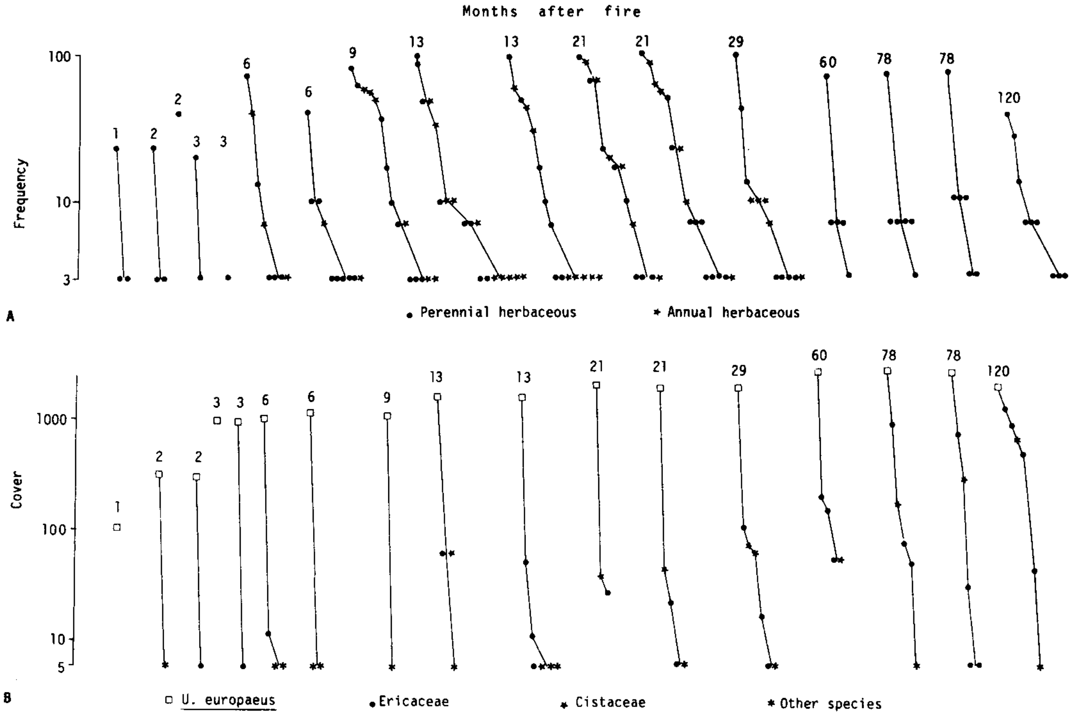

FIGURE 8.- A: Dominance-diversity curves for herbaceous species in U. europaeus shrublands after fire. All data from a single sampling site (From CASAL et al., 1986 b)

B: Dominance-diversity curves for woody species in U. europaeus shrublands after fire (From CASAL et al., 1986 b)

Dominance (Simpson index) was very high after spring fires and low after autumn fires, decreasing gradually in both cases during the remaining six years of the study. After spring fires, shrub cover spreads rapidly, with low diversity and evenness, whereas the spread of shrub cover is slower and the diversity and evenness are greater after autumn fires (Figures 9A and 9B).

To sum up, the diversity of the post-fire community is heavily dependant upon the pre-fire community, and in particular upon its dominant species, which control both woody and herbaceous subsystems by limiting the population on controlling the implantation of other species; also it is influenced by the timing of the fire. DALDA (1978) remarks that the poor diversity of Galician shrubland is due to their repeated burning.

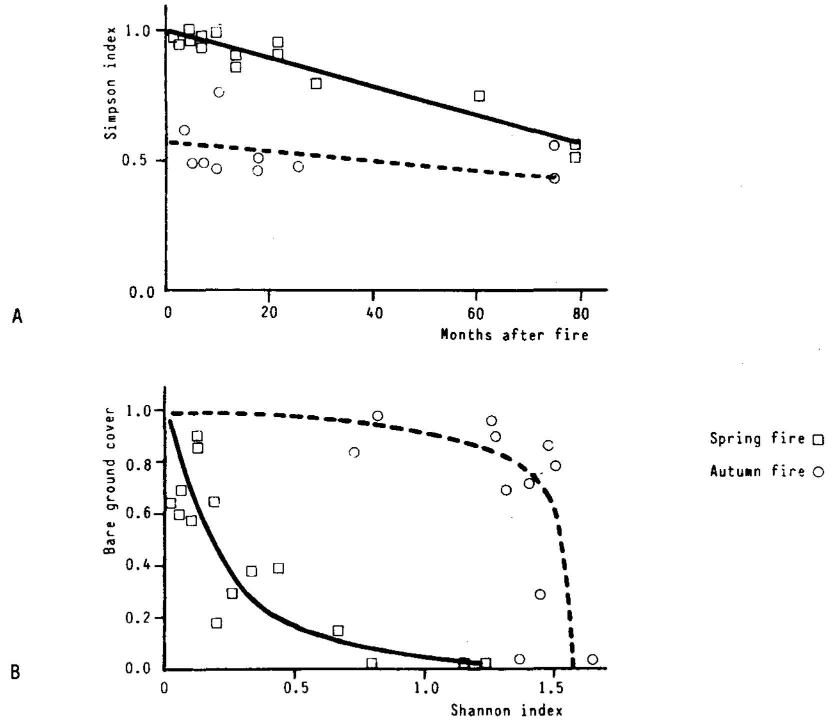

FIGURE 9.- A: Evolution of dominance. Simpson index calculated from cover data of woody species in *U. europaeus* communities (From CASAL & GARCIA NOVO, 1985)

B: Relationship between bare ground cover and diversity. Shannon index calculated from cover data of woody species in *U. europaeus* shrublands (From CASAL & GARCIA NOVO, 1985)

SYNTHESIS

The post-fire dynamics of an ecosystem is determined by the marked reduction in the biocoenosis structure and the alteration of the physical environment caused by fire. The degree of destruction depends on the intensity of the fire and the state of the ecosystem's biomass composition and distribution at the time of the disturbance. The type of vegetation and its condition influence both the intensity of the fire and the mode and rate of regeneration.

Where fire or other disturbances are recurrent, pre-fire species may be replaced by others, depending on their vital attributes (NOBLE & SLATYER 1977, 1980). A cycle of degradation and regeneration is repeated, in which the relatively stable intermediate phase is a low formation proper to the initial stages of an undisturbed succession (FRANQUESA 1986). Such communities are maintained by periodic burning : in absence of fire, they would undergo progressive colonization by trees while recurrent burning would cause rapid degeneration of vegetation and soil, accelerating erosion, hampering the regeneration of woody species and facilitating the establishment of greater numbers, with greater abundance, of herbaceous species and undemanding shrubs (TRABAUD & ALLAG 1976, DEBUSSCHE *et al.* 1980, CASAL *et al.* 1984).

In the case of communities dominated by Papilionaceae, post-fire dynamics fit CONNELL & SLATYER's (1977) inhibition model. Some communities, however, undergo an autosuccession similar to that observed in Mediterranean shrubland communities (GARCIA NOVO 1977, TRABAUD 1980, MARTIN VICENTE 1982, GRANADOS *et al.* 1986, TARREGA 1986). In some other examples, different stages can be distinguished in the secondary succession (DALDA 1978, DEBUSSCHE 1978, CASAL *et al.* 1984, CASAL *et al.* 1986b).

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Structure and organisation of a Quercus coccifera garrigue after fire.

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SUMMARY - After fire, recolonization implies a process of structural and functional organisation to reach a new state of equilibrium with the environment.

Vertical structuration can be analysed through the evolution of the total leaf area distribution in the vertical profile. The change through time in the vertical gradients of the leaf mean area and the stomatal density and size, among other things, are useful too to interpret this process.

Space occupation patterns can be analysed through the evolution of the foliage projective coverage and the species spatial dependence (regarded as contacts between pairs of species), considering a reticle covering the surface to be studied. A set of models may be useful to carry out this analysis. Changes in time of the basal area, density and stem diameter, and diametric classes distribution of stems are also useful to analyse and interpret the changes taking place after fire.

Some theoretical problems in relation to these different approaches are discussed in this paper. Examples focusing on a Quercus coccifera garrigue chronosequence are presented to illustrate some remarkable features.

KEYWORDS : Post-fire recolonization, chronosequence, garrigue, space occupation, vertical structure.

INTRODUCTION

The repeated fires in the Mediterranean ecosystems constitute an abrupt release of the energy which had been incorporated in the system in the form of organic structures. Burning causes a sharp loss of the structure and functional organisation of plants above-ground biomass. Recovery of the vegetation is much slower than its destruction and is related to environmental limitations and to the species capacity to face them up.

The capacity of Mediterranean species to overcome the phenomenon of fire is well known (Naveh 1974, 1975). The success of the strategies adopted (KEELEY 1977, KEELEY & ZEDLER 1978, MALANSON 1985, MALANSON & WESTMAN 1985), which can be briefly summarized as the capacity to resprout or to germinate from seeds, is linked up to the characteristics of the phenomenon and the circumstances in which it is produced (TRABAUD & LEPART 1981, TRABAUD 1984, KRUGER 1983) frequency, intensity (inversely related to the former), season of the year, predators...

When regeneration is by resprouting the organisation and structuration process of the community does not start from scratch. There is a subterranean space occupation which determines a speeding out of reorganisation and structuration of above- ground components.

The community organisation patterns are reflected in the horizontal as well as in the vertical structure. Therefore, it is interesting to make a detailed analysis of the main parameters which are modified in this process of secondary succession. An analysis of their spatial (horizontal as well as vertical) distribution through time has been done. As an example, we shall focus on a typical Mediterranean community: a Quercus coccifera garrigue. The study was made in different stands constituting a fire-induced age gradient. All of them are situated in the karst massif of Garraf near Barcelona (SABATE 1986, SALA 1986).

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VERTICAL STRUCTURE

The community growth implies a vertical process of structural and functional organisation, reaching a state of equilibrium with the environment and permitting an optimal acquisition and use of limited resources. At structural level, this organisation can be analysed through the evolution of the total leaf surface distribution in the vertical profile (leaves are the main production centres and, at the same time, the part of the plant where most of the water loss takes place). At a more functional level, the change in the vertical gradients of certain leaf morphological parameters through time, reflects the response and the degree of adaptation to external factors as well as to the microenvironment which is created within the community while it is maturing.

Leaf production and leaf area index (LAI)

After fire, the increase in coverage and biomass is logically accompanied by an increase in the total leaf area per ground surface unit. This increase is limited in time and, during the earlier stages of regeneration, it will be more marked in those communities dominated by resprouting species than in those dominated by seeding ones.

The characteristic decrease of the production/biomass ratio, found in any secondary succession process, is reflected in the vegetation by a decrease in the leaves/wood - or total-biomass ratio. In the Mediterranean shrub communities, which regenerate after fire by resprouting, this ratio decreases during the first 5-15 years of growth (RUNDEL & PARSONS 1979, Fig. 1), due to the large initial production of leaves.

Fig 1: Leaf/total shrub biomass ratio along a gradient of stand age since fire in a Quercus coccifera garrigue.

This tendency shows that leaf production is limited. In the Mediterranean ecosystems, which are submitted to water deficits in summer, and quite often, to a low availability of minerals, leaf production depends on these factors and on the investment required to obtain increments of these resources. This is why the most frequent LAI values in shrub communities generally fluctuate around 3 and why it is possible to achieve higher values (around 4) only in mesic sites (with higher water availability), where a lower investment in root materials is required. In any case the LAI increases, which represent an increase in production, require at the same time, a higher degree of shade tolerance. This tolerance represents a limit above which the LAI increases no longer signify an increase in production (MILLER 1983).

In the communities dominated by sprouting plants, the existence of a preformed root system which occupies the subterranean area means that the bare ground created by fire is soon recolonized by a low stratum of vegetation. From that time until the community has re-established itself and has grown in height, it is expected that a biomass or foliage redistribution would occur in the vertical profile. In other words, the lower leaves strata (those nearest the ground) will be progressively substituted by the higher strata (TRABAUD 1983, RUNDEL & PARSONS 1979).

Besides the implication that a determined LAI distribution through the vertical profile can have for the light regime, the annual water balance of the community can also be modified according to this distribution. In this sense MILLER & STONER (1979) verify, from simulations to estimate the temperature and transpiration of Ceanothus greggii and Lithraea caustica, that, maintaining the LAI constant and varying its vertical distribution only, the annual rate of transpiration increases when the leaves are concentrated at the bottom of the canopy or when the leaves are distributed in the canopy as it actually occurs. In these cases, a significant quantity of leaves remain protected from direct insolation. These leaves get less warm, and when water is available, transpiration is always possible until depletion of soil water occurs. In other words, fewer leaves are subject to drying out and the environmental control over the water loss may be less effective. Therefore, the accumulation of leaves in the higher strata ("heliophylum" structure) is associated with a greater environmental control over water loss.

As can be observed in Fig. 2, an inversion in the distribution of the leaf area

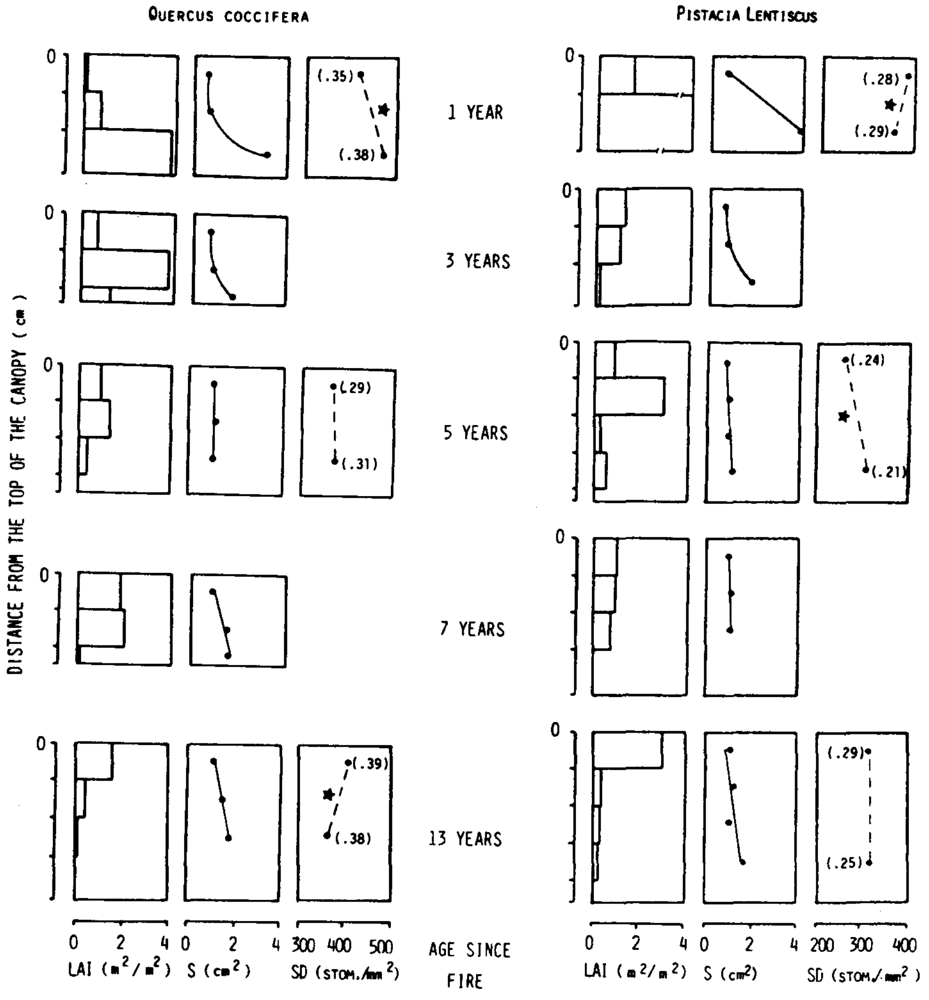

Fig 2: Vertical structure of Quercus coccifera and Pistacia lentiscus along a gradient of stand age since fire. Leaf area index (LAI) and leaf mean surface (S) are measured by 20 cm-thick canopy layers. Vertical profiles of leaf mean surface (solid lines) are theoretically extrapolated. Stomatal density (SD) is only measured at the uppermost and at the lower canopy layer. Between brackets, the percentage of the total pore area. (* : significant differences, $p < 0.001$).

occurs in Quercus coccifera as well as in Pistacia lentiscus, along the stand age gradient. It goes from an accumulation of the total leaf area in the lower strata at the earlier stages of development, to an accumulation in the upper layers as the community matures. It can be seen, in fact, that in a few years the community reaches its typically "heliophylum" structure in which upper stratum of leaves intercepts the major part of the incoming solar irradiance.

Morphological gradients

At the same time that the architecture of individual plants is being modified and the community continues to grow and becomes denser, the microenvironmental conditions inside are being modified too. An increase of shade occurs, which not only affects the LAI distribution, but which is also reflected in the vertical leaf morphological and anatomical gradients. The more the community grows and becomes denser, the more exposed leaves will morphologically differ from those which are in the shade, in the same way that sun leaves differ from shade leaves.

As SPECHT (1972) suggests it succession leads to a greater utilisation of soil water by the vegetation. Thus, taking into account that the vertical differenciation in the canopy is generally accentuated when there is a stress (strong winds, hydric stress,...), it is to be expected that the increasing conditions of summer water stress through time (until the community has established itself) will also favour vertical differenciation.

During the first year, when the community is still open, there are no microclimatic gradients in the vertical profile. However, vertical morphological gradients can be found between the earlier leaves grown after fire and those from the shoots which had sprouted later. This may be interpreted as a response to the disturbance suffered by the system. This is the case, for instance, for the leaf mean surface. The improvement in water relations and mineral availability which may occur after fire, can significantly favour the growth of the newly emerged leaves which are typically larger (ARIANOUTSOU-FARAGGITAKI & MARGARIS 1981, Fig. 2). In resprouting species the reserves obtained from the pre-formed root system can additionally favour the growth of the earliest leaves. With time, the effects of fire diminish and the vertical gradients of the mean leaf surface reproduce the microclimatic gradients which are established within the canopy (they become less pronounced). In Fig. 2 it can be observed that, as the community matures, the leaf mean surface gradients do not become more pronounced in response to the increasing conditions of shade inside the community. This fact shows that the vertical microclimatic gradients are very slight. This is because plants are small (1 m high maximum).

Other differential parameters between sun and shade leaves are the stomatal density and size. Both parameters are very sensitive to light conditions and water availability during leaf development. By increasing the light (BOARDMAN 1977) and decreasing the availability of water (GINDEL 1973) an increase in the stomatal density is noted, generally accompanied by a decrease in the stomata dimensions. As a general rule, the increase in stomatal density is proportionately greater (in the same species) than the decrease in stomatal size. This means that the total porous area per unit of leaf area increases with the stomatal density, and logically, this modifies the maximum conductance capacity.

Taking into account the stomatal frequency and size changes in response to the water and light regime, it is to be expected that when the community matures, a vertical differenciation occurs between both parameters as a response to the increasing conditions of shade inside and to the greater competition for water as the coverage and biomass increase. This differenciation is associated with a progressive redistribution of the foliage in the vertical profile. It can be very important for the global regulation of water use in the community, due to the direct involvement that the stomatal apparatus has in the control of water transport along the soil-plant-atmosphere continum.

As many authors have already noted, it appears paradoxical that in conditions where the risk of losing water increases, the stomatal density also increases. The first question which arises is what advantages can this fact represent for the control of water loss. In this sense it seems that an increase in the stomatal density is related to a higher sensitivity to changes in the environmental humidity (EL-SHARKAWY et al. 1985). A greater number of stomata per unit of leaf area allows an increase in the water conductivity of the epidermis. This means that in conditions of water deficit, the epidermis can dry out more rapidly (with the consequent activation of the hydroactive and hydropassive mechanisms of stomatal occlusion) without the water content of the mesophyle) being too affected.

As can be observed in Fig. 2, Quercus coccifera shows a direct parallelism between LAI in a given layer and stomatal density, throughout the process of the community structuration. This is logical since a greater leaf area index allows higher rates of evaporation and consequently new leaves will be in conditions of greater water deficit. The result, however, could be very effective since a greater stomatal density makes it possible (in favourable conditions) to reach higher conductance rates in those strata where most foliage accumulates. At the same time, it allows a more effective control of water loss in conditions of water stress, where the risk of water loss by transpiration is greater.

In contrast, in Pistacia lentiscus (Fig.2) the same parallelism is not observed. The vertical differenciation of the stomatal density does not seem to respond to any definite factor. This seems to indicate that Pistacia lentiscus exercises a less effective control of water than Quercus coccifera. This is in agreement with the secondary role of Pistacia lentiscus in the community (much less dominant).

HORIZONTAL STRUCTURE

Space occupation after fire can be studied from two different points of view. The first one would be to consider the system as above-gradient projection of the different species (considering the bare ground too). From this perspective, we can study the mosaic formed by the species and its evolution. The second perspective would be to consider space occupation from the point of view of population, by means of analysing the stems present in the community.

Analysis of the system

In a first approach, the presence of the species can be summarized according to their relative coverage (Fig. 3). In this example we are presenting, the coverage reaches a value of 90%

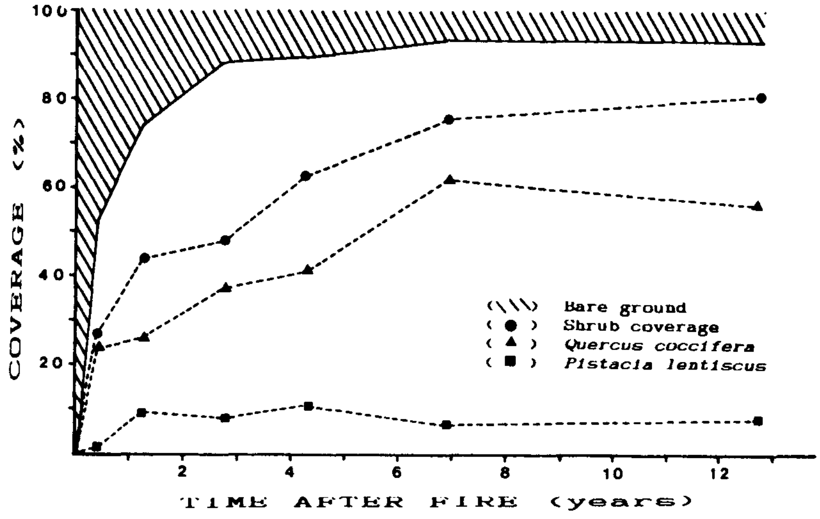

Fig. 3: Coverage (%) along a gradient of stand age since fire in a Quercus coccifera garrigue. The rapid occupation of bare ground in the first three years is basically due to Quercus coccifera. Pistacia lentiscus stabilizes with far lower alues (near 10%). The surface which is not occupied by shrub species is colonized by herbaceous species, mainly Brachypodium retusum.

between the second and third year. This high value is basically due to the dominant species of the garrigue: Quercus coccifera. This approach constitutes a filter in which a lot of information is lost. It considers the community as infinitely mixed, and does not inform neither about spatial distribution of the foliage projective cover through space (mosaic's dimensions or fragmentation), nor about the relative position of the species (spatial relationships).

In a second approach we can consider the sequential order in which individuals from different species are presented along a transection. From such a transection, it is possible to calculate the probability to move from species i to species j for (i,j) moving one step along the transection. The transition matrix between all the possible pairs of species can be regarded as markovian matrix associated to the spatial Markov chain. The application of the Markov chain model to space is equivalent to extrapolating the number of steps in the transection to infinity. If the species are arranged along a gradient, the two transition matrices obtained moving along the two different directions in this gradient will be different and so, the equilibrium coverage (which they tend to reach by infinite steps according to the Markovian model) will be different. As far as the assumptions of the Markovian model are concerned, a simple Markovian transection would not imply any local deviations or depressions in the diversity of a poor spatial structure, which could be translated into a "rectangular" spectrum of diversity (a very mixed or homogeneous distribution of species). On the other hand, a strong spatial organisation could be translated into a segregation and hierarchisation of the species distribution ("obliquous" spectrum) MARGALEF 1977, 1980, in prep.).

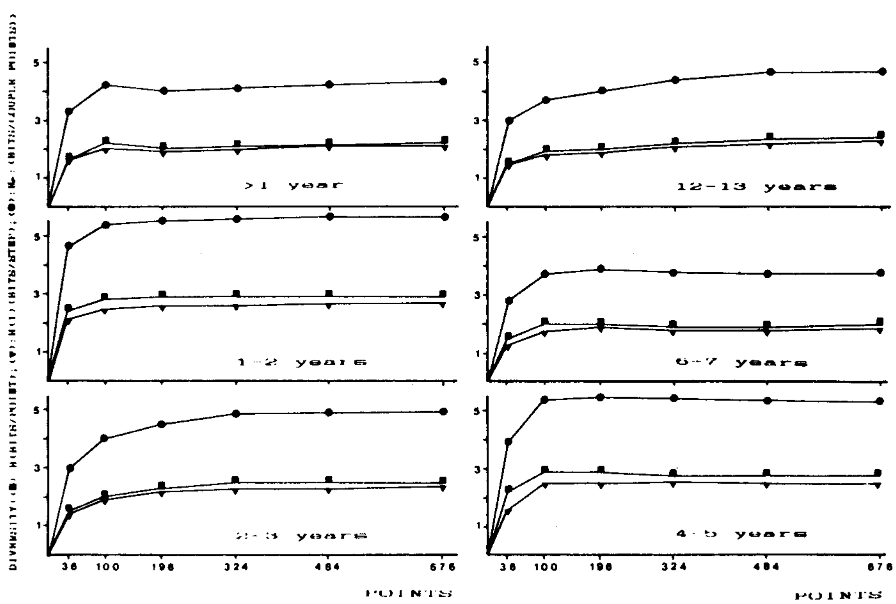

Fig. 4: Diversity spectra of stands with increasing age after fire in a Quercus coccifera garrigue. Age since fire:

$$H = -\sum_{i} P_i \operatorname{Log}_2 P_i \quad \text{(bits/step)}$$

$$H\,(1) = \sum_{i=1}^{n} P_i H_i = -\sum_{i=1}^{n} \sum_{j=i}^{n} P_i P_{ij} \operatorname{Log}_2 P_{ij} \quad \text{(bits/step)}$$

$$H_F = -\sum_{i} \sum_{j} P_i P_{ij} \operatorname{Log}_2 P_i P_{ij} \quad \text{(bits/couple-points)}$$

In a third approach, one can consider a reticle covering the surface to be analysed. One can obtain the four transition matrices along the four different directions in the space (GRACIA 1978, LABO 1983, SABATE 1986). If there are not any gradients in the area under study, there will not be any significant differences among these four matrices and so, all of them can be summarized in a total transition matrix. This matrix contains the total information of the spatial pattern. If we compare the evolution of such a matrix at different times after fire, it gives us a time and space integrated view. Fig. 4 shows the diversity spectrum of the stands studied. All of them present a very clear rectangular shape. This shape is due to the low structure of the mosaic of the garrigue after fire. This low structure results from the presence of a few dominant species, basically Quercus coccifera. Due to its resprouting capacity, the space distribution seems to be already established before fire (TRABAUD & LEPART 1980).

In order to analyse the information contained in this kind of matrix we can use difference conceptual tools (KHINCHIN 1957, MARGALEF 1957, 1977, PIELOU 1975):

The Shannon index (1), which is obtained from the coverage distribution of the present species. It gives us the average information associated to the species that occurs at each point of the reticle (in bits/point).

$$H = - \sum_{i} P_i \, \mathrm{Log}_2 \, P_i \qquad (1)$$

where P_i is the probability of appearance of the species i.

The spatial diversity (2), which is obtained when we consider the space transition matrix. It gives us the information associated to the displacement from one point to the next in the reticle.

$$H\ (1) = \sum_{i=1}^{n} P_i H_i = - \sum_{i=1}^{n} \sum_{j=i}^{n} P_i P_{ij} \, \mathrm{Log}_2 \, P_{ij} \qquad (2)$$

Finally, the diversity obtained from the matrices of frequencies (3 and 4), H_F (the sum of H and H (1)), which considers as events (or "species"), the pairs of species which can be formed in the space.

$$H_F = - \sum_{i} \sum_{j} n_{ij}/N \, \mathrm{Log}_2 \, n_{ij}/N \qquad (3)$$

Which is equivalent to:

$$H_F = - \sum_{i} \sum_{j} P_i P_{ij} \, \mathrm{Log}_2 \, P_i P_{ij} \qquad (4)$$

because:

$$P_i = N_i / N \quad \text{and} \quad P_{ij} = n_{ij} / N_i$$

The spatial diversity will reach a maximum when the spatial distribution of each species is independent from the others. The contacts between pairs of species are explained by their relative abundance. In other words, when $P_{ij} = P_j$, then $H = H(1)$. The higher the spatial dependency between species is the lower the spatial diversity will be, and the values of H_F (equations 3 and 4) and H (equation 1) will be closer. As can be seen in Fig. 4, a tendency towards segregation is not observed in the garrigue, and the diversities (1) and (2) are practically equal.

In addition to the research about spatial dependencies, we can summarize the structure of the mosaic with different parameters. The fragmentation index (5) (SABATE 1986), is an example that gives us a good idea about the degree of continuity in the coverage of each species: continuous or discontinuous patches differing in size.

Its evolution through time indicates how the community is recovering and reorganising after fire. Let us consider F_{ij} step frequency from i to j; F_{*j}, frequency with which j has been reached from whichever species; then $C_{ij} = F_{ij}/F_{*j}$ will be the probability that if the species "j" has been reached, the point of departure was the species "i". In this way, we will have a ponderation of the contact taking place between two species in space. The fragmentation is obtained by:

$$F = \Sigma\, C_{ij}\, P_i\, P_j / \Sigma\, C_{jj}\, P_j^2 \qquad (5)$$

Thus, F is an average measure of the dispersal or compactness of the patches of the species coverage, which appear during the recovering of the vegetation after fire. It is the ratio of interspecies and intraspecies contacts. Fig. 5, shows how during the initial stages, F has low values. This is due to the fact that the bare ground "event" give a very homogeneous character to the surface. However, it rapidly breaks up when the species begin to reappear on the surface (in the case of garrigue, this basically occurs by resprouting), and reaches a maximum value of fragmentation, between the second and the third year, when all the species of the community are at the same level. The closure of the community, due to the growth of the shrubs plants, reduces the number of species in the higher strata, and Quercus coccifera becomes the most abundant in relation to the other shrubs species. The growth of its patches gives a much more homogeneous character to the mosaic, and the index of fragmentation decreases again.

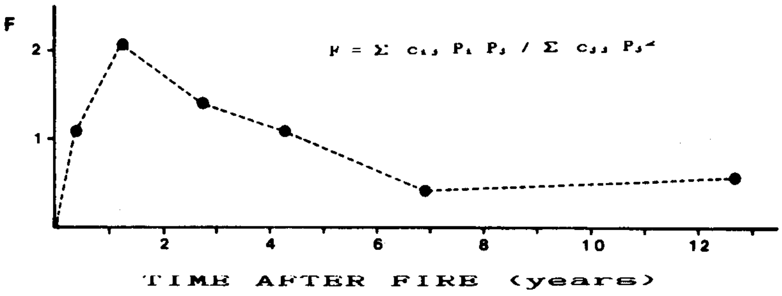

Fig. 5: Values of the fragmentation index "F", of the Quercus coccifera garrigue stands of increasing age since fire.

Population analysis

We can approach space occupation from the perspective of population, through the analysis of the stems which are present in the community. A first approach would be the evolution of their basal area. Observing how the basal area is distributed or partitioned between the species, one can see the role of each of them and their relative importance in relation to the structure of the community. Fig. 6 shows how, in the garrigue, the increase in the community space occupation is due to the growth of Quercus

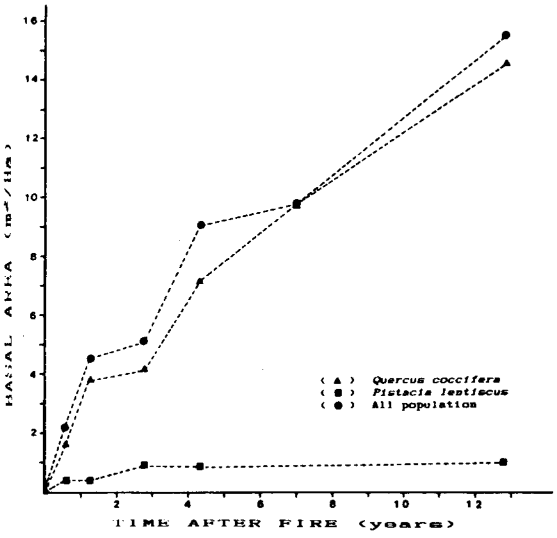

Fig. 6: Values of the basal area (m^2/Ha) of the Quercus coccifera garrigue stands of increasing age since fire.

coccifera. Pistacia lentiscus remains at lower values, and there is no increase in its basal area, despite the fact that individual stems become larger with time.

A second approach would be to analyse how many individuals or stems perform this basal area. This will give us the stocking density of stems per unit of ground area (see Fig. 7).

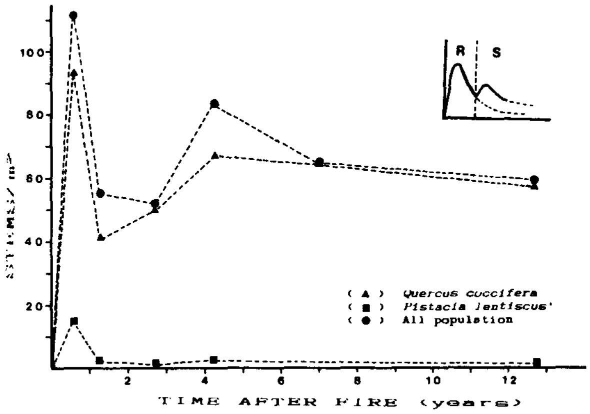

Fig. 7: Density in stems m^2 of the Quercus coccifera garrigue stands of increasing age since fire. An interpretation is presented at the upper righthand part of figure. The density fluctuations are divided into two phases. The first is dominated by the incorporation alone of stems of resprouting of individuals already present before the fire (R). Field observations suggest that some months later (between 33 and 55) an input of new individuals could take place (S). Because acorns are quite abundant, these new plants could probably be generated by germination.

An initial stage with a high stem density can be observed. This is due to the vigourous resprouting of the species, of Quercus coccifera in particular, which were present in the community before fire. A fall in density is noted during the first months: all the stems are not successful and there is a process of selection among the resprouts. An increase in density is noted between the second and the third year and especially between the 33rd and the 51st month. This is interpreted as an entry of new individuals which occupy new bare spaces in the area studied, or as the fact that new individuals appear below pre-existing sprouting individuals. After this rise in stem density a fall is again observed. Although it is not so pronounced now, it can be attributed to the growth of stems and their process of selection, as it is normal in other communities (HARPER 1984, WHITE 1980, LONG & SMITH 1984). Another question is whether these new stems come from the seed germination produced by stems of shoots already present in the community or if they come from new resprouts from individuals already present.

A third approach consists of analysing how this basal area is distributed between stems of different sizes, which can be grouped into diameter classes. In time, with the growth of stems, individuals of larger classes will appear more and more. The entry of new young individuals will also be reflected in an increase in the lower diameter classes. Fig. 8 shows the growth of young individuals between the 33rd and the 51st month, which corresponds to the increase in density.

Field observations often place these new stems below those already existing. Moreover, it has been observed that Quercus coccifera produces acorns before the 33rd month. Acorns were found on the ground after the 55th month. One can observe that many of these acorns are attacked by larvae of Coleopterae. All these observations suggest that this input of new individuals does not take place by resprouting only but probably by seeding too.

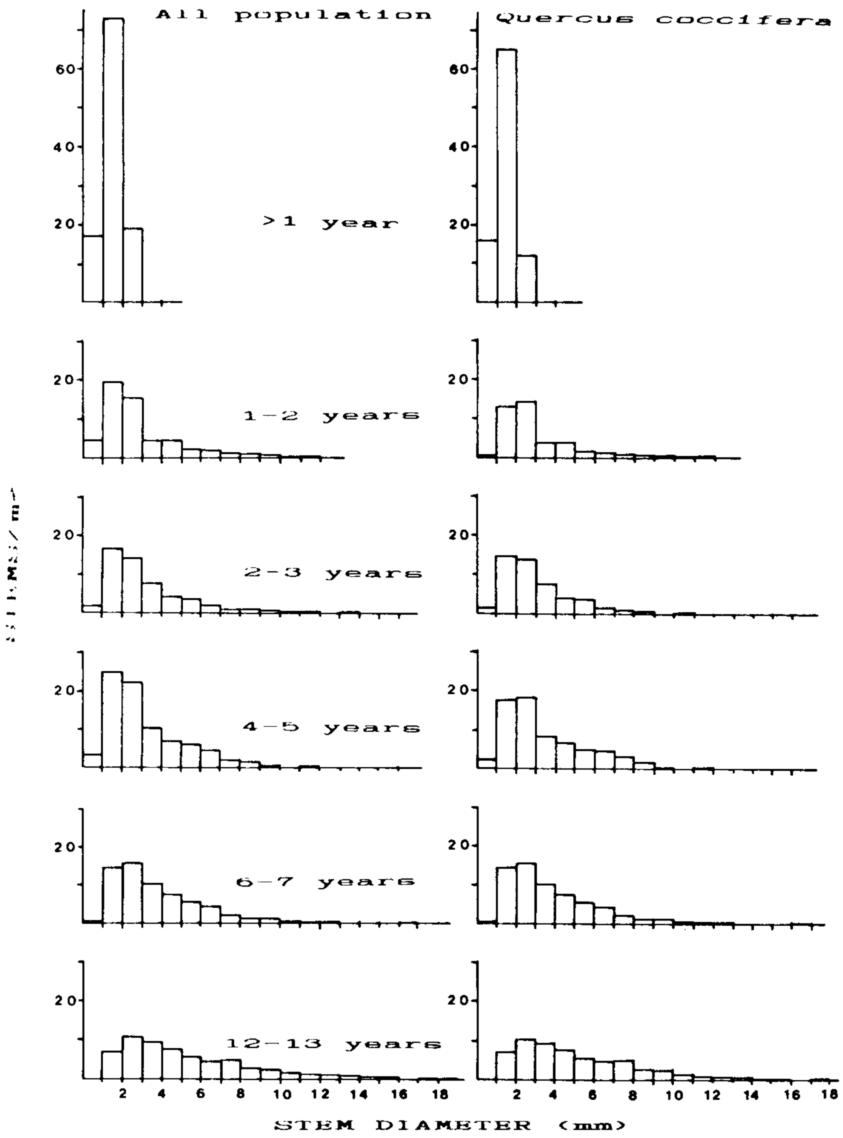

Fig. 8: Distribution of stem density in the Quercus coccifera garrigue of increasing age since fire. Diameter classes are presented in mm. The left-hand column represents the whole population of shrub stems and the right-hand column, the Quercus coccifera only.

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Effect of fire and associate heating wave on the physicochemical parameters related to the soil potential erodibility.

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SUMMARY - The common experience shows that the soils over which fire has passed appear largely degraded and profoundly eroded. The aim of this paper is an attempt to understand the primary reasons for this degradation and erosion by means of a study focused on the modifications induced by fire, and the associate heat wave on those soil parameters such as organic matter content, particle size distribution, aggregation, cementation, permeability, porosity and plasticity all associated with soil erodibility.

The high temperature developed during the fire indeed destroys all the surficial organic matter, produce a temperature gradient along the soil profile and induce an accumulation of hydrophobic substances in subsurficial layers in so reducing the water infiltration and increasing the runoff. The passage of a fire produces changes in soil texture increasing the sand and silt fractions and decreasing the clay fraction, in addition, it reduces the soil porosity and increases the bulk density.

To clarify the effect of the heat wave that accompanies the passage of a fire on the physico-chemical parameters related to the soil erodibility two soils were subjected to an artificial heating in controlled conditions.

Heating up to 220°C had only little effect on organic matter content, particle size distribution, plastic and liquid limits. Beyond this value the organic matter was burned up, the soils lost their plasticity, the sand fraction increased while the soil porosity decreased.

The aggregate stability of both soils increased continuously also when the generally acknowledged cementing agents, such as the gel forms, the organic matter and the organometallic cements were destroyed. This means that the heating promotes new forms of cementation and hardening very similar to soil laterization.

A field situation of this type, with single soil aggregates very hardened but not plastic is certainly not desirable because it may lead to massive landslides and produce more erosion than in a less cemented but more plastic and elastic soil.

KEYWORDS : Soil burning, soil heating, soil erodibility.

Vast areas of the world are burned annually by both natural and man-set fires. A large proportion of these areas is under forest cover extending from the equator to the limits of tree growth in both southern and northern hemispheres.

Burning is a widely practised method of land management in the humid and dry tropics. In both forest and savanna zones, burning is essential under traditional farming systems, to rid the land of the great amount of plant debris following a fallow period and to clear the land in order to obtain a suitable seedbed for cropping.

In Mediterranean areas the burning of stubbles and crop residues is a very common practice used for economic reasons to facilitate the seedbed preparation and particularly because many farmers believe they obtain a positive response from burning wheat stubble in terms of crop yields and reduced phytotoxicity arising hypothetically from the decomposition of the residues. This practice may be the starting point for dangerous forest fires where agricultural and forest lands are contiguous.

Another well-known and largely applied practice in the Mediterranean areas is the burning of brushland in order to convert the Mediterranean chaparral to grassland and to favour grazing.

In recent years, a new forest and brushland management technique has spread out throughout the world, especially in the Mediterranean areas : prescribed burning. It is used in order to reduce fuel and associate fire hazard.

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As mentioned above, uses of fire to modify the environment are numerous and it is not surprising considering how fire has accompanied the history of human settlement throughout the world. Fire is thus an essential factor in the development of various ecosystems. Man-influenced ecosystems, at least as we know them today, have been largely designed and modelled by fire. Nevertheless, the fact that fire is such an important factor within the ecosystems does not mean that it is always beneficial. The common experience shows for instance that the soils over which fire has passed appear largely degraded and profoundly eroded. Of course, this is an "environmental disease" that we must avoid. The aim of this paper is an attempt to understand the primary reasons for this degradation and erosion.

DIRECT EFFECTS OF FIRE ON SOIL

The most obvious effect of fire is the removal of the protective cover of vegetation and of litter from the soil surface. This increases sediment production because surfaces which were formerly protected are now exposed to all eroding forces, particularly to falling raindrops (splashing). However, this is not the only fire effect; changes in many soil physico-chemical parameters also occur and these may also play a significant role in increasing sediment production after fire. A study focusing on the modifications induced by fire and the associate heat wave on different soil parameters such as organic matter content, particle size distribution, aggregation, cementation, permeability, porosity and plasticity, all parameters associated with soil erodibility, seems however, largely appropriate.

EFFECTS OF WILD FIRES

Practically all the literature shows that the most striking effect of the passage of fire is on the soil organic matter content. Organic matter is generally acknowledged as the first anti-erosive agent because it binds the individual soil particles together, increasing aggregate stability and reducing soil erodibility (MARTIN et al. 1955, HARRIS et al. 1966, ALLISON 1968). The higher the temperature induced by the burn and the greater the frequency of fire, the greater is the change in organic matter that can be expected. Surface temperatures of 690°C have been found to destroy all surface litter and 99% of soil organic carbon (RAISON 1979). Soil temperatures during forest and brush fires may reach and go beyond this value at the soil surface, as reviewed by DE BANO et al. (1979). Grassland burning produces temperatures generally lower than 200°C but high enough to destroy the humic acids (WEST 1965).

Another very interesting aspect, partly correlated with the dynamics of organic matter, is the effect of fire on soil wettability. This topic has been largely investigated by DE BANO (1966, 1970, 1979, 1981) who observed that brush fires decreased water infiltration by producing a water repellent layer. On burnt areas, indeed, a water repellent layer is frequently found below and parallel to the soil surface whereas soil at the surface may be wettable. From laboratory and field observations, DE BANO and his staff have developed a tentative hypothesis regarding the formation of hydrophobic layers. The process depends on a group of organic chemicals that are hydrophobic. These chemicals derive from the decomposition of fresh organic litter (SHOLL 1971, 1975); in soil they may intermix with the soil mineral particles, clog the interstitial pore spaces and form a layer that is relatively impermeable to water (Fig. 1-A). When a fire occurs, the litter and the upper soil layer are exposed to very intense heating. Temperatures in the plant canopy may soar to 1100°C (COUNTRYMAN 1964), temperatures at the soil surface are lower but may reach 850°C (DUNN & DE BANO 1977). These temperatures totally destroy the non-wettable property of the soil (DE BANO et al. 1966) : at these temperatures the hydrophobic constituents of the soil organic matter partly evaporate and are consumed by the flame of the fire, or partly become even more fluid and are forced to move downwards along the temperature gradient developed in the soil profile until they meet cooler soil particles on which they can condense and form a new layer of accumulation of hydrophobic substances (Fig. 1-B). Therefore, after fire, a water repellent layer

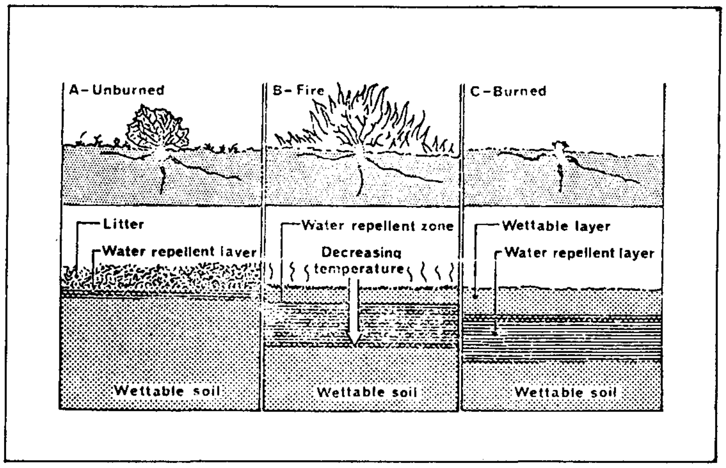

Fig. 1 - The formation of the subsurface water-repellent layer. (DE BANO 1981).

is present below and parallel to the soil surface on the burnt area (Fig. 1-C). In this situation, soils are very sensitive to erosion, particularly on steep slopes (Fig. 2). Indeed, the layered arrangement allows rainfall to infiltrate only to a limited depth before the wetting front reaches the water repellent layer. When the infiltration of water is impeded or temporarily slowed, the thin mantle of wettable soil becomes saturated. Water then flows laterally and runs off. The surface runoff provides the moving force for soil erosion and the flowing water takes away particles of soil from the upper wettable layer with some portion of the water repellent layer below. This mechanism can explain gully erosion and rill formation, commonly encountered in burnt soil (WELLS 1981).

In a recent paper we demonstrated that the hydrophobic substances behaved as cementing substances, with an aggregating power quite similar to that of organo-metallic cements (GIOVANNINI et al. 1983). Their disruption by fire at the surface decreases the aggregate stability of the surface layer which, as a consequence, becomes more erodible, whereas their accumulation in a deeper layer increases the possibility of rill formation and of gully erosion. All these indications were found in an experimental burning performed on a Sardinian chaparral, never burned before (GIOVANNINI & LUCCHESI 1983). The most noticeable results are reported in Table 1. After fire, there is a decrease in soil organic matter content in the upper layers of the soil connected with disappearance of resistance to wetting and with lowering of aggregate stability. The upper part of the underlying B horizons showed, on the contrary, an accumulation of organic matter and an increase in aggregate stability.

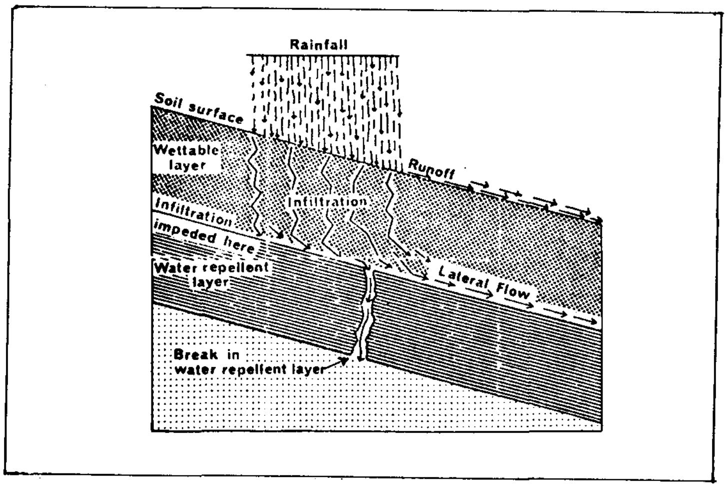

Fig. 2 - The effect of rainfall and the origin of runoff. (DE BANO 1981).

Table 1 - Soil organic matter (O.M.) contents and water stability index (W.S.I.) of aggregates.

Prof.	Horizon	Before fire		After fire	
		O.M.%	W.S.I.	O.M.%	W.S.I.
1	A_1	8.6	52.3	5.5	45.4
	B_{12}	2.2	15.6	3.0	19.2
	B_{22}	1.3	10.6	1.3	9.4
	B_3	0.9	7.2	0.9	8.0
2	A_1	7.5	50.5	4.8	42.8
	B_2	2.0	15.2	3.0	18.4

In opposition to previous findings, no clear evidence for soil hydrophobicity was detected, although the translocated organic matter presented some of the characteristics of hydrophobic substances, such as the extractability in benzene, the contribution to the aggregate stability and the particular shape of the differential thermal analysis (DTA) curve with a second exothermic peak at 440°C (Fig. 3) recognized as the diagnostic for hydrophobicity in a previous paper (GIOVANNINI et al. 1984). The discrepancy may be explained by the fine texture of the tested soil. Indeed, coarse textured soils can acquire a thicker and more substantial water-repellent layer than finer textured soils and the difference may be partly due to the amount of surface particle area covered with the translocated organic substances (DE BANO et al. 1970). The situation described here demonstrates that, such as in the Sardinian chaparral, the passage of fire had enhanced erosion hazard by weakening the internal resistance of the soil aggregates to the disruptive action of water.

In addition to the above-mentioned parameters, fire affects some other physico-chemical parameters related to soil erodibility such as particle size distribution, bulk density, and porosity.

Fig. 3 - DTA of the Sardinian soil.

A - Horizon A before fire. C - Horizon B_{21} before fire.
B - Horizon A after fire. D - Horizon B_{21} after fire.

The results of many studies indicate that the passage of fire produces a change in the soil texture of the surface layers, with an increase of the sand and silt fractions and an obvious decrease of the clay fraction (RAISON 1979, DONAHUE 1972). Once again, this is a negative effect. Pioneer works performed by LUTS (1934) and BOUYOUCOS (1935) suggested that erodibility was directly proportional to the percentage of sand and silt and inversely proportional to the clay fraction.

Following severe burns, it has been noted that macropore space at the surface is reduced and bulk density is increased (BEATON 1959, MOEHRING et al. 1966, TARRANT 1956). Compaction of surface soil layers can occur as a result of exposure to the beating action of raindrops (FULLER et al. 1955). The net results of both the reduction of large pore space and the compaction, is that infiltration rates decrease and at the same time, there is an increased susceptibility to erosion, according to the slope and the intensity and duration of precipitation. When this phenomenon is added to the above-mentioned formation of subsurface water repellent layer, erosion hazard may become really relevant.

EFFECT OF HEATING

All the reported results appear to be largely influenced by the burning conditions, by the development of fire and by the amount and the nature of fuel, that is by the temperature developed during fire at the soil surface. In order to ascertain the primary mechanism of the transformation induced in the soil and clarify the real effect of the temperature, we must consider that in a real fire two components may be recognized : the heat input and the supply of ashes. In the field, the effects of these factors are confounded making thus the identification of individual causes of changes in soil properties difficult. In order to ascertain the effect of heat input and of correlated temperature level, we have investigated the effects of an artificial heating, performed under controlled conditions, on the soil parameters related to soil erodibility.

For this study two soils were selected according to their original different texture : a Silty Clay soil and a Sandy Loam soil.

The arrangement of the heating programme was done on the basis of a previous differential thermal analysis (DTA) so that each temperature corresponded to genuinely occurring thermal reaction in the soil.

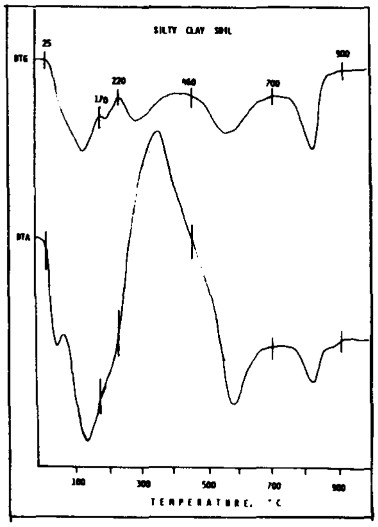

Fig. 4 - DTA & DTG of Silty Clay soil.

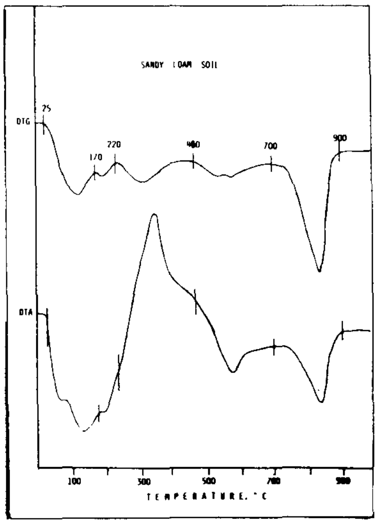

Fig. 5 - DTA & DTG of Sandy Loam soil.

The soil samples were heated in this way for one hour, in an oven or a muffle furnace, at the following temperatures : 25°C (control), 170°C (complete dehydration of the samples), 220°C (dehydration of the gel forms), 460°C (combustion of organic matter), 700°C (loss of OH groups from clays) and 900°C (decomposition of carbonates) (Fig. 4 and 5). The following parameters were determined : particle size distribution, soil porosity, organic matter content, liquid and plastic limits, water stability index of the aggregates and relevance of the organo-metallic cements.

As heating increased, soil sand content increased while silt and clay contents decreased simultaneously. These changes are attributed to the fusion of finer particles into sand-sized particles.

Soil porosity decreased sharply in relation to the dehydration of the gel forms in the case of the sandy loam soil, and to the combustion of organic matter in the case of the silty clay soil.

The effect of heating on soil organic matter was similar in both soils. There was practically no effect until 170°C, a little decrease occurred at 220°C; after 460°C, and more conspicuously over 700°C and 900°C, the soil samples absorbed even more water but did not form any plastic paste.

Water solubility index of the soil aggregates showed in both soils a continuous increase after all the thermal reactions and also after the combustion of the organic matter generally considered as the most important cementing agent.

The organo-metallic cements appeared efficient, and not influenced by heating up to 220°C, their removal always causing the same percentage decrease of the stability index; at higher temperatures they were effectively destroyed, as well as one of their components, organic matter. The other components, (i.e. polyvalent metals) so released, probably contributed to new forms of cementation because the aggregates increased their water stability index as well as the untreated aggregates. The results obtained at this point may be summarized in a uniform description.

As the heating of the soil progresses many of the generally acknowledged cementing agents, such as the gel form, organic matter and organo-metallic cements, are effectively destroyed. The stability of soil aggregates, nevertheless, increases continuously and this evidently occurs because of the development of new forms of cementation which replace the disrupted ones and harden the aggregates even more; but they become at the same time even less porous, plastic and elastic. A field situation of this type, with single soil aggregates very hardened but not plastic, is certainly not desirable because it may lead to massive landslides and produce more erosion than in a less cemented but more plastic and elastic soil.

CONCLUSIONS

In this paper we presented some evidence of how fire can damage soil properties and induce soil erosion. Nevertheless, the fact that man-set fires are so numerous and in constant increase indicates that farmers, timber managers and shepherds continue to find beneficial effects in using fire.

From the strict protection point of view, we must recommend the total exclusion of fire from the environment. Nevertheless, the recognition that the use of fire may be profitable for some groups of workers suggests that this recommendation will remain only utopian. So we must accept to live with fire but we must prevent that the use of fire becoming too dangerous and destructive. This may be achieved through new interdisciplinary studies aiming at obtaining a more precise knowledge of the necessary conditions to distinguish between the beneficial and the detrimental effects of fire on soil.

This may be important to include in the preparation of guidelines suggesting, to the various operators, times and methods for the practical and not-dangerous application of man-set fires in the various ecosystems.

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Wildfire effects on soil erosion.

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SUMMARY' - Wild fire has been causing soil losses by erosion for a very long time in the Mediterranean Basin. In fact, wild fire is an ecological factor common to all Mediterranean-type ecosystems and very often its effects result in a marked change in the soil hydrological parameters which last for several years.

Periodic and catastrophic floods and debris flows after wild fire in some regions of the Mediterranean Basin have shown the dramatic effect of large and severe fires. Regions on the border between Mediterranean and Atlantic climate, like Galicia (in northwestern Spain) in the past were generally free of fire and suffered little erosion. In recent years, however, wild fires have destroyed thousands of hectares of forest. These areas now largely unprotected and under a high rainfall climate are suffering noticeable soil losses.

The Mediterranean climate is in itself a very important factor in the erosion process after fire. Its tendency to concentrate the annual rainfall in a short time, generally in a few high intensity storms, is an adverse feature which enhances the erosive pulses. This paper summarizes available information about effects of wild fire on soil losses, reports some recent research and points out some research needs.

KEYWORDS : chaparral, erosion, soil losses, wild fire runoff

INTERCEPTION

Fire affects the erosion process in several ways. One effect is the destruction of the protective influence of the vegetative cover and the forest litter.

Studies on how the interception by vegetation is modified by fire are not very abundant. ANDERSON et al. (1966) found a 17 percent increase in stormflow after the burning of an Hawaii watershed covered by fern. The increase was associated with a reduction in interception storage. In the case of forest stands, both surface and crown fire may kill trees, but fire intensity is a major factor controlling how they die. Canopy removal can be very fast and can last for several months or years. Consequences are quite different. MITSUERA et al. (1984) recorded an annual interception of 3.6 percent of precipitation in a burnt forest of Pinus densiflora and 14 percent in the unburnt forest. The difference in interception corresponded closely to the difference in stream discharge. VEGA and BARA (in preparation) found differences between a Pinus pinaster stand burned by an intense crown fire and other stands burned by a moderate surface wild fire. Foliage was burned in the first case, and the annual interception the first year after fire was 14 percent. In the moderately burnt stand, interception was 21 percent. In the unburnt stand, it was 29 percent.

Dead leaves falling from scorched tree canopies can effectively protect the soil surface from direct impact of raindrops by making a mat covering soil exposed by fire. In their Pinus pinaster study, VEGA and BARA (above) found dramatic differences between burnt areas in the surface erosion generated by fire. In the crown fire stand, erosion the first year after fire was 15 times greater than in the scorched crown stand. In the same study on the effects of fire on erosion, two adjacent plots were laid out in a Eucalyptus globulus stand burned by an August wild fire. In one of them, where the fallen leaves from the scorched canopy were removed in September and October, soil losses were 5 times greater than in the plot with leaves remaining on the soil.

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EVAPOTRANSPIRATION

Removing the vegetation results in other important modifications. Soil evaporation is increased but obviously plant evapotranspiration (ETP) is drastically reduced. In many cases, the final result is that ETP is reduced in the first year after fire. Hence, the amount of water volume stored in the soil profile tends to be greater and more of the rainfall is available for runoff. Thus, KLOCK & HELVEY (1976) found a volume of additional water stored in the soil profile almost equivalent to the average annual runoff measured before an intense fire in 3 watersheds in Washington, because of the greatly reduced transpiration losses. Nevertheless, when fire stimulates rapid regeneration of the initial forest, there may be increases in water yield a few years after burning and yield may subsequently drop as the vigorously growing forest increases the demand for water. LANGFORD (1976) found this result in a burnt Eucalyptus regnans area.

SURFACE RUNOFF

If more surface runoff is generated after a wild fire, soil particle movement can be accelerated. Data show how overland flow increases markedly after fire. In Mediterranean chaparral areas of California, RICE (1974) observed that overland flow is generally less than 1 percent of precipitation. The year after fire, he found that flow averages were of 10 to 15 percent of the precipitation, ranging up to 40 percent. ANDERSON (1949) revising data from different authors on changes in runoff after fire in chaparral, found that runoff was between 1.35 and 14.8 times greater in burnt than in unburnt plots of either Manzanita (Arctostaphylos sp.), Manzanita and oaks or Manzanita, oak, pine and shruboak vegetation, but there was no significant difference in runoff from sparsely covered Chamise plots (Adenostoma fasciculatum) before and after fire. WELLS (1981) also indicated increases in surface runoff on burnt plots of chaparral the first year following fire; runoff was 7 times greater in burnt than in unburnt plots. SETTERGREN (1969), summarizing the data from several authors, pointed out that fire effects on surface runoff appeared to increase with annual precipitation and to be greater on the heavier textured soils. Conversely, as runoff and slope steepness of the unburnt plot increased, response of the site to fire relative to the unburnt control decreased. Regardless of this relative decrease, total annual runoff increased after fire. Nevertheless, WRIGHT et al. (1976) found that overland flow after an intense prescribed burning on six watersheds was directly related to slope steepness but was not affected by fire on level watersheds. On moderately steep slopes, flow was greater on burnt than on control plots for up to 18 months after fire, and on very steep slopes it was still greater than controls after 30 months. WRIGHT et al. (1982) showed how annual overland flow increased from 0.36 - 1.83 mm on untreated watersheds to about 25 mm for the first year after burning. In Western Montana, DEBYLE & PACKER (1972) observed that surface runoff from snowmelt was up to 8 times greater from logged and burnt plots than from unburnt ones. MEGINNIS (1935) found an overland flow 8 times greater on annually burnt plots of scrub oak than on unburnt.

VEGA and BARA (in preparation) found that the ratio of annual surface runoff to precipitation in an unburnt Pinus pinaster stand was only 0.31 percent. After an intense fire (about 5000 Kcal.$m^{-1}s^{-1}$), this ratio was 8.5 percent, i.e. 21 times greater the first year following fire, and 3.75 times greater four years after fire. After a moderate fire (about 600 Kcal.$m^{-1}s^{-1}$), this ratio was 11 times greater than in the unburnt control the first year following fire and 4.5 times greater the second year.

Depending on the kind of prevalent erosive mechanisms on each site, runoff works in different ways. Sometimes, an increased overland flow does not directly increase sheet erosion. Sometimes, an increased flow does provoke landsliding and falling of dry ravel deposits which were previously accumulated near or on the stream channels.

BURNT SOIL ERODIBILITY

Soil erodibility can be altered for several reasons. Fire consumption of organic matter is partially related to its intensity, especially by the persistence of the glowing combustion phase. Organic matter plays an important role in soil by improving water holding capacity, infiltration rate, and aggregate stability, all related to soil erodibility.

In laboratory tests, DURISCOE & WELLS (1982) showed that fire temperatures can change the particle size distribution of certain chaparral soils. Temperatures above 400°C caused an apparent reduction in the clay fraction of certain soils. ALMENDROS et al. (1984) found that in a forest soil subjected to various temperatures in a muffle furnace, the clay percent dropped with higher temperatures. BARA & VEGA (1983) also measured lower clay contents in 42 forest soils affected by wild fire in northwestern Spain. DIAZ FIERROS et al. (1982) found a coarser texture in burnt forest soil in the same area.

Many of the factors which affect infiltration rate (like vegetative cover type, percentage of ground cover, percentage of organic matter and organic layer depth) may be drastically affected by wild fire. There are experiments which showed that repeated burning decreased infiltration rates (AREND 1941, AUTEN 1934); while other authors found no change in this rate (VEIHMEYER & JOHNSON 1944, HODGKINS 1957) or little effect (BURNS 1952).

Fire has been shown to create water repellent layers in the soil. Initially the phenomenon was observed in chaparral areas of California. These layers, which developed in the first centimetres below the soil surface, preclude infiltration. Thus, when the soil mantle above this repellent layer becomes saturated, there is an increase of overland flow and erosion as this mantle is carried away by surface runoff. Water repellency problems have been extensively studied (KRAMMES & DE BANO 1965, DE BANO & KRAMMES 1966, DE BANO et al. 1976). DYRNES (1976) in Oregon observed a thick water repellent layer in the upper 23 cm for 5 years following wild fire. CAMPBELL et al. (1977) found a similar layer, persisting for 4 years in a burnt area in Arizona. Infiltration was reduced from 68 mm/hr in an unburnt site to 26 mm/hr on severely burnt sites. In Europe, GIOVANNINI & LUCCHESI (1984) found the same problem in burnt forest soils. They are studying the organic compounds that appear to cause repellency.

The effect of burning on soil structure is unclear. Some authors claim that aggregation is favoured (SCOTT et al. 1956, TARRANT 1956, LEBORGNE & MOUNIER 1959, IBANEZ et al. 1983) while others reported a slightly diminished stability only (SAN ROQUE et al. 1985, GIOVANNINI & LUCCHESI 1979, GIOVANNINI et al. 1983, DE BANO et al. 1979, DYRNESS & YOUNGBERG 1957). In a study with four burnt soils subjected to the action of a rain simulator and the two aggregate stability tests of HENIN and EMERSON, DIAZ FIERROS et al. (1987) measured an increased stability of the burnt soils in the rain simulator and HENIN's test; there were no significant differences with EMERSON's test. On the other hand, splash losses measured in this experiment were greater in the burnt soils; This is probably due to the sensitivity of the ash layer to the impact of raindrops. Conversely, VEGA & BARA (in preparation), using the HENIN test to study the aggregate stability of 42 burnt and unburnt forest soils in the same area, in northwestern Spain, found a lower stability in recently burnt soils. This effect was generally attenuated after two years following wild fire, but burnt soils affected by a new fire exhibited a new drop in stability. In another experiment, (DIAZ FIERROS et al. op cit) in which samples of soils were heated in a muffle furnace, stability of organic matter measured with a rain simulator increased slightly up to 110°C but declined sharply with further increases in temperature. These authors pointed out that their results are in agreement with those of ALMENDROS et al. (1984), who found that stability and maturity of organic matter peaked between 100°C and 160°C. They concluded that the differential response of soils to the two kinds of erosion and the non linear variation of stability with temperature may perhaps partially explain the apparent discrepancies found in the literature.

SOIL LOSSES

Soil losses after wild fire have been reported by many investigators. It is clear that the effects of wild fire on flooding and sediment yield under unfavourable climate circumstances are common in Mediterranean ecosystems.

As WELLS (1981) noted, the so-called fire-flood sequence is well documented in California's chaparral. ANDERSON (1949) reported that 41 watersheds in southern California produced sediments at an average rate of 1.77 to 28.8 m^3/ha per year. After one fourth of the watersheds were burnt, the estimate flow of sediments from a 100-year flood was 29.2 to 333.3 m^3/ha.

SINCLAIR & HAMILTON (1955) estimated erosion after a wild fire at San Dimas

Experimental Forest at 162.4 m^3/ha. The normal erosion rate for that drainage is about 5.8 m^3/ha per year. ROWE et al. (1954) estimated that sediment yields in the first year after fire can be as much as 35 times the normal. SCOTT (1971) observed that more than 590 m^3/ha was measured in individual debris basins after a flood following a wild fire.

DAVIS (1977) reported that sediments make up to 25 percent of the total flow before fire and 30 to 60 percent after fire. Many post-fire flood events are sediment flows rather than water runoff.

WELLS (1981) reported that the sediment yield increased from 0.67 m^3/ha per year in unburnt plots to 19.1 m^3/ha the first year after fire. WELLS (1983) gave the rate of 658 tonnes/ha per year for post-fire sedimentation compared with the normal rate of 226 tonnes/ha per year. WELLS (1982) reported that sediment production on burnt areas was 278 m^3/ha per year compared with the normal rates, which ranged from 16 to 59 m^3/ha per year. RICE (1963) estimated the erosion rate of 118 m^3/ha during the 24 months following the fire, i.e. about 6.5 times the normal rate.

Dry ravel is sometimes the predominant erosive process. KRAMMES (1960) estimated that such erosion, directly related to fire, ranged from 5 to 55 tonnes/ha for the first year, 10 times the preburn rate. About 89 percent of this material came from surface sloughing during the dry season. Many times, most of the increase in sediment does not result from fresh erosion but rather from remobilization of existing deposits. Like in California, the chaparral of Arizona also suffers periodic destructive fires. GLENDENING (1959) reported 140 tonnes/ha over a 6 month period after an intense fire. PRICE (1962) found 77.5 tonnes/ha over a 21-month period, i.e. about 100 times the normal rate before fire.

In Nevada, a watershed severely damaged by wild fire produced 8.75 tonnes/ha of sediment after a single storm, (COPELAND 1965).

In other habitats, a severe fire on a Pinus ponderosa watershed in Arizona produced 74 tonnes/ha of sediments after a single storm (RICH 1962). In another P. ponderosa stand in Arizona, CAMPBELL et al. (1977) estimated losses of 4.30 tonnes/ha for the first year after a fierce fire. MEGAHAN & MOLITOR (1975) studied the sensitivity of soils derived from acid igneous parent material to surface erosion and showed how the erosion rate is controlled by fire intensity. On two small adjacent watersheds covered with P. ponderosa, a high intensity wild fire on one of them caused soil losses of 1.30 m^3/ha for the first year, whereas a moderate fire on the other caused no soil loss.

HELVEY (1980, 1985) in Washington found that 1.4 to 3.8 tonnes/ha were lost the first year after a very severe fire, which was about 175 times more than in the control. LEITCH et al. (1983) working in a Eucalyptus forest burned in Australia, estimated the soil losses at 22 tonnes/ha for the first year following fire. In the same habitat, BROWN (1972) recorded 0.23 to 22.99 tonnes/ha of suspended sediments per day, 100 times more than the control. In the Rocky Mountains, POTTS et al. (1985) estimated that the soil losses after wild fire ranged between 0.35 and 6.5 tonnes/ha per year; the control sediment losses were 0.35 to 2.7 tonnes/ha per year.

In northwestern Spain, DIAZ FIERROS et al. (1982) measured soil losses in 29 small plots (20 m^2). An average of 56 tonnes/ha of soil loss was found. In the same region, on two plots of 0.04 ha installed in a Pinus pinaster stand affected by an intense fire, VEGA et al. (1982) measured soil losses of 22 tonnes/ha the first year after burning, compared with only soil traces lost from the control site.

For other Pinus pinaster stands burned by a moderate wild fire (VEGA & BARA in preparation), sediment yield was only 1.50 tonnes/ha for the first year following fire. The same authors, studying Eucalyptus globulus plantations burned by a wild fire, found soil losses ranging from 0.14 to 4.80 tonnes/ha per year. In Valencia (Eastern Spain) SAN ROQUE et al. (1985) estimated soil losses after three wild fires to range between 16 and 76 tonnes/ha per year.

RECOVERY

Recovery depends on several factors. Among them, fire intensity and speed of plant regrowth after fire are the most important factors, but rainfall intensity and distribution after fire are critical too.

Generally, fire effects on erosion do not last long. However, ROWE et al. (1954) estimated that 8 to 10 years were necessary for watersheds to return to pre-fire conditions.

Anderson & Trobitz (1949c) predicted differences in deposition until 15 years after fire. A similar period of time is indicated by TIEDEMANN et al. (1979). Nevertheless, BROWN (1972) indicated that 4 to 5 years were necessary in *Eucalyptus* watersheds. WRIGHT et al. (1982) showed stability was reached in 4 to 5 years after intense prescribed burning in Texas. WELLS (1981) found that 3 years were necessary for the chaparral in California. VEGA & BARA (in preparation) found that recovery in northwestern Spain took only 1 to 2 years. DIAZ FIERROS et al. (1987), working in the same region, measured negligible soil losses 1 year after fire.

RESEARCH NEEDS

To a great extent, the literature indicates that processes, factors and mechanisms which affect soil erosion are not well understood. Some of the research needs are pointed out :

- A better understanding of relations between fire intensity and soil losses.
- A more detailed investigation on the effects of fire on aggregate stability.
- A research effort to develop models which will accurately predict erosion after fire.
- More studies on nutrient losses from areas disturbed by fire.
- Research on the influence of repeated fires on soil degradation and desertification.

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ECOLOGIA MEDITERRANEA - Tome XIII (fascicule 4) - 1987

Fire, bird conservation and land management in the North-Mediterranean area.

R. PRODON*

SUMMARY *Despite the common perception of wild fires as a major ecological catastrophe, their effects on the avifauna are not uniquely negative. Concerning the protection of the most typically Mediterranean bird species in Southern France, the progressive disappearance of the herbaceous formations from the landscape might represent in the future a more serious problem. The maintenance of these formations (for example by prescribed burning), facilitating the fight against large wild fires, would reconcile the exigences of avifauna and forest protection in Mediterranean fire-prone areas.*

KEYWORDS : *fire ecology, bird protection, postfire successions, Mediterranean landscape.*

INTRODUCTION

The Mediterranean Basin was the scene both of the first technical revolution of mankind, the domestication of fire (with one of the oldest fire sites on the French Riviera) and of the appearance and development of scientific thought. Is it not a paradox (the first of several we will find in this paper) that the bulk of scientific research on the ecology of fire was conducted outside this area ?

In contrast with the large number of ecological data accumulated in North America, South Africa and Australia since the thirties, we enter the last quarter of the 20th century almost without any data on the fire impact on fauna in the Mediterranean Basin. That is why, stimulated by the example of L. TRABAUD, we began to study the impact of the wild fire of July 1976 (Massif des Aspres, Pyrénées-Orientales) on the avifauna. This research was progressively extended on both sides of the crystallo-metamorphic chain of Les Albères, and organised as a collective research including other groups of animals (small mammals, soil arthropods, ants...). The first results being already available (PRODON & FONS 1980, PRODON & LEBRETON 1983, PRODON et al. 1984a, PRODON et al. 1984b, 1987, ATHIAS-BINCHE et al. 1987) and waiting for a general synthesis of our 10-year study still to be made, we chose to develop here, after a short summary of our results, some aspects concerning the interplay between fires and avifauna conservation (see also PRODON 1987).

I - QUANTITATIVE EVALUATION OF THE IMPACT OF FIRE ON THE AVIFAUNA

Before any ecological evaluation, the first problem is to choose one or more criteria. There are of course several possibilities, both at the community level or at the species level.

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1.1. Bird density

The census methods used until now in our programme do not provide absolute densities, but only relative abundances. Nevertheless, some rough estimates can be made. The postfire variations of bird numbers are complex, and differ from one formation to another. In grassland, there are no clear variations, perhaps a slight increase. In the maquis, a decrease the first year is followed by values above normal from the 2nd year onwards. In Quercus suber forests, conditions quantitatively not far from normality are reached as early as the second year. In Quercus ilex forests, a decrease the first year is followed by a transitory increase the 2nd to 4th years (but always below the normal values) then by a decrease. During the years to come, we hope to obtain some absolute densities in selected plots of burnt forests using the time-consuming mapping method.

1.2. Avifauna richness

The avifauna richness, i.e. the number of bird species in a given area, is often intuitively supposed to suddenly decrease after fire, then to increase slowly, to reach the normal number again only after a long time. In fact, in our area, the variations of richness are somewhat parallel to those of abundance. In the burnt heather maquis, between the 2nd and the 4th year after fire, the number of species can be 50% higher than in the control. In the burnt cork oak forests, the number of species is close to normal from the second to the third year, but decreases afterwards. In the burnt holm oak forest, richness seems to remain below normal for several decades.

1.3. Avifauna composition

By far the most important aspect of the impact of fire on avifauna is the turn-over of species which results both from the fire itself and from the postfire regeneration of vegetation. A certain constancy of the global quantitative parameters of the avifauna may correspond to a nearly complete substitution of species.

There are nearly as many types of postfire recolonisation patterns as there are different species. Let us examine a set of 5 species taken as examples (Fig. 1); the curves refer to all the plant formations (grassland, maquis, forest) of the area taken as a whole, the relative weights of the different formations being equalized by calculation:

- Oenanthe hispanica : this open landscape species logically increases its frequency after fire as it is able to colonize some burnt maquis previously unsuitable because they were too thick.
- Sylvia undata and S. melanocephala : these species typical of the Mediterranean shrubland disappear almost completely the first spring after fire. Later, in the regeneration maquis, they can widen their distribution so as to exceed their pre-fire number.
- Fringilla coelebs : for this forest species, the apparent stability (if not slight increase) of frequency after fire is highly surprising. This can probably be explained by a strong philopatry of this bird. Later on, the species, which usually forages on the forest ground, cannot withstand the closure of the secondary undergrowth and tends to disappear.
- Erithacus rubecula and Regulus ignicapillus : immediate (R.i.) or deferred (E.r.) disappearance, without recolonization during the time of the study.

1.4. Postfire successions

The interspecific differences in the response to fire result in the phenomena of succession. As it is impossible to keep in mind or to represent a general picture of the year to year variations of some 50 to 60 species in the about 80 burnt stations of the study (plus the corresponding controls) during 10 years, it is necessary to have recourse to synthetic indices.

A previous study of the vegetation gradient in the Mediterranean zone of the Albères chain (Pyrénées-Orientales), a gradient resulting from the spontaneous evolution of the vegetation after land neglect, led to index any bird species of the area on a structural axis representing the transition from grassland to forest (PRODON & LEBRETON 1981).

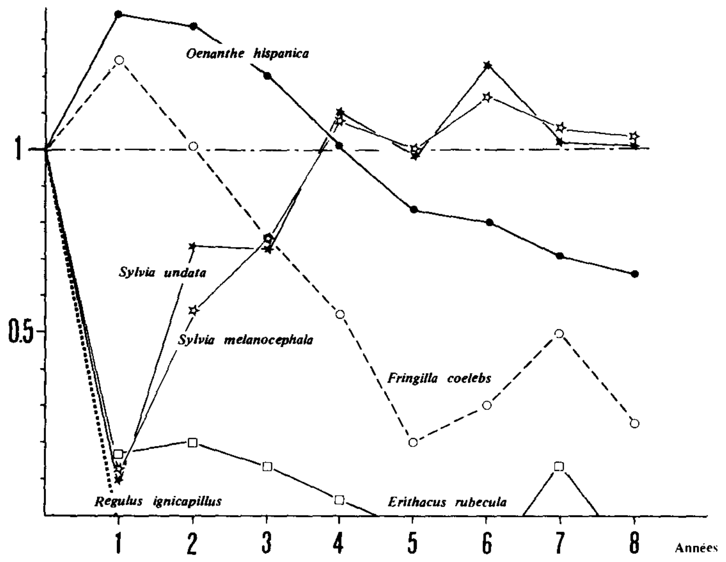

Figure 1 - Postfire relative variations of frequency for 6 common bird species of the Mediterranean zone of the Albères chain. (prefire frequency = 1).

This indexation, and its correlation with the vegetation cover profile, resulted in two indexes, IAG (Index of Avifauna Gradient) and ISG (Index of Structure Gradient). The former allows to index any "relevé" (or station) on the axis from its bird species list, the latter allows to index the same "relevé" (or station) from its vegetation profile. By definition, under undisturbed conditions, ISG must be approximately equal to IAG apart from random variations. In this way, the specific composition of the avifauna and the structure of the vegetation can be numerically expressed with the same units. In terms of "maturity", i.e. degree of evolution towards climatic stages, the higher the index, the more "mature" the system.

Example : impact of fire on the avifauna of two Mediterranean forests (Fig. 2).

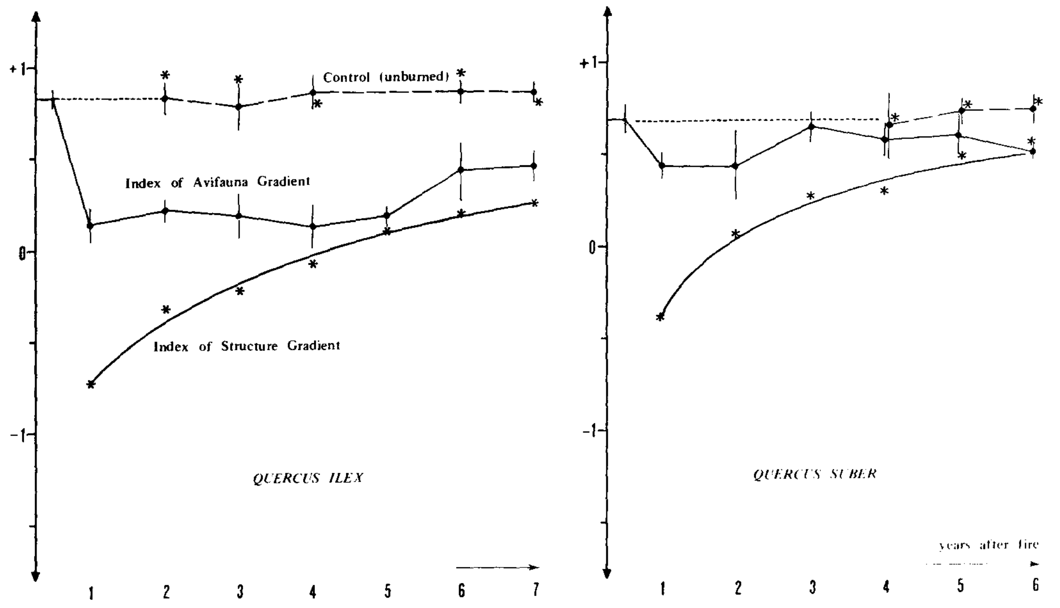

Figure 2 - Variations of the avifauna composition after fire in two different types of Mediterranean forests, compared with the regeneration of vegetation by using gradient indices.

The above indices were calculated every spring the first years after fire in two sets of stations, in cork oaks (Quercus suber) and holm oak (Q. ilex) forests respectively. In the holm oak forests, the destruction of the canopy by the fire (even if many dead branches persisted) resulted in the sudden and logical decrease of ISG. The decrease of IAG is surprisingly smaller because of :

- the persistence of several forest bird species, despite the extreme disturbance of the biotope (philopatry or site tenacity related to the territorial behaviour of birds);
- a certain reluctance or inability of the open vegetation bird species to colonize immediately the burn.

This double phenomenon results from a sort of inertia in the behaviour of one component of the system (i.e. the avifauna) to adapt to the variations of the other component (i.e. the vegetation structure). Despite this inertia, the graph clearly shows that the return to a normal forest avifauna is a very long process which will probably require several decades.

The same data processing applied to the cork oak samples show that, in this case, the inertia is greater and that the return to a sub-normal avifauna is much faster. This is obviously correlated with the great speed of canopy regeneration of this fire-resistant tree.

1.5. The rarity of the species : an important criterion for the conservationist

Fire induces important species changes in the avifauna of most of the woody Mediterranean formations. Can these changes be qualified as positive or negative from the protectionnist's point of view ? One of the most important criteria to be considered as regards species protection is rarity. There are at least two different aspects for species rarity : numerical rarity, and geographical rarity. If the former is very difficult to measure for birds (except for some colonial or large-sized bird species), the latter is easy to quantify since distribution indices have been published in atlases. For any species i an index of rarity can be defined as follows :

$$IR_i = 1 - n_i/N$$

where N is the total number of geographical units covered by the atlas and n_i is the number of units where the species i is present. The index of rarity of a "relevé" is the average of the IR of all present species.

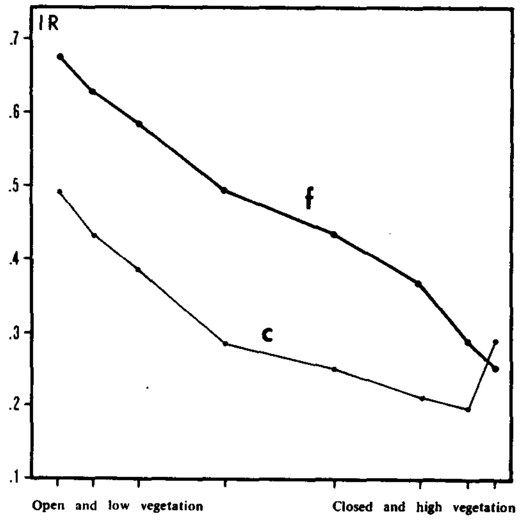

Figure 3 - Variations of the average index of avifauna rarity along a vegetation structure gradient in the Mediterranean zone of the Albères chain (f: index of rarity calculated from the French atlas of breeding birds, c: index of rarity calculated from the Catalan atlas).

Let us first consider the variations of the mean rarity of the avifauna along the structure gradient. The calculations of IR area come from the data of Yeatman (1976). The figure shows a monotonic and highly significant decrease of IR along a vegetation gradient, from the rocky and dry grassland to the closed holm oak forests. This decrease is essentially due to the fact that the evergreen Mediterranean forest shares the same basic avifauna as the other forest types of the western Paleoarctic, and that most of the Mediterranean species are restricted to open vegetation (BLONDEL 1986). If the same calculations are made from the data of the Catalonia atlas (MUNTANER *et al.* 1983), a rather similar graph is obtained, although the IR values are lower.

Does the opening of the vegetation by fire induce a similar increase of mean rarity ? The comparison of 46 point counts in burnt cork oak forest with 21 in unburnt stands confirms (t test; $p < 1\%$) the inverse relationship between average bird rarity and vegetation cover. Similarly two of the rarest birds in France, namely the Black wheatear (*Oenanthe leucura*) and the Thekla lark (*Galerida theklae*) are restricted to the open formations; they cannot withstand the closure of the scrub, but rapidly colonize the burnt maquis.

These observations can be summarized by a paradox : the mature Mediterranean forest of *Quercus* has now become a rather scarce formation in continental North Mediterranean areas, and thus needs careful protection. But nevertheless its avifauna is composed of very common European bird species. In contrast, some of the barest, rockiest and driest Mediterranean landscapes, which are generally perceived as degraded and desolate, are the refuge of some of the rarest species of the French fauna, not only of birds, but also of other animal groups (e.g. reptiles, ants, coleoptera, orthoptera, ...).

II - CURRENT TRENDS OF LANDSCAPE EVOLUTION (Fig. 4).

As in most of the hilly and rugged areas of North Mediterranean countries, the abandon of cultivation and cattle breeding results in a general progression of woody vegetation. "This regeneration is slowed down, but not arrested, by forest fires" (LE HOUEROU 1981). On a crystallo-metamorphic substrate as in the Albères chain, the colonization of herbaceous formations by shrubs (*Genera Ulex*, *Cistus*, *Daphne*, *Calycotome*...) is rather rapid, and the very low density of grazing cattle is unable to prevent the closure of the maquis, except on some small remaining areas (e.g. resting places). At the same time, the successive fires inexorably reduce the forest areas.

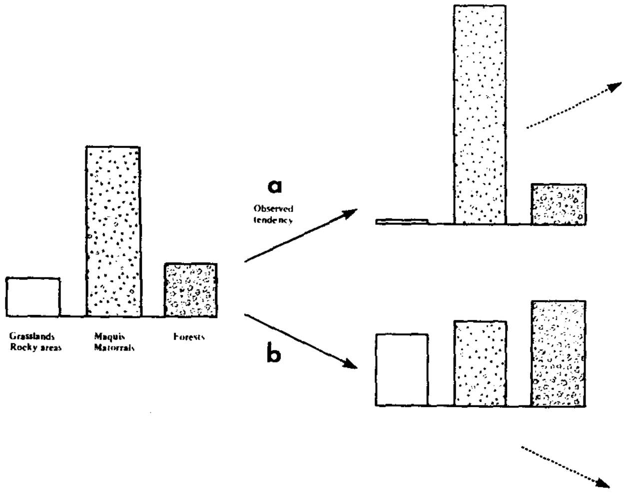

Figure 4 - Schematic tendencies of the evolution of the landscape in the Mediterranean zone of the eastern Pyrénées. a: extrapolation of the observed evolution, b: recommended evolution using land management techniques of periodical prescribed burning of selected areas.

Because of the erratic location of fires, the frequency of burning in any site of the landscape is too low to prevent the colonization by woody vegetation, and too high to permit the reconstitution of the forest. So, the landscape tends to be constituted by an almost continuous brush cover with reduced diversity of vegetation and, subsequently, reduced diversity of the avifauna. At the same time, this woody cover allows the periodical development of huge fires, uncontrollable when the conditions of dryness and wind are severe. In this general context, the traditional strategy practised until now in the study area which consisted only of fire fighting, fire prohibition and tree planting, is an ecological cul-de-sac.

III - FUEL BREAKS AND BIRD PROTECTION

The necessity to cut the continuity of woody vegetation with firebreaks and buffers has been recognized by firemen for a long time. But classical firebreaks are expensive to create and to preserve, and are generally too narrow to play a decisive role during the largest wild fires, and to have a significant action on the distribution and diversity of macro-fauna. To be effective, firebreaks should be large enough (i.e. hundreds of hectares, or even more), and include, if possible, the hillcrests. Their vegetation should be kept in a young stage "since younger less flammable growth checks the progress of fires burning in older growth" (MINNICH 1981). Prescribed burning, with a burn cycle short enough to keep the dead-to-live fuel ratio at a minimum, in association with grazing when possible, is the only economical method to prevent these areas from being invaded by woody plants. It is also, the less harmful technique for both fauna and flora (chemical techniques should be absolutely banned in these areas where so many rare species are to be protected). Unlike the recommendations of several North American authors (e.g. MINNICH, op. cit.), it would not be opportune to make any rotation of the burning sites and "to develop a patchwork of different aged stands" because :

1) it takes a rather long time to completely eliminate the woody species by repeated burning (see the experiments of TRABAUD 1980),

2) most of the rare and endangered species concerned by this management are sedentary, or strongly philopatric.

Prescribed burning sites should ideally be restricted to judiciously chosen areas, in order to further the progression of the remaining non-cultivated landscape towards the forest. An integrated management linking traditional fire suppression strategy and prescribed burning in carefully selected places would, at the same time, reduce the hazards of fire fighting, decrease the occurrence of very large wild fires, and satisfy the requirements for both forest and rare species protection.

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ECOLOGIA MEDITERRANEA - Tome XIII (fascicule 4) - 1987

Study of ecological influence of fire on fauna in mediterranean ecosystems(soil and above-ground layer).Patterns of post-fire recovery.

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SUMMARY - The effects of wild fire on the fauna (soil arthropods and small mammals) were studied in the Albères chain (Pyrénées Orientales, France). After fire, the vegetation structure tends to go back to the pre-fire situation. Retrogressive ecosystems (i.e. grasslands, maquis) rapidly return to the pre-fire situation, while forest restoration is very slow.

Fire destroys the litter layer and induces strong changes in the humus biological activities. It induces an immediate loss of animals, and also a delayed mortality. But almost all species may recolonize their habitat.

In open habitat, characterized by a high primary production where the herbivores/predators system dominates, the recovery is rapid. On the other hand, in woodlands where the necrômass dominates (decomposers system), the disturbances are very strong and the recolonization takes a very long time with many transitional stages.

KEYWORDS : post-fire recolonization, litter, soil fauna, ants, small mammals, herbivore/predator system, decomposer system.

INTRODUCTION

Most of the mediterranean hill landscapes exhibit progressive successions because of the rural drift from the land about 80 years ago. These progressive successions are at present oftenly receded by wild fires, which appear to be a consequence of the land desertion. Despite the ecological importance of the wild fires in the mediterranean regions of the Old World, only a few research studies dealt with effects of fire upon the fauna. The present paper is devoted to the short or mid-term effects of the fires; it mostly concerns the results of the study made by our staff and associated students in collaboration with researchers from other institutions (Prof. C. FELIU, Barcelona; Dr. P. GREENSLADE, Canberra; Prof. S. MAS COMA, Valencia; Dr. L. TRABAUD, Montpellier). Our data are supplemented by results of other authors when possible, in order to constitute a "state-of-the-art" on knowledge of the ecological influence of fire on the fauna in the Mediterranean Basin.

Our samplings were mainly carried out in the Albères chain (Pyrénées Orientales) because they give a good example of the complex retrogressive/progressive dynamics of mediterranean landscapes. Three main layers were distinguished : 1) soil and litter, 2) surface and grass layer, 3) shrubs and canopy. Three animal groups were selected as bio-indicator in each layer, resp. : 1) soil arthropods, 2) small mammals, 3) birds. The synthesis of the results summarizes the patterns of ecosystem recovery, especially : the evaluation of influences of fire on animal populations and dynamics according to vegetation structure, ecological successions and comparison of the different patterns of recovery (PRODON FONS & PETER 1984, ATHIAS-BINCHE & SAULNIER 1986, SAULNIER & ATHIAS-BINCHE 1986, ATHIAS-BINCHE 1987). The data concerning the avifauna are presented in a separate paper (PRODON 1987).

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RESULTS

2.1. Vegetation structure

Three main types of landscape can be distinguished in the present Albères chain :

- woodland, more or less evolved or retrogressive. Typically holm oak woods or coppices (Quercus ilex) and cork oak stands (Quercus suber).
- maquis : a brushy retrogressive pyrophilous formation, equivalent to the mattoral or the chaparral.
- grassland : either a retrogressive landscape on eroded soils or a transient stage in woodland after fire.

The post-fire dynamics of the holm oak ecosystems were described in detail by TRABAUD (1986). In old forests, the canopy is very dense and consequently the understorey is poor. After fire, the upper layer disappeared in favour of the grass layer. After about 3 years, brush began to dominate, then the community slowly evolved towards its pre-fire structure. The closer the community is to the climax, the longer it will take to return to the pre-fire situation. Thus, the restoration of the maquis will be faster than that of the coppice, but the coppice will return to its pre-fire situation more rapidly than a forest of tall trees.

The restoration is rather rapid in cork oak stands, because the vegetation structure is more closely related to tall trees on maquis understorey than to closed forest (due to the light canopy of Q. suber), and the fire burns only a few parts of the trees because of the protective effect of the cork bark. Thus, the regeneration starts from the canopy (3 to 6 months after fire), and the landscape retains an aspect close to a forest. One to 2 years after fire, the soil is covered by herbaceous plants (mostly Leguminosae); then 3-4 years after, the understorey is dominated by a Cistus maquis (ATHIAS-BINCHE & SAULNIER 1986, ATHIAS-BINCHE & BRIARD 1986).

In low maquis, the germination of Cistus, Ulex and herbaceous plants, give a grassland the first spring following fire. Later, shruby species tend to eliminate the herbaceous cover, and the ecosystem returns rapidly to the pre-fire structure (PRODON 1986). In high maquis, the situation is comparable the 1st spring to the situation of low maquis. Later higher bushes (Erica, Daphne, Calycotome) dominate a grass cover from which emerge young Cistus and Ulex.

In grassland, after the first rainfall following fire, germination rapidly reconstitutes the herbaceous cover. Two springs after fire, the flowering of annual and perennial plants is more important than in the unburnt places (PRODON 1986). This type of ecosystem is the fastest to return back to its pre-fire situation.

2.2. Effects of fire on soil and fauna

2.2.1. Soils, edaphic Myriapoda and Uropodid mites, mean-term effects

*Introduction, sampling sites, methods.

Data on short-term effects of the fire upon Uropodid mites and Myriapoda was published in previous papers (ATHIAS-BINCHE & SAULNIER 1986, SAULNIER & ATHIAS-BINCHE 1986, ATHIAS-BINCHE 1987). The present work deals with their dynamics during the 4 years following fire.

Uropodina and Myriapoda were studied in a cork-oak stand (Valmy, Argelès-sur-Mer, Albères chain) which burned in June 1982. Sampling was made in a burnt plot and in a control plot. The burnt plot had burned 10 to 15 years ago. The sampling period covered 4 years (from June 1982 to May 1986). Six samples of 250 cm^2 were made in each plot every fortnight; the litter was separated from soil layers (0 to 5, 7 or 12 cm, according to the proximity of the bedrock). Some samples were also taken in an old cork oak stand situated on deeper soil, which was supposed to have never burned (ATHIAS-BINCHE 1987). Fauna was extracted in Berlese funnels.

*Postfire evolution of litter and soil

The restoration of the canopy was rapid in the Valmy cork oak stand. During the first 2 years after fire, the understorey was only composed of herbaceous plants (mostly Leguminosae), which were afterwards eliminated by a secundary maquis. During the first years, the soil surface was characterized by patches of bare soil, mosses, grass clumps, soil under maquis shrubs, or thickets of surviving cork oak trees. (ATHIAS-BINCHE & BRIARD 1986).

Fire completely destroyed the litter, which constitutes both a trophic resource for soil organisms and plants (nutrient cycling), and a protective layer regulating the edaphic microclimate. The amount of litter began to increase in the 2nd winter following fire (Fig. 1), but 4 years after fire, it had still not regained its pre-fire values. The distribution of this litter was very heterogeneous, as it is estimated by the variation coefficient ($CV\% = 100.s/\bar{x}$). In addition, the litter situated under the maquis shrubs was decomposed with great difficulty by soil organisms. Thus, despite the increase of litter weight, the main biologically active layer (fragmentation layer) may require more than 10-12 years for its reconstitution.

On the schist bedrock of the Albères hills, the soil is of the acid-brown mediterranean type. In the Valmy site, the soil is never deep (0 to 30 cm) on gentle slope. In the control plot, the active brown humus layer rarely exceeds 7 cm. These soils are not very developed probably because of old cultural practices. The soil activity is higher in the old stand, which is situated in a shadowy talweg.

The fire affects mostly the upper layers of the soil. It induces a loss of carbon and nitrogen, and a disequilibrium in the base exchange capacity (Table 1). Just after fire, the decrease in carbon and nitrogen contents was mainly due to the combustion of the litter.

	SB	%	SNB	%	SAD	%
Ca meq%	3.5	54.9	10.5	71.1	6.8	62.1
Mg meq%	1.66	26.0	2.75	18.2	2.61	23.8
K meq%	0.301	4.7	1.046	6.9	1.157	10.5
Na meq%	0.911	14.2	0.550	3.6	0.337	3.4
S	6.372		14.846		10.940	

Table 1 - Soil chemical components in the burnt cork oak stand (SB), in the control (SNB) and in the cork oak stand on deep soil (0-3 cm). Base exchange (meq), their percentage (%) and total bases (S).

Carbon rate increased slightly during the 3-6 months following fire in the 0-2 cm layer (because of the fall of some dead leaves, the occurrence of dead soil organisms). Then, it tended to decrease in relation to the development of the maquis and to erosion losses and increase slowly up to the situation in the control plot (Fig. 2). Losses in nitrogen were marked in the upper layer, but the nitrogen content distinctly increased 3 months after fire, because of the symbiotic activity of the Leguminosae; then it decreased in the maquis stage, increased again slowly to reach the level of the control plot, and finally that of the old stand. The abundance of available plant nutrients in ashes produces a rapid - but temporary - fertilization of the soils; then, losses might be important for several years. Erosion losses seemed to be exported more in debris than in runoff water (DE BANO & CONRAD 1976, DE BANO 1982).

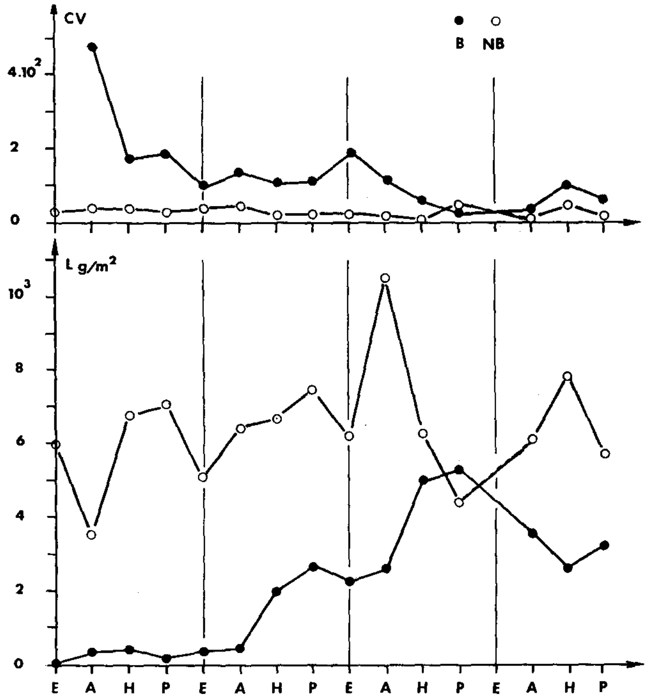


Figure 1 - Valmy cork oak stand, seasonal variations of the litter quantities in the burnt site (B) and in the control (NB) (dry weight) in g/m² during the 4 years following fire, and variation coefficient (CV, %). E: summer, H: winter, s: seasons).

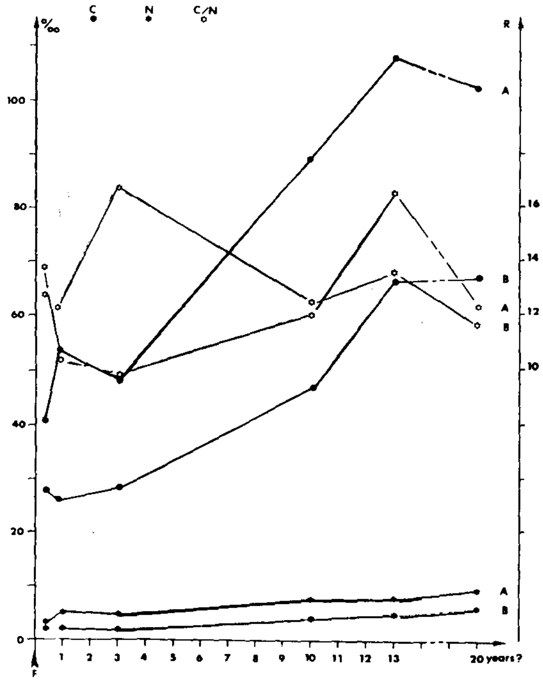


Figure 2 - Variations of nitrogen and carbon rate in the Valmy cork oak stand resp. 3 months, 1 and 3 years after fire (in the burnt plot), 10 and 13 years after fire (in control plot, supposed to be burnt about 10 years ago), and in an old cork stand on deep soil. C: carbon content (‰), N: nitrogen, C/N : C/N ratio, A: 0-2 cm soil layer, B: 2-5 cm layer.

Fire induces also changes in the soil organic matter properties. In central Spain, ALMENDROS et al. (1984) demonstrated that the pyromorphic humus became more stable, due to an increase of the aromaticity and the condensation degree of the humic acids. This type of humus was less decomposable by soil organisms that might have damaging effects on soil fertility.

Soil microbial activity, measured as deshydrogenase activity, nitrification and CO_2 release, might not be very perturbed after fire, at least in phryganic ecosystems, degraded by fires or old cultural practices (ARIANOTSOU-FARAGGITAKI & MARGARITIS 1982). Nitrification process could be even more intense just after fire. This increase in the total soil metabolism, which was higher in the first years after fire, might be correlated with the overgrowth of herbaceous plants.

In soils, fire induces both the disappearance of the litter cover, and a sudden -but short- increase in nutrient content and microbial activity in the humus layer, rapidly followed by longer-lasting losses in organic matter and nutriments.

*Studied arthropods

- Uropodina

The Uropodina (Anactinotrichida) are slow moving mites feeding on living substances (animal, fungi or microorganisms). Thus, they are situated near the end of the trophic chains in the soil sub-system, and heavily depend on the efficiency of the matter/energy exchanges at the soil surface. Any disturbance of the soil activity induces strong effects on their number and their community structure (ATHIAS-BINCHE 1981, 1983).

The Valmy community is composed of only 3 species. The present paper only deals with Olodiscus minimus, a small-sized euryecic and parthenogenetic species (450 µm long), largely widespread in Europe and North Africa, which represented here more than 90% of the total community. In contrast to many other Uropodina, it does not so much depend on the biological activity of the litter as it is sometimes able to feed on plants.

- Myriapoda

The groups found in the Valmy site were as follows (cf. also SAULNIER & ATHIAS-BINCHE 1986) :

Cl. Diplopoda (Millipedes) : - Glomeridae : Large litter dwelling decomposers, very rare in the control plot, absent after fire (not taken into account in the present work). - Blaniulidae and Iulidae. Body cylindrical, living mostly in litter, but good burrowers. Decomposers. Only small species occurred at Valmy (max. body length 50 mm). - Polyxenida (=Penicillata). Small, probably microphageous decomposers (max. length 5 mm).

Cl. Chilopoda (Centipedes) : - Geophilomorpha : Geophilidae. Predators. Long but narrow and fast moving Myriapoda (max. length 60 mm), very good burrowers. - Scolopendromorpha : Cryptopsidae. Litter dwelling predators (max. length 40 mm). Very scarce at Valmy, especially after fire, not taken into account herein. - Lithobiomorpha. Fast moving predators, unable to burrow their way (max. length 40 mm).

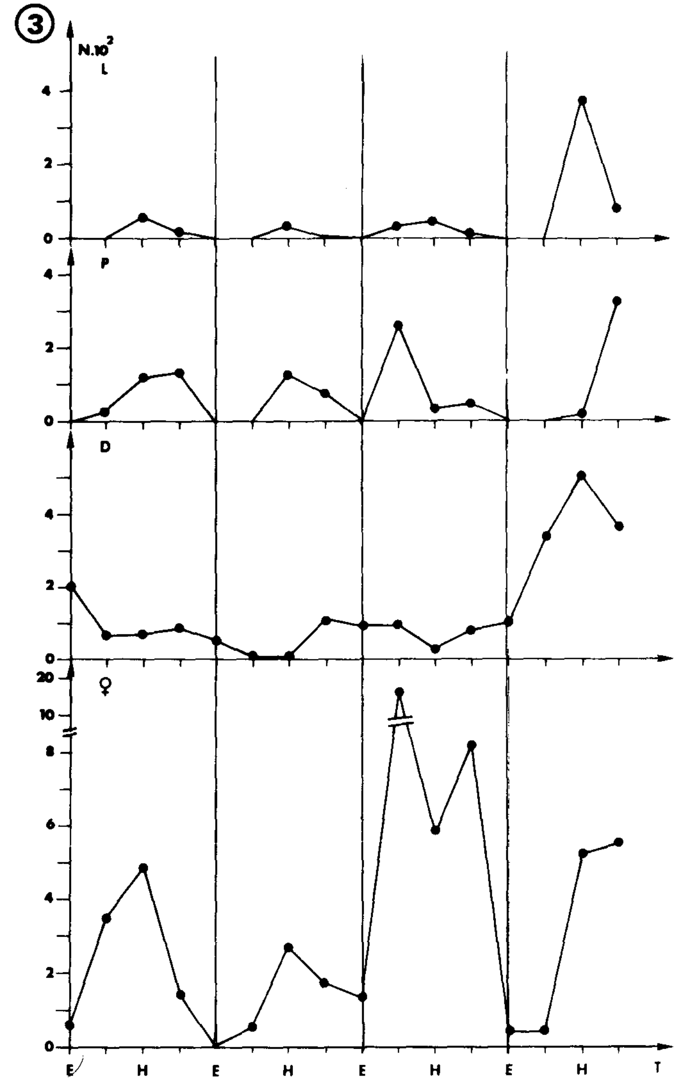
Cl. Symphyla. Detritivores, fungivores, phytophagous or microphagous (bacterial feeder). Not burrowing, but able to use soil crevices, root channels or galleries (max. length 10 mm).

Cl. Pauropoda. Considered fungal feeders (max. length 1 mm).

* Phenology of O. minimus, and recolonization pattern (ATHIAS-BINCHE 1986).

The generation duration of O. minimus requiring only 8 to 11 months (ATHIAS-BINCHE 1985), instead of several years in many Myriapoda, the study of its phenology during the 4 years following fire allowed to understand the major trends of recolonization processes.

In the control, immature stages (larvae and protonymphs) occured only during wet seasons, (Fig. 3); while deutonymphs (pre-adult instar) were less influenced by seasons, and were present in summer. Females were more abundant during the rainy seasons. In the burnt plot, the immature densities were strictly correlated to the climatic factors, especially temperature (Fig. 4), larvae being only produced in winter. Deutonymphs occured all through the year, while females were mostly found in autumn and winter.



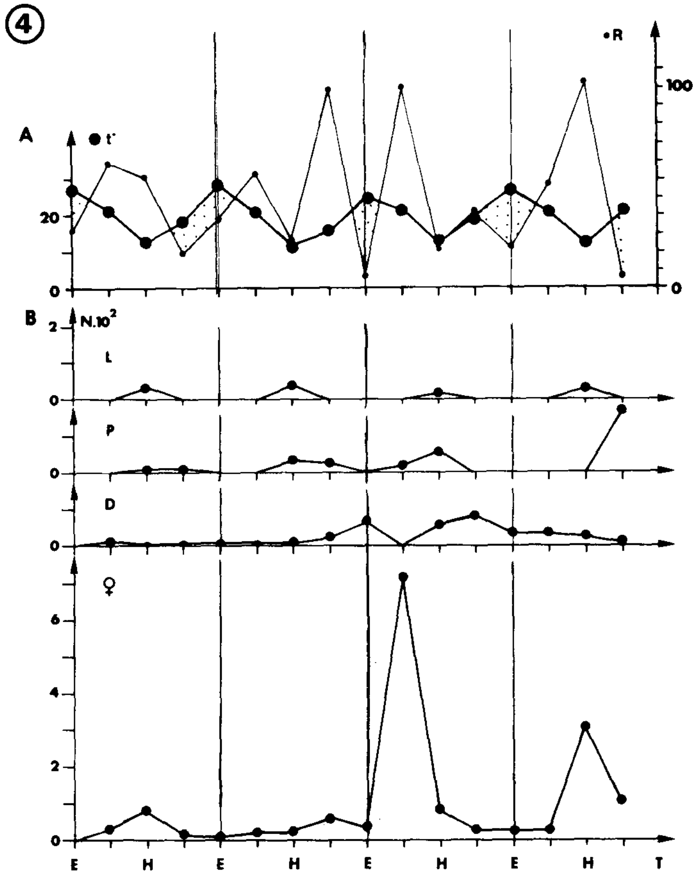


Figure 3 - Seasonal phenology of O. minimus in the control (N: ind. $10^{-2}/m^2$). L: larvae, P: protonymphs, D: deutonymphs, D: deutonymphs, ♀: females, E: summer, H: winter.

Figure 4 - A: air temperature (t°: maximum temperature in °C, seasonal averages) and seasonal rainfall (R: mm) measured at the Centre d'Ecologie Méditerranéenne, Banyuls-sur-Mer. B: Seasonal phenology of O. minimus in the burnt plot. Same symbols as in Fig. 3.

The deutonymph , which is the most resistant instar, is able to stop its post-embryonic development in case of unfavourable conditions (ATHIAS-BINCHE 1981b, 1985). In Mediterranean climate, it emigrates down in soil during the warm season and enters estivation, a period of low activity. The estivation will end with the decrease of temperature, usually associated with the beginning of the rain; then, the deutonymph rapidly moulds, and the young female lays eggs, producing cyclical recruitments of immature stages. In Valmy, the recolonization of _O. minimus_ is mostly a demographic process : deutonymphs and some old females would undergo the summer temperature and dryness in the deep layers of the soil or sometimes in ant nests. Afterwards, with the autumnal coolness and moisture, they moved up to the soil surface for reproducing. Then, the survival of the immature stages tended to increase in relation to the litter reconstitution (Table 2), this layer gradually protecting the soil against heavy temperatures. The recovery would be very long, not only because of lack of resources, but also because the microclimate of open habitats did not allow the development of more than one generation per year, whereas in wetter forest habitats, the populations could reproduce twice a year (ATHIAS-BINCHE 1985).

	J	DN	F		J	DN	F
B1	3.2	3.4	13.8	NB1	86.2	104.2	258.2
B2	14.2	11.1	23.8	NB2	57.5	43.3	121.5
B3	21.7	17.7	214.0	NB3	110.8	75.3	795.7
B4	48.7	24.5	114.0	NB4	200.7	85.5	171.0

Table 2 - _O. minimus_, average densities (ind./m^2) of the juveniles instars (J), the deutonymphs (DN), and the females (F) in the burnt (B) and in the control (NB) during the 4 years following fire.

*Vertical distributions and densities of Uropodina and Myriapoda

- Pre-fire situation

The main trends of the vertical distribution of each group were estimated by the seasonal percentages of animals collected in the control litter and the average of these percentages for the 4 years (Fig. 5 : L%).

Among predators, the percentage of Geophilids in litter was low, especially during the dry seasons, i.e. spring and summer. On the contrary, Lithobiomorpha were frequent in litter almost all the year round. Within the decomposers _sensus lato_, the Symphyla were mostly observed in soil, especially in summer. Pauropoda could be also regarded as soil dwellers, but they moved in litter during the wet season. Penicillata were in between with an average of 42% of animals in litter, their vertical distribution did not vary very much in relation to the season. Seasonal migrations were well marked in Uropodina : less than 1% of the animals was found in litter during the summer estivation, compared with 41 to 45% the rest of the year. Iulidae are littericolous as opposed to the other decomposers, they were slightly more abundant in litter during the dry season.

These results allowed to classify the different groups in relation to their feeding habits and their migratory behaviour, especially in summer, which is not only the dryest season, but also the main period of fire occurrence. The groups may be ranked as follows (Fig. 5) :

- Predators : litter dwellers (Lithobids); indifferent but humicolous during summer (Geophilids).

- Decomposers : strict litter dwellers (Iulids); indifferent (Penicillata); indifferent but humicolous during summer (Uropodina); soil dwellers (Pauropoda, Symphyla).

	1		2		3		4		MEAN	
	B	NB	B	NB	B	NB	B	NB	B	NB
Symphyla	8	16	2	13	80	54	44	23	34	27
Pauropoda	7	13	2	72	40	47	148	312	49	111
Penicillata	18	113	39	78	67	62	16	343	36	149
Iulids	1	65	3	68	8	164	34	167	12	116
Lithobids	4	26	6	21	2	34	16	85	7	42
Geophilids	7	6	2	24	10	39	97	14	29	21
Myriapoda	47	248	54	283	208	412	398	177	177	480
Uropodina	43	531	68	253	190	788	115	798	103	592

Table 3 - Valmy stand, densities of the Uropodina and the Myriapods (N/m^2) during the 4-year study. Years following fire (1, 2, 3, 4). Densities in burnt (B) and unburnt (NB) plots. Mean for the 4-year study.

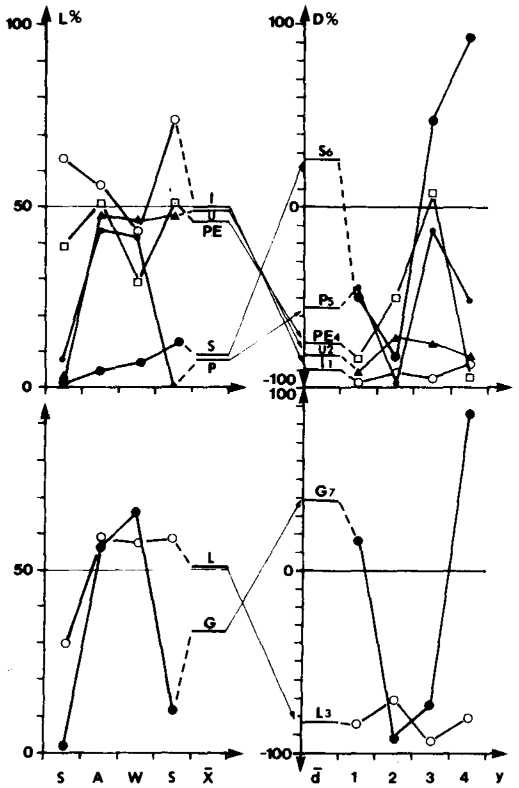


Figure 5 - Valmy stand, soil fauna.

A) Seasonal percentages of animals sampled in litter (L%) and average percentage for the four-year study (X). S: summer, A: autumn, W: winter, S: spring. Upper diagram : decomposers <u>sensu lato</u> : I: Iulids, U: Uropodina, Pe: Penicillata, Pa: Pauropoda, S: Symphyla; lower diagram : predators : L: Lithobids, G: Geophilids.

B) Algebric differences of the densities between burnt and control plots (D% = NB-B/NB) for each of the four years following fire (y) and average of the differences for the sampling period (d); the groups are ranked by decreasing percentages $\bar{d}$ from I1 to G7.

- Postfire situation

The sensitiveness to fire was evaluated by the differences between densities in control and burnt plot ($D\% = - \left[(NB-B)/NB\right]$), where NB and B are respectively the abundance in the control and in the burnt site; In Fig. 5, d is the average of D% for the 4 studied years.

Since the immediate effect of fire was the disappearance of litter, it affected at first the litter dwellers. D% and $\bar{d}$ were the highest for the Iulids and the Lithobids, which lost more than 80% of their population. Uropodina were also very sensitive ($\bar{d} = 82\%$), whereas they are not strict litter dwellers. This might be due to their dependence on the efficiency of the primary trophic chains in the soil sub-system. Penicillata were also heavily affected, despite the fact that they were not strict litter dwellers. The Pauropoda lost 55% of their density, whereas they were humicolous, but the annual variations of D% indicated that they were mostly sensitive to fire effects during the first 2 years.

*Population growth estimation

Population growth was estimated by the logistic curve, which is a very general model. The logistic equation, $dN/dt = rN(K-N/K)$, can be written in the integral form, $N_t = K/1+e^{a-rt}$ (N_t : population size at time t; K : upper asymptote, i.e. max. value of N; r : rate of population growth *per capita*; e : base of natural logarithms; a : constant of integration defining the position of the curve relative to the origin). The linear form of this equation : $\log_e(K-N/K) = a-rt$ (coordinate y: $\log_e(K-N/K)$, x : time (here in months), a : intercept, r : slope) was used to fit the logistic model to the observed data of densities in the burnt plot, using correlation coefficient and regression equation (KREBS 1978). For *O. minimus*, K equals 1500 ind/m^2, i.e. the average winter density in the control during 4 consecutive winters, using winter densities in the burnt forest as observed data (ATHIAS-BINCHE 1986). For Iulids, Penicillata and Lithobids, K was equal to the average density in the control. For Pauropoda, K was equal to the maximum annual density in the control whereas some values in the burnt site exceeded the average density in the control. For Symphyla and Geophilids, where the mean density exceeded that of the control, K was equal to the maximum density in the burnt plot (Table 4, Fig. 6).

The model did not fit very well the Myriapods, because it is not applied to a population but to a group. Nevertheless, the logistic curve gave a good estimation of the delay needed for recolonization, using only simple count data. From an ecological point of view, the calculated curves corresponded to the main characteristics of the groups observed in the field.

As it was shown above, fire induced the highest losses in the group of litter dwellers and their return to the pre-fire situation would depend on the litter restoration. The logistic model indicates that it will probably require more than 20 years for Uropodina and Lithobids to reach the maximal density K. These biolytic feeders (=feeding on living material) depend both on the recovery of the litter and of the energy transfers across trophic chains. Recolonization could be slightly more rapid for litter dwellers feeding more directly on dead plant material or fungal hyphae (e.g. Iulids, Penicillata). The growth may require less than 10 years for Pauropoda, which are also decomposers, but not strict litter dwellers. On the other hand, soil dwellers were less affected by fire (e.g. Symphylids and Geophilids). Such herbivores or predators even could be favoured by the transitional development of grass and by the mortality affecting their competitors. However, this growth would also be transitional, and their density would decrease and reach the pre-fire level after the restoration of oak litter and related fauna.

*Discussion

The sensitiveness to fire effects depends on the characteristics of the groups : trophic level, demography, seasonal activity or vertical distribution. In Mediterranean conditions, the ability to migrate downwards during summer is an advantage. Since fire is destroying the litter, animals able to live in the deeper layers of the soil, even temporarily, have better probabilities to survive than strict litter dwellers. The example of Uropodina and Myriapods demonstrates that the restoration of the soil communities is a long process.

	Regression	R	K
Symphyla	Y=-0.657T+3.46	-0.817	70
Pauropoda	Y=-0.139T+6.15	-0.857	192
Penicillata	Y=-0.057T+6.96	-0.943	149
Iulids	Y=-0.088T+4.13	-0.998	116
Lithobids	Y=-0.036T+1.81	-0.786	42
Geophilids	Y=-0.249T+7.92	-0.786	97
Uropodina	Y=-0.044T+2.97	-0.997	1500

Table 4 - Logistic growth of Myriapods and Uropodina. Regression equation in the linear form of the logistic, $Y = \log_e ; ((K-N)/N) = -rT+a$, where T : time (in months), R : correlation coefficient for the 4 paired values, K : maximal value of the densities (upper asymptote of the logistic curve, number of individuals/m^2).

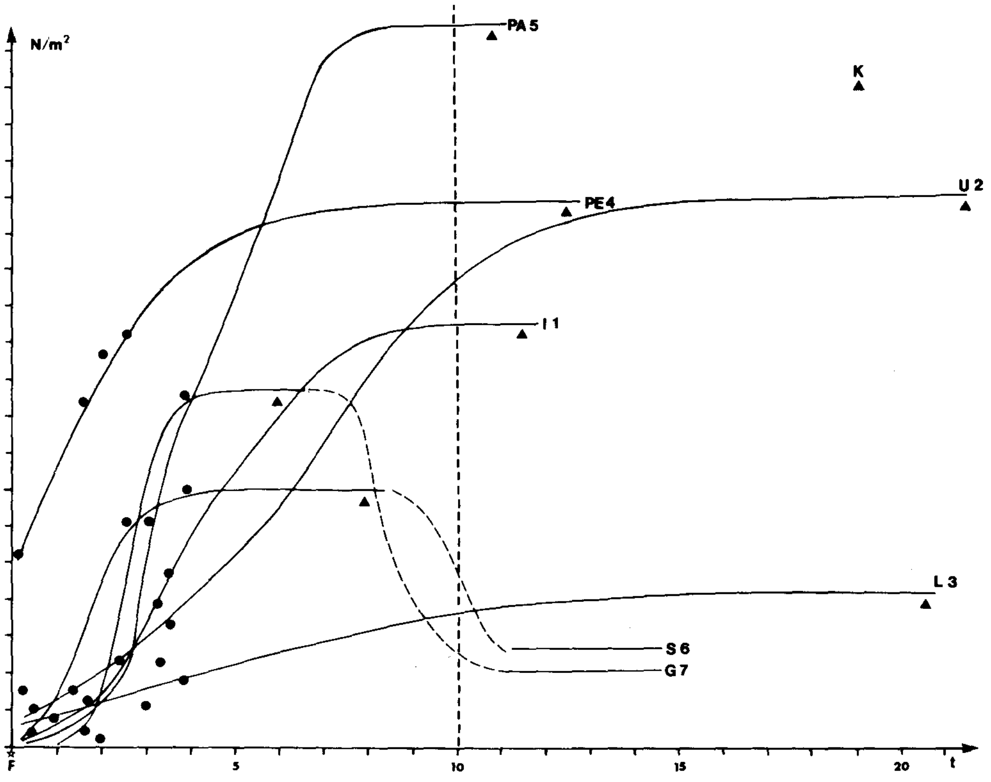


Figure 6 - Logistic theorical growth curves of Myriapoda and Uropodina (see text and table 5). N/m^2: densities (N/10/m^2 for Uropodina). F: fire, t: time (years), arrows indicate the time for which the upper asymptote K is attained.

The biolytic feeders are the most affected because they rely on the primary food chains of the soil sub-system. For such organisms, the recolonization may be very long, depending not only on the litter recovery, but also on the restoration of the whole soil biological activities. True decomposers and detritivores, feeding more directly on dead plant material or fungi, could recover more rapidly than the biolytic feeders. The mortality is lower in the humicolous forms. The recolonization is indigeneous for the soil arthropods. It starts from the deep horizons in the case of good burrowers or small animals able to migrate downwards, especially in summer. For the litter dwellers, the recolonization is a swarming process, coming from unburnt places, or small refugiae (unburnt area between rocks, foot or trunks, etc.), the population aggregate gradually, following the litter reconstitution.

2.2.2. Effects of fire on ants (SOMMER 1986)

*Introduction, methods

Ants are social soil arthropods, but they are not true decomposers, because they mostly feed on above ground resources. They are active during warm periods. Four types of habitats have been sampled : - Grassland on eroded ranker burned in 1978 and in 1981, Madeloc (Port-Vendres) - Maquis burned in 1982, Pouade valley (Banyuls-sur-Mer) - Maquis burned in 1982, Madeloc. - Cork oak stand burned in 1982, Valmy. Ant nests were counted in one or several sample plots of 5 m^2 each.

*Results (Table 5)

- Grassland.

One week after the fire, death rate reached almost 100%. Fire appeared to be very devastating on shallow soils where ants could not move downwards. After 1 year, all species usually dominant in grassland were present again, but about 1/3 had disappeared and nest number had decreased to reach 25% of the pre-fire situation. Omnivores and granivores dominated the community, and specific diversity decreased markedly. The recolonization seemed to start from unburnt areas of maquis. Two years after fire, species richness and nest densities decreased; this might be due to the post-fire evolution of vegetation from a grassland to a low maquis.

SITES	FIRE	DATES				
GRASSLAND	1981	CONTROL	SEPT 82	SEPT 83		
T		41.7	31.6	18.9		
S		21	14	10		
MAQUIS (Pouade)	1982	CONTROL	AUTUMN 82	SPRING 83	AUTUMN 83	
T		12.2	7.5	3	7.4	
S		7	8	4	6	
MAQUIS (Madeloc)	1982	CONTROL	AUTUMN 82	SPRING 83	AUTUMN 83	
T		3.9	1.7	6.6	6.8	
S		8	2	8	10	
CORK OAK (Valmy)	1982	CONTROL	JULY 82	SEPT 83	JUNE 83	SEPT 83
T		13	1.5	1.4	5.5	5.4
S		11	4	3	11	9

Table 5 - Ant post-fire dynamics. Sites, date of fire, sampling dates. T : total number of ant nests/100 m^2, S : number of species.

	VALMY		LES ABEILLES	
	BURNT	CONTROL	CONTROL	MAQUIS
Sminthuridae				
Sphaeridae sp.	0	10*	0	3
Prorastriapes sp	5*	0	0	0
Deuterosminthurus sp	0	0	0	16*
Sminthurinus sp	1	1	0	0
TOTAL	6	14	0	19
Entomobryidae indet.	4	47*	101	51
Entomobrya multifasciata	1	0	0	13
Entomobrya sp	6*	0	0	0
Lepidocyrtus curvicollis	0	4*	12*	1
Orchesella sp	0	3	12*	4
Tomocerus sp	1	19*	13	10
Tomocerus longicornis	0	0	1	0
TOTAL	12	73	139	80
Isotomidae spp	2	1	3	2
Xenylla sp	0	0	0	1
Poduromorpha	130*	44	35	4
Taxa	8	9	7	11

(*) : significant.

Table 6 - Results from pitfall traps, Collembola.

	VALMY		LES ABEILLES	
	BURNT	CONTROL	CONTROL	MAQUIS
Protura, Campodea	0	0	4	0
Dermaptera	10*	1	0	12*
Orthoptera	1	0	0	0
Psocoptera	1	4	1	2
Pseudococcid	0	0	3	1
Hemiptera (other)	0	1	1	0
Thysanoptera	1	2	1	0
Lepidoptera	0	0	1	0
Lept. larvae	0	0	0	1
Diptera	21	36	47	45
Dipt. larvae	0	3	2	0
Hymenoptera (excl. ants)	0	2	3	1
Coleoptera	2	1	5	7
Coleop. larvae	0	0	0	1
indet. larvae	1	1	5	0
Total Pterygota (excl. ants)	37	51	69	70
Araneida, Phalangida	6	3	9	12
Pseudoscorpionida	2	0	2	0
Total Arachnida	8	3	9	12

Table 7 - Results from pitfall traps, of soil arthropods densities (excl. Acarina and Collembola).

- Maquis.

In the Madeloc site, one week after fire, the myrmecofauna almost completely disappeared, but the number of nests was already low before fire. After 1 year, more than 12 species were sampled but they were represented by a small number of nests. Only 3 species were present before fire, the other ones were presumably coming from more open habitats. In the Pouade site, one week after fire, all species present before fire were found again (Table 6), but many nests contained numerous dead individuals. Since the soil was deeper than in the Madeloc maquis, there was a better nest protection against fire. One year after, densities tended to decrease, but recover during summer. The recolonization started from neighbouring areas because no exogenous species were found.

- Cork oak stand (Valmy).

Species richness and numbers were low as in all forests. One week after fire, species richness and densities decreased markedly; many dead individuals were found in the nests. All the small species living in superficial nests had disappeared. Two months after, the myrmecofauna was even reduced. One year after, the recolonization restarted. One could distinguish 2 groups : one corresponding to the usual cork oak fauna and the other was composed by species characteristic of open habitats. The number of nests remained rather low.

*Discussion

For ants, the recolonization may occur either by terrestrial movements or by swarming winged females, which could come from rather distant areas. These migrations can explain the importance of exogenous species in some studied plots.

2.2.3. Collembola and other soil arthropods (excl. Acari), preliminary results
(GREENSLADE & ATHIAS-BINCHE 1986).

*Introduction, sites and methods

Collembola (Apterygota) mostly feed on fungal hyphae or dead plant material. In Mediterranean climate, fauna is composed by two groups : a winter community, active during the wet season and passing the summer in egg stage (mostly Entomobryidae), and a community more resistant to dryness, present all the year but migrating into deeper layers during dry season (POINSOT-BALAGUER 1984). These two behaviours can explain some differences in their post-fire dynamics.

Four sites were studied; two at Valmy which consisted of a burnt (3 years after fire) and an unburnt plot of cork oak stands. The other two were near the Mas des Abeilles (Banyuls). One of them was beside a creek with holm oak stands and the other was about 50 m away on a hill side which carried a dense maquis and had been burned in 1976 (9 years ago). At each site five pitfall traps were run for six days each (1.8 cm diam., 3/4 filled with an alcohol/glycerol mixture).

*Results

- Collembola (Table 6). There were some differences in pitfall catches between sites which could be accounted for by the fact that the sites were more open and by the lack of developed humus and litter layer. For instance, Prorastriapes cinquelineata was only found on the burnt site at Valmy and Lepidocyrtus sp. was virtually confined to the unburnt wooded sites.

The total number of species and individuals trapped on each site did not differ significantly, but Poduromorpha were more abundant in the Valmy burnt plot, although Entomobryidae were more numerous in unburnt habitats, especially in the control plot of green oak. Fire which induces indirectly an increase in the dryness of the habitat, may favour the "summer resistant fauna" to the detriment of the "winter" fauna. In addition, the life cycle being very short for Collembola (8 days to a few months), the recolonization may be faster than for other soil arthropods.

- Other arthropods (Table 7). The other soil arthropods were more numerous in the holm oak stand even in maquis; the total number of Pterygota differed significantly. The predaceous Arachnida appeared, on the contrary, to be more abundant in open habitats.

These results are in agreement with those of ATHIAS-BINCHE (1986 in press), which tend to demonstrate that the soil fauna is more abundant in holm oak forest soil, which is less degraded than in cork oak stands, oftenly planted on vineyards or poor soils. Generally, decomposers are more sensitive to fire than predators. These latter can be even favoured by the arrival of phytophagous arthropods in the herbaceous cover.

2.3. Thecamoebae (COUTEAUX 1976, 1977)

*Introduction

The Thecamoeba (Protozoa, Testacea) are predators of soil bacteria. Their dynamics is strongly dependent on soil moisture and mineral salt content. The effects of fire on the Thecamoebae had been studied in a submediterranean site (Banne forest, Ardèche), in a mixed *Castanea sativa* and *Pinus maritima* stand, with a heather understorey. The sampling was carried out respectively one week, 7 months, 1 and 2 years after the fire, in a control and a burnt plot.

*Results

Fire affected mostly the litter dwellers; 2 years after fire, the density remained lower than in the control plot. The recolonization of the newly fallen litter probably started from the humus layer. In the humus (0-1 cm), only a weak decrease was observed 6 months after fire. This was due to transitional chemical changes, to drainage of water containing mineral salts from ashes, and probably to the increase of the pH. However, density returned to its pre-fire value 2 years after. Despite a rather rapid recolonization, the community structure was strongly affected by long-term effects of fire (high spècific diversity, but low densities and decrease of species richness), and some species replacements were observed.

*Discussion

As for the arthropods, the littericolous forms were most affected by fire. The recolonization is endogenous, originating from the humus. The recovery of the density may be rapid, but the community structures remain unbalanced. In addition, chemical factors have a strong effect on the thecamoeban fauna.

2.4. Effects of fire on small mammals (ground living species) and their endoparasite helminthofauna (FONS 1986, FONS & GRABULOSA in press, TORREGROSA-ORTS *et al.* in press).

*Introduction, methods

As usual in Mediterranean ecosystems, species richness of small mammal communities is rather low in the studied areas (Albères hills in France and Spain, Aspres hills, France). Three species, the Woodmouse (*Apodemus sylvaticus*), the Wild mouse (*Mus spretus*) and the White-toothed shrew (*Crossidura russula*) were found in the studied areas. The House mouse (*Mus musculus*) and the Garden dormouse (*Eliomys quercinus*) were scarcely found, usually in association with human settlements.

The sampling method was the standard trap line (SPITZ 1969). A sample consisted in 2-3 lines of 55 traps, at about 3 m intervals, constituted by 4 types of alternating traps (SHERMAN, INRA, snap traps, rat traps), left for about 72 hours (330-495 trap nights). Four types of ecosystems were sampled : cork oak stands, green oak forests or coppices, and maquis with both control and burnt plots for each type.

*Results

- A demographic population model : the Woodmice in cork oak stands. A. sylvaticus was the dominant species in the cork oak, as in many other forest habitats. In La Junquera, burned in 1978 (Albères hills, Spain), population remained rather stable in the control during the 5 years following fire. In the burnt site, after a first period with almost no capture, density increased to recover the control density about 1 year after fire. Afterwards, the population of the burnt plot exceeded that of the control for about 3 years, and then decreased. During the period of high density, population structure was unbalanced by a heavy excess of sub-adult wandering males, while the female number increased slowly. A similar pattern was observed in an other cork oak stand burned in 1976 (Llauro, 16 Km NE of La Junquera). The study began only 2 years after fire, but covered the 5 following years. Two years after fire, the densities did not differ significantly between burnt and control, although the figures were slightly higher in the burnt plot for the next 2 years, the sex ratio being in favour of the males. The recovery to a normal population structure began 4 years after fire (occurrence of lactating females, marks of reproduction). The densities became comparable between burnt and control plots from the 5th year; then the sex ratio equilibrium was observed from the 7th year onwards.

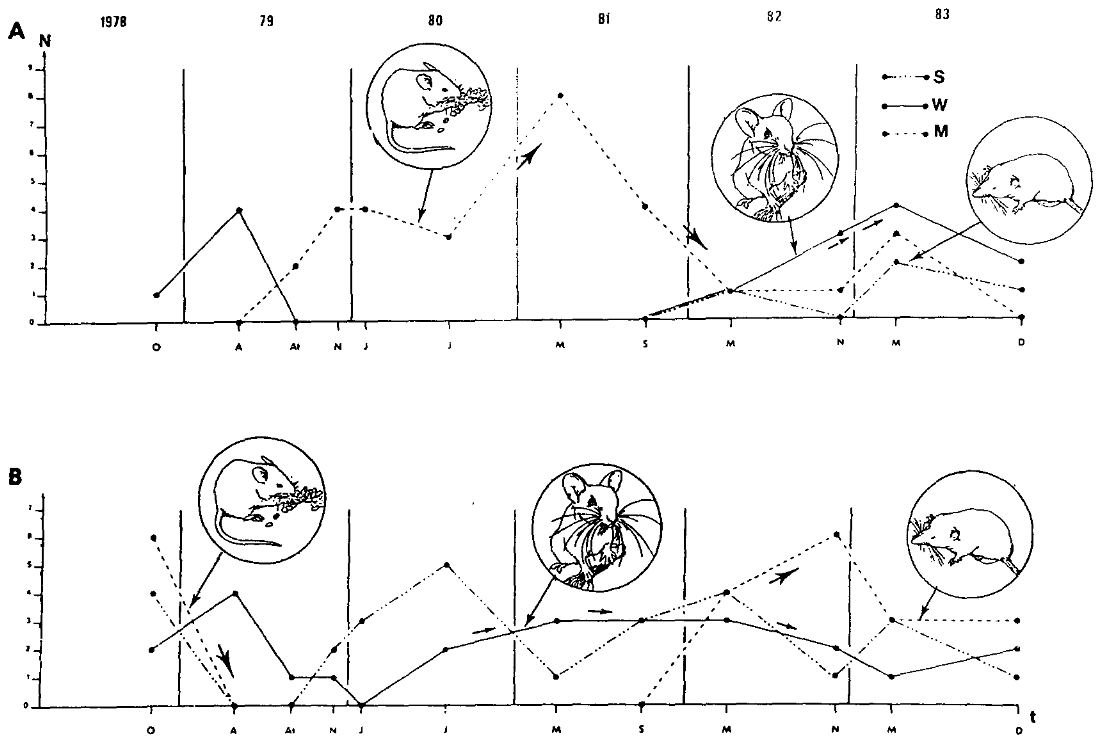


Figure 7 - Postfire succession of small mammals in a low maquis (Joan peak, Banyuls), in a burnt plot (A) and in the control (B), number of individuals/55 traps (N). Note the sudden disappearance of the wild mouse in the control, its arrival in the burnt site and the concommitant absence of the woodmouse. On the contrary, woodmouse population grows regularly in the control, precisely during the period where there is a lack of wild mouse. Just after the re-appearance of the wild mouse in the control, the woodmouse population will decrease, for dominating afterwards in the burnt area. M: wild mouse, S: shrew, W: woodmouse, t: time (months).

Observations of small mammal densities higher in burnt than in control plots have been made by several authors (BENDELL 1974). This phenomenon seemed to be a response to the increase of the herbaceous production, especially in the Mediterranean ecosystems where the mature vegetation is rather sclerophyllous and unattractive. The sex ratio imbalance might be due to the higher dispersal ability of the males. Thus, the post-fire increase of the population is mainly due to emigration of young males, often sexually active, from unburnt sites or refugiae, rather than to reproduction of surviving individuals.

- Postfire evolution at the community level.

- Maquis. A low maquis burnt in 1978 (Banyuls pass, Albères hills) was sampled 2-3 times a year during 5 years (Fig. 7).

- Mus spretus : Only a few individuals were captured during the 1st year following fire - either survivors or early colonists. Then, a sudden increase was observed, reaching its maximum 2 years after fire. This peak was followed by a rapid decline during the 3rd year, then there is apparently, a complete disappearance during the 4th year.

- Apodemus sylvaticus : Only few individuals were found during the 1st year. An acme, followed by a decline was observed 4 years after fire.

- Crossidura russula : This population seemed to completely disappear after fire; this situation remained during the 3 following years. Then, the density tended to increase in the 4th year.

A similar pattern of post-fire succession (peak of wild mice, together with almost no woodmice, followed by the dominance of the woodmice 4-5 years after the fire, and reappearance of the shrew later) was observed in an other maquis (Joan peak, Banyuls). In this case, population variations between the burnt and the control plots tended to be inversely correlated, suggesting some population movements between the two sites.

- Forest-type ecosystems. In a burnt cork oak stand and its control, the results of a 16 month sampling confirmed the succession of Mus-Apodemus-Crocidura (Fig. 8). A longer post-fire study (Llauro, cork oak and holm oak sites) suggested the reappearance of the shrew after 4 years.

Some patterns of the post-fire successions can be understood by the ecology of observed species. The wild mouse, ecologically distinct from the closely related species Mus musculus (ORSINI et al. 1982) seems to prefer rather open and dry habitats, and it is the reason why it is not found in green oak forests of tall trees. This explains its occurrence in the early stage of the succession, and its disappearance as soon as the dense maquis develops during the 2-7 years following fire. The increase of A. sylvaticus density, following a transitional period of very low number, is probably due to the attractiveness of the herbaceous cover developing after fire. However, the lack of pregnant or lactating females is more difficult to explain, even if the high proportion of males could be attributed to their better ability to colonize vacant habitats. Thus, fire strongly affects long-term population dynamics. The shrew, which is unable to burrow, seeking food and shelter in the litter and stone piles, is heavily dependent on the recovery of litter habitats. Thus, as for soil Arthropoda, the recolonization may require a longer time.

*Helminthofauna of A. sylvaticus

The parasite species recorded in the studied sites belonged to the classical fauna of A. sylvaticus in the paleoarctic region. The endoparasite community was composed of 4 species of Cestoda and 6 species of Nematoda in the burnt plots and 1 species of Trematoda, 4 species of Cestoda and 7 species of Nematoda in the control sites. However, the infestation percentage was lower in burnt sites than in the controls. The lack of Trematoda in burnt areas demonstrated the important effect of fire on Pulmonata (Gastropoda). The edaphologic conditions after fire seemed to be unfavourable to the dispersal of the myracidia and the cercariae, instars living out of the host, usually in invertebrates. The heteroxenous Helminta (=needing intermediate hosts) were also not very abundant in the woodmice of the burnt habitats, this fact demonstrates that the intermediate hosts (above-ground invertebrates, acari and other soil arthropods) were deeply affected by fire, especially the litter dwellers.

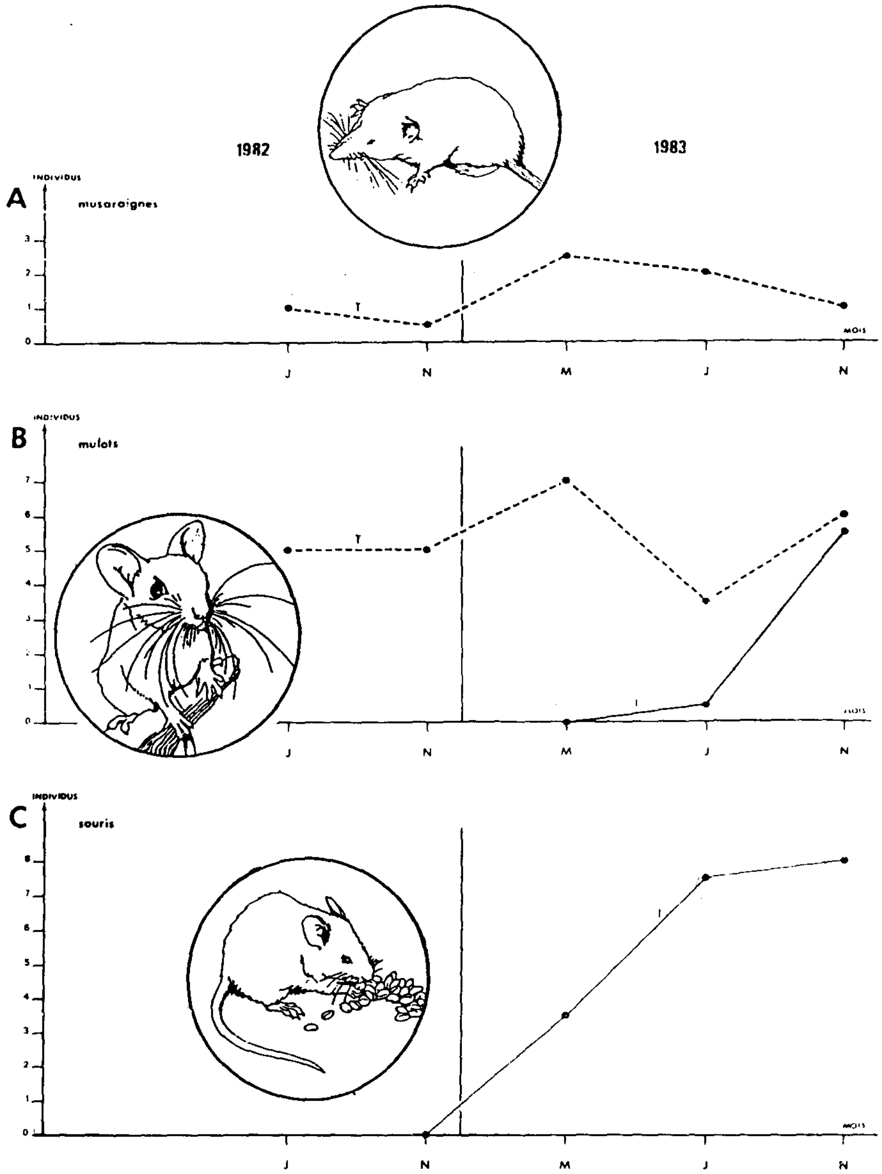


Figure 8 - Postfire dynamics of small mammals in a cork oak stand burnt in June 1982 (Valmy) in the control (dashed lines) and in the burnt area (solid lines). A: shrew, B: woodmouse, C: wild mouse. A: the shrew is completely absent in the burnt site. B: stable population of the woodmouse in the control; no individual in the burnt area during the 1st year, then proliferation of wandering males, at the beginning of the first year after. C: No mice in the control, but a strong development in the burnt site till Marsh 1983.

*Discussion

The small mammal models led to the finding of three different types of recolonization patterns. Fires immediately induced a strong death rate and then a delayed mortality (injured animals, lack of resources), but after about 1 year, the different following patterns could be observed :

- a quasi soil fauna pattern : as for the litter dwelling Arthropoda, the growth of shrew populations would depend on the litter recovery, but this growth required a shorter time (about 4-7 years) because shrews are only dependent on litter as habitat, and not on its biological activity.
- a demographic and dispersal model : for the woodmice, the recolonization might be rapid in some ecosystems, but it is only due at first to wandering males. Thus, the recovery seemed to be fast, but the normal population structure required more than 5 years to be restored.
- a competitor model : the wild mouse, a Mediterranean species adapted to open dry habitats, appeared to be temporarily favoured by fire effects, but, in some cases, its mean-term dynamics might be complicated by competitive exclusion together with the post-fire dynamics of the woodmouse. The competition occurrence seemed to depend on the pre-fire situation, the type of ecosystem, and maybe, the season of the fire occurrence.

CONCLUSION

After fire, vegetation structure tends towards the pre-fire stage. Retrogressive landscapes (grasslands, maquis) return rapidly to their prefire situation (2-5 years), while forest restoration could be very slow. Fire induces an immediate loss of animal, more or less marked in relation to the group, then a delayed mortality, but it finally never produces the complete disappearance of any species, if fire is not too frequent. The main processes involved in the recovery patterns of soil and above-ground fauna may be summarized as follows :

- Speed of restoration. The recolonization may be rapid (1-4 years), and can even outgrow the control densities of wildmice, woodmice and of some soil invertebrates (humicolous Myriapods and Thecamoeba, spiders, Collembola, some ants). Population and community structures may be unbalanced, at least temporarily. The growth is longer (5-10 years) in litter dwellers (shrew, Millipedes), while it may be very slow in some soil arthropods, especially the biolytic feeders (Uropodina, Lithobids).
- Dispersal and migrations, demographic features. The recolonization depends on the dispersal ability and the amplitude of migratory movements. Migrations may be indigenous, starting either from the limits of the burnt site (mammals, ants, soil arthropods), or from refugiae or deep layers of the soil (Uropodina, some Myriapods, some ants, Thecamoeba). It may also originate from exogenous sites for some ants, and perhaps wild mouse. The migration amplitudes range from about 1 km-100 m (ants, small mammals, some phytophagous insects) to only a few cm (Thecamoebae, Uropodina, humicolous arthropods).

Demographic strategies play an obvious role in the recovery phase, especially for life duration. Collembola, which reproduce rapidly, recolonize their habitats faster than many other soil arthropods. Behaviour, physiology and dispersal ability of some instars play also an important role (estivation of uropodid deutonymphs, dryness-resistant Collembola, woodmouse males, winged adult ants). Reproduction has the major part in the restoration of the populations, but sometimes, during the first stages of post-fire dynamics, population growth can be due to migrating instars (woodmice, Geophilids). Usually, instars able to burrow, to migrate down in humus, or laying their eggs in soil, especially in summer, are less affected by fire than surface dwelling animals.

- Situation in the trophic chains. The major short-term effect of fire is a temporarily development of herbaceous cover, which favours at first granivores, herbivores or root-feeders (mice, woodmice, ants, some insects, Dermaptera, and Symphylids), and then their predators (spiders, Geophilids).

The litter dwellers - predators or decomposers - feeding more or less directly on the litter, or on its organisms, are markedly reduced by fire, either by the disturbances of their habitats or of the food chains (shrews, Millipedes, Pulmonata). Soil biolytic-feeders, depending on long trophic chains are the most affected (Uropodina, Lithobids). The changes in the pyromorphic humus also induce a more or less temporary disturbance of the soil biological activity (Bacteria, Thecamoebae).

- Species richness, species replacements. Some stenotopic species disappear after fire, and may be replaced by more eurecic forms, often characteristic of open habitats. These replacements, usually transitional, are frequent in soil (Collembola, Uropodina, Myriapods, ants, Thecamoebae). The reciprocal replacements of woodmice by wild mice might also be complicated by competitive processes. In the herbivores/predators system, the species richness decreases temporarily, but the situation may recover rather rapidly, even with an increase, as for the herbaceous species. On the other hand, species richness is strongly affected in the decomposer system.

- Mesological factors. Fire induces an increase of the amplitude of climatological parameters (temperature, evaporation), that influence the demography, survival and behaviour of many invertebrates and small mammals. Erosion, slope and soil thickness, nutrient losses and chemical changes play a role in the soil organism dynamics. The season of fire occurrence is also important : winter fires having more devastating effects than summer fires, many animals reproducing during wet seasons.

As demonstrated by only a few examples, the recolonization patterns are very various and depend on many factors. However, it is possible to draw some major trends. In open habitats, characterized by a high primary production, and where herbivores and predators dominate (above-ground system), the recolonization is rather rapid, with a few disturbances in the energy pathways. On the other hand, in forest-type ecosystems, where the necromass constitutes the main resource, and is dominated by the decomposer system (below-ground fauna), fire induces strong changes in densities and community structures. The recovery will be very long, with many transitional stages, according to the gradual recovery of forest structure and functioning.

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Shrub responses to experimental fire. First phases of regeneration.

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SUMMARY - This study is devoted to the knowledge of the regeneration of shrub vegetation community after the impact of fire. Four zones have been controlled, and they are situated in the Leon province, in the north-west of Spain, between the border of influence of the evergreen-oak tree and a typical shrubland of heath in mountain region. In each of the study zones one sampling plot of 100m² was burned, after their structural composition had been evaluated by means of an analysis of their cover. Another plot which was totally cleared by cutting was used to evaluate the biomass of the different species. The first phases of regeneration of the most important species have been evaluated subsequently. The volume has been estimated, utilizing two models of approximation which answer to the geometrical figure of half of one ellipsoid and the paraboloid of revolution.

KEYWORDS : Shrub, experimental fire, regeneration, vegetation cover, heaths.

INTRODUCTION

The Advisory Commission of Spanish Technical and Scientific Research (CAICYT) during the last few years has promoted some Special Programmes and Mobilization Plans with the aim of orientating research towards priority areas or towards those insufficiently covered. The special programme of Research and Development (I + D) in Agroenergetics was decided in 1982 in reply to the energy crisis of the Seventies and began in July 1984. One of the aims is the evaluation of the potential energy of the conventional and alternative sources of the biomass. The work we present is part of that programme, having as its primary aim the evaluation of the biomass in the forms of shrub in the Spanish region of Castilla - Leon. One part of this study is devoted to the knowledge of the regeneration of the vegetation community after the impact of several degenerative and drastic processes such as rooting up, cutting or fire. This communication will be centred on the effects of the latter.

The superficies of land covered with shrub in Spain in any of its types is about 20%, corresponding to the climax communities in the supraforestal border, extended to lower altitudes because of the elimination of the forest and the effects of abusive sheep grazing or else as parts of derived regressive formations, due to the degradation of arboreal climax.

The characteristics of shrubby formations of Mediterranean climates which are adapted to more or less periodical fires, are numerous, and in some instances they even depend on fires to maintain themselves in optimal conditions of structure and functioning. The Californian chaparral (BISWELL 1974, PARSONS 1976, KEELEY & KEELEY 1981) and the garrigues in France (TRABAUD 1980, TRABAUD & LEPART 1980), in Israel (NAVEH 1974) and in Greece (PAPANASTASIS 1977, 1978), are some of the examples. In Spain, heath and Cistus form degraded communities with a good response to fire, either due to sprouts as in the case of the heath (Erica) or to seeds as in the rockroses (Cistus).

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Table I.- Average and total cover values of the shrub found before burning.

San Isidro woodland

Date of sampling JUN 1985
Date of burning MAY 1986

	n	$\bar{x}$	σ	$\bar{X}$
Genista scorpius	60	23.3	19.1	13.9
Rosa sp.	13	8.8	5.1	1.1
Halimium umbellatum	7	2.7	1.5	0.1
Crataegus monogyna	6	18.3	11.0	1.0
Quercus rotundifolia	2	10.5	9.5	0.2
Dorycnium pentaphyllum	1	15.0	-	0.1

Majada of Setibar scrub

Date of sampling JUL 1985
Date of burning JUL 1985

	n	$\bar{x}$	σ	$\bar{X}$
Erica australis	89	35.2	23.2	31.4
Calluna vulgaris	85	25.2	17.7	21.4
Erica umbellata	70	25.1	13.7	17.6
Arctostaphylos uva-ursi	67	65.0	30.5	43.5
Halimium alyssoides	53	6.8	5.9	3.6
Quercus pyrenaica	34	11.5	8.2	3.9
Halimium umbellatum	7	2.8	1.0	0.2

Cota Isestil scrub

Date of sampling JUL 1985
Date of burning JUL 1985

	n	$\bar{x}$	σ	$\bar{X}$
Erica australis	100	78.6	25.5	78.6
Calluna vulgaris	2	10.0	0.0	0.2
Halimium alyssoides	1	10.0	-	0.1

San Isidro sky station

Date of sampling JUL 1985
Date of burning JUL 1986

	n	$\bar{x}$	σ	$\bar{X}$
Calluna vulgaris	100	86.6	10.5	86.7
Vaccinium myrtillus	98	2.5	1.4	2.4
Daboecia cantabrica	8	10.0	5.0	0.8
Erica australis	7	29.2	22.9	2.0

In any case they can be considered as processes of autosuccession (GRANADOS et al. 1986) because of the rapidity with which the structure and the floristic composition recover. In the Northwest of Spain similar responses can be observed for the Ulex communities and Erica - Ulex communities (CASAL et al. 1984).

MATERIAL AND METHOD

DESCRIPTION OF THE ZONES OF STUDY

Four zones have been observed. The San Isidro woodland is situated to the north of the city of Leon, in the bordering zone of influence of the evergreen-oak grove forming a type of community characteristic of the regressive state of that arboreous dominant. Its altitude is of about 800 m. Genista scorpius and Halimium umbellatum appear as the characteristic species, accompanied by Rosa sp. and Crataegus monogyna, typical of the border of the forest, and Dorycnium pentaphyllum as nitrophilous indicator.

Near to the mountain area are situated two other zones of study. One of them, (known as Cota Isestil scrub) next to the small village of Palacios de Rueda at an altitude of 1000 m above sea level, could be phytosociologically considered as a variant of the Arctostaphylo crasifoliae-Ericetum aragonensis community, in which Erica australis subsp. aragonensis of great size appears as the dominant accompanied by Lavandula stoechas subsp. pedunculata, Halimium alyssoides, Thymus zygis, Erica umbellata and Calluna vulgaris, with a very scarce representation of all of them. It is found on very degraded marly soil within the climatic dominion of the oak tree. In this same climatic dominion and only 5 Kms away from the former is Majada of Setibar scrub, at an altitude of 1150 m. It is different from the previous one because Erica australis subsp. aragonensis, Erica umbellata, Calluna vulgaris and Arctostaphylos uva-ursi have a significative representation. They are accompanied by Chamaespartium tridentatum and Halimiun alyssoides. The sprouts of Quercus pyrenaica show a lesser degree of regression.

The fourth zone of study is situated in the San Isidro sky station, completely in the mountainous region at an altitude of about 1600 m. It is a typical shrubland of heath, very humid, and is characterized by the presence of Daboecia cantabrica, Erica australis subsp. aragonensis and Calluna vulgaris, accompanied by Vaccinium myrtillus and Cytisus scoparius, all of them with a height under 50 cm and with a very homogeneous tapestry pattern cover. Phytosociologically it is included in the Daboecio - Ericeto aragonensis - Cytisetosum scoparii community.

SAMPLING

Four sample plots of about 100 m^2 were defined in each of the study areas in the zone of shrub possessing the most homogeneous characteristics. One of them was left as a witness and the other three received different treatments after their structural composition had been evaluated, by means of an analysis of their cover in terms of their vertical projection over the ground, characterized independently in units of 1 m^2. The plot which was totally cleared by cutting was used to evaluate the biomass of the different species by means of fresh weighings immediately after each cutting. Other plots were burnt and the regeneration of the different species was evaluated at posterior dates, using various methods depending on the characteristics of the vegetation and the evolution of its growth. For those plots and for the species where the size of the sprouts was appreciable each of the plants was evaluated separately. Three conventional measurements were taken (height, maximum distances in the axis N - S and in the perpendicular). The volume was estimated with these measurements involving two models of approximation (half of one ellipsoid and paraboloid of revolution).

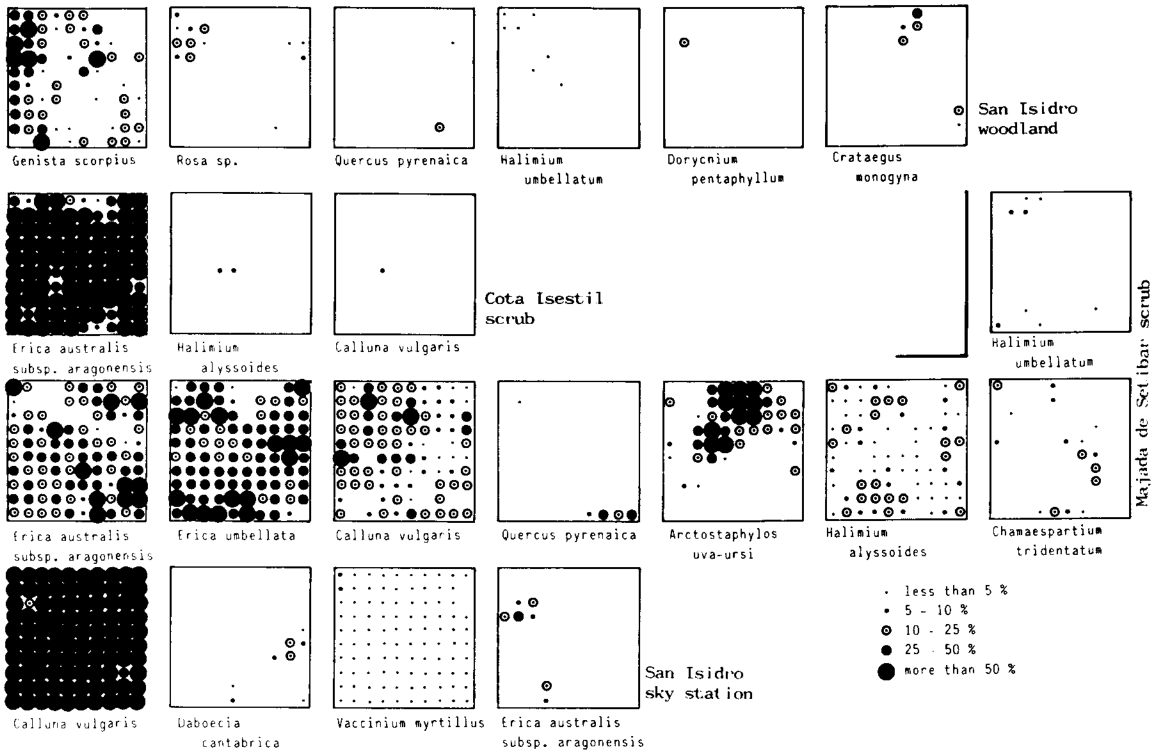

Fig. 1.- Spatial distribution and semiquantitative values of cover of the shrub species found before burning in the four controlled sampling plots.

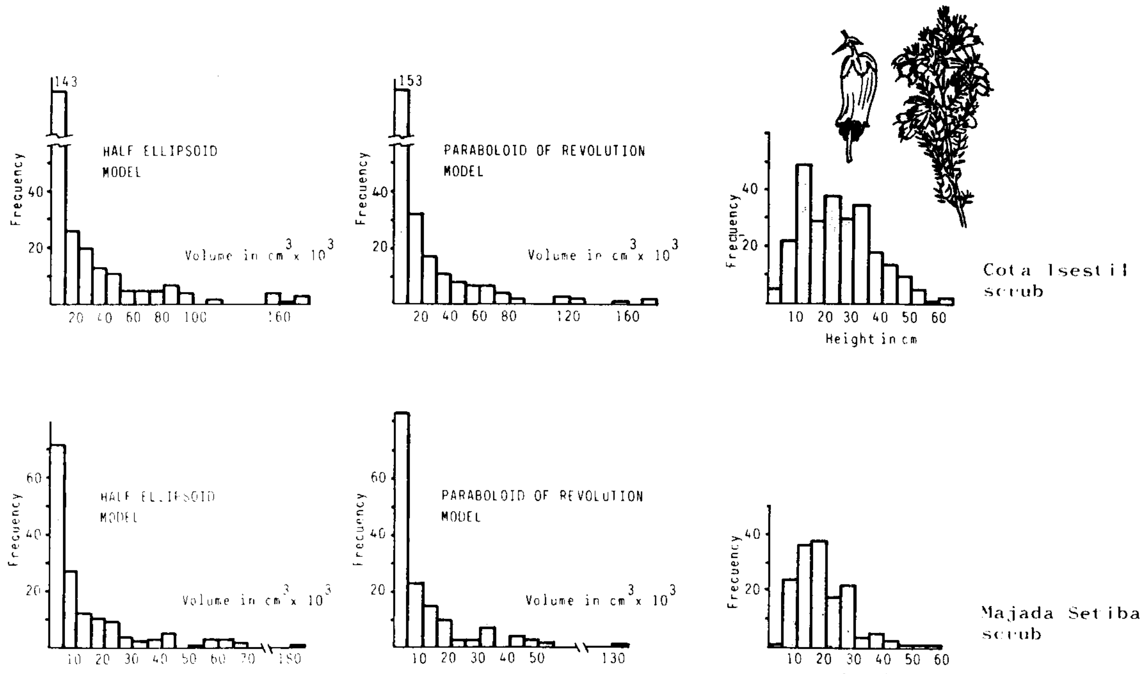

Fig. 2.- Estimated volume by different models and distribution of heights of Erica australis subsp. aragonensis in two controlled zones.

RESULTS AND DISCUSSION

ORIGINAL SITUATION

The plots burned in each of the study zones offered a semi-quantitative disposition given in Figure 1. The average cover in units in which the species are present, and the total cover of species of shrub found are shown in Table I. In the San Isidro woodland there is a clear domination of Genista scorpius, both in frequency and in abundance. The rest of the accompanying species are represented on only a small scale so that only Rosa sp. exceeds 1% of the total cover. In the Cota Isestil scrub, the species Erica australis appears in all sampling units of 1 m, reaching a cover of 78,6% in the plot, while the two accompanying species are represented by a very low presence and cover. On the contrary, in the Majada of Setibar scrub the values of importance are more divided. Erica australis, Arctostaphylos uva-ursi reaching the highest cover, with outstanding values also for Erica umbellata and Calluna vulgaris. In the San Isidro sky station Calluna vulgaris is evenly distributed while at the same time its cover is high. On the other hand, Vaccinium myrtilus, also distributed in almost all the plots, only represents 2.4% of the total cover.

Table II - Biomass values of the shrub species

	Weight in Kg/100 m^2
San Isidro woodland	
Cutting JUL. 1985	
Genista scorpius	37.7
Rosa sp.	1.0
Dorycnium pentaphyllum	0.3
Cota Isestil scrub	
Cutting JUL. 1985	
Erica australis subsp. aragonensis	165.1
Majada of Setibar scrub	
Cutting JUL. 1985	
Erica australis subsp. aragonensis	31.4
Erica umbellata	13.8
Calluna vulgaris	11.9
Arctostaphylus uva-ursi	6.8
Halimium alyssoides	1.3
Chamaespartium tridentatum	1.2
Quercus pyrenaica	0.5
Halimium umbellatum	Inap.
San Isidro sky station	
Cutting JUL. 1985	
Calluna vulgaris	117.7
Daboecia cantabrica	21.7
Vaccinium myrtillus	0.1

As far as biomass is concerned, and taking into account the species found in the plots with cleared undergrowth, the weight of the above-ground part was calculated, the values being shown in Table II. On the whole there is a good correlation between the biomass and the cover of both plots, with the only outstanding exception being *Arctostaphylos uva-ursi* in the Majada of Setibar scrub, due to the special manner of growth and distribution of this species. The importance of *Erica australis* in the Cota Isestil scrub is outstanding because of its great biomass, whilst at the time it appears in this plot as the only representative species. In the San Isidro sky station, with values also over 100 Kg, *Calluna vulgaris* shares with *Daboecia cantabrica*, although the latter to a lesser extent, practically the totality of the biomass. In the San Isidro woodland, *Genista scorpius* is the only significant representative, whilst in the Majada of Setibar scrub the proportion of weight as well as of cover are more evenly distributed amongst the different species.

RESPONSE OF REGENERATION

After the initial sampling, the measurements in successive years are to check that the regeneration of the shrubs has been carried out. Because the response in each of the zones was very different, the method of evaluation was also different and basically determined by the degree of response.

In the San Isidro woodland the presence of sprouts of any of the species was almost imperceptible. The only species in the plot that up to now show some small sprouts are *Dorycnium pentaphyllum*, *Quercus pyrenaica*, *Cistus laurifolius* and *Daphne gnidium*. In any case, due to its scarcity it can be considered that the biomass is almost non-existant.

In the Cota Isestil scrub, only small sprouts of *Erica australis* appear, basically from the base of the trunk of each of the stalks burnt, as re-sprouts of the stock, together with some much smaller sprouts, which might have come from seeds. A first sampling carried out in April 1986, taking into account the 100 units of 1 m^2 and evaluating their cover gave for *Erica australis* the following values :

n	$\bar{x}$	σ	$\bar{X}$
77	9.5	5.3	7.3

In relation to the sampling prior to the fire this means a proportion of 0.09 with regard to the total cover. The maximum height reached at that time was 40 cm, highlighting the fact that most of the burnt shrubs were re-sprouting.

A second sampling, performed in October 1986, was carried out measuring each stalk individually. Taking as a starting point the values of the projection over the ground of each stalk, the total cover has been estimated taking as the approximate model of projection the surface of an ellipse. The value thus obtained reaches a surface of 24.11 m^2, which in relation to the cover prior to the burning means a proportion of surface regeneration in the second year of 0.31. Nevertheless it must be taken into account that in the independent surface measurement per stalk, superposition has not been considered, and therefore the real proportion would be considerably smaller.

In the same way, the volume was estimated according to the models of the ellipsoid and the paraboloid of revolution. The volumes thus obtained have been grouped for both models in size types whose graphic expression is reflected in Figure 2. Taking into account the model of the ellipsoid we obtain by simple addition a total volume of 5.98 m^3 for the plot of 100 m^2, with a minimum volume per unit of 4.3 cm^3 and a maximum of 0.17 m^3. Using the model of the paraboloid, the volumes obtained are slightly lower with a total volume of 4.55 m^3. In any event the median value between these two values could be estimated. In the same way, if the initial volume prior to the fire was estimated by taking into account the cover of the plot and the average height (1.20 m), the volume which would be obtained would reach 94.32 m^3, and consequently the proportion of regeneration, considering the mean of both models, is 0.05 in volume. The distribution according to size, also expressed graphically in Figure 2, points out as the most common values those defined in the range 10-15 cm, stalks with a height inferior to 5 cm being scarce. The highest value is 61 cm.

In the Majada of Setibar scrub, as for the Cota Isestil scrub, two samplings were also carried out after the burning. In the first one performed in April 1986, the results of which are given in Table III, only two species show a significant cover. One is *Arctosthaphylos uva-ursi*, with elongated and creeping stems and persistant, coriaceous leaves, which because of its aspect and morphological characteristics resists fire well. With a total cover of 11.01; its proportion in relation to the initial situation is 0.25. The other is *Erica australis*, with a total cover of 3.22%, which therefore as regards the situation prior to burning means a proportion of 0.10. The re-sprouts are still very low, with a maximum height of 20 cm. The presence of other species is not significant.

The second sampling performed in January 1987 was carried out taking into account each stalk independently. The total surface estimated, according to the model of the ellipse, was 11.2%. The proportion as regards the original situation was 0.36 although the same conditions as for the previous plot applied regarding the superposition of the projections. For *Arctostaphylos uva-ursi* the relation is more real, due to the creeping growth, in such a way that the estimation in this sampling of a total cover of 15.0% means a proportion of regeneration which equals 0.34.

Table III. Average of total cover values of the first sampling made in the Majada of Setibar scrub.

	n	$\bar{x}$	σ	$\bar{X}$
Erica australis	57	5.64	4.49	3.22
Arctostaphylos uva-ursi	54	20.38	15.17	11.01
Erica umbellata	1	15.00	-	0.15
Halimium alyssoides	15	4.26	2.79	0.64
Chamaespartium tridentatum	1	2.00	-	0.02

The volume for *Erica australis* was also calculated in the same way starting for the two geometrical models above-mentioned (Fig. 2), obtaining for the ellipsoid a total volume of 2.07 m^3, with a maximum of 0.18 m^3 and a minimum of 12.6 cm^3. A total volume of 1.59 m^3 results for the paraboloid. The relation between the average value of these estimations and the value of the initial volume, for an average height of 0.80 m, is 0.07, a value which is quite similar to the one obtained for this same species in the previous plot.

The distribution according to height (Fig. 2) defines as the most common values those included in the range 15 to 20 cm, with only one specimen inferior to 5 cm. The maximum height reached was 59 cm.

In the San Isidro sky station, burned in July 1986, in October of that same year there were only three small sprouts of *Erica australis* with a maximum height of 5 cm.

All the plots will be checked in successive years until they reach a structure similar to the witness plot of the zone. The response of plots of similar characteristics to clearing and uprooting is referred to in other complementary works.

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ECOLOGIA MEDITERRANEA - Tome XIII (fascicule 4) - 1987

Ecological impact of prescribed winter burning on fuel breaks in french mediterranean forests. First results.

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SUMMARÝ - Changes in plant communities, litter and soil were simultaneously studied for two years after prescribed winter burning, scrub clearing and wild fire in French Mediterranean forests. Prescribed fires did not fundamentally change community structure, understorey specific composition and soil microclimate, which contrasts with heavy consequences observed after wild fire on the same ecosystem components. Yet, consequences on soil microclimate, soil arthropods and recovery of the understorey were slightly different after burning or clearing. In particular, grasses were generally more depressed and shrub vegetation recovered faster after burning than after clearing.

KEYWORDS : Prescribed burning, wild fire, scrub clearing, understorey vegetation, soil microclimate, littler, arthropods.

INTRODUCTION

Various experimental prescribed winter fires were carried out on fuel-breaks in some French Mediterranean forest communities. These fuel-breaks are generally cleared of scrub every 3-5 years. The ecological impact of prescribed burning is compared with the effect of understorey mechanical clipping, which is the commonly used method to reduce fuel loads on fuel-breaks.

The research was conducted during two years in two forest communities which are typical of those occupying the large calcareous areas of the French Mediterranean area. Three sites belong to the same community dominated by an overstorey of Quercus pubescens. In the Artigues site, three methods of prescribed fires were carried out on the same underlayer vegetation dominated by the grass Brachypodium pinnatum. At Ollières, two fires were carried out in two adjacent sites : site A with dominant grass vegetation, site B, with dominant shrub vegetation of Phillyrea latifolia. At the La Roque Anthéron site, the overstorey is dominated by Pinus halepensis and the understorey by a scrubby layer of Quercus coccifera and the grass B. ramosum. Experimental fires were carried out in March and April 1984 and 1985.

Similar studies were carried out after a wild fire (July 1984) in mixed Pinus halepensis - Quercus pubescens forest at the Trets site to compare the consequences of wild fire and prescribed burning on the components of similar ecosystems.

Changes in plant communities, aboveground arthropods, litter, soil (microclimate, mesofauna and organic matter) were studied.

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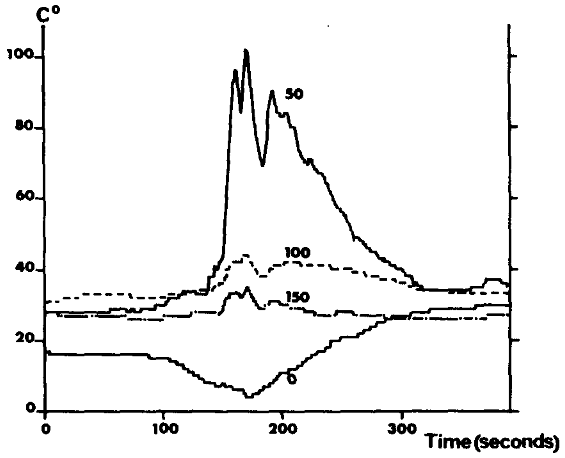

Fig. 1 Temperature at the soil surface (0) and at 50 cm, 100 cm and 150 cm heights during the prescribed fire in plot 3 at Artigues (March 1984).

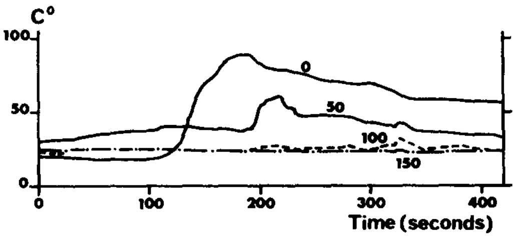

Fig. 2 Temperature at the soil surface (0) and at 50 cm, 100 cm and 150 cm heights during the prescribed fire at La Roque d'Anthéron (March 1984).

	Cleared			Prescribed burnt								
				plot n°3			plot n°4			plot n°5		
	1	3	4	1	3	4	1	3	4	1	3	4
Brachypodium pinnatum	17	23	22	99	55	61	30	32	33	71	58	53
Other grasses	12	15	14	16	32	28	24	17	11	27	38	44
Total grasses	29	38	36	115	87	89	54	49	44	98	96	97
Forbs	7	10	17	2	11	4	5	17	16	3	1	2
Total herbaceous species	36	48	53	117	98	93	59	69	60	102	105	103
Total shrub species	4	7	7	7	3	5	0	3	4	1	0	5
TOTAL	40	55	60	124	101	98	59	72	64	103	105	108

Table 1 - Phytomass index of the understorey (number of contacts in a 10 m line-transect) in adjacent plots subjected to scrub clearing and prescribed fires (plots n°3, 4 and 5) before treatments in March 1984 (1), and after treatments in June 1985 (3), June 1986 (4) at Artigues.

CHARACTERISATION OF PRESCRIBED WINTER FIRES

According to the burning technique used and climatic conditions, the mean rate of spread was 10-30 m h^{-1} at Artigues, 10 m h^{-1} at La Roque d'Anthéron and 150-180 m h^{-1} at Ollières.

Spatial variability in heating during fires occurred vertically and horizontally. During the burning at La Roque d'Anthéron, the highest temperatures were recorded at ground level reflecting the fact that the main fuel was leaf litter. At Artigues, the highest temperatures occurred at 0.5 m above the ground, meaning that this was a partial ground cover burn rather than a low level litter burn.

In Quercus pubescens forest (Artigues), fuel weight was 14 to 18 t ha^{-1} and temperatures reached 40°C to 124°C at 0.5 m above the ground, and 30°C to 40°C at ground level. Time during which temperatures exceeded 70° (lethal temperature) at 0.5 m above the ground lasted 24 to 70 seconds (Fig. 1). In the Pinus halepensis forest (La Roque d'Anthéron), fuel weight was 9 t ha^{-1} and temperatures reached 50°C to 60°C at 0.5 m height and 90°C to 305°C at the soil surface. Time during which temperature at ground level exceeded 70°C lasted more than 2 minutes (Fig. 2).

These vertical temperature profiles show that prescribed fires did not much affect the trees directly. But fires need to be defined in terms relevant to their impact on the soil, especially on the surface horizons where reserves of organic matter and nutrients and biological activities tend to be most concentrated. We lack data on temperatures below the soil surface during prescribed fires.

IMMEDIATE EFFECTS OF PRESCRIBED FIRES

Prescribed fires only affected the understorey, the soil litter and maybe the top 1-2 cm of the soil in some cases.

The above-ground herbaceous vegetation was entirely burned during prescribed fires and only the largest shrub stems remained after burning.

The litter layer was partly burned. During two different prescribed fires in the Quercus pubescens forest at Ollières, 40% and 60% of the total litter weight burned, mainly the recently fallen litter layer (65% and 75%).

Another evidence of low soil heating was given by soil microarthropods. When prescribed burnings were carried out, they were numerous and active (winter). No significant change in their number could be noted before and after burning.

The direct effects of wild fire at Trets were much more intense : the whole above-ground vegetation and soil littler burned. There only remained ashes on the soil and blackened trunks and branches of trees and shrubs after this fire.

CONSEQUENCES OF PRESCRIBED FIRES

1 - On the plant community

a) Trees

In the Quercus pubescens community at Artigues, total tree mortality during two years after the treatments was greater in the three burnt plots (8%, 14% and 17%) than in the mechanically cleared plot (4%). Mortality mainly affected the little trees (12-16 cm in circumference at breast height). However, the growing rate of the surviving trees was similar after clearing and burning.

In Pinus halepensis forest (La Roque d'Anthéron), no trees died during the two years after burning and their growing rate was similar in all treatments.

	Cleared				Burnt			
	1	2	3	4	1	2	3	4
Brachypodium ramosum	68	54	49	57	139	81	82	110
Other grasses	2	0	1	0	0	0	0	0
Total grasses	70	54	50	57	139	81	82	110
Forbs	4	10	9	11	5	5	11	14
Total herbaceous species	74	64	59	68	144	86	93	124
Quercus coccifera	27	2	13	22	63	18	60	70
Other shrub species	4	5	3	3	4	1	4	12
Total shrub species	31	7	16	25	67	19	64	82
TOTAL	105	71	75	93	211	105	157	206

Table 2 - Phytomass index of the understorey in adjacent plots subjected to prescribed burning and scrub clearing before treatments in March 1984 (1) and after treatments in June 1984 (2), June 1985 (3) and June 1986 (4) at La Roque d'Anthéron.

	Cleared			Burnt		
	1	2	3	1	2	3
Brachypodium pinnatum	58	18	17	84	30	45
Other grasses	34	34	50	18	3	12
Total grasses	92	52	67	102	33	57
Forbs	5	14	3	7	15	5
Total herbaceous species	97	66	70	109	48	62
Total shrub species	0	0	0	1	5	7
TOTAL	97	66	70	110	53	69

Table 3 Phytomass index of the understorey in adjacent plots subjected to prescribed burning and scrub clearing before treatments in March 1985 (1) and after treatments in June 1985 (2) and June 1986 (3) in site A at Ollières.

	Cleared			Burnt		
	1	2	3	1	2	3
Phillyrea latifolia	1182	204	459	885	58	293
Ligustrum vulgare	113	35	67	171	33	124
Acer monspessulanum (shoots)	90	87	194	401	78	258
Quercus ilex (shoots)	62	0	37	48	9	14
Quercus pubescens (shoots)	5	16	43	55	208	575
Other woody species	0	2	2	3	27	2	8
TOTAL Woody species	1452	343	802	1586	388	1272

Table 4 Phytovolume of woody understorey in dm^3 by 10m^2 (0,5 m x 20 m transect) in adjacent plots subjected to prescribed burning and scrub clearing before treatments in March 1985 (1) and after treatments in June 1985 (2) and June 1986 (3) in site B at Ollières.

b) Understorey

After each of the prescribed fires, understorey vegetation recovered rapidly. Two months after fires, vegetation cover was 48% of the pre-fire level in Quercus pubescens forest (Ollières site A) and 50% in Pinus halepensis forest (La Roque d'Anthéron) (tables 2 and 3). The shrub understorey of site B at Ollières reached 24% of its pre-fire volume (table 4). Then, phytomass progressively increased from growing season to the other. Total initial phytomass recovered in plots 4 and 5 at Artigues only after two growing seasons (table 1). In the other cases, recovery was still not complete after three growing seasons.

In the presented dominant grass understorey at Artigues and in site A of Ollières, forbs presented a clear increment during the first year, mainly due to legumes. Then, grasses progressively recovered their pre-fire dominant place; however, some changes in their relative abundance were still to be observed after three growing seasons. Further measurements have to be made.

Most woody species of the understorey grew very quickly after prescribed fires. For instance Quercus pubescens shoots took up a volume 4 times higher than initially after the first growing season at site B. Two years after fire, they reached 10 times their pre-fire level. Quercus coccifera recovered its pre-fire phytomass during the second growing season and exceeded it after only three years. Other shrubs were generally characterized by lower growth rates.

In fact, prescribed fires changed the floristic composition of the community for only a very short time.

c) Comparison with mechanical scrub clearing

Two months after clearing global understorey cover was slightly greater than after burning. Thus, 68% of the initial phytomass was found in site A and at La Roque d'Anthéron against 48% and 50% respectively after burning (see above).

Herbaceous vegetation is less affected by clearing than by burning. Two months after treatments, 68% and 86% of the initial herbaceous phytomass were found in cleared plots in site A and at La Roque d'Anthéron respectively, against 44% and 60% in burnt plots. Afterwards, herbaceous biomass, mostly composed of grasses, higher after clearing than after burning. Shrub species, on the contrary, are more affected by clearing. Actually, Quercus coccifera which recovered completely after two growing seasons in the burnt plot, recovered only 80% of its pre-fire biomass after 3 years when cleared. Identically, after the second growing season, the cleared woody understorey of site B took up only 55% of its initial volume against 80% for the burnt plot.

d) Comparison with wild fire at Trets

Wild fire completely modified the forest structure at Trets. In the unburnt forest, tree and shrub cover is more than 100% of the soil surface and herbaceous ground cover is only 6%. In burnt area, the overstorey completely disappeared and ground vegetation covered 38% of the soil surface one year after fire, and 67% after two years.

The burnt vegetation consisted of shoots and seedlings from trees and shrubs and an abundant herbaceous layer. Seventeen of the 38 species belonging to the pre-fire community grew again during the first year after fire, and 20 during the second year. Moreover, many new species appeared in the burnt area, but sometimes for one year only. Thus, 52 species were found in July 1985, one year after fire, and 54 in 1986 of which 42 are common between the two years. This reflects the changing composition of the ground vegetation after wild fire.

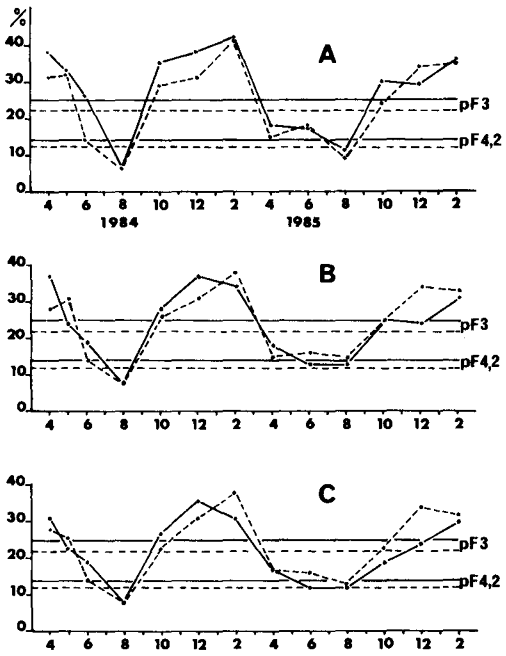

Fig. 3 Soil moisture content (% g water/g dry soil) of 0-3 cm (A), 3-6 cm (B) and 6-10 cm (C) soil layers from adjacent plots subjected in March 1984 to prescribed burning (solid lines) and scrub clearing (dashed lines) at La Roque d'Anthéron. Differences between plots were always insignificant (F-test, P < 0.05).

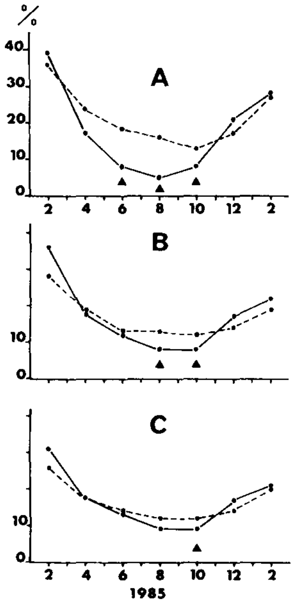

Fig. 4 Soil moisture content 5% g water/dry soil) of 0-3 cm (A), 3-6 cm (B) and 6-10 cm (C) soil layers from area subjected to wild fire in July 1984 (solid lines) and control area (dashed lines) in the forest of Trets. Differences between plots were always insignificant (F-test, P< 0.05) except for some sampling dates (triangle symbols).

2 - On soil microclimate

The moisture content of 0-10 soil layers from prescribed burnt plots was not significantly lower than that of soil from cleared plots during two years after treatment (Fig. 3). In contrast, for the same depths, soil temperatures on burnt plots were higher during summer and lower during winter than on cleared plots. Thus, during the summer after treatment, the observed increases in maximum temperatures were 2-5°C at 2cm, 3-6°C at 10 cm depth at Ollières (fig. 5).

Soil microclimate was more affected after wild fire at Trets (fig. 4). During summer, the moisture content was significantly lower than in the control plot and maximum temperature increased by 17°C at 2 cm and 6°C at 10 cm depth.

The lack of consistently significant differences in moisture content between prescribed burnt plots and cleared plots means that differences in diurnal temperature patterns measured after prescribed fires may be due to increased exposure to radiation rather than to changes in the heat storage or heat transmission characteristics of the soil as seems to be the case after wild fire.

3 - On soil litter

Changes in the forest floor and leaf decay were measured during one year after treatment in sites A and B at Ollières. The mass of litter on the forest floor was measured by harvesting 10 x $0.1m^2$ quadrats randomly located on cleared and burnt plots every four months. Loss in leaf weight was measured by the litter bag method.

In site A, where the understorey is mainly herbaceous, prescribed fire burned 60% of soil litter mass. From April after burning and clearing, to August without any litter input, soil litter weight decreased in the two plots, but significantly less in the burnt plot; thus post-fire difference between plots decreased during this period (Fig. 6). From August to December, with inputs of the starting litter fall, soil litter weight increased in two plots, so post-fire differences between plots completely disappeared. Afterwards, from December to April, one year after treatment, inputs of litter fall balanced disappearance of forest floor, and soil litter weight remained relatively constant in the two plots.

In site B where the understorey is dominated by shrub vegetation, prescribed fire burnt only 40% of soil litter mass. During the first 4 months, litter weight decrease was greater in burnt than in cleared plot, so post-fire difference between plots was emphasized. In December, with the beginning of litter fall, the same quantities of soil litter were found in the two plots. But, from December to April, the rate of disappearance of forest floor was again greater in burnt than in cleared plot, so one year after treatment, the same post-fire difference in soil litter weight was observed.

Leaf decay measures gave the same kind of results (Fig. 7). In site A, loss in weight of leaves in litter bags was slightly less on burnt than on cleared plot during the first months after treatment; afterwards, it was the same in the two plots. In site B, loss in weight of leaves was always greater than in cleared plot.

Two opposite kinds of burning consequences on soil litter disappearance were observed after these two different prescribed fires in two adjacent understorey communities. One fire directly affected forest floor so that litter decay rate was reduced for only some months. The other fire burnt less litter; consequently litter decay rate was emphasized during all the following year. Apparently conflicting conclusions about fire effects on soil litter decomposition need further studies.

In the Trets forest, there was no difference in loss in weight of leaves in litter bags between burnt and unburnt areas (fig. 7). Forest floor was accumulating from April 1985 to April 1986 in the unburnt forest; in the same way, soil litter increased but remained very poor in the burnt area during the two studied years after wild fire.

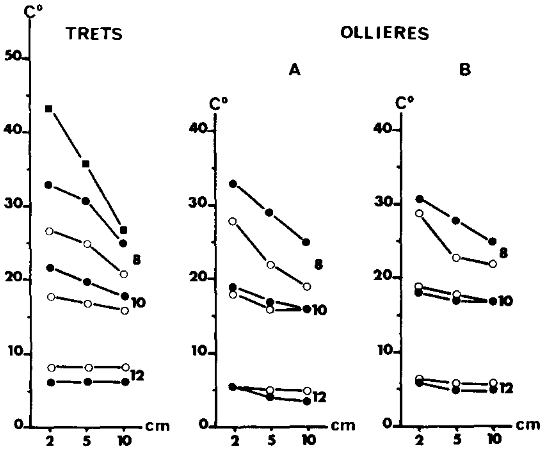

Fig. 5 Soil temperatures at 13.00 h in shady situation at 2 cm, 5 cm and 10 cm depths in burnt plots (closed circular symbols) and unburnt plots (open circular symbols) in August (8), October (10) and December (12) 1985. In burnt area of Trets, soil temperatures were also measured in typical sunny situation (closed square symbols). Measurements were taken 13,15 and 17 months after wild fire at Trets and 4, 6, 8 months after prescribed burnings at Ollières.

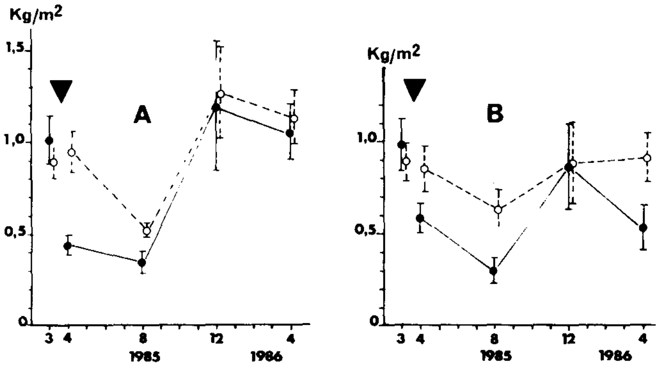

Fig. 6 Changes in soil litter weight from adjacent plots subjected to prescribed burning (closed symbols) and scrub clearing (open symbols) in sites A and B at Ollières. Treatment date is indicated by triangle symbol.

	Burnt	Cleared
Locusts	112	33
Plant hoppers	41	19
Buprestid	33	8
Hymenoptera	14	13
Plant bugs	7	14
Spiders	141	151
Centipedes	13	19
Cockroaches	5	6
Harvestmen	16	66
Crickets	11	50
Diptera	23	56
Ground beetles	40	120
Rove beetles	65	147
Other beetles	80	347
Pill millipedes	7	153
Julid millipedes	406	718
Moths (larvae)	35	405
Others	9	8
Total number	1 058	2 233

Table 5 - Total number of trapped arthropods. Data concern ten permanent pitfall traps settled during a 5 months period (from 18.06.1985 to 18.11.1985) both in burnt and cleared plots in site A at Ollières (treatments in April 1985).

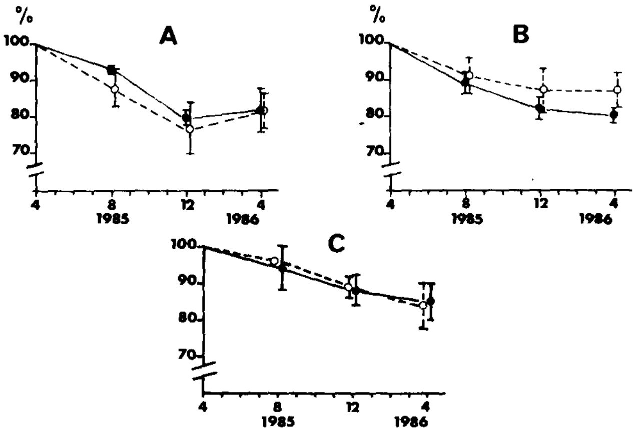

Fig. 7 The loss in weight of Quercus pubescens leaves in burnt plots (closed symbols) and unburnt plots (open symbols) on sites A and B at Ollières and in forest of Trets (C).

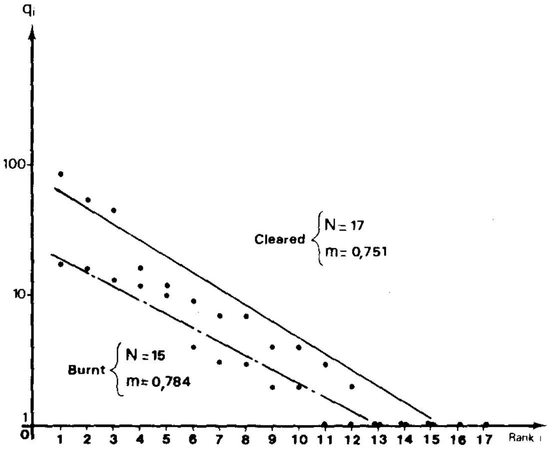

Fig. 8 Adjustment to a Motomura's model of insect number distribution both in burnt and cleared plots in site A at Ollières. Data concern May 1985 period, two months after the burning process. i : rank of species, q_i : number of individuals in rank i of species , N : total number of species, m : environmental Motomura's constant.

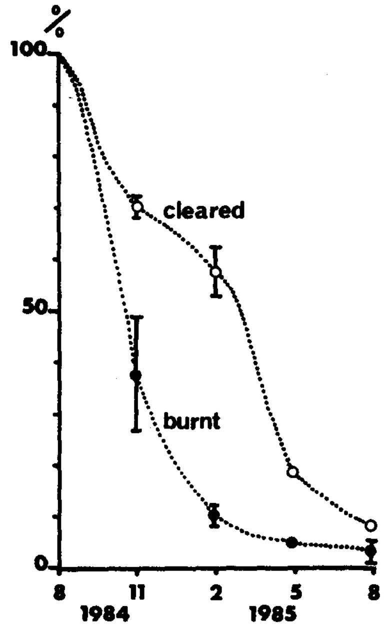

Fig. 9 The loss in weight of cellulose samples, deposited in soil in August 1984, in adjacent plots subjected to prescribed burning (closed symbols) and scrub clearing (open symbols) in March 1984 at Artigues.

	Burnt	Spared
Locusts	270	0
Plant hoppers	118	11
Plant bugs	103	9
Lepidoptera	24	2
Buprestid	19	0
Harvestmen	187	6
Hymenoptera	25	7
Spiders	178	203
Diptera	27	35
Pill bugs	2	16
Pseudoscorpion	3	17
Cockroaches	13	38
Pill millipedes	17	61
Julid millipedes	9	43
Rove beetles	13	63
Crickets	12	178
Other beetles	147	26
Total number	1 180	782

Table 6 - Total number of trapped organisms. Data concern ten permanent pitfall traps settled during a 5 months period (from 18.06.1985 to 18.11.1985) both in burnt and spared forest of Trets (wildfire in July 1984).

4 - On above-ground anthropods

Modifications of litter quantity and vegetation structure consecutive to burning process involve several microclimate alterations at the soil surface (temperature, luminosity, moisture) which can modify environmental functioning. Arthropods integrate these adjustments very well; so that an exhaustive analysis of their number and of variation in their organisation structure would give good information on the immediate and long term effects of burning on above-ground environment.

Two complementary methods were used. The first consisted in settlement of sets of ten pitfall traps, both in burnt and in adjacent cleared plots. This method was used during the first summer after the technical interventions were carried out at the experimental sites, and after wild fire in the Trets forest. The traps, filled up with a preserving liquid (picrate), intercepted the surface moving fauna. The second trapping method consisted in dung-baited pitfall traps for flying insects, the taxonomic group collected depending on the type of lure used. Three standard traps for dung beetles were used in each experimental plot during two consecutive years after each prescribed winter burning. Each seasonal trapping period lasted one or two weeks.

Table 5 shows that modifications did occur in the moving fauna consequently to a prescribed winter burning. Total number of organisms diminished by half, and their general organisation was modified. However the global faunistic composition was not altered. Both heliophilous and phytophagous insect groups were favoured, as they either feed on young plant tissue or they oviposit inside burned wood. It is the case of locusts, bugs and buprestids. On the contrary, the microclimate changes due to partial removal of leaf litter damage many predatory and detritivorous groups. This is the case for ground beetles, rove beetles, crickets and pill millipieds (i.e. Glomeris marginata). Abundance of black snake millipedes (Iulus) both in burnt and cleared plots reveals their high migratory activity from the edges of the fuel-breaks. However, they seem to avoid partly burnt areas.

It is of particular interest to examine the case of moths which were scarce in burnt plots but abundant in the cleared plots. These insects were trapped as caterpillars and probably fell down from the trees onto the ground. One may question whether the burning process had an influence upon tree insects. At least, the environmental changes did not at all affect other Arthropods, such as Arachnids.

There is some interest to examine the effects of a controlled burning on the structure and organisation of a specific insect guild, as that of dung beetles. All the species have indeed similar feeding habits but show different responses to microclimate and to site openness.

In some cases, i.e. plots in the Quercus pubescens forest of Artigues and Ollières, a controlled winter burning was shown to affect significantly the level of Scarabeid populations only while a short decrease of the most forestal or hygrophilous species was observed. It was the case of Onthophagus coenobita and verticicornis. On the other hand, an increase of the most xeric and heliophilous species (i.e. O. lemur)was registered.

Figure 8 shows that, after two months, the curves representing the insect number distributions (Motomura log-linear model), respectively in the burnt and cleared plots, became parallel and very close to each other; in the same way, the values of "m" (Motomura's constant) were very similar (0.751 and 0.784 respectively). One year later, the differences between burnt and cleared plots were still significant, but less important.

Wild fire in a forest is very different. Despite the fact that one year after the fire took place more organisms could be trapped in the burnt areas than in the unburnt, fauna appeared to be strongly unbalanced. In the unburnt forest, the soil surface moving fauna is indeed mostly constituted of woodlice, millipedes, cockroaches, crickets, rove beetles and arachnids (80% of total trapped organisms). Most of them are detritivorous or predatory groups linked to leaf litter (Table 6). One year after the fire, most of the groups limited to leaf litter, except the arachnids, had greatly decreased. The heliophilous and phytophagous groups replaced them, arriving simultaneously with the newly growing plant species. The most common insects were locusts, plant bugs and buprestids as in controlled burnt plots.

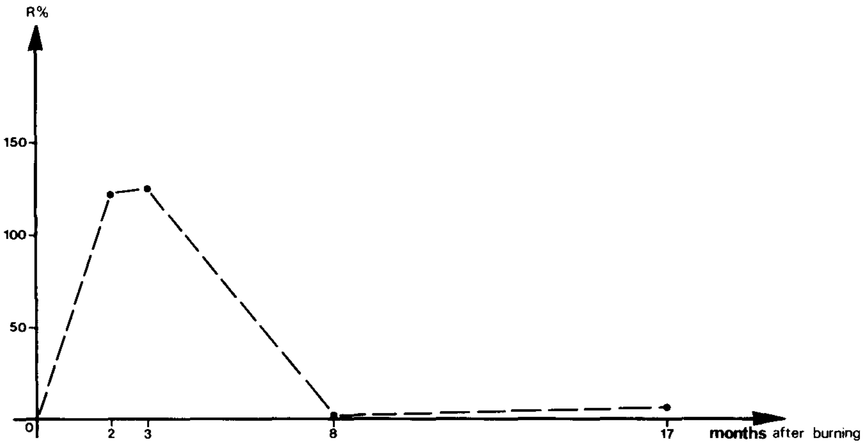

Fig. 10 Changes in soil mesofauna after treatments at Artigues: differences between plots expressed as $R = \% \frac{Nb - Nc}{Nc}$

(N_b = total number of individuals in burnt plot, Nc = total number in cleared plot).

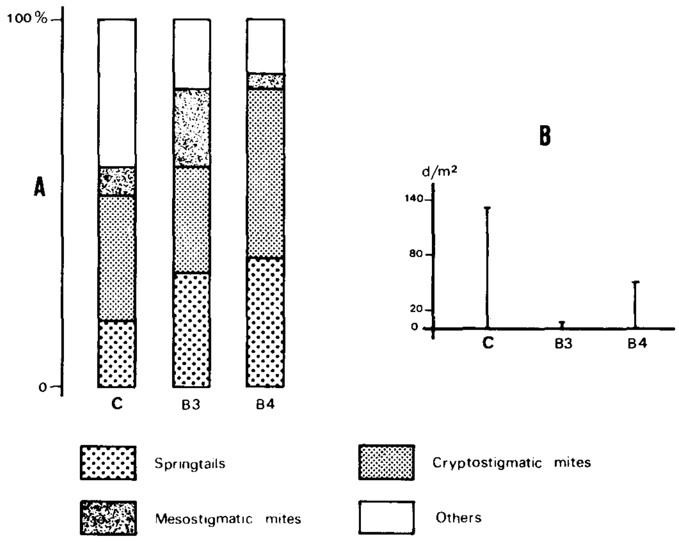

Fig. 11 Litter mesofauna 17 months after treatments at Artigues.

A : Relative percentage of the main taxonomic groups in cleared (c) and burnt plots (B3 and B4)

B : Number of insect larvae in cleared (c) and burnt plots (B3, B4).

5 - On soil cellulolyse activity

Cellulose samples (without ash cellulose paper), enclosed in wire bags, were deposited in the soil (5 cm depth) of studied plots. In all cases, loss in weight of cellulose samples was faster in prescribed burnt than in cleared plots (fig. 9).

Cellulose degradation is mainly due to soil microorganisms. Stimulation of their activity may be due to several causes:

. burning may reduce food resources for microorganisms but may also stimulate plant growth and rhizosphere activity by ash deposition.

. the amplitude of soil microclimatic variations is emphasized by the reduction of the litter; this may stimulate microflora activity.

6 - On soil organic matter

No difference was found in soil organic matter and soil nitrogen (0-10 cm) between prescribed burnt and cleared plots during two years after treatment. In contrast, compared with unburnt forest levels (12-18%) lower rates of soil organic matter and nitrogen were measured in the burnt area during one year after the wild fire in the Trets forest (8-11%).

7 - On soil mesofauna

Soil mesofauna (mites and springtails) are unable to escape from disturbances and to shelter themselves from new mesologic conditions. Thus, it is usually admitted that microarthropod numbers depend on resources level, and community structure on climatic conditions. They therefore give information on changes in soil environment conditions caused by valuable fires.

In April 1985, 9 months after the Trets wild fire, soil arthropod populations were 20 times less numerous than in unburnt forest. This means that wild fire must have caused immediate mortality for a great number of soil arthropods. They therefore give valuable information on changes in soil environment conditions caused by fires. Afterwards, pioneer groups ("r-selected" organisms), able to develop their populations from a small number of survivors especially increased. Thus, one month later the rate of increase of springtail populations was 30% for cryptostigmatic mites, 50% for endeostigmatic and 150 to 200% for prostigmatic mites (numerically important in open sites).

Controlled winter burning did not induce such a mortality. On the contrary, 2 months after fire, edaphic populations were more numerous than in the cleared plot at Artigues. Afterwards, differences between prescribed burnt and cleared plots decreased and 8 to 10 months after treatment, these differences were no longer significant (fig. 10). At this time, there was no difference in community structure between the two plots.

Litter mesofauna was more directly affected by prescribed burning than soil mesofauna and remained unbalanced longer. Thus, 17 months after the Artigues burning, populations density was still greater than in cleared plot and community structure was different. So, a greater proportion of springtails and cryptostigmatic mites was registered than in the cleared plot and, on the contrary, a lower rate of insect larvae (fig. 11).

CONCLUSION

1 - Prescribed burning versus wild fire

The similar measures carried out during the same period after prescribed burns and wild fire showed that:

. Prescribed fires only concerned the understorey and a part of the soil litter while wild fire affected the whole forest community and the total forest floor.

. Heat from prescribed fires did not affect soil organic matter and soil nitrogen as observed after wild fire in the Trets forest.

. Consequently, these prescribed fires did not fundamentally change community structure, understorey specific composition and soil microclimate which contrasts with the heavy consequences observed after the Trets wild fire on the same ecosystem components. Disappearance of trees with, as a consequence, lack of leaf litter for several seasons, bare and unprotected soil and total openness of the site, modify the functioning of the ecosystem and the inter-relationship between its associated edaphic and surface soil fauna.

2 - Prescribed burning versus scrub clearing

French Mediterranean fuel-breaks are generally scrub cleared every 3, 4 or 5 years, (studied understorey is resulting from this repeated treatment). The same measures carried out on these particular forest communities after prescribed fire and clearing showed that:

. The recovery rate of understorey was slightly different after burning and clearing : grasses were generally more depressed after burning than after clearing; on the contrary, shrub vegetation recovered faster in burnt than in cleared plots. However, there were fundamentally the same pretreatment communities that tended to recover and the observed differences in recovery were mostly due to initial composition rather than to treatment.

. Scrub clearing did not affect soil litter but prescribed fires partly burnt this litter. Soil microclimate fluctuations, mainly surface soil temperatures, were slightly greater in burnt than in cleared plots and may be due to a reduction in depth of the litter layer. This may have important effects on biological processes in these soil horizons; thus, mineralization of cellulose was increased in burnt plots; decay rate of leaf litter and accumulation patterns of soil litter were modified after burning.

. Microclimatic changes appear to be the major cause for the main disturbances observed after the burning event in surface soil fauna and litter mesofauna.

Repeated prescribed fires would probably have other consequences on the forest ecosystems by cumulative losses of volatile elements, by reduction in the content of organic matter in the surface soil with detrimental effects on soil physical characteristics, and by reduction of food supply of the litter and soil fauna and microflora. If prescribed burning is used in the management of forests to reduce fuel and prevent wild fires, the impact of repeated low intensity fires on these factors has to be studied.

On the whole, these prescribed fires led to a depression of grasses and emphasized shrub growth, making future grazing problematic. Thus, it seems that fires used to facilitate grazing could induce more and more successive fires. Further studies are needed about the effects of grazing pressure on the recovery of forest understorey after successive burning.

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Preliminary results of a study on short term effects of prescribed fire in pine stands in NW Spain.

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SUMMARY - Short term effects of prescribed burning experiments carried out in P. pinaster stands in the N.W. of Spain on fuels; soil nutrients, soil microflora and forest floor mesofauna were studied.

Fire intensities range from 21 to 420 $Kw.m.^{-1}$ *The mean maximum temperature recorded on the litter duff interface was 408°C; the temperature was greater than 60°C for about 5 minutes (mean value). On the duff-soil interface, 59° was reached and 21°C was measured 2,5 cm below the top of the mineral soil.*

Fires consumed 32 per cent of the total fuel and 55 per cent of the available fuel. Two years after the fires, the total fuel was practically equal to the preburn load but the available fuel represented 64 per cent of its initial load.

Changes in soil (0-5cm layers) nutrients were very small. Soon after the fires a little increment for the organic matter content, a small drop in pH and an increment in K, ca, Mg and P were detected. Nitrogen content was practically equal to pre-fire level. Two years later, all the nutrients remained at the same initial level, except for Ca which was 65 per cent higher than before burning. The organic matter per cent was a little smaller than before fire but pH was higher.

Some groups of the soil microflora, especially aerobic free living nitrogen fixers, were enhanced by the fire. The others remained at the same level as in unburnt areas.

Forest floor mesofauna populations were affected by fire and depending on groups, duff heating and duff consumption, their recovery lasted from six months to three and a half years.

KEYWORDS : Prescribed fire, prescribed burning, effects of prescribed fire, fuel reduction, soil nutrients, soil microflora, forest floor mesofauna, fire temperatures.

INTRODUCTION

Many very destructive wild fires have been affecting the North West of Spain in recent years. In 1985 more than 100,000 ha of forested and unforested lands were burned in Galicia. Because of this region's mild temperature and high precipitation tion, reforestation has one of the fastest growing rates in Europe, including very fast growing rates for some understorey species too.

Fuel accumulation has become a problem, especially in many forest plantations. *Pinus pinaster* has been the main species for reforestation. Even though most of its stands remain very dense, its open canopy lets light and water reach the ground level. Consequently, some shrub species such as *Ulex Erica australis, E. arborea,* and *Rubus* sp., which are basically components of the seral communities of heathlands existing before the pine plantations, can continue to develop fairly well under *P. pinaster* closure.

For many years there has been a policy of total exclusion of fire and grazing so that necromass and biomass have accumulated to a high level. Prescribed burning and grazing are now being tested on an experimental scale as a means to reduce fuel load, decrease fire hazard, to enhance palatability of vegetation regrowth, and to create microhabitats for domestic and wild animals.

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This paper summarizes some preliminary data about short term effects of prescribed fire on soil nutrients, biomass and necromass loading, and soil microflora and mesofauna populations in P. pinaster stands in Galicia.

STUDY SITES AND METHODS

Six locations were selected for this study. Two of them were in Pontevedra province, on soils derived from granitic bedrock. The area has a mild Atlantic climate with an annual mean temperature of 15°C., 1600 mm of annual precipitation, and one month of drought (sensu Gaussen). Pteridium aquilinum cinerea, and Daboecia polifolia were present. The other four sites were in Lugo province. Two of them are in the SE corner, on soil with schist parent material in a Mediterranean climate area, receiving about 850 mm of annual rainfall and three months of summer drought. Chamaespartium tridentatum is the dominant shrub species in the understorey. Arbutus unedo, Cistus populifolius, Cistus psilosepalus, Lavandula stoechas and Phyllirea sp. are other common species. Finally, the other two sites are situated in the province's central plateau (Terra cha'), on hydromorphic soils with a conglomerate bedrock. The area has a mean annual temperature of about 11°C, 1100 mm of rainfall and approximately two months of drought. Major understorey woody species are Ulex gallii, Erica australis, Chamaespartium tridentatum, Calluna vulgaris, Erica arborea and Rubus sp. The overstorey species in all cases is Pinus pinaster, more than 30 years old, with densities of 800 to 1600 trees per hectare.

Six plots were set up in each location, three for burning and the other three for control. Plots were quadrangular, from 900 to 1300 square metre in size. Prior to the first, a fuel inventory was carried out on the plots to be burned. A two-stage (systematical and random) ranked sampling (McINTYRE 1952) was used for biomass sampling and dead and down fuels. Thirty six samples per plot were selected and a 1 square metre quadrat was used to collect the fuel at each sampling point. This was classified by composition, oven dried, and weighed. Fuel was separated into biomass and dead and down material. The litter was separated, and twigs, limbs, bark portions, logs, and cones were weighed by size classes in 1, 10 and 100 hours time-lag fractions, i.e. dead fuel fractions with 1, 10 or 100 hours of time-lag. Fuel sampling was completed with a duff inventory. A 25 x 25 cm quadrat was used on a systematically selected point inside each square metre plot. After the litter was picked up, duff was removed, air dried, sieved to segregate small stones, oven dried, and weighed. A subsample was burned in a muffle furnace to calculate the mineral soil particle weight and recalculate the real organic duff weight. Fuel inventories were repeated immediately after the fires, and two years later, for the eighteen burnt plots.

Before the fires, fifty soil cores, 0-5 cm in depth and 8 cm in diameter, were removed from plots to be burned, following a grid over the plots. Each ten collected cores were bulked to give five composite samples per plot. More samples were collected close to the preceding points immediately after the first, and also two years later. The samples were air dried, then passed through a 2 mm sieve, and pH was measured (1:2.5 soil-water mixture). Total Nitrogen was determined by Kjeldahl semimicro; exchangeable potassium, calcium and magnesium were determined by atomic absorption spectrophotometry.. Available phosporus was determined by Bray-2, and organic matter was determined by the Walkey and Black method. Analyses were carried out immediately after burning, and two years later.

Three sites were selected for the soil microflora study. One site (Cerponzones), on a granitic substrate, had plots burned in a moderate low linear intensity range (60 - 105 Kw/m). The other (Vilacha'), on a schist bedrock, was burned with a moderately high level of intensity (400-600 Kw/m). The third site (Pedrido), on a soil derived from conglomerates, was burned with a moderately low intensity (60 - 105 Kw/m). At each location, two burned and two unburned plots were chosen. Control plots were selected at random. The criterion for selecting the burned plots was that they were to have two levels of fuel consumption. These plots were seasonally sampled after the fires for two years. Eight samples per plot were randomly taken and were bulked to give one composite sample per plot. A previously sterilized metal cylindrical soil sampler (5 cm depth and 8 cm diameter) was used.

Litter was removed, and duff and the first centimetres of soil were collected. Soil containers were also sterilized, and the samples were taken to the laboratory in a portable refrigerator. The microorganism populations were determined using the Pochon and Tardieux (1962) suspension-dilution plate counting method. Selective solid or liquid culture media were used, except for the total microflora where yeast extract broth (ACEA 1981 and 1985) was chosen. Czapeck-Dox agar medium for fungi and soil-agar extract for bacteria were used. Wilson's solution was utilized for algae. Pochon and Tardieux's media were used for aerobic and anaerobic nitrogen fixed free living organisms, starch decomposers, anaerobic cellulose decomposers, pectin decomposers, protein decomposers, and ammonifiers. Alexander and Clark's method was used for Nitrosomonas and Nitrobacter, Focht and Joseph's for denitrifiers, and, finally, Eggins and Pugh's for aerobic cellulose decomposers.

Forest floor and soil samples (0-5 cm depth) were taken periodically for mesofauna population countings on each site. Depending on the site, from to to four plots were sampled. Fifteen samples per plot were randomly collected with a metal 625 square cm (25 x 25cm) quadrat, and the litter, duff, and soil layers were separately taken from the field, packed, placed in Berleśe-Tullgreen funnels, and the animals were extracted into an alcohol-water mixture. Depending upon the sites, these samples were taken at different times after the fires. At two of the sites, the samples were taken immediately, six months, and one year after burning. At the rest of the sites samples were collected one and two years after the fire. On all sites, plots were sampled three years after the fires. The specimens kept in the alcohol vials were observed with a stereoscopic microscope and grouped into Collembolans, total Acari, Acari immatures, Diptera larvae, Coleoptera larvae, and Thysanoptera. Poduromorpha within Collembolans and Cryptostigmata and Mesostigmata within Acari were separated.

Data on fuel moisture contents, meteorological conditions, fire behaviour and temperature distribution during prescribed burns were recorded. For each plot, twenty samples for moisture content of vegetation, litter 1h, 10h and 100 h fuels, duff, and soil (0-5 cm depth) were collected immediately prior to the fires. Air temperature and relative humidity were measured every ten minutes during the fires. Wind velocity and direction, at 1.50 m height for 10-minute periods, were also recorded in the stand. Flame lengths were estimated by comparing them with stakes, and taking pictures during the fires. Rate of spread was calculated from the time the fire needed to run a known distance between stakes in a 6 x 6m grid on the plot; a mean value was obtained.

Temperatures were measured with chromel-alumel thermocouples connected to a recorder; from three to six points per plot were measured. At each point, three thermocouples were set up. One of them (0.8 mm diameter) was positioned at the litter-duff interface, another (0.8 mm diameter) at the duff-soil interface, and the third (3.18 mm diameter) was located 2.5 cm below the top of the mineral soil.

Fires were carried out between winter 1982 and spring 1984. The ignition technique was basically heading fire. In some cases, a backing fire was used. Linear intensity was calculated from flame length (BYRAM 1959).

RESULTS AND DISCUSSIONS

Some data on the fire behaviour during the burning can be seen in Table I. In general, air temperature, humidity, and wind velocity were pretty low. The vegetation moisture content was very small, probably because most of the fires occur during the dormant season of the plant. On the other hand, litter had a high moisture content. It is to be noted that in some cases the duff had a rather low moisture content; soil moisture content in the same cases was low. Broad ranges of spread rate and linear intensity were obtained. The intensity varied from low to moderately high.

TABLE I

FIRE BEHAVIOUR, METEOROLOGICAL CONDITIONS AND FUEL MOISTURE CONTENT FOR PRESCRIBED BURNING EXPERIMENTS

Meteorological data	Interval	Mean value
Air temperature (°C)	6.00 - 18.00	12.50
Wind velocity ($m.s^{-1}$)	0.09 - 1.16	0.60
Air rel. humid (percent)	45.00 - 70.00	58.00

Moisture content	Interval (percentage)	Mean value
Vegetation	64 - 99	78
Litter	12 - 36	27
Twigs (1 h time lag)	27 - 39	36
Duff	32 - 275	141
Soil (0-5 cm)	17 - 53	33

Behaviour parameters	Interval	Mean value
Rate of spread ($m.min^{-1}$)	0.20 - 2.7	0.60
Flame length (m)	0.30 - 1.25	0.74
Linear intensity ($Kw.m^{-1}$)	21 - 420	38

Temperatures during the fires were very variable (Table II). Mean data show that the lethal temperature (60°C) was reached at the duff-soil interface only for 2'; lethal temperatures were achieved for about 5' at the litter-duff interface. There were many differences related to varying characteristics, but the moisture content appears to be an important variable that affects duff and soil heating. As far as soil is concerned, the average maximum temperature 2.5 cm below the duff-soil interface was only 21°C.

TABLE II

TEMPERATURES DURING PRESCRIBED FIRES

-Mean values from the mean data of eighteen experiments-

PLACE	INITIAL TEMP (°C)	MAX.TEMP (°C)	TIME,T>60°C	TIME,T>100°C
Litter-Duff interface	11.6 ± 1.2	408 ± 104	4.88' ± 1.41'	3.17' ± 0.94'
Duff-Soil interface	11.5 ± 1.2	59 ± 27	1.93' ± 1.10'	1.25' ± 0.81'
2,5 cm undersoil	11.3 ± 1.3	21 ± 4,3	4" ± 7"	0.6" ± 1.8"

In Fig. 1, fuel reduction by the fire and the short term change in the fuel components loadings are shown. The mean total fuel load is about 41 tonnes/ha. The duff makes up 47 percent of this amount. The litter (fresh and partially decomposed) and biomass represent 43 percent of the total fuel; and the vegetation is 11 percent.

Immediately after the fires, the reduction was similar for biomass (55%) and fresh litter (55%), and greater than the reduction for 0-5 mm diameter dead fuels (50%). The fire consumed 32% of the total fuel, but considering the sum of the biomass, fresh litter and 1 hour time-lag dead fuels as a "rough" estimation of the available fuel consumption was at least 55%. If partially decomposed litter is included as available fuel; the consumption was 47 percent. Anyway, this reduction is important for diminishing the intensity of a potential wild fire over this area. The other dead fuels suffered lesser changes. The 6-25 mm in diameter dead fuels decreased at around 29 percent. No change was detected for the 26-75 mm and 14 percent for partially decomposed litter. Duff consumption was low (17 percent).

HOUGH (1965) measured a reduction in the biomass, after prescribed burning, of 65 percent of the initial load on one site, and from 58 to 62 percent on other sites. VEGA et al. (1983) found a slightly larger reduction in brush and fine fuels, and reductions of 67 and 41 percent respectively for the litter and duff layers were reported by SACKETT (1980), and between 40 and 60 percent for fuels less than or equal to 5 mm in diameter. These values are similar to those obtained in our burnings. He also measured a reduction of from 24 to 46 percent for fuels from 6 to 25 mm in diameter and from 45 to 54 percent for those of 25-75 mm from 6 to 25 mm in diameter and from 45 to 54 percent for those of 25-75mm. He reported a reduction from 14 to 47 percent for larger woody fuels.

Two years after the fires the total fuel is practically equal to the preburn load; however, the available fuel represents 64 (or 84) percent of its initial load. The biomass is 49 percent of the amount before burning, the fresh litter 68 percent; and the finest dead twigs (0-5 mm) have practically rebuilt to their preburn level, leaving the 6-25 mm fuels still reduced. A marked increase can be observed for the load of the other dead fuels. This increment varies from 61 percent for the fuels within 26-75 mm in diameter to 72 percent for partially decomposed litter. VEGA et al. (1985) found the same trend for two years after prescribed fires in Galicia and the same effect has been reported by LOTTI (1960), GORDON (1967) and BRENDER (1968). Finally, duff recovery was also very fast, and after this fire the quantity was similar to that before burning.

Soil analysis data are summarized in Fig. 2. Soon after the fires, an increment for the organic matter content was detected, probably due to charcoal residue or organic compound condensation coming from the duff. Two years later, the organic matter content is a little smaller (1 percent) than before the fires. Nitrogen is practically equal to pre-fire levels. A small drop in the pH was found immediately after the fires. Conversely, it was a little higher two years later. Potassium, calcium, magnesium, and phosphorous were higher soon after the fires. The major relative increment was for the calcium (169 percent) and magnesium (97 percent), but two years later, all the nutrients remained at their initial level, except calcium which was 65 percent higher than before burning. Thus the trend for calcium can be verified as being retained at a higher proportional level than magnesium in the burned soil, and magnesium a higher proportional level than potassium.

In other prescribed burnings in Galicia, VEGA et al. (1983) one year after fires did not find significant differences in levels of these nutrients, on burnt plots compared with control plots. On other sites, different authors have found similar short term results for nutrient contents (METZ et al. 1961, WELLS 1961, CHRISTENSEN 1977, ST. JOHN & RUNDEL 1976, MOHERING et al. 1966, REGO et al. 1983, 1987, McKEE 1982). In another study, HUNT & SIMPSON (1985) reported a significant decrease in the potassium, sodium, and organic carbon on very poor soils in Australia.

In this study, mean seasonal values from two years of soil microflora analyses show that there were no significant differences in the number of micro-organisms in any one group between burned and unburned plots in Pedrido. Significant changes (at $p < 0.05$) were detected for total microflora, bacteria, fungi, aerobic free living nitrogen fixers, starch decomposers, and pectin decomposers, between one of the burnt plots and its control in Cerponzones (Fig. 3), the populations being greater on the burnt plot.

Fig. 1 PRESCRIBED BURNING SHORT-TERM EFFECTS ON FUEL LOADINGS

- Mean values from 18 plots -

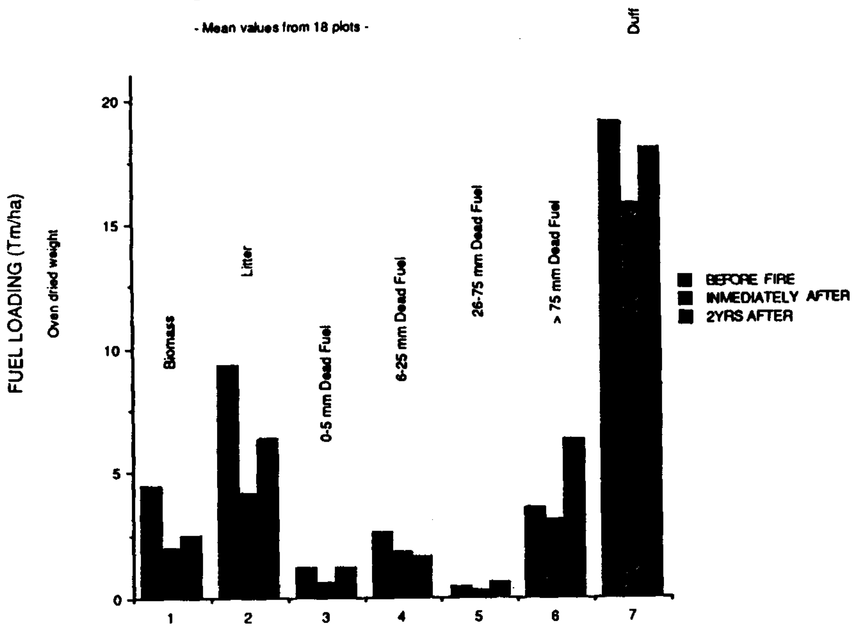
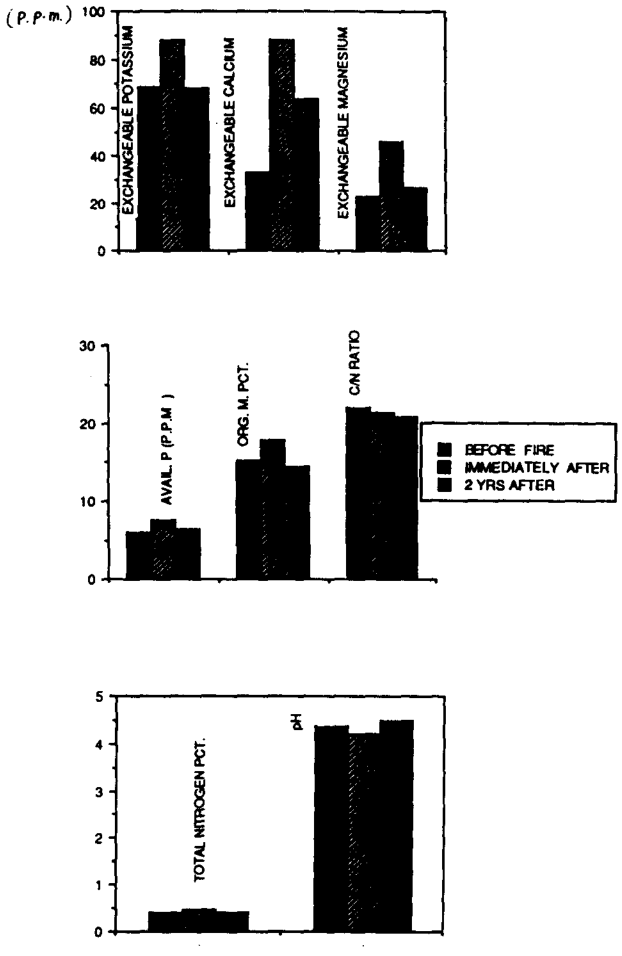


FIG. 2 SHORT-TERM CHEMICAL CHANGES IN SOIL FOLLOWING PRESCRIBED BURNING

(0-5 cm layer), mean values (18 plots)



These differences were found on a plot burned with a lower intensity, but more litter and duff were consumed in it.

In Vilacha there was a significantly higher number ($p < 0.1$) of bacteria, aerobic Nitrogen fixers, and ammonifiers on both burnt plots than on unburnt plots (fig. 3).

This increment in the number of aerobic Nitrogen fixers on two sites agrees with other findings (LUNT 1951, REMEZOV 1941, FOWELLS & STEPHENSON 1933, GIL 1982, GRICAL & McCOLL 1977, VIRO 1974). This increase could be related to the pH elevation because Azotobacter is very sensitive to it, and it appears to be one recovery response to the nitrogen lost by heating. VEGA et al. (1983) found that these microorganisms show a trend, although not a statistically significant one, towards higher populations over prescribed burned areas than unburned ones.

Other authors have also found increases in the total microflora after wild fire (ARCARA et al. 1975, GIL 1982, VELASCO et al. 1986) or prescribed burning (TOLEDO 1987).

BERRY (1978) did not observe differences between areas treated with periodic prescribed burning for 50 years, and JORGENSEN & HODGES (1971) and VEGA et al. (1983) found the same thing. , Also, increases in bacteria were reported after prescribed burning (WRIGHT & TARRANT 1957, AHLGREN & AHLGREN 1965, NEAL et al. 1965, NEAL et al. 1965, MARGARIS 1977).

It has been generally reported that the number of fungi decrease after fire (NEAL et al. 1965, NISSEN 1976, MARGARIS 1977, ARCARA et al. 1975, GIL 1982, VELASCO et al. 1986) in relation to the increase of pH, but some authors (JORGENSEN & HODGES 1971; VEGA et al. 1983) did not find significant differences. In this study, there were no significant differences on two sites, and an increase was detected in Cerponzones on one burnt plot, perhaps because the pH increment was not important enough to affect the fungi adversely.

Increases in starch decomposers and ammonifiers agree with trends found by DUNN et al. (1979), VEGA et al. (1983), and GIL (1982).

Forest floor mesofauna was affected by the fires due to the litter consumption, duff heating and partial consumption. On the sites sampled soon after the fire, populations of all groups were reduced for the litter and duff layers on the burnt plots, as compared with the controls. Though maximum temperatures during the fires were not very high on the duff-soil interface, lethal temperatures were reached for several minutes on the litter-duff interface, so that mesofauna populations in the duff suffered the impact of the wave of heat. Generally, samples on recently burned plots showed that live animals moved into the first cm of soil, probably to take refuge from the adverse conditions in the remaining duff, especially the temperature and water content. This was also observed by METZ & FARRIER (1971) on areas burned with prescribed fire, and by MANSILLA et al. (1985) on sites affected by wild fire.

On the other hand, soil animals appeared not to be affected by the fires and the level of populations was the same for burned and unburned plots, immediately after the fires.

Recovery time was different among the groups studied (Table III). But differences are not obvious when data are grouped by linear intensity levels or percentage of litter and duff weight consumed (see first part of Table III). It appears that a better definition in the lower ranges exists if some measurement of the relative litter and duff consumption is taken into account along with the temperature gradient inside the duff. Both parameters affect the original populations; one fire can be very fast so that a high temperature can be reached on the duff top, but consumption can be small. On the other hand, a glowing combustion probably will not reach a very high temperature peak, but duff consumption can be important. Furthermore, for the effect on microfauna, perhaps the absolute peak temperature is not the only main variable. Temperature gradient appears more meaningful. So we considered the percentage of little and duff consumed multiplied by the mean maximum temperature reached on the littler-duff interface, divided by the initial duff depth for each plot.

FIG. 3: CHANGES IN 0 - 5 cm SOIL AND DUFF LAYER MICROFLORA POPULATIONS FOLLOWING PRESCRIBED BURNING (MEAN VALUES FROM SEASONAL ANALYSIS FOR 2 YEARS AFTER BURNING)

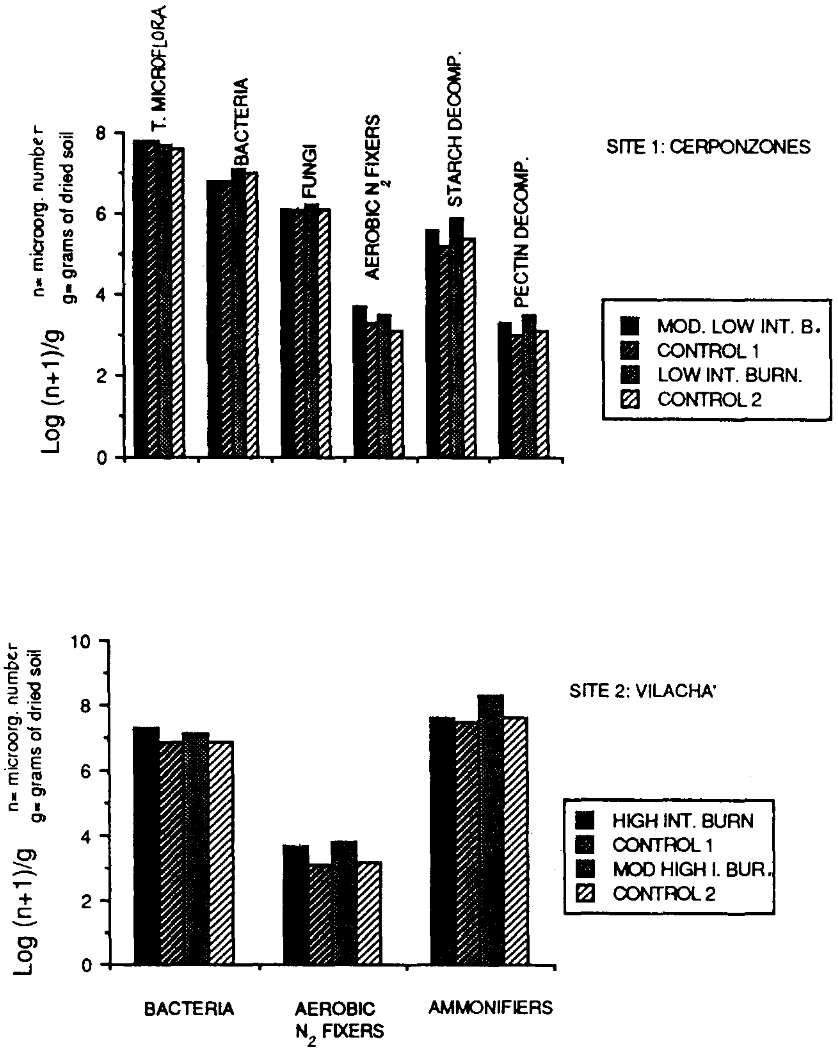


TABLE III

RECOVERY TIME (YEARS) OF SOME FOREST FLOOR LAYER MICROFAUNA ORGANISMS AFTER PRESCRIBED BURNING BY A) LINEAR INTENSITY, B) DUFF CONSUMPTION AND HEATING

	A) Linear intensity (Kw/m)		
	High	Moderate	Low
Group	420 - 630	250 - 300	21 - 125
Collembola (Total)	3	1 - 3	0.5 - 3
Poduromorpha	3.5	1 - 3.5	0.5 - >3
Acari (Total)	2.5	1 - 2.5	0.5 - 3
Cryptostigmata	2	1 - 2.5	0.5 - 3
Mesostigmata	3.5	2,5 - 3.5	0.5 - 3.5
Immature Acari	1.5	1 - 1.5	0.5 - 1.5
Diptera larvae	1.5	1 - 1.5	0.5 - 2
Coleoptera larvae	3	>2 - 3	0.5 - 3
Thysanoptera	3	2 - 3	0.5 - 3

	B) Litter and duff consumption and heating*		
	High	Moderate	Low
Group	120 - 220	19 - 45	3 - 16
Collembola	3	0.5 - 3	0.5 - 1
Poduromorpha	3.5	0.5 - >3	0.5 - 1
Acari (Total)	2.5	0.5 - 3	0.5 - 1
Cryptostigmata	2	0.5 - 3	0.5 - 1
Mesostigmata	3.5	0.5 - 3	0.5 - >2
Immature Acari	1.5	0.5 - 1	0.5 - 1
Diptera larvae	1.5	0.5 - 2	0.5 - 2
Coleoptera larvae	3	0.5 - >2	0.5 - >2
Thysanoptera	3	0.5 - 2	0.5 - 2

* See text for this measurement

In the case of the higher range of heating and/or consumption, the total number of Acari appears to recover before that of Collembola. Within Collembola, Poduromorpha take 3.5 years to rebuild their initial level, being a heat sensitive group. Within Acari, the abundant Cryptostigmata need less time than Mesostigmata, which are predators, to equal populations in the unburned plots. Perhaps the decrease in the phytophagous populations induces a reduction in the predators, which in addition have a longer reproduction cycle. It is surprising how fast the immature acari recover to a level similar to the control populations.

Diptera and Coleoptera larvae, show a different recovery time for the higher range of duff heating/consumption. Diptera larvae need only 1 1/2 or 2 years, but Coleoptera larvae, 3 years. Thysanoptera exhibit a similar recovery time pattern to the Coleoptera larvae.

These above data appear to agree with other data from METZ & FARRIER (1971). They found that 3.5 years were necessary for the mesofauna recovery on periodically burned plots. REGO et al. 1983, 1987, (this volume) have reported from 3 to 4 years, depending on groups. GILLON et al. (1987, this volume) give a shorter time while ATHIAS BINCHE et al. (1987, this volume) found a generally longer recovery time, but their data are from areas burned with wild fires. Probably although the total number of animals has recovered after this period, changes in the specific compositions and interspecific relations could remain for a longer time.

CONCLUSIONS

Some short term effects of prescribed burns have been studied for two or three years after the fires in pine stands in Galicia. Total fuel was reduced by the fires, but special significance is that its architecture was altered; though its recovery is fast, the available fuel, two years after the fire, is less than before burning. Soil exposure was nil, and the duff layer remained, protecting the soil in all cases.

Only small changes were detected for the nutrients in the 0-5 cm soil layer soon after burning. Two years later, these changes were minimal.

Some groups of the soil microflora populations were enhanced by the fire and their activity increased. Others remained at the same level as on unburned areas.

The forest floor mesofauna populations can be altered immediately after fire, and, depending on groups, duff heating and consumption; their recovery period can last for six months to three and a half years. Soil mesofauna appeared not to suffer any change in their populations even immediately after fires.

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ECOLOGIA MEDITERRANEA - Tome XIII (fascicule 4) - 1987

Prescribed fire effects on soils and vegetation in Pinus pinaster forests in northern Portugal.

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SUMMARY - The importance of fire in the Mediterranean Basin is stressed and influence of man in relation to fire is discussed. A succession diagram in deciduous oaks climax series is proposed, and the successional role of pine forests is analysed. It is concluded that pine forests evolved, in many cases, with fire, but not necessarily with destructive crown fires. More frequent light underburning is natural in various pine ecosystems. This natural process can be replaced by prescribed burning in the management of pine forests. The use of prescribed fire in Pinus pinaster forests of northern Portugal is discussed from the first accounts dating back to 1836 to the experimental fire management programme started in 1982. Research on the ecological effects of prescribed fire showed that soil fertility and tree growth are not significantly affected; understory vegetation is very well adapted to fire, and forage quality and productivity are enhanced. Recommendations for the use of prescribed fire are suggested.

KEYWORDS : Prescribed fire, fire management, Pinus pinaster, ecological effects.

INTRODUCTION : FIRE IN THE MEDITERRANEAN BASIN

Fire has been a very important factor fashioning numerous vegetation types throughout the world. Its antiquity is clear : fire is probably as old as terrestrial vegetation (KOMAREK 1973). However, the action of fire has been particularly stronger in regions with a Mediterranean type of climate because of the existence of a hot and dry season and the dominance of sclerophyllous vegetation that is highly flammable and combustible due to the preponderance of species with a high content of resins or essential oils (TRABAUD 1982, CASTILLO 1985). The influence of man is of extreme importance in those regions. Old and permanent human settlements established "natural barriers" to fire such as agricultural fields or roads (TRABAUD 1981). Also fire fighting operations contributed to a reduction of the burnt areas. However, man causes fire, directly or indirectly, and natural fires lightning are of minor importance at present, representing in the Mediterranean Basin only 2.4% of the burnt area. This complex influence of man explains that for the past 25 years in Europe the number of fires has increased while the burnt area remained about the same or slightly diminished (LE HOUEROU 1981).

Thus, fire which has been the principal natural force in the biological evolution of the Mediterranean Basin is now transformed into an artificial force through its use by man (TRABAUD 1981).

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MAN AND FIRE

Man's ancestors probably arrived in the Mediterranean region about one million years ago but until fire was mastered, man's impact on vegetation and environment was very slight (LE HOUEROU 1981). The conquest of fire by Homo erectus took place between 500,000 and 400,000 B.P. (LE HOUEROU 1981, CASTILLO 1985) or about 300,000 B.P. (PERLES 1977) making it possible to improve the techniques for hunting and gathering. Man has voluntarily used fire for hundreds of thousands of years, he has transported it and preserved it without knowing how to "create" it. This important event took place only some 40,000 to 50,000 years ago (TRABAUD 1982). Domestication of animals took place between 12,000 and 7,000 B.P. but 6,000 years ago, Neolithic peoples still depended largely on hunting. They must have noted that the animals they fed upon were attracted by burnt zones and therefore fire has been used extensively. The nomadic agriculture that followed included the use of fire before or after tree felling, then reclamation and cultivation. Documentation on the use of fire is available in historic books as the Bible and the Talmud (NAVEH 1974) or the Iliad and the Georgics (LIACOS 1974). But fire, already used by prehistoric man to defend or to pillage territories, was also extensively used as a war weapon in various periods of the later European history. Nowadays, fire has been less and less used for its traditional purposes (land reclamation, etc.). There is an exception for shepherds who continue to light fire to promote the growth of new shoots of palatable species (TRABAUD 1981), especially in less developed areas as in Portugal and in other southern European countries.

In Portugal there are documents on the use of fire that date back to the XIIth century. The negative consequences of fire on erosion in the Modego River Basin in central Portugal were known by the XVth century when laws were enacted to eliminate the use of fire near rivers (NEVES 1981). Currently forest wild fires burn an average of 43.7 thousand hectares per year, about 1 percent of total forest and shrubland area. This value is higher than the average for Mediterranean Europe as a whole, with almost 0.3 percent (LE HOUEROU 1981). Fires burn forest at a rate superior to the normal afforestation capacity (SILVA 1981). This last decade was considered to be "exceptional" with severe extensive forest fires. This is probably related to the buildup of fuel in the understorey resulting from the abandon of grazing and harvesting for fuel and livestock litters after the drift from the land in the sixties (SILVA 1984). Therefore from the point of view of wild fire hazard reduction and improvement of grazing in forest, prescribed burning of the understorey has potential in the management of these vegetation types.

FIRE IN RELATION TO PINE FORESTS

Climax vegetation of northern Portugal, transitional between the mediterranean and temperate zones, is dominated by the deciduous oaks Quercus robur and Quercus pyrenaica (BRAUN-BLANQUET et al. 1956). These forests were extensively replaced by conifers, mainly Pinus pinaster, that withstand more frequent fires and are among the most flammable ecosystems in the region (TRABAUD 1981). For example, Pinus halepensis forests represent about one third of the area burned in Greece, Spain, France, and Italy even though this type of forest occupies only 17%, 7%, 4%, and 3% of the forest areas in the four countries respectively (LE HOUEROU 1981). In Portugal Pinus pinaster forests represent more than half of the area burned, but occupies only 30% of the total forest and shrubland area. In other words Pinus pinaster forests burn almost twice as easily as the average for all forest and shrublands in Portugal. Therefore, it is important to understand the successional relationships and the seral position of pine forests in relation to fire (Fig. 1).

After a wild fire, pioneer grasses and shrubs together with tree seedlings colonize the site. If a new fire occurs every 10 to 15 years the community evolves to a shrubby stage (GUILLEN cited by RICO 1978) where the Ericaceae are dominant (BRAUN-BLANQUET et al. 1956, CASTILLO 1985). More frequent fires result in the perpetuation of an herb stage.

If a wild fire occurs every 25 to 35 years the pine stand is mature and has fertile cones (GUILLEN cited by RICO 1978) that often open during fires, disseminating seeds that find mineral soil, abundant nutrients and weak competition, which facilitates establishment. At that fire frequency pine forests may perpetuate themselves indefinitely (TRABAUD 1981).

Clearly, if fire frequency is of one hundred years or more, and if seed source is available, primeval oak and chestnut forest can reestablish (GUILLEN cited by RICO 1978). However, if a fire occurs in mixed oak-pine stands, fire temperatures can be sufficiently higher under pines to ensure elimination of the oaks in the vicinity of adult pines (WILLIAMSON & BLACK 1981). In that case, succession is interrupted and thus, at a fire frequency of less than 100 years the community evolves to a fire climax dominated by pines.

This discussion so far assumes that all forest fires are destructive to crown fires. However, in similar pine forests in other parts of the world this is not the case. In *Pinus ponderosa* forests of North America for example natural underburning by light and creeping surface fires with a variable frequency of 5 to 20 years is known to have been of extreme importance (BISWELL 1973, HALL 1977, DAVIS *et al.* 1980, WRIGHT & BAILEY 1982).

To ensure the stability of pine forest ecosystems, adequate management (natural or artificial) is necessary (CASTILLO 1985). Natural underburning can be replaced by prescribed underburning in the management of pine forests. This is the ecological rationale for the use of prescribed fire in northern Portugal pine forests.

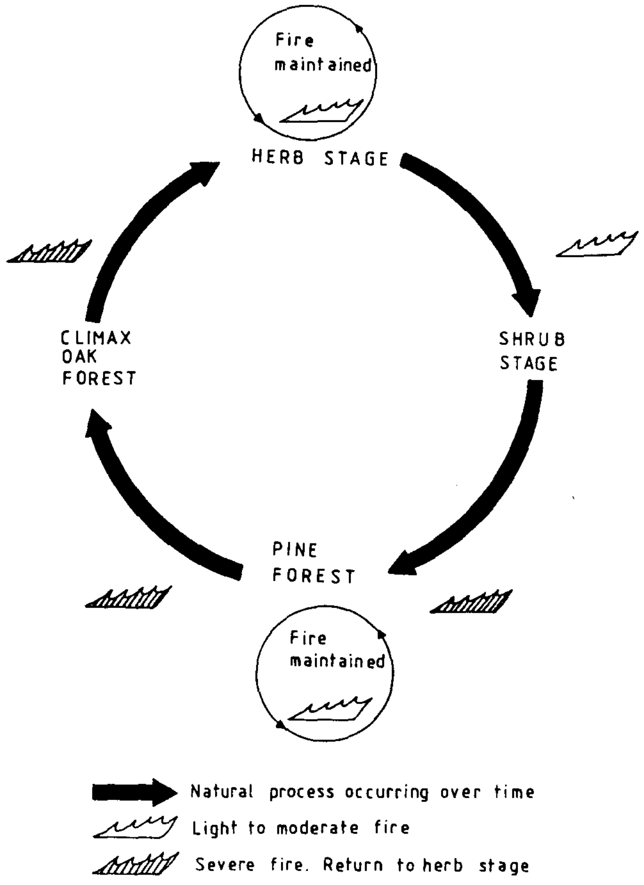

Fig. 1 Forest succession in deciduous oak climax series adapted from CASTILLO (1985) using the succession diagrams of DAVIS *et al.* (1980).

THE USE OF PRESCRIBED FIRE

For a long time in Europe, policies of fire exclusion and suppression were implemented. Natural fire was replaced by grazing and cutting of understorey vegetation in the maintenance of low understorey fuels. A new equilibrium was reached. With the decrease of harvesting because of emigration and the prohibition of grazing in forests until 1974, that equilibrium was lost in Portugal. Prescribed fire seems to be the wise way to restore more natural conditions in the pine forests (BISWELL 1973), as it is a very effective tool in decreasing wild fire hazard. In *Pinus ponderosa* forests KALLANDER *et al*. (1955) and WEAVER (1957) showed a 94 and 90 percent reduction in the areas burned by wild fires on previously burnt communities.

In Portugal the first account on the use of prescribed fire in *Pinus pinaster* forests dates back to 1836 when Friderico Varnhagen, an officer responsible for the management of government forests wrote that "after the pine forest is 20 years old there is a safe way to prevent it from being burned by summer wild fires : burning the pine litter in dry winter days will not damage the tree roots, and if that operation is repeated annually there will be no danger of summer wild fires that damage the roots and kill the pines. (...) The pine stands that get used to seeing their floor burned annually grow much faster and the benefit is great in every sense". He further elaborates on some of the most important principles for the technique of prescribed underburning : "it is obvious that for a safe operation, even in wintertime, the pine forest should not have tall shrubs in the understorey. (...) The burn must be done with appropriate winds, and fire should be lit on the opposite side of the wind".

This pioneer of prescribed fire was completely forgotten, and prescribed fire was almost unknown in Portuguese forestry until Dr. Edwin KOMAREK came to Portugal in 1976. A very important step in the history of Portuguese silviculture was then made (SILVA 1984). After that, some attention was paid to prescribed fire in Portugal. At the same time prescribed burning began to be studied in Spain (VEGA 1981) and fuel management plans were started (VELEZ 1985).

The research efforts on the effects of prescribed fire concentrated on areas considered important by scientists and land managers :

(1) the effects of fire on the productivity, structure and diversity of the plant and animal communities;

(2) the effects of fire on litter accumulation, decomposition, and nutrient concentration (KICKERT *et al*. 1976, AGEE 1982).

THE BASIC HYPOTHESIS

An experimental fire management programme was started in northwestern Portugal under the initiative and responsibility of Eng. MOREIRA DA SILVA and at his request research on the ecological effects of prescribed fire was undertaken. Dr. Stephen BUNTING came to Portugal in 1982 to help to initiate research. The results of this study are presented in detail in REGO (1986).

The basic hypothesis was that prescribed fire could be introduced as a management tool in Portuguese forests to achieve the following objectives :

1. Reduce wild fire hazard
2. Improve forage quality and quantity.

This should be done without significant detrimental consequences on soil, understorey communities, and tree growth rates.

CONCLUSIONS OF THE RESEARCH

From the previous sections, it became apparent that fire was an important natural factor in stabilizing pine forest ecosystems and that the active fire suppression policy and the lack of adequate management resulted in building up an excess of fuel in the understorey which caused devastating crown fires.

This research concluded that there were various effects of prescribed fire on the forest surface soil. A decrease in the organic matter content was the single most important effect causing similar decreases of the closely positively related chemical parameters : available K, Cation Exchange Capacity, and the exchangeable cations Mg, K, and Na. Conversely there was an increase of the negatively correlated soil pH. Calcium content was also increased by fire, while total nitrogen and available phosphorus were not significantly altered by fire. Soil mesofauna was harmed by fire. Recovery of collembola took 3 years while oribatei seem to depend upon the reconstitution of organic matter to recover fully.

Frequency of understorey plant species was almost unaffected by fire while vegetation cover tended to regenerate the preburn species at different frequency rates. The nutritional quality of Chamaespartium tridentatum, the most abundant shrub species, was enhanced by fire. A short-lived increase in cellulose and hemi-cellulose, a corresponding decrease in lignin, and a longer-lived increase in protein (1.5% increase approaching the preburn level after 4 years) and digestibility were observed.

It was also apparent from the study that the use of prescribed fire did not cause any relevant effects on tree growth rate.

In summary, this research concluded that the introduction of prescribed fires of light intensity did not significantly affect soil fertility or tree growth; understorey vegetation is fairly well adapted to fire and recovers quickly without major modifications of floristic composition; and forage quality and productivity are enhanced by fire. Reduction of understorey fuel probably result, if prescribed burning is applied to large enough areas, in a significant reduction of wild fire hazard.

Simultaneous research on the relation of bark beetles to prescribed fire showed that burning would favour beetle attack on old dense stands where beetles were already present at lower levels. Therefore, it was concluded to avoid burning in those situations (CABRAL Personal Communication). This was one important consideration that must be taken into account in future fuel management plans.

It is important to note that the conclusions of this research cannot be extended beyond the limits of the study. They can safely be applied to the Pinus pinaster forests of the study area and for the fire intensities applied. The conclusions can probably be further extended to the mature pine forests of northwestern Portugal with light to moderate prescribed fires.

Another important consideration is that fire frequency should be determined in relation with the specific site characteristics and the relative importance of the various objectives to achieve with prescribed burning in these multiple-use forests. However, natural underburning in similar areas (at a frequency of 5 to 20 years) and the traditional fire practices by shepherds in the region (with a frequency of 3 to 7 years) suggest that prescribed fire could be used every 5 to 7 years as suggested by BISWELL et al. (1973) for pine forests in the United States.

This recommendation for the use of prescribed fire in portuguese pine forests is probably going to find some resistance among many forest managers that are uneasy about the possibility of an effective and safe use of prescribed fire. However it is important to stress again that fire is a part of the ecology of the pine forests regardless of what man may wish or attempt to do about it. In the absence of prescribed burning, wild fires will occur (KILGORE 1972).

The studies conducted demonstrate the ecological soundness of prescribed fire as a management tool in Portuguese forestry and I hope that forest managers will finally agree with the clever Finnish proverb : "fire is a bad master but a good servant" (VIRO 1974).

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