EUFIRELAB

EVR1-CT-2002-40028

D-04-03

http://eufirelab.org

EUFIRELAB:

Euro-Mediterranean Wildland Fire Laboratory, a “wall-less” Laboratory for Wildland Fire Sciences and Technologies in the Euro-Mediterranean Region

Deliverable D-04-03

Wildland Fires Impacts: a State of the Art

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with contributions from partners

P009, P019, P020, P021, P026, P030, P032, P034

February 2004

CONTENT LIST

SUMMARY

This document presents an exhaustive literature review centred especially in the Mediterranean ecosystems, but covering also the boreal forests.

Five chapters are dedicated to the Mediterranean ecosystems:

- plants analysed at an organismic level (structural and functional impacts), at the population level, and at the community level (plant composition and abundance, diversity, spatial arrangement of individuals, biomass production, nutrient balances);
- animals: chronological and geographical distribution of fire studies, animal groups studied (arthropods, vertebrates, other groups), sampling designs, parameters relevant for the fauna at population level and community level, conservation value;
- soils: mode of action of fire on soil, thermal reactions of soil, effect of heating on soil physical parameters, effect of heating on soil chemical parameters, ashes, ash leachate (effects on soil erodibility), impact of fire on soil physico-chemical parameters, impact of fire on soil micro-organisms, effect of fire on nutrient cycling and nutrient budgets, effect of fire on soil fertility, resilience of burned soils, soil water repellence, soil erosion (usle – musle – rusle), European soil erosion model (eurosem), soil erosion (water erosion prediction project – wepp), estimation of soil erodibility factor k, vegetal protection of soil, estimation of the vegetal covering factor c, prediction of post-fire soil erosion (Promethus), prevention of the post-fire soil erosion, and schematic diagram;
- aquatic ecosystems; and
- landscapes

The sixth chapter is dedicated to the Boreal forest ecosystems and wildland fires impacts

The last chapter gathers 368 bibliographical references related to wildland fires impacts on the different components of Mediterranean and Boreal ecosystems

GLOSSARY

None

ASSOCIATED FILES

None

1 GENERAL INTRODUCTION

Unit 04, was originally moderated by Margarita ARIANOUTSOU P021, José MORÉNO P020, Roger PRODON P019 and Ramon VALLEJO P009, has been in charge the activities related to WP04.

The current leader is Ramon VALLEJO, and Juli PAUSAS P009 administrates the section within the EUFIRELAB web site.

1.1 PLANNED ACTIVITIES DURING THE CONTRACT

The participants:

- describe the existing methods for characterising the fire impacts on the different strata and components of Euro-Mediterranean ecosystems,
- sort critically among the existing methods the most valuable for analysing the relationships between fire, ecosystems composition and structure, taking into account the particularities of Euro-Mediterranean ecosystems,

1.2 FIRST RESULTS ACTIVITIES DURING PERIOD 1

An exhaustive literature review has been done, centered especially in the Mediterranean, but covering also the boreal forest.

Here is presented a first, preliminary review, covering plants, vegetation, soils, aquatic ecosystems and landscapes from the Mediterranean Region, and the boreal forest as a whole.

2 MEDITERRANEAN ECOSYSTEMS: INTRODUCTION

Numerous studies have been done in the Mediterranean region addressing the impacts of fire.

A division of work was made early on, whereby the various groups involved in this task would follow their own approach (plants, soils, animals landscapes, boreal forest).

With regard to the first group, given the large number of papers and the variety of approaches and methods used, and in order to serve as preparatory work for other tasks within the section, the emphasis of the review was to concentrate on the on methodological issues.

This is, the aim was to address issues like: what has been done, where, how, how many times, using what procedures, etc.

A division of task was made back in June, 2003, among the two responsible partners, JM MORENO and M ARIANOUTSOU.

Following it, tables were prepared in which various papers were revised in regard to their methodological approaches.

The papers reviewed were a selection among those published in the international literature that were thought to better reflect the various methodological approaches used.

This revision is, therefore, limited in scope, but will be the basis for compiling a database of fire impacts on plants and the vegetation in the Mediterranean region, and for further work within this section.

The tables for the various items is downloadable from the site under Ecology /Directories/ working_documents, page
http://www.eufirelab.org/prive/directory_list.php?genre_id=4&rep_id=64

The other two groups have each followed a different approach but emphasis was also made on methodological issues, where appropriate.

3 MEDITERRANEAN ECOSYSTEMS: PLANTS

3.1 ORGANISMIC LEVEL

3.1.1 Structural impacts

3.1.1.1 Impact on roots and subterranean structures: (A CRUZ, UCLM)

Some studies have focused on survival of subterranean structures after fire in Mediterranean ecosystems.

Most of the analysed studies used simulations of fire by means of propane torch in order to control for burn intensity, an aspect of fire regime considered as very important in terms of plant survival.

Moreover, this method ensures that all studied plants were alive before the fire, which is not possible in a wildfire.

Most studies are focused on one or two species; *Erica* species are recurrent in these studies. In general, survival was monitored during the two first years after the fire or burning.

3.1.1.2 Impact on seeds (B LUNA, UCLM)

Two main methodologies have been developed in the Mediterranean Basin to study the fire effects on seed germination.

1.- Study of seed germination under specific conditions in chambers with temperature and humidity control in the laboratory.

Previously to incubation seeds are exposed to different treatments which simulate direct and indirect fire effects.

Among these treatments, it is worth to point out the heat-shock treatment to different temperatures ranging from 60 to 150ºC and different times of exposure.

In some works, the effects of light and nitrate addition are tested as indirect fire effects.

The number of replicates is very variable using between 100-500 seeds per species and treatment.

Recount of germination is mainly every 1-3 days and duration of experiment is around 2 months.

Most of the works are focused on one or a few species and the plant families of special interest have been: *Cistaceae*, *Leguminosae* and Ericaceae, but other important families like *Labiatae* or *Compositae* have been also analysed.

Although most of the growth forms have been used, nanophanerophytes studies are more abundant, maybe because of its high rate between the families of interest.

The analysed papers are concentrated in the Iberian Peninsula, France and Greece.

2.- Study of fire effects on soil seed banks in greenhouse and field experiments.

Samples of unburned and burned soils are collected and incubated.

This basic study is usually completed with direct estimate of soil seed banks by means of physical separation and results are contrasted with germination in field.

Soil samples are divided in two depth fractions (0-2; 2-5 cm).

Experiments last a long period around one year or until not more germination is observed.

All plant species of the community are usually studied, but some works are focused on several species, like those of Pinaceae or Ericaceae families.

The analysed papers are also concentrated in the Iberian Peninsula, France and Greece.

3.1.2 Functional impacts

An exhaustive search for publications related to the impact of fire in functional attributes of plants from the Mediterranean Basin produced few results.

Generalisation about techniques used for determining the impacts of fire on functional attributes are hard to obtain.

Consequently, it will be recommendable to extend the search for publications related to other Mediterranean-type areas of the world. In addition, we also recommend the inclusion of studies in which the complete clipping of aboveground tissues was employed to simulate some of the impacts caused by the fire.

3.1.2.1 Photosyntehseis and respiration: A CRUZ, UCLM

In all the studies portable gas-exchange systems have been used for determining assimilation rates.

Sampling has been made during the first year after the fire, commonly at different seasons of the year (spring is always included).

Measurement were performed at midday, and sometimes also in the morning. *Quercus ilex* is a species found in all studies.

Experimental designs usually include comparisons between burned and unburned sites.

3.1.2.2 Water relationships: (A CRUZ, UCLM)

Predawn and midday water potentials were commonly measured for determine the water status of plants.

A Scholander-type pressure chamber is the common method for measuring plant water potentials.

Stomatal conductance was sometimes measured with portable gas-exchange systems.

Measurements were commonly made during the first year after fire, usually during summer.

Experimental design usually include comparisons between burned and unburned sites.

3.1.2.3 Nutrients: (A CRUZ, UCLM)

These studies usually include many species, and comparisons among them with respect to nutrient strategies is a recurrent objective.

Sampling is commonly restricted to leaves.

Analytical methods are diverse, and nitrogen is the mineral nutrient most commonly analysed.

3.1.2.4 Carbohydrates and other reserves: A CRUZ, UCLM

Studies related to the impact of fire on plant carbohydrates reserves are lacking in the Mediterranean Basin.

In the only study available, sampling was made on a single shrub species, and measures were obtained in subterranean organs up to two years after burning.

An enzymatic method was used for determining carbohydrate concentrations.

3.1.2.5 Growth: B PÉREZ UCLM, D KAZANIS NKUA-BIO

The measure of the individual growth after fire depends on species.

In these studies, a variable number of individuals is selected (usually by random), marked and monitored during months or years.

The most common measures are the elongation of stems, number of resprouts and height of individual plants.

In the case of resprouter species, the first two are more frequently used than height.

However, when the analysed species is a seeder, the individual height is preferred.

Sometimes, one or two diameters of each individual are measured too.

When the species present numerous stems, as *Erica* spp. or *Quercus* spp., the study can be restricted to limited area and number of stems per plant, which are selected by different criteria.

There are some studies where, apart from the height, the basal stem diameter was measured.

This approach is applied in species producing dense populations after the fire, arising questions related to intra-specific competition.

The measures are taken with variable frequency but, at least, during or at the end of growth season.

In most cases, the results include statistical analysis with Anova, correlations and regressions but also indirect ordination methods have been applied when different communities/stands have to be analysed in parallel.

Also, there are some studies (Italy, Greece) where stem growth and plant height development has been followed on a frequent basis in burned and adjacent unburned ecosystems, in order to estimate the growth rate of the same plants and plant groups in different successional stages.

In these cases, also, classical statistical and direct ordination methods have been applied.

The measures are taken with variable frequency but, at least, during or at the end of growth season.

In most cases, the results include statistical analysis with Anova, correlations and regressions.

3.2 POPULATION : M ARIANOUTSOU NKUA-BIO
E ZUAZUA, UCLM

Despite the many studies done on fire impacts in Mediterranean ecosystems, our understanding of fire impacts on plant populations and demographic dynamics is still poor.

Very few detailed demographic studies are available.

Most studies provide global figures on the effect of fire on plant mortality rather than data on changes in the population structure of resprouting plants induced by fire or other post-fire factors.

Several different approaches have been used to analyse fire impacts at the population level.

In most instances studies are based on wildfires with the corresponding difficulties of associating death of a plant with its status prior to the fire or with aspects of fire such as fire intensity and subsequent severity.

Distinguishing the causes of plant mortality is important in order to allow predictions on the outcome of factors associated with changes of the fire regime and characteristics.

Some of the analysed studies used simulations of fire by means of a propane torch (burning of individual plants) in order to control for variables such as pre-fire plant status and size, and burn intensity (temperature and/or time).

This has the drawback that quite often a subset of plant sizes is only included.

Experimental fires, with precise measurements of fire characteristics, and knowledge about the state of the plant prior to fire are still scarce.

Some of these studied, and the subsequent monitoring of seedlings have provided the best evidence of the factors that control seedling germination and mortality (see QUINTANA et al. 2003).

With regard to the sampling design and arrangement of plots, studies are highly variable.

Replicates are not always present, although in recent studies these are more usual.

Experimental designs sometimes include comparisons between burned and unburned sites (or between burned and clipped individual plants).

The plot size differs from studies.

Usually the size and the number of the plots is related to the form of the species in question, (e.g. for herbaceous legumes or for *Cistus* spp. or for *Pinus halepensis*) and the surface burned.

Usually the minimum plot area varies from 1/4 m^2 or even less, in the case of seedlings, to 1 m^2 or larger sizes (in the range of 1/10 ha or more, when trees are involved).

Similarly, the number of the plots is extremely variable, between 10-50, but can be as large as several hundreds (QUINTANA et al 2003).

The arrangement of the plots is also varying between studies: elsewhere a transect line is followed while in other studies a random location of studied plots is mentioned.

Usually the plots are permanent and they are monitored for at least three consecutive years.

Several cases were revisit of permanent plots in later stages has been referred.

Recovery of plant species that regenerate vegetatively is a rather straight forward process, which depends on the meteorological conditions prevailing on the site, particularly during the first year after fire, the productivity of the site (available nutrients in the soil), competition between recovering plants and external factors such as grazing.

Populations of these species generally consist of one age cohort, that of the initially regenerated plants.

The situation is different with the seeding species, as their recovery is strongly dependent on the specific life histories of the plants, that is whether they are short living plants, such as the annual herbaceous legumes, or relatively short living plants, such as the rockroses or long living plants, such as the pines.

The analysed papers are concentrated in Spain, Portugal and France.

Usually, they are developed in matorral, machia, garrigue, and other shrubland formations.

Plant families of special interest have been: Cistaceae, Leguminosae and Ericaceae.

From the few demographic studies in post-fire communities it is shown that seedling emergence is larger in the first year, but this depends on the species concerned and on the climatic conditions following fires.

Actually, second year germination can even be greater than that of the first year.

Survival, however, appears to be highly restricted to the first year, but this may vary depending on site productivity and other interactions.

In addition, time of germination might be critical.

Yet, first germinants not necessarily survive best, although, age of the seedling does determine the outcome of survival.

The juvenile phase for most of the woody seeding species, such as the rockroses, lasts for only two years that is they reach maturity and consequently reproductive age quite soon.

After that, establishment is possible but little is known on the factors that control it, and can vary from species to species.

This can be deduced from the different size structure of individuals that is commonly found in any stand.

3.3 COMMUNITY

3.3.1 Plant composition and abundance: B PÉREZ. UCLM

The analysis of the postfire changes in plant species abundance and composition of the community are included in many studies.

However, the objectives of these studies are very different and because of that they show a great methodological heterogeneity.

We have analysed studies made in Portugal, Spain, France, Italy, Algeria, Greece and Israel.

Usually, they are developed in matorral, machia, garrigue, phrygane and other shrubland wildfires and forest fires in pine woodland, mainly with *Pinus halepensis* and *P. pinaster*.

In general, works describe the dynamic of the cover (total and groups like herbaceous, woody and different life forms), species richness and diversity with a diachronic and/or synchronic methods.

Monitoring of these variables can vary between two and twenty or more years and it can be regular or irregular in temporal spacing.

Of great values are those studies in which replicated chronosequences are used and monitoring is continued during several years, but these studies are scant.

The selection of plots depends on the general objective of the study.

When this is to describe the postfire dynamic of the vegetation, usually the plots are randomly selected inside the burned area, sometimes using as control the unburned area, with several restrictions due to slope, aspect, type of soils, etc.

More complex designs with blocks are not frequent.

Replicates within a burn are not always present, although in recent studies these are more usual.

True replicates, this is, using different fires are scarce.

Plot size differs enormously from one study to another.

To estimate or measure the analysed variables linear transects and quadrates of different sizes are used.

In general, the studies employ linear transects to measure or to estimate cover of woody species and quadrates, associated or not to the linear transects, to measure or estimate the cover of herbaceous species.

In the last case, most commonly each work uses a specific scale.

Number of species that are present is one of the most common measures in the analysed studies.

The use of statistical analysis and replicates follow the same temporal pattern.

In the earlier works these were absent, but are common in recent works in which use of Anova, Ancova, multiple comparisons, a posteriori test, simple and multiple regressions with controlling factors and multivariate methods are employed.

When is needed, data are transformed to improve statistical assumptions.

3.3.2 **Diversity:** F FERNÁNDEZ-GONZÁLEZ, UCLM

As with other components of biodiversity, plant diversity can be estimated in several ways.

Apart from structural or functional components of biodiversity, that will be commented in another section of this report, plant diversity is usually interpreted in a compositional sense, and its measurement is commonly based on the identification of plant species.

A common routine is to express plant diversity as species richness, or species density.

For comparative purposes the number of species recorded must be referred to a sampling unit, that may be

- 1 a plot with fixed area and dimensions (length and width), or a set of plots per sampling unit,
- 2 a line-intercept or a point-intercept transect, or a set of them with fixed lengths, intervals and distances per sampling unit,
- 3 a fixed number of individuals per sampling unit, corresponding to true individuals (genets) or to arbitrarily defined ones (ramets, e.g. shoots independently rooted in the ground surface)

The species richness values gathered through each one of these sampling methods have some advantages and other disadvantages.

(1) and (2) are by far the more widely used in post-fire studies concerning plant diversity.

The third one requires more sampling effort and has been rarely used in studies concerning plant diversity.

Species richness is additive, and species can be subdivided into species groups defined by different plant attributes (e.g. life forms, post-fire regenerative traits, distribution ranges, other functional typologies) allowing for instance more feasible comparisons among different geographical territories.

Nevertheless, reliable comparisons are only possible if the same support has been used in the sampling.

Unfortunately, there are no clear preferences toward any of the sampling alternatives at this respect, neither the dimensions or number of plots or transects per sampling unit.

Even in the same study one sampling approach may be used for a group of plants and one another for other group.

Measurements of species richness based on area know an interesting variant in which plots of different area sizes are sampled in order to evaluate the rate of increase of species richness as a function of area.

The species-area relationships obtained in this way can be adjusted to mathematical equations and compared across a range of spatial scales.

Also in this approach there are no general agreements about the dimensions and shapes of the plots representing the different area scales, and these latter may be nested in different degrees, influencing the comparability of the results.

A still different concept concerning species richness may be derived from accumulative curves of species, when several plots or transects are sampled within each sampling unit. In that case, besides the mean species richness per plot area or transect length it is possible to extrapolate the accumulated species richness for the sampling unit or some estimate measuring the species turnover between plots or transects (e.g. Jaccard or Sorensen similarity indices, or the named heterogeneity or beta-diversity indices).

The comparability of estimations is again depending upon the support used to obtain the data.

Species richness does not take into account the relative abundance of species, that are an important element for understanding diversity patterns.

The estimation of abundance introduces other source of diversity concerning sampling protocols.

Plant species abundance can be measured in several ways:

- 1 frequency in a set of plots (in general of small area in relation to the area of the stand being sampled),
- 2 density of individuals per plot,
- 3 cover, expressed as percentage and derived either from visual estimation on a plot of fixed area or from the number or length of the intercepted points or segments, respectively, along a transect,
- 4 trunk or basal area, derived from individual measurements referred to a sampling surface and mainly applied to trees or tall shrubs,
- 5 biomass, usually expressed as aerial dried phytomass per unit area and estimated either directly, hence using destructive methods, or by means of surrogates like basal areas or crown dimensions and the proper allometric functions of transformation.

Choice of the abundance measurements is usually depending upon the research purposes and the sampling resources and time availability.

Although some measures are related among them, direct transformations do work only under some severe assumptions.

All the abundance measures are scale-dependent, but for some purposes the values obtained using different supports can be compared if their coverage of the community or sampling unit can be considered representative enough.

When abundance measurements are available, plant diversity can be estimated through the named indices of “ecological diversity” or “proportional diversity”.

A huge number of such indices have been proposed, although only a bit more than half a dozen of them are commonly applied in plant diversity studies.

All of them are sensitive to the type of the abundance measures used, and even some of them were defined to be used only with certain abundance measures.

Hence proper comparisons only can be based on similar abundance measures.

The different behaviour of these indices is also related to their sensitivity to abundance variations in species groups with different abundance ranges.

Evenness indices would be more sensitive to variations in dominant species, while other indices are governed by variations in intermediate-abundance species.

3.3.3 Spatial arrangement of individuals: I TORRES UCLM, P ANDRIOPOULOS NKUA-BIO

In this section the works carried out in the Mediterranean region regarding the impact of fire on the spatial arrangement of individuals are reviewed.

A total of ten publications have been reviewed, taken into works performed in either Eastern or Western Mediterranean basin.

Methodologies described below show a considerable degree of heterogeneity.

There are several approaches that have been used to analyse the spatial pattern of individuals after fire.

One of them is the establishment of a small number of plots (1-3, occasionally 12) on burned areas (PAUSAS et al. 2003, MATEU et al. 1998, FARACO 1998, GLOAGUEN 1993, BAUTISTA and VALLEJO 2002).

Size of plots varies from 100 – 1000 m^2, but the most common size is of approximately 100 m^2.

In some works plots are divided in quadrats of size 1x1 to 0.3 x 0.3 m, depending on properties sampled.

Position of individuals is noted in those plots that do not include quadrats.

In plots that include quadrats, position of quadrat is noted.

Properties sampled are position of individuals (dead trees, live or dead seedlings), cover of species and in some works other properties such as biomass or plant size is recorded.

Age of stands varies from immediately after fire to 11 years post-fire, the most common being 1- 4 years post-fire.

Duration of studies ranges from 1 to 11 sampling campaigns.

Analysis of data comprises Ripley's K-funcion and some similar tests (PAUSAS et al. 2003, MATEU et al. 1998), Block variance methods (GLOAGUEN 1993), and Geostatistical techniques (calculation of variograms and cross-correlograms) (FARACO 1998, QUINTANA 1999; BAUTISTA and VALLEJO 2002).

The works described here have been carried out in Spain and France.

A quite different approach consists on the definition of distance zones from seed source (ESHEL et al. 2000, IZHAKI et al. 1992, NE'EMAN et al. 1992, NE'EMAN and IZHAKI 1998, BAUTISTA and VALLEJO 2002, Benidorm site).

In most cases these zones are the different distances to the dead tree, divided in near zone, the inner half of the dead tree canopy, the far zone which is the outer half of tree canopy and the area between canopies.

Transects through the zones are established and quadrats sampled along these transects, or quadrats are established on each zone directly.

Size of quadrats is of 0.5 x 0.5 m or 1 x 1 m.

The property sampled in most cases is density of seedlings of species under study, sometimes also size of seedlings or of tree trunk is recorded.

Stand post-fire age in these works is usually one year, but one of the works considered post-fire ages ranging from 2 to 20 years.

Duration of studies takes no longer than 18 months.

The most studied species in this kind of approach is *Pinus halepensis*.

Statistical analyses used are non-spatial, and comprise several ANOVAs and other classical tests such as Tukey's test, t-test and linear regression.

All works described here have been carried out in Israel, with the exception of one carried out in Spain (BAUTISTA and VALLEJO 2002).

One of the works analysed (BAUTISTA and VALLEJO 2002), apart from the methodology described above, uses a slightly different approach, sampling vegetation presence/absence in 25 m transects at 10 cm intervals.

Analysis performed is two-term local variance method (TTLV), a variant of block variance methods.

Generally, CORONA et al. (1998) suggested that, after using the relative efficiency method to compare the efficiency of each size and shape, the early regeneration of Aleppo pine proved to be best assessed by 8 m x 2 m plots.

The need for such wide plots depends upon the typical spatial arrangement of the seedlings whose clustering is associated with the presence of seed bearing trees

3.3.4 Biomass production : B CÉSPEDES and H FERNÁNDEZ UCLM

After a wildfire or experimental fires post-fire regeneration was studied and compared to unburned sites (controls sometimes with different ages).

Resprouting, mainly by woody species, and seedling, mainly from the herbaceous species during the first years after the fire, with environmental factors constitute the main critical factors that determine different patterns of recovery.

Several papers analyse the effect of burning frequency and seasonality on phytomass production.

To determine the above ground biomass samples of subplots are collected (1 m^2 each) randomly along the burned and unburned plots at the same time.

In general, the existing vegetation is cut down to ground level, separating each woody species (sometimes were separated photosynthetic and non-photosynthetic fractions or were separated into different fractions of fuel categories) and considering herbaceous ones as a whole.

The dry weight is measured after oven-drying at 60-100ºC to constant weight (around 12-48 hours according to temperature).

In regeneration studies (density and cover of dominant species) different number of permanent quadrates with diverse size were situated randomly or along a transect in the plots.

For studying shrub seedling recruitment and herbaceous, additional quadrates (generally subplots of 0.25 m^2) are located along different plots.

In general, although biomass of herbaceous species increased on first years in burned plots, above-ground phytomass decreased with increasing fire frequency and intensity, mainly due to the lower biomass of woody plants that regenerated in the mature phases of the community regeneration.

The analysed papers are from the Iberian Peninsula, France, Portugal and Greece.

3.3.5 **Nutrient balances:** B CÉSPEDES and H FERNÁNDEZ UCLM

One of the main processes associated to fires concerns the effect of burning on the nutrient cycle. Quantitative data on the impact of fire on biogeochemical cycles in the Mediterranean basin are scant.

There is more information available on the nutrient changes after fire in other Mediterranean-type ecosystems such as in California (DE BANO and CONRAD 1978) or in others countries.

In this search only 3 papers has been found to be related geographically to Mediterranean Europe.

We can split the papers in two groups:

The first group analyses the effects of fire at two times: pre-fire and post-fire (TRABAUD 1994): nutrient levels are determined in field collected pre-fire vegetation and combustion residues.

Nutrient losses are calculated using pre-fire concentration and the biomass consumed by the fire.

In the second type of studies the effect of fire in nutrient changes using temporal sequence (CARREIRA and NIELL 1992).

The reference of GILLON et al. (1999) can fit into the second group; one part is a study of the immediate fire effect on nutrient changes and the type of fire (downhill and uphill fires) and in the other part they aim at determining the short-term effects on the cycles of nutrients.

All vegetation is separated in different classes because their flammability and nutrient content are different.

Nutrient analysed changes since only N and P (GILLON et al. 1999), and N, C and cations (TRABAUD 1994).

Much attention has been paid to nitrogen pools and dynamics associated with fire (three articles analyse this nutrient) because N often limits primary productivity in natural ecosystems.

In addition, N is easily lost during fuel combustion because N volatilises at relatively low temperature.

Methodology for chemical analyses is very variable because there are many methods for measure nutrient contents in biomass, but the results are consistent.

4 MEDITERRANEAN ECOSYSTEMS: ANIMALS

This chapter has been compiled by R Prodon, EPHE

4.1 INTRODUCTION

Studies on the ecological impact of fire – either wildfire or prescribed burning – on animals were sufficiently numerous in North America, Australia and South Africa, to have been the subjects of several reviews.

One may quote, among others, the syntheses of Koslowski and AHLGREN (1974), CONRAD and OECHEL (1982), WRIGHT and BAILEY (1982), KAUFMAN et al. (1990) for different parts of North America, BRADSTOCK et al. (2002) for Australia, BIGALKE and WILLIAM (1984) and WILGEN et al. (1992) for South Africa.

Concentrating ourselves on southern Europe – i.e., the countries of the Mediterranean Basin, including its surrounding mountains in particular the southern side of the Alps – we tried to review all the papers dealing explicitly with fauna in this area.

We excluded the case of the European boreal forest (see GRANDSTROM 2001) and the case of central Europe (see GOLDAMMER and PAGE 1997).

Despite the ancientness of the fire problem in the Mediterranean, no research program on the impact of fire on the fauna seemed to have been launched before the end of the seventies.

Fortunately, in the last 20 yrs, things have been progressively changing, and the animal component of the ecosystem began to be taken into account in research programs. I

In our review, we will do no distinction between wildfire and prescribed burning, as the latter were often used by animal ecologists as an experimental substitute of the former.

4.2 CHRONOLOGICAL AND GEOGRAPHICAL DISTRIBUTION OF FIRE STUDIES ON ANIMALS

The problem of fire was mentioned since a long time, among other environmental factors, in ecology-related publications on the Mediterranean fauna, but papers dealing explicitly with this topic and including field data seem to have appeared only on the beginning of the eighties, initially at the slow pace of ≤1 paper.year^{-1}.

Since then, the number of papers regularly increased at the rather modest pace of about 4 papers.year^{-1}, to reach presently the total of about 90 (without taking into account unpublished material like theses or reports).

This fairly regular increase in the number of studies must not conceal their extremely uneven geographical distribution in the Mediterranean, since about 70% of these papers come from 5 study areas only (Eastern-Pyrenees, Mount Carmel in Israel, Attica in Greece, Barcelona area, and Lago Maggiore area in Swisserland).

It seems to be a complete lack of data for the whole North Africa.

4.3 ANIMAL GROUPS STUDIED

4.3.1 Arthropods

The works on the impact of fire on soil fauna deals mainly with micro-artropods, i.e., Pseudo-scorpionida, Acari, Geophilomorpha, Polyxenida, Symphila, Collembola, Thysanura, Protura, Diplura, and various insects and insect larvae (Hemiptera, Psocoptera, Coleoptera, Dermaptera, Diptera, etc.).

These groups, that all can be sampled with a common methodology (dry-way extraction with Berlese's funnel), are often studied collectively.

The case of certain taxa may be detailed, e.g., among mites (Acari; a very large group in terms of both individual and species numbers), the Oribatids and the Uropodids, and among insects, the Collembola.

Among terrestrial above-ground arthropods, data are available on spiders and Carabid beetles, that constitute relatively classical material in landscape ecology, and that can be sampled with a common methodology (pitfall traps).

The case of the macro-arthropods of the litter (centipedes, millipedes, spiders, insects) is sometimes considered on its own; sampling such animals may require either Berlese's funnel or pifall traps, or both.

Ants, an insect family both well-diversified and functionally important in the Mediterranean ecosystems, generally require specific sampling designs and methods.

There were, at least in France, several field studies on grasshoppers (Orthoptera) – an insect order that plays an important role in grassland ecosystems – after controlled burning, but most of them remained unpublished; sampling grasshoppers require relatively specific methods, some of them giving "absolute" densities (although with an unknown efficiency), other ones giving only relative abundances.

4.3.2 Vertebrates

The majority of fire studies on animals deals with birds.

At least during the breeding season, a large part of the avifauna can be sampled collectively by methods (quadrats, transects, point censuses) that use both visual and acoustic clues.

The corresponding analyses at the community level represent the majority of the available publications on the topic.

They address a variety of issues related to the impact of fire on species dynamics and landscape patterns (see below).

Much rarer are the studies of birds at the population level.

Published data seem to concern only the Pyrenean gray partridge, several *Sylvia* warblers, and the case study of a wintering passerine.

The scope of the available data on mammals also result from the methodology of the sampling.

The efficiency of the widely used live trapping is limited to the small mammals.

Consequently, the data only concern mice of the genus *Mus*, field-mice (g. *Apodemus*), dormice (g. *Eliomys*), shrews (g. *Crocidura*), and – in the eastern Mediterranean – gerbilids (g. *Gerbillus* and *Meriones*).

In one area (eastern-Pyrenees), the parasites of the small mammals were intensively studies.

It seems to exist only one study dealing with the interaction of fire with grazing by a large herbivorous mammal, the mountain goat (*Capra aegagrus*).

Available studies on reptiles are limited to the tortoise *Testudo hermanni*.

Only one publication deals with the case of the European pond turtle (*Emys orbicularis*); this work seems also to be only one that concerns an aquatic animal.

4.3.3 Other groups

Data on other animal groups are limited to the case of the terrestrial molluscs.

4.4 SAMPLING DESIGNS

It would not make sense to attempt a review of the methods concerning animals since these methods are not only group-specific, but sometimes even species-specific.

Some of them give data at the community level.

It must be kept in mind that in this case, the word “community” has a methodological rather than a biological meaning.

Among them, certain sampling methods (e.g., most of the sampling methods for insects) imply the death of the collected animals.

In certain cases, even the environment of the fauna must be collected (extraction of soil fauna with Berlese’s funnel), obliging the observer to change the precise sampling site from one session to the other, and so adding a source of variability in case of long-term monitoring.

These methods are well adapted to arthropods that are generally difficult to identify precisely.

At the other extreme, certain methods do not require any direct physical contact with animals, and can be repeated on the same place at even short time intervals (breeding bird censuses for the avifauna).

Non-destructive, fast, easy, these methods are particularly well adapted to long-term monitoring.

But, as the probability to record an animal present is generally less than one (and often much lower), the sampling procedure is worthy to be repeated several times within each sampling session if the aim is a precise estimation of certain parameters like species richness or species turnover.

The case of methods (mist-net for birds, live-traps for small mammals) implying the capture of animals that can be released afterwards is specially interesting.

These methods offer the possibility to mark individually the animals, and to recapture them later.

Such capture-mark-recapture techniques give access to precise demographical analyses leading to unbiased estimations of parameters, like survival rates, that are of paramount importance when assessing fire impact on animals.

Published works used either synchronic or diachronic designs, in certain cases both of them.

As far as we know, the longest duration of published diachronic studies is 8 years after fire (even if publications of much longer series are currently in preparation), and synchronic studies mention plot ages as high as 38 years.

In the case of ancient fires, the problem is often the reliability of the information concerning the limits of fire and its severity.

4.5 PARAMETERS RELEVANT FOR THE FAUNA

4.5.1 Population level

Species abundance may be either relative or absolutes.

In the former case, the estimations directly resulting from the result of sampling are generally species- or group-specific, and direct between-group quantitative comparison have to take into account this problem.

Even in case of within-species between-samples comparison, the problem of possible variations in animal detectability has to be considered.

This is particularly evident in case of pre-/post-fire or burned/unburned comparisons, where the disappearance or reduction of the plant cover often results in an increase (but sometimes in a decrease) of the efficiency of the sampling method.

This issue is too rarely considered and does not admit any easy remedy.

To call for absolute abundance measurements is not always possible; moreover, they may also be sensitive to variations of efficiency.

The use of capture-recapture methods with individually-marked animals must theoretically lead to better estimates.

It is particularly true for the estimation of demographic parameters.

For example, estimating a mortality rate must take into account the presence of undetected animals both before and after the fire.

We will deal more extensively with this problem in next deliverables.

But there remains the impossibility to make the difference between animals that were killed by the fire and animals that definitely left the study area.

Measuring immigration and emigration, and identifying the source of animal recolonisation of burned areas, is one of the challenges of fire studies.

4.5.2 Community level

Diversity indices seem to be of limited interest in the measurement of the impact of fire.

Parameters relative to species number (species richness), and species turnover are affected by sampling methods and efficiencies.

Concerning species composition, useful parameters stemming from the use of multivariate statistics may be of great interest in measuring the impact of fire and the resilience of animal communities.

We also will deal later more extensively with these problems, for which we are currently testing available methods in our lab.

4.5.3 Conservation value

The concept of “conservation value” is not strictly an ecological one, but rather concerns the communication with general public or stakeholders.

Several *ad hoc* measurements may be proposed for a quantitative evaluation of the impact of fire at the community level in term of conservation.

The only published one concern the impact of prescribed burning on birds.

4.6 DISCUSSION

Data on the impact of fire on fauna are still extremely incomplete in the Mediterranean region.

The published works form a scatter of case studies making difficult any attempt of synthesis.

Whole important groups are still awaiting field studies (e.g., earthworms; amphibians and reptiles other than tortoises; insects other than Orthoptera, Coleoptera, ants, or soil insects; large mammals; aquatic animals; etc.).

The study of animal-vegetation relationships in a fire context (e.g., structure-dependence, seed dispersal, grazing) is at its very beginning.

Due to the variety of the sampling methods and to their taxonomic specificity, it would be much premature to look for unified sampling procedures for animals, except perhaps in few favourable cases (soil fauna, birds).

But it remains essential to get on unified procedures of measurements of demographic parameters or parameters related to community dynamics able to precisely quantify the ecological impact of fire

5 MEDITERRANEAN ECOSYSTEMS: SOILS

This chapter has been compiled by R VALLEJO CEAM, based on contributions from G GIOVANNINI and S LUCHESI CNR-IES-SCS.

Fire is a very complex phenomenon in which many components act simultaneously; it, in addition, affects the whole ecosystem.

Even focusing our attention only on the impact on soil, the phenomenon remains complex enough with the simultaneous presence of two fire's components: the heat input and the ash layer deposition, and these two components affect in different ways the various soil domains.

Field researches and the derived literature reports, very often, deal with the whole problem, thus making the understanding of the mode of action of fire on soil difficult.

We feel that a rationale way for correctly understand the impact of fire on soil is the specific examination of the effects of the various fire's components on the different soil domains, according to, for instance, the following chapters:

- 1 The mode of action of fire on soil
- 2 The thermal reactions of soil
- 3 The effect of heating on soil physical parameters
- 4 The effect of heating on soil chemical parameters
- 5 The ashes
- 6 The ash leachate
- 7 The impact of fire on soil physico-chemical parameters
- 8 The impact of fire on soil micro-organisms
- 9 The effect of fire on nutrient cycling and nutrient budgets
- 10 The effect of fire on soil ferility
- 11 The resilience of burned soils
- 12 The soil water repellence
- 13 The soil erosion : usle – musle - rusle
- 14 The soil erosion : eurosem
- 15 The soil erosion : wepp
- 16 The estimation of soil erodibility factor k
- 17 The vegetal protection of soil
- 18 The estimation of the vegatal covering factor c
- 19 The prediction of post-fire soil erosion : prometheus
- 20 The prevention of the post-fire soil erosion
- 21 The prediction of post-fire soil erosion : schematic diagram

5.1 THE MODE OF ACTION OF FIRE ON SOIL

The soil environment, during and immediately after a vegetation fire, is directly affected by input of heat and ash.

In the field, the effects of these two factors are concomitant, thus making identification of the individual causes of changes in soil properties difficult.

This suggests the need to study separately the two effects in order to understand the mode of the action of fire on soil.

Characteristics of the soil environment are altered both as sudden modifications induced by the passage of the fire and also as delayed changes derived from the simultaneous modifications of the soil physico-chemical composition, of the plant covering capacity and of the biological spectra.

Sudden modifications are caused by both the heat wave that accompanies the fire and by the ashes deposited on soil surface as consequence of fire, they are very striking and immediately perceptible, but the delayed changes leave their mark on the soil and determine its future evolution.

The resilience of burned soil too is ruled, once again, by both the thermal shock suffered by soil and the input of ashes, or their leachate.

Each of these two factors is predominant in relation to the elapsed time since the occurrence of fire and to the seasonal and climatic conditions.

The resilience of the soil ecosystem, in addition, represents the crucial point for a correct management of the burned soil.

It is of fundamental importance to ascertain the time elapsed since the passage of fire in order to define the real level of evolution occurred in soil after the fire.

An exact knowledge of the causing mechanisms and of their temporal sequence, therefore, appears as the sole successful way to define a correct policy of management of burned soils.

5.2 THE THERMAL REACTIONS OF SOIL

The current literature reports many articles on the artificial heating of soil (BETREMIEUX et al. 1960, NISHITA and HAUG 1972, SERTSU and SANCHEZ 1978, KANG and SAJJAPONGSE 1980, KITUR and FRYE 1983, GIOVANNINI and LUCCHESI 1988, GIOVANNINI et al. 1990) .

All report programs of heating graded in constantly increasing temperature intervals of fifty or a hundred degrees.

This should imply a linear response of the soil to the heating; on the contrary, the soil, as the heating increases, undergoes distinct and well defined thermal reactions: in other words the soil responds to the heating according to a "discrete step" model.

The Differential Thermal Analysis (DTA) is a useful tool to discover the thermal reactions really occurring in soil when the heating increases (GIOVANNINI and LUCCHESI 1984)

The standard DTA of soil is characterised by:

- a very broad endothermic peak, beginning just as the heating is applied and ending at 170° C, due to the dehydration of the sample,
- a very small endothermic peak starting at 170° C and ending at 220° C, due to dehydration of the gel forms,
- a well-defined exothermic peak starting at 220° C and ending at 460°C, due to the combustion of the organic matter. The thermal reactions interesting rearrangements and transformations of cristallinity of iron and aluminium oxides usually occur in this temperature interval but are masked by the intense exothermic effects derived from the organic matter combustion. They may be revealed operating under nitrogen atmosphere,
- a very sharp endothermic peak starting at 460° C and ending at 700° C, due to the loss of OH groups from the clays,
- a well-defined endothermic peak starting at 700° C and ending at 900° C, due to the decomposition of carbonates.

The DTA, in addition, is able to reveal the presence and the thermal behaviour of other materials present in soil like: pollutants, specific organic compounds, hydrophobic materials and so on.

Suggestion is made to refer always to the above-mentioned thermal reactions and to the relative temperature interval of occurrence, when discussing about the heating of soil, in order to have a clear and physically significant reference.

5.3 THE EFFECT OF HEATIG ON SOIL PHYSICAL PARAMETERS

Heating the soil has variable effects on physical parameters ((BETREMIEUX et al. 1960, SIVARA et al. 1962, NISHITA and HAUG 1972, SERTSU and SANCHEZ 1978, GIOVANNINI and LUCCHESI 1988).

The increasing heating has a negligible effect on the plastic and liquid limits of soils up to 170°C, a little decrease is noted at 220°C while after 460 °C and more conspicuously after 700° and 900°C the soil adsorbs even more water but it does not form any plastic paste.

In correspondence of the combustion of the organic matter the soil loses its plasticity and elasticity.

Heating up to 170°C has only a small effect on the particle- size distribution but after 220°C the sand fraction increases sharply while the silt and the clay fractions decreases.

These changes are more pronounced in those soils that have higher initial clay content.

These changes are attributed to the fusion of clay particles into sand-sized particles probably due to calcination in which iron and aluminosilicate are involved.

The water stability index of the soil aggregates shows a continuous increase after all the thermal reactions and also after the combustion of the organic matter, generally considered the most important cementing agent.

The more pronounced increase occurs in the range 220–460°C, when the thermal transformations of the iron and aluminium oxides also occur.

The consideration that the internal reorganisation and recristallisation of the iron and of the aluminium oxides contribute to increase the resistance of the soil aggregates, particularly in concomitance with the combustion of the organic matter, suggests that during the heating process, the soil undergoes a kind of laterisation.

The increasing heating produces different modifications of the porosity of the soils.

In the clayey textured soil, the porosity increases continuously up to 460°C, after this point the porosity sharply decreases as a consequence of the loss of the OH groups from the clays and the disruption of carbonates.

On the contrary, in the sandy textured soil, the porosity decreases continuously with the most striking evidence in the temperature interval between 170 and 220°C.

5.4 THE EFFECT OF HEATING ON SOIL CHEMICAL PARAMETERS

Heating the soil has variable effects on chemical parameters (NISHITA and HAUG 1972, SERTSU and SANCHEZ 1978, KANG and SAJJAPONGSE 1980, KITUR and FRYE 1983,GIOVANNINI et al. 1990).

Soil heating at 220°C causes a pH decrease, at 460°C an increase at the initial level occurs, while heating at higher temperature, 700°C and 900°C, increases the pH greatly by 4 to 5 units.

The causes of the initial decrease could be attributed to the oxidation of certain elements, the exposure of new surfaces, the dehydration of colloids and consequent decrease of the buffer action (COLES and MORRISON 1930, BETREMIEUX et al. 1960, RUSSEL et al. 1974).

The sharp increase of the pH at high temperatures, on the contrary, may be due to the loss of the OH groups from the clays and, finally may be ascribable mostly to the formation of oxides of several elements derived from the disruption of the carbonates.

The Cation Exchange Capacity decreases progressively with increasing temperature.

The aggregation of finer particles promoted by the heating can cause this decrease, however this is not the sole factor.

The simple drying of the colloidal particles at lower temperatures may have an influence as well as the combustion of the organic matter.

At the higher temperatures the dehydration of the mineral crystal lattice and the resulting breakdown of the lattice can also contribute to the Cation Exchange Capacity reduction.

The effect of heating on soil Organic Matter content is well defined in all soils.

There is no detectable effect until 170°C, a little decrease occurs at 220°C, whereas at 460°C the combustion is practically concluded.

These results are in total agreement with the results of the thermal analyses.

The total Nitrogen shows similar behaviour content.

A little decrease occurs up to 220°C, whereas a very pronounced decrease takes place in concomitance with the combustion of the organic matter: after this combustion the content of total Nitrogen is really very low.

The NH_4^+-N, on the contrary, increases greatly with the heat treatment up to 220°C, then decreases above this temperature and after the 460°C is barely detectable.

The increase in NH_4^+-N with heating is due to the mineralisation of organic NH_4^+- containing complexes in the soils; the decrease at higher temperatures is due to fixation or volatilisation.

The heating promotes a mineralisation of the Organic Phosphorus and a continuous decrease of this form accompanied by an equivalent increase of the inorganic form is detected.

After 460°C the organic form is totally destroyed and the Phosphorus is present only in the inorganic form.

The amount of Available Phosphorus gets a remarkable increase with increasing heating up to 460°C followed by a sharp decrease after this temperature, which suggested that the available P might come from the mineralisation process of the organic P.

5.5 THE ASHES

The ash is the residual material remained after burning of fresh vegetation or dry litter.

Useful information is reported in the cited references: JORDAN 1965, DAUBENMIRE 1968, LEWIS 1972, VIRO 1974, GIOVANNINI 1997.

The amount of deposited ash depends on the weight and spatial distribution of vegetation, its degree of combustion and the subsequent transport of burned residues.

Returns of ash vary from 2-9% for wood and 13-20% for grasses.

The ash properties depend on the burning conditions.

If the combustion is not complete the ash is black because it still contains residual organic matter and charcoal materials.

Where the combustion is complete the ash is white-grey and has practically a mineral composition.

The composition of ash depends on the composition of the original vegetal material.

This topic has been largely investigated but unfortunately some of the ashes analysed were obtained by muffling which is likely to result in an ash of higher mineral content than would be produced during field burns.

Ash residues are generally dominated by carbonates of the alkali and alkaline earth metals with variable amounts of silica, sesquioxides, phosphates and small amounts of organic and inorganic nitrogen.

Data reported in literature demonstrate the great variability in composition of ashes depending from the source, so we can find variability of N content from 0.03 to 1.5%, of P content from 0.03 to 3.0%, of K content from 0.3 to 20%, of Ca content from 2.5 to 25% and Mg content from 1.5 to 15%.

Generalising we can say that most of organic constituents are burned and dispersed in the air while many cations, previously linked with organic constituents and incorporated in greater complexes, are rendered water soluble and immediately available for plants.

In some experiments of artificial leaching of litter and ash, it has been found that fire increased the solubility of the various cations while the solubility of nitrate and phosphate was identical in burned and unburned litter samples.

Compared to biological decay of plant remains, burning rapidly release some nutrients into a plant available form.

High levels of available nutrient in surface soil can persist for a long time after the ash deposition.

After burning of forest and shrub vegetation the amount of C and N in soil may increase as consequence of addition of partly ashes residues.

The amounts of P, K, Ca and Mg released by fire and accumulated on soil surface in the ashes are up to 10 times higher in respect to both the total and available quantities of these elements in soil.

All these considerations show that the ash beds deposited after the passage of fire can be considered as a reservoir of nutrients that improve the soil fertility and that facilitate the plant growth.

5.6 THE ASH LEACHATE : EFFECTS ON SOIL ERODIBILITY

Ash residues are generally dominated by carbonates of the alkali and alkaline earth metals with variable amounts of silica, sesquioxides and phosphates.

Leaching of ash produces the hydrolysis of the contained basic cations and the formation of an alkaline residue, which may have a pH exceeding 12 that can increase the pH of the soil .

The extent of the increasing of soil pH, depends on the buffer capacity of the soil that can be decreased by the heating during the fire.

It is well known that the alkaline solutions are effective in solubilising soil organic matter and that the organic matter, holding together soil mineral particles, promotes the aggregation.

On the other hand ash leachates bring into the soil a great density of electrical charges that can favour the flocculation of the dispersed clays.

The addition of ash leachate to the soil can therefore influence the potential soil erodibility that is decreased by the flocculation and accelerated by the dispersion.

Although the structure of surficial soil is determined by aggregation, the degree of flocculation or dispersion of the clay fraction becomes more important in the subsurface.

Specific experiments were performed to clarify this topic.

Fresh vegetation and dry litter collected in the Mediterranean maquis were ashed at 585°C leaving a well formed bed of white ashes that, without muffling, were leached with water in the ratio 1:50.

The pH of the leachate was 11 and the cationic composition indicated the presence of Potassium (1600 ppm) Sodium (860 ppm), Calcium (20 ppm) and Magnesium (2 ppm).

Ten grams of soil were treated with 500 cc of ash in the first six hours the alkaline solution was able to disperse the soil particles at the same extent to that produced by the addition of Sodium hexametaphosphate.

Increasing the contact time, on the contrary, the ash leachate promoted the aggregation of the finer clay particles into silt-size particles and after a minimum contact time of 30 hours an increase of the silt fraction of about 30% was obtained.

The flocculation of the clay particles continued up to a contact time of 78 hours and the increase of the silt fraction, to the detriment of the clay fraction, reached the 78%.

During the experiment no changes were observed for the sand fraction that remains unaltered at the values obtained after the sudden dispersion produced by the alkaline solution.

In another experiment the sub-surficial compact fill layer of various little plots were sprayed with a similar ash leachate.

After spraying, the plots were allowed to stand and finally treated with a rain simulator for 36 minutes.

The results showed that ash-leachate treated plots had 36% less erosion than non-treated plots.

The reduced erosion on treated plots is most likely due to the flocculation of the clay fraction induced by electrolytes from the ash-leachate.

Electrolytes, and cations in particular, allow the negatively charged clays to form flocks, making the clay less detachable and more resistant to erosion.

Interesting references on: DE SERRA and SCHNITZER 1972, HALCOMB and DURGIN 1979, GIOVANNINI et al. 1987, GIOVANNINI 1997.

5.7 THE IMPACT OF FIRE ON SOIL PHYSICO-CHEMICAL PARAMETERS

The passage of fire causes many modification in soil physico-chemical parameters.

Extent and degree of the modifications are a function of the heat developed during the fire.

Heat transfer in soil is mainly by thermal conduction, and the conductivity increases with the moisture content.

Thus, heating of dry soil cause a greater rise in surface temperature, but less penetration of heat compared with moist soil, therefore, considering that in Mediterranean area wildfires occur mainly in the hot summer season when soils are more dry, the greater modifications are to be expected in the upper layer of soil (0-2.5 cm) (DE VRIES 1975, GIOVANNINI and LUCHESI 1983, GIOVANNINI et al. 1987, GIOVANNINI 1997, GIOVANNINI and LUCHESI 1997).

Experimental data show that burning causes an increase of soil bulk density and a decrease of soil porosity.

The combustion of organic matter implies the decrease of its volume fraction and an increase of the volume fraction of minerals that have a higher density and a lower porosity.

The particle-size-distribution undergoes, with the increasing temperature, a continuous increase of the sand fraction corresponding to a simultaneous decrease of the clay fraction.

The final content of silt fraction normally results as a statistic compensation of the above-mentioned fluxes and depends on the initial silt content in soil.

The soil aggregate stability increases with two rapid increments more pronounced when there is denaturation of the gel forms and transformations of the iron oxides, however, the cementation remains unaltered on combustion of the soil organic matter.

These results reaffirm the relevant role of iron and aluminium in the cementation process and the occurrence of a kind of laterisation during the heating of the soil.

Soil pH decreases with the increasing temperature up to 400°C owing to the lowering of the buffer action associated with denaturing of the colloids and the combustion of the organic matter.

Then increases thanks to the loss of OH groups from the clay minerals.

Cation exchange capacity decreases progressively with the increasing temperature.

The aggregation of the clay particles into sand-sized particles with the consequent decrease of the reactive surface area and the combustion of the organic matter with the disruption of the exchange sites seem to be responsible for this.

Soil organic matter content decreases following a sigmoid curve with a rapid decline in the interval 210°-400°C.

The overall trend in organic matter content agrees with the responses of the differential thermal analysis of soils and an eventual incomplete combustion of the organic matter is ascribable to the short residence time of the fire.

Total soil nitrogen content after fire results as a compensation between the decrease due to volatilisation and the increase derived from the incorporation of N-containing compounds of the deposited ashes.

The ammonium nitrogen and nitrates normally are volatilised and lost.

Soil organic phosphorous decreases continuously in parallel with the combustion of the soil organic matter, whereas, the available phosphorous increases, in the same temperature interval confirming that the available phosphorous is the outcome of the mineralisation processes of the organic phosphorous.

All these data are in good or close agreement with the previous results on the effects of the artificial heating of soil, and confirm that the modifying processes occurring in soil are strictly related to the temperature reached during the fire at the soil surface, whereas, at least immediately after fire, the ashes do not affect these processes.

Immediately after the fire, the ashes appear as a solid phase super imposed on the solid phase soil; the two phases are clearly separated.

The ashes can develop their action only in the form of leachate, that is, when the rain leaches the soluble ash-components and brings them, as solution, into the soil pores.

So the effects of the ashes are to be expected only after the rains, but the sudden modifications induced in soil physico-chemical properties are clearly ascribable to the heat that affects the soil during the fire.

5.8 THE IMPACT OF FIRE ON SOIL MICROORGANISMS

The passage of fire front besides to induce modifications in soil's physico-chemical characteristics can sterilise the upper soil layer.

Temperature above 127°C sterilise the soil, and a 10 minutes exposition at 70°C will kill non-spore forming fungi, protozoa and some bacteria.

After fire an appreciable decrease in number and activity of most microorganisms is normally detected.

The extent of the effects on various species is influenced by fire intensity and moisture conditions.

Lethal temperatures for heterotrophic bacteria in chaparral and forest soils are reached at 210°C in dry soil, but most are killed by temperature above 150°C.

In wet soil, rapid death begins at 50°C and no bacteria survive beyond 110°C.

Nitrifying bacteria appear to be somewhat more sensitive to soil heating than typical heterotrophic bacteria.

Nitrosomonas and *Nitrobacter*, indeed, are killed in dry soil at temperature of 140°C, but at only 75° and 50°C, respectively, in wet soil.

Actinomycetes behave very similar to bacteria with death at 125°C in dry soil and 110°C in wet soil.

However, some studies have reported that these microorganisms are generally more resistant to heating than bacteria.

However, reinoculation by windblown dust and debris soon follows; when moisture is sufficient and/or after the first rainfall, the microbial population abruptly increases for some weeks until a new equilibrium is reached.

Insofar as burning changes the soil properties, the micro-organisms having advantage in a burned soil will differ from those having the advantage before burning, since for most part effects of fire are indirect through the physical and chemical changes induced.

These changes vary in degree and duration by intensity of burn, soil and climatic characteristics of the site and kind of vegetation that invades the area after fire.

All this accounts for the great variability of the strategies of the bacterial recolonization met in the different burned areas.

For instance, it is well know that the passage of fire increases significantly the soil pH and this favours the bacterial population growth in respect to fungal population growth.

It has been frequently hypothesised that the high soil nitrification rates commonly observed following fire result from increased activities of population of *Nitrosomonas* and *Nitrobacter*.

However in some experiments in chaparral soils these populations remained very low in the first 12 months following fire, justifying therefore the new hypothesis that heterotrophic nitrifiers have a greater and primary importance in these soils in the first year following fire.

In another experiment on the recolonisation of the ash-bed soil produced by burning a log pile, it has been detected that bacteria rapidly recolonised the soil and shortly after the fire their number exceeded those of unheated soil, whereas actimycetes and fungi recolonised the ash-bed soil more slowly.

The early bacterial and fungal recolonizers included many types not detected in untreated soil about a year was required for return to the pre-fire situation.

Literature references on: AHLGREN and AHLGREN 1960, AHLGREN and AHLGREN 1965, PETERSEN 1970, DUNN and DEBANO 1977, DUNN et al. 1979

5.9 THE EFFECT OF FIRE ON NUTRIENT CYCLING AND NUTRIENT BUDGETS

Fire has the potential to change ecosystem nutrient capital and mobility, and hence to affect plant growth and survival in situations where the availability of nutrients is a major factor controlling these processes.

(BENTLEY and FENNER 1958, JORDAN 1965, LEWIS 1974, SERTSU and SANCHEZ 1978, BIEDERBEEK et al 1980, KANG and SAJJAPONGSE 1980, KITUR and FRYE 1983, GIOVANNINI et al 1990, CHANDLER et al. 1991).

Fire is likely to have most effect on systems, which depend for their long-term stability on efficient nutrient accumulation, retention and recycling processes.

Such situations are often characterised by low soil nutrient reserve with a large proportion of nutrient capital contained in the biomass; and rates of decomposition the soil-litter subsystem may regulate the productivity of the entire plant community.

Burning leads to some direct nutrient loss from the system, either by volatilisation and convective transfer of ash, or by loss of ash due to the action of wind or water.

Some of the nutrient in smoke may be transported away from the fire's site and lost.

Nutrient loss is usually small in relation to the total soil and biomass reserve.

However, it may be more relevant to express nutrient losses as a proportion of the nutrient in the biomass since not all nutrients in the system are equally mineralisable and hence available for plants.

Most attempt to estimate nutrient loss due to volatilisation during fire in the field have been based on the differences between nutrient content of vegetative material before burning, and that of the residues after fire.

This approach probably presents a low accuracy especially for slash burns because it is difficult to estimate the nutrient content of large logs, and to collect all the residues remaining after fire.

Another way to estimate the loss is the direct measure of nutrient in both particulate and gaseous form after the vegetation burning.

However, unless burning is carried out under controlled conditions, quantification of losses is virtually impossible.

Large losses of some nutrients, particularly N, can occur during slash burning, therefore this practice appears senseless in nutrient deficient sites, in sites where the replacement of losses is slow and in situations where the retention of the fire-mobilised chemicals by the soil is poor.

Apart from the direct effects of fire on nutrient cycling, there are potential longer term effects such as changed patterns of nutrient uptake or N fixation resulting from mortality or alteration to the plant community occupying a site.

In forests where fire occurs frequently, the over storey often grows in association with a number of leguminous species which are adapted to survive burning and the seed of which is stimulated to germinate by heat.

The legumes may replenish N lost during fires and are surely important to the N economy of the poor forest soils.

5.10 THE EFFECT OF FIRE ON SOIL FERTILITY

The concept of soil fertility is universally utilised, but its simple, concise and exhaustive definition appear still difficult.

In 1973 the Soil Science Society of America (Soil Science Society of America) proposed the following definition: "Soil fertility is the status of a soil with respect to the amount and availability to plants of elements necessary for plant growth".

This definition, anyway, implies that amount of growth is a variable dependent on the level of soil fertility but that other factors such as type of plant and growing conditions may also significantly affect growth. (PEECH et al. 1947)

Soil fertility, therefore, is not a simple but a "complex" concept and sometimes the sub-concepts as "physical fertility", "chemical fertility" and "microbiological fertility" are used (COOK 1939, COTTENIE 1978, GIOVANNINI et al. 1988, GIOVANNINI et al. 1990, GIOVANNINI and LUCHESI 1997, GIOVANNINI et al. 1997).

In previous chapters we have seen that fire can influence soil physical, parameters, but heating up to 500°C induce in soils modifications that are neither irreversible nor detrimental for plant growth; and the 500°C is the maximum temperature reached at the soil surface in most forest fires.

The passage of fire can cause moderate losses of some nutrients but in the same time mineralise many elements previously linked with organic constituents and incorporated in greater complexes, rendering them water soluble and immediately available for plants.

Plants can take advantage from this large availability of nutrients.

Even very low temperatures induced by fire can sterilise the upper soil layer, but microbial re-inoculation is very easy and after the first rains the microbial population abruptly increases, in so favouring the nutrients uptake by plants.

After a fire, when the moisture content is just adequate a rapid re-vegetation is observed in burned soils.

The final conclusion, therefore, is that a single fire event is not detrimental but, in some cases, may be beneficial for soil fertility.

The situation appears totally different when repeated fires occur; in this case the ecosystem has not time for the normal resilience, the detrimental effects become additive and no more equilibrated by the beneficial effects.

In case of repeated fires, in addition, the soil erosion is enhanced causing the shortening of the first requirement for soil fertility: the soil .In these cases the decrease of soil fertility appears extremely evident.

5.11 THE RESILIENCE OF BURNED SOILS

The sudden modifications occurring in soil during the passage of fire are strictly related to the temperature developed at the soil surface whereas the ashes do not affect these processes.

Some days after fire, anyway, the vegetative processes start again accompanied by the revival of microbial life.

Three months after fire it is already possible to detect a moderate increase of the Organic Matter, Organic and Available Phosphorus contents and of the Cation Exchange Capacity, after the drastic decrease of these parameters produced by fire.

This trend continues in the following months up to the first important rainfalls.

Moreover this process appears more pronounced and the regain of the pre-fire values more enhanced where the fire had been more intense and had induced a major decrease.

This early accumulation in soil is clearly ascribable to the biological decay and to humification processes of the remains of shot-living plants just re-vegetated.

When the first important rainfalls begin this trend to regain the pre-fire values appears as stopped and the acquired values do not increase anymore but follow only seasonal fluctuations at the same extent of the unburned soils.

The vegetative processes with the decay and the microbial decomposition of vegetal remains, and all the normal processes occurring in soil continue, of course, even during the season of the rains but the products of new accumulation in soil are probably washed away and eroded by the rains that, therefore, behave as an equaliser and a maintainer of the acquired equilibriums.

With the fist rains, anyway, a new interesting process starts: the leaching of the ashes.

The ashes contain remarkable amounts of N-containing compounds and many basic cations rendered soluble by the fire.

The rains are able to leach the N-containing compounds and to incorporate them in the soil, so after rains a great increase of Nitrogen and Ammonium may be detected in burned soils.

Rains, in addition, produce the hydrolysis of the basic cations and form an alkaline solution which may have a pH of 11 that is able to increase the pH of the soil.

The extent of soil's pH raise depends on the buffer capacity of the soil that have been decreased by the heating suffered during the fire.

The effects on soil pH and Nitrogen compounds, as produced by the leaching of ashes, may continue up to 2 years after fire; then the falling rains begin to wash out the products of the ash-leachate and the derived effects are lowered or lost (GIOVANNINI and LUCHESI 1983, GIOVANNINI et al. 1987, GIOVANNINI and LUCHESI 1997, GIOVANNINI 1997, GIOVANNINI 1997, GIOVANNINI et al 1998).

5.12 THE SOIL WATER REPELLENCE

The surface layers of forest soil show, very often, a slowing-down of the water adsorption process, and in some cases they appear as really water repellent.

On burned areas, on the contrary, 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 the hydrophobic layers both surficial and sub-surficial.

The process depends on a class of organic chemicals that are hydrophobic.

These chemicals derive from the decomposition of the fresh organic litter; in soil they may intermix with the soil mineral particles, clog the interstitial pore spaces and form a surficial layer that is relatively impermeable to water.

When a fire occurs, the litter and the upper soil layer are exposed to very intense heating.

In some cases the temperature at the soil surface may reach 500°C and may remain relatively high for a long time.

This temperature totally destroys the non-wettable property of the surface soil, the hydrophobic constituents of the soil organic matter in part evaporate and are consumed by the fire in the flame, and in part become even more fluid and are forced to move downward along the temperature gradient developed along the soil profile until they meet cooler soil particles where they can condense and form a new layer of accumulation of hydrophobic substances.

Therefore, after the fire, a water repellent layer may be present below and parallel to the soil surface on the burned area.

This situation appears strongly hazardous for soil erosion, particularly in soil on steep slopes: indeed the layered arrangement allows rainfall to infiltrate only at 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 found in burned soil.

Basic information may be found on references: DE BANO and KRAMMES 1966, DE BANO et al. 1970, SCHOLL 1971, SCHOLL 1975, DE BANO et al. 1976, DE BANO et al. 1979, DE BANO 1981, GIOVANNINI and LUCHESI 1983, GIOVANNINI and LUCHESI 1984, WALLIS and HORNE 1992.

5.13 THE SOIL EROSION: USLE – MUSLE – RUSLE

Many people believe that the vegetative cover is the single most important factor in soil erosion control and that the post-fire soil erosion is caused by the reduced protection offered by the post-fire residual vegetation.

This, of course, is true, but not completely because it does not account for the moderate erosion suffered by non-burned soils protected only by reduced vegetation.

A serious prediction of the post-fire soil erosion risk, therefore, must keep in account both the reduction of the protective vegetal cover and the degradation induced in soils by the passage of fire.

The problem of prediction of soil erosion is a complex procedure that needs continuous refinements.

In 1965 WISCHMEIER and SMITH (WISCHMEIER and SMITH 1965) proposed a Universal Soil Loss Equation (USLE) to predict the erosion losses from cropland.

In 1971 the equation was adapted to predict the erosion of construction sites and in 1974 (WISCHMEIER et al. 1971, WISCHMEIER 1974) it was further extended to undisturbed areas.

In 1975 WILLIAMS (WILLIAMS 1975) developed a Modified Universal Soil Loss Equation (MUSLE) which uses different procedures to obtain an average value of the various factors for small agricultural watersheds.

Recently the algorithms to calculate the individual factors have been changed significantly by RENARD et al. (RENARD et al. 1991) in the Revised Universal Soil Loss Equation (RUSLE).

Despite its simplifications and modifications the USLE continues to be the workhorse of soil erosion prediction (WISCHMEIER and SMITH 1978).

The USLE in the original structure may be represented as a multiplicative sequence of factors:

$$\mathbf{A = R \times K \times C \times L \times S \times P}$$

where

- *A* is the average annual soil loss;
- *R* is the rainfall and run-off erosivity index and expresses the ability of the erosive agents, like rain, to cause soil detachment and its transport;
- *K* is the soil erodibility factor, it is a soil characteristic and measures the soil's susceptibility to detachment and transport by the agent of the erosion;
- *C* is the vegetative cover factor and accounts for the protection offered by the vegetal cover;
- *L* is the slope length factor,
- *S* is the slope steepness factor, that is topographic parameters, and
- *P* is the influence of the eventual erosion control practices.

It is evident that the passage of fire can affect only the soil erodibility *K* and the vegetative cover *C* factors, therefore the attention, for burned areas, is to be focused on these two.

The USLE seems an appropriate tool to predict erosion even in burned soils, at least for homogeneous areas describable with the same S and L factors.

5.14 THE EUROPEAN SOIL EROSION MODEL : EUROSEM

The European soil erosion model (EUROSEM) is the result of a collaborative research programme involving scientists from ten European countries and USA (MORGAN et al. 1992, MORGAN et al. 1994, MORGAN et al. 1997, QUINTON 1997).

EUROSEM is an event-based model designed to operate for successive short time steps within a storm.
The model simulates erosion on a single slope plane or segment and can therefore be used to predict soil loss from individual fields.
These segments can then be linked together in a cascade to simulate hill slopes.
By joining these with channel elements small catchments can be represented.
Each of the segments are assumed to have uniform properties.

The model computes soil loss as a sediment discharge, defined as the product of the volume of runoff and the sediment concentration in the flow, to give a volume of sediment passing a given point in a give time.

EUROSEM deals in turn with the interception of rainfall by the plant cover; the volume and kinetic energy of the rainfall reaching the ground surface as direct through fall and leaf drainage; the volume of stem flow; the volume of surface depression storage; the detachment of soil particles by raindrop impact; detachment and deposition of soil particles by runoff; and the transport capacity of the runoff.

EUROSEM differ from most other process-based soil erosion models in its treatment of effects of soil, concentrated overland flow and vegetation.
Soil erodibility is represented through the use of soil cohesion and an index expressing its detachability by raindrop impact.
Rill and inter-rill processes are modelled explicitly with water and sediment routed from inter-rill area.
Vegetation or crop is modelled through its effects on the volume and energy of the rain reaching the ground surface, infiltration, roughness imparted to the flow, and reinforcement of soil cohesion by the root system.
Soil conservation measures can therefore be simulated by describing the soil, micro topography and vegetation conditions associated with each practice.

This approach requires a considerable number of parameters – over 30 per model element in the current version of EUROSEM – many of which are difficult to determine over field areas, for example saturated hydraulic conductivity.
Attempts to reduce predictive uncertainty in model simulation are in progress.

5.15 THE SOIL EROSION: THE WATER EROSION PREDICTION PROJECT - WEPP

The Water Erosion Prediction Project (WEPP) hill-slope model is a physically based hydrological and erosion model developed by USDA for the quantitative prediction of erosion from hill-slopes and small to medium-sized basins (SAVABI et al. 1995, TISCARENO et al. 1995, GHIDEY and ALBERTS 1996, SOTO and DÍAZ-FERROS 1998).

Though the WEPP model is starting to replace the empirical models previously used, like USLE model, it is still in the testing and evaluation phase, it should thus be used with caution until the types of environment for which it gives reliable results have been clearly identified.

The Wepp model consists of various components, namely (a) a stochastic weather generator that permits simulation of daily climatic data on the basis of existing records, (b) an infiltration/runoff component based on a modified version of the Green-Ampt infiltration equation, (c) a soil–water balance component based on the corresponding component of the SWRRB model, (d) a vegetation growth component based on the EPIC model, (e) a component describing the decomposition of plant residues, and (f) an irrigation component.

Data inputs is divided into four categories: management, slope, soil, and climate.
The model has two run modes, depending on whether the area under study is defined as cropland or rangeland.
There are also different run modes for continuous and single-storm simulation.

The WEPP model has been tested to predict the erosion of areas burned both with experimental fires and with wildfires in North-western Spain.

In general, WEPP predictions of total runoff volume over test period were acceptable; erosion losses were likewise predicted with reasonable accuracy, though the model showed a consistent tendency to under-estimate the soil loss after wildfires.

5.16 THE ESTIMATION OF SOIL ERODIBILITY FACTOR K

The soil erodibility factor *K*, in the USLE, measures the influence of physical and organic properties on a soil's susceptibility to erosion (Wischmeier and Smith 1965, Wischmeier et al. 1971, Wischmeier 1974, Giovannini et al. 1998, Giovannini et al. 2001).

An estimate for an unknown *K* can be calculated from the regression equation proposed by Wischmeir and Smith (1978):

$$K=2.1\cdot10^{-6}\cdot M^{1.14} x\ (12 - a) +3.25\cdot10^{-2}\ (b - 2) + 2.5\cdot10^{-2}\ (c - 3)$$

Where: M = (% Silt + % fine Sand) (100 - % Clay)
a =% Organic Matter
b =Soil-structure code: 1 very fine granular; 2 fine granular; 3 medium granular; 4 massive
c =Soil permeability class: 1 rapid; 3 moderate; 5 slow

The passage of fire alters profoundly the soil physico-chemical parameters and the alteration is always related to the temperatures developed at the soil surface, which are an indirect index of the fire intensity.

As the heating temperature increases, the percentage of clay and silt decreases whereas the percentage of sand, both coarse and fine, increases.

The increase in heat promotes the aggregation of finer particles of clay and silt into greater sized particles of sand even more resistant to the disrupting action of water.

In the same time the larger particles are organised in a different way in the space and the soil structure passes from fine granular to coarse granular up to blocky or massive.

During the passage of fire the flames burn the litter and the woody fuel and the soil surface is exposed to the very high temperatures that promote the distillation of the soil organic matter.

The volatile compounds are burned in the flame, whereas the non-volatile compounds move downward along the soil temperature profile and are newly condensed when they meet cooler soil portions.

If the combustion conditions and the soil moisture content are appropriate, a sub-surficial hydrophobic layer may occur.

The global result of this mechanism is a very clear post-fire decrease of the soil organic matter content in the superficial soil.

This is associated with an increase of the water permeability of the superficial layer.

In the eventual presence of a sub-surficial hydrophobic layer, on the contrary, the whole soil permeability dramatically decreases.

All these considerations bring us to consider the application of the above reported regression equation as the most appropriate to estimate the erodibility of burned soils.

5.17 THE VEGETAL PROTECTION OF SOIL

First and foremost, vegetation protects the soil from erosion by intercepting raindrops and absorbing their kinetic energies.

Some water may be evaporated from the leaves, but most reaches the ground surface either by steam-flow or by reforming into droplets which, if the vegetative cover is close-growing, have little chance of picking up speed and gaining further kinetic energy.

In addition to the passive protecting action there are many interactive processes between a plant and its soil that affect erosion.

Some of these processes include the following: the physical binding of soil by plant stems and roots; electrochemical and nutrient bonding between roots and soil; reduction of runoff by stalks and organic litter; improved infiltration along root channels; greater incorporation of organic matter into the soil resulting in better structural and water-holding qualities; increased fauna and biological activity leading to better soil structure.

Of course fire burns, totally or partially, the vegetal cover and, after fire, the soil is less protected and more exposed to the erosive action of rain.

However, it would be a mistake to regard vegetative cover as the panacea for soil erosion control.

It is true that a mature, natural forest cover is a perfect shield against erosion processes and that erosion under undisturbed forest is usually always negligible.

The temptation, therefore, is to promote afforestation as a universal conservation measure; however, in some cases, the planted trees have accelerated the erosion rate.

What are the causes of the ambivalent effects of vegetation? Three can be identified.

First, the height of the vegetative cover above the ground surface is important.

Droplets, often larger in mass than in the original rainfall, may reform on leaves.

In falling to the ground, they may accelerate sufficiently to have a sizeable kinetic energy.

Second, tall-growing vegetation may reduce ground cover completely.

Unimpeded surface runoff, coupled with the bombshell effect of large droplets falling from leaves, can cause significant erosion.

Third, ambivalent effects of vegetation have also been demonstrated in laboratory rainfall simulator tests.

It has been found that a cover of grass reduced erosion on slopes under five degrees, but above eight degrees, the rate of erosion exceeded the rate on bare soil because the higher slopes generate turbulent eddies downstream of grass blades that erode more soil (Sreenivas et al. 1947, Barnet and Rogers 1966, De Ploey et al. 1976, Giovannini et al. 1998).

5.18 THE ESTIMATION OF THE VEGETAL COVERING FACTOR C

The relevance of the vegetative cover in the soil erosion process has been formalised by WISCHMERIER and SMITH (1978) including the factor *C* in the Universal Soil Loss Equation:

A = R x K x C x L x S x P

Derivation of *C* values is a complex and intricate procedure for areas prone to soil erosion. However, a

- theoretical value *C*=1 may be attributed to areas bare of vegetation and however very prone to erosion, whereas
- a value *C* = 0.001 may be attributed to the well-covered areas, as in permanent pasture.

The contribution of trees and bushes, which also intercept and dissipate the erosive energy of rainfall, will have intermediate values.

On the basis of data reported by WISHMERIER and associates (WISCHMEIER and SMITH 1965, WISCHMEIER et al. 1971, WISCHMEIER 1974, WISCHMEIER and SMITH 1978) we (GIOVANNINI et al. 1998) have estimated, for a certain percentage of ground cover and for each of the protecting layers, the theoretical values reported in following table

Estimating the contribution of the vegetal cover in the various layers (height: 0-5 cm; 5-50 cm; 50-100 cm; 100-200 cm; greater than 200 cm) to the protection of the soil surface, or in other words the capacity of the various layers to intercept the rain, and utilising the theoretical values reported in the table it is possible to estimate the value of the *C* factor of an area.

5.19 THE PREDICTION OF POST-FIRE SOIL EROSION : PROMETHUS

The prediction of the post-fire soil erosion risk has been tackled in the frame of PROMETHEUS Project (GIOVANNINI 1996, GIOVANNINI 1999).

The logical scheme of the original structure of the USLE has been followed, not with the aim of predicting the exact amount of soil loss in Kg/ha per year, but only to predict the magnitude of the post-fire erosion.

This is in order to have information on whether the expected events will be catastrophic, disastrous or merely acceptable.

The adopted logical scheme may be followed in the chapter reporting the Post-fire Soil Erosion Schematic Diagram.

Considering the great relevance of the soil characteristics in response to the fire and to the erosion processes, we start with the evaluation of the site's pre-fire soil erodibility *K*, calculated by means of the regression equation previously reported.

Both Fire-line Intensity that is the energy released per unit of time and per unit of length of the fire edge and Burn-out Time that is the time taken for all fractions of the fuel to burn out may define the characteristics of the fire.

Combining the Soil Erodility with Fire-line Intensity and Burn-out Time we can define the Post-fire Soil Erodibility that, considering the response of all the parameters determining the soil erodibility to the passage of fire, will increase in respect to the pre-fire value.

The Vegetal Covering Capacity represents the percentage of protection of the soil surface against the aggressive rain impact; combining the Vegetal Covering Capacity with the parameters characterising the fire: Fire Line Intensity and Burn-out Time, we can evaluate the Residual Vegetal Covering Capacity.

It is evident that after the passage of fire the vegetal protection will be lower than before fire.

Another parameter relevant to the erosive processes is the Pre-fire Surface Hydrophobicity, very common on forest soil surface As a consequence of the passage of the fire; the fate of the hydrophobic layer may be variable.

The Post Fire Soil Hydrophobicity may be: totally absent because destroyed during the fire, present in the surface layer because not affected by light fire, or present in a sub-surface layer when the fire intensity and soil thermal properties are adequate.

The combination of the Post-fire Soil Erodibility, Residual Vegetal Covering Capacity and Post Fire Soil Hydrophobicity determines the Site's Potential Erodibility, that is the intrinsic possibility of a burned site to be eroded in the case of a rain event.

At this point of the evolution of the logical scheme, the two factors related to the topographic conditions must be kept in account.

The slope factor S, which represents the influence of the angle of slope on soil erosion and the slope length factor L, representing the length of slope in which deposition of eroded material does not occur.

Now all the characteristics of the burned or burnable area are determined and the soil erosion will be dependent only on the characteristics of the rain.

The Rainfall Erosivity represents the ability of the erosion agent to cause soil detachment and transport.

It is due to the direct raindrop impact and to the runoff that rainfall generates.

The final combination of the Site's Potential Erodibility, Landscape Characteristics and Rainfall Erosivity allow us to predict the Expected Erosion that will have 3 possible levels of magnitude:

- Low when the post-fire soil erosion is up to 5 times greater than the pre-fire value,
- Medium when it reaches up to 25 times and
- High when it is greater than 50 times.

All these parameters, with their definitions and classes, have been organised, in the frame of PROMETHEUS Project, in a computer programme so that introducing the appropriate values of the various independent parameters, it is possible to predict the magnitude of the expected post-fire soil erosion and if it is the case to plan an appropriate remedy.

% of Ground cover

	25	50	75	100
large trees (crown>2m high)	0.970	0.930	0.800	0.730
medium trees (crown 2m high)	0.870	0.750	0.630	0.500
high bushes (crown 1m high	0.830	0.650	0.470	0.300
low bushes (crown 0.5m high)	0.790	0.580	0.370	0.160
grass/litter (0.05m high)	0.200	0.100	0.012	0.003

5.20 THE PREVENTION OF THE POST-FIRE SOIL EROSION

If the prediction of post-fire soil erosion is for a low acceptable value, no particular preventive measures are to be planned as in the case of a response for a high, catastrophic event.

Considering that it is utopian to consider the total exclusion of fire from the forest environment, the problem becomes that of converting the fire effects from disastrous to acceptable.

In the case of post-fire soil erosion risk this may be achieved playing with the relevant parameters.

From the close examination of the Universal Soil Loss Equation it is evident that neither the site's topography nor the meteorological events can be changed, whereas some modifications are possible for the parameters Soil Erodibility and Vegetal Covering Capacity.

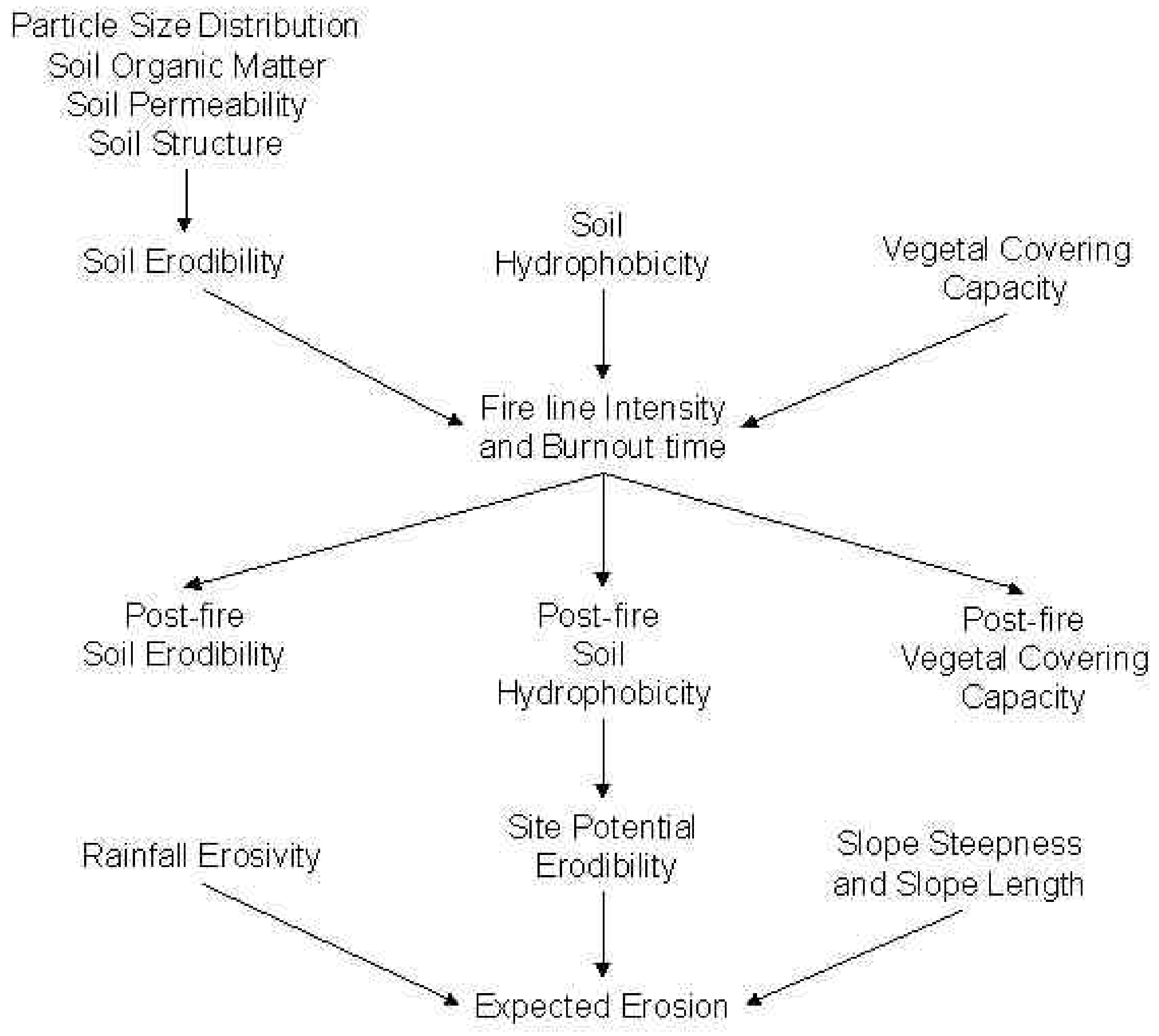
Experimental data show the greater relevance and the long lasting persistence of the increased Soil Erodibility with respect to the reduced Vegetal Covering Capacity on the soil erosion process, so that the major attempts to reduce soil erosion risk must be focused on the former parameter.

Examining the soil erosion scheme we see that the post-fire Soil Erodibility is dependent on the pre-fire Soil Erodibility, that is a characteristic of the soil and therefore hardly modifiable, and on the fire intensity.

A very mild fire, indeed, induces a very low increase in soil erodibility, contrary to a severe fire, which induces conditions for extreme soil susceptibility to detachment and transport by erosional agents.

The reduction of the post-fire soil erosion risk, therefore, corresponds with the reduction of the fire intensity and this may be achieved with an appropriate manipulation and management of both the dead and living fuel.(Giovannini 1996, Giovannini 1999).

5.21 THE PREDICTION OF POST-FIRE SOIL EROSION: SCHEMATIC DIAGRAM



6 MEDITERRANEAN ECOSYSTEMS: AQUATIC ECOSYSTEMS

This chapter has been compiled by D Angeler UCLM

6.1 INTRODUCTION

An exhaustive search for publications related to the impact of either wildland fires or fires for management purposes in aquatic ecosystems yielded so far a total of 176 citations from peer reviewed international publication sources.

In regard of the Euromeditarranean region, the results from this search indicate that the impact of catchment perturbations on streams, lakes and wetlands in form of fires has so far been neglected by European scientists.

Only a single study (BELLIAS and RODA, 1993) was found to be related geographically to Mediterranean Europe.

6.2 RESULTS AND DISCUSSION

BELLIAS and RODÁ reported on the effects of heathland management, using prescribed burns, on the water quality of selected streams in Catalonia (NE-Spain).

The methodology employed is summarised in Table 1.

The main finding of BELLIAS and RODA (1993) was that prescribed burns resulted in statistically insignificant increases of nitrate concentrations in stream water, concluding that stream water quality is negligibly affected by catchment perturbations in form of fire.

These results agree with others obtained outside the European Community (e.g., WRIGHT 1976, DAVIS 1989), but disagrees with others (HAUER and SPENCER 1998).

The contradictions in the literature are currently not easy interpretable, because of the lack of consideration of catchment characteristics (fire severity and extension, vegetation characteristics, soil characteristics etc) and biotic integrity of fire-impacted aquatic ecosystems.

Given the projected increase of fire intensity and severity with global change, an urgent scientific consideration of fire impacts in aquatic ecosystems, especially in fire-prone areas, such as Mediterranean Europe, seems necessary.

Region	Ecosystem type	Parameters	Methods	Reference
NE Spain	Streams	Na, Mg, NO_3, SO_4, pH	-) Flame emission spectrometry -) Atomic absorption spectrometry -) Ion chromatography	Belillas & Roda (1993)
MN USA	Streams/Lakes	Na, Mg, Ca, P, K, electrical conductivity, alkalinity	-) Atomic absorption spectrometry -) Colorimetric methods	Wright (1976)
AZ USA	Streams	NO_3, SO_4, Cl, Na, Mg, Ca, K, electrical conductivity, pH	-) Atomic absorption spectrometry -) Automatic analyzers	Davis (1989)
MT USA	Streams	N, P	-) Colorimetric methods	Hauer & Spencer (1998)

Table 6-1. Comparison of methodological approaches used to determine effects of fire on selected aquatic ecosystems. Note that for reasons of comparisons studies form outside Europe are included.

7 MEDITERRANEAN ECOSYSTEMS: LANDSCAPES

This chapter has been carried out by I GITAS, A KLAASE and G MITRI (MAIC); P MARTIN and R ROMERO-CALCERRADA (CSIC) and O VIEDMA UCLM

7.1 INTRODUCTION

Fire impact is described as the immediate evident effect of fire on the ecosystem in terms of biophysical alteration (e.g. crown scorch, soil exposure, depth of burn, fuel consumption) (KAFKA et al., 2001).

Fire impact is determined by fire intensity, the duration of fire and by the response of the ecosystem.

Fire impact describes the immediate effect of fire on the ecosystem.

An immediate effect is the removal of vegetation by fire.

Fire effects are any change(s) on an area attributable to a fire, whether immediate or long-term, and on-site (within the burned area) or off-site (e.g. atmospheric pollution).

Fire impact does not take into consideration the role played by fire and its impact on the biota in terms of short-term effects (producing successional change and structural and floristic changes in the vegetation) and long-term effects (changes in the vegetation of a region producing what has been called the fire mosaic) (FOX and FOX, 1987).

Fire impact only measures direct (first order) effects, which are considered to be those changes brought about by the fire itself, arising from mainly the burning of organic matter and the heating of the surface layers of the soil.

Examples are fuel consumption, tree mortality and smoke production.

Fire impact does not consider indirect effects such as tree regeneration, hydrological response and vegetative succession.

One can also distinguish detrimental and beneficial effects, such as erosion after fire and temperature condition alterations.

Besides fire impact, a number of terms exist to describe the effects of fire.

Examples are fire severity, fire damage and more specifically, vegetation mortality.

Since fire effects are very complex, usually only a part of the fire effects is considered.

Fire severity is a general term that most commonly describes the combined effects of both flaming combustion and smouldering combustion on either a wildfire or prescribed fire site as manifested in various fire behaviour characteristics.

Fire severity can be used to assess fire effects.

Fire severity is a very general concept; however, it is often defined by less general terms such as fire impact, which in fact only take into account the biophysical alterations of the ecosystem caused by fire.

For example, plant mortality is a fire impact and can be used as a measure of fire severity, although in this way, other effects such as atmospheric pollution due to smoke are left out of scope.

Usually, fire severity is described by considering only part of the fire effects.

The result is that fire severity is a concept that is interpreted and defined in many different ways.

Concerning the impact on the ecosystem (vegetation) the definition of WHITE et al. (1996) is most commonly used: fire severity is a descriptive term that integrates the physical, chemical and biological changes on a site as a result of fire.

AGEE (1993) wrote that fire severity is a qualitative measure of the immediate effects of fire on the ecosystem.

It relates to the extent of mortality and survival of plant and animal life both aboveground and belowground and to loss of organic matter.

RYAN and NOSTE (1985) characterised fire severity by classifying the combined effects of fire intensity (indicating aboveground fire effects) and the depth of ground char.

Fire severity is related to both the heat released aboveground and belowground (determined by both fire intensity and duration) and the response of the ecosystem.

Sometimes fire severity is defined as the combined effect of fire intensity and duration, so, without considering the response of the ecosystem.

To avoid further confusion, we introduce the term heat exposure.

Heat exposure describes the combined effect of fire intensity and duration, so the total heat released.

Fire severity depends on both the characteristics of the fire and on the response of the ecosystem.

One can say that heat exposure determines the *potential* effect of fire on the ecosystem and describes fire behaviour, while fire impact and severity describe the *actual* effects (changes) resulting from fire.

7.2 DRIVING FACTORS OF FIRE IMPACT AND THEIR SPATIAL AND COMPOSITIONAL PATTERNS AT LANDSCAPE LEVEL

Interactions between fire and landscape are complex and uncertain.

Fire impact is related to the characteristics of fire itself (described by fire regime) and the characteristics of the environment (the landscape).

The landscape consists out of abiotic (the physiography and weather) and biotic elements (the vegetation).

Together, the fire regime and the landscape are the driving factors of fire impact, determining its spatial and temporal pattern and highly influencing each other.

On the whole, one can say that fire impact is determined by on one hand the total heat exposed by a fire and on the other hand the response of the ecosystem.

In the following paragraphs the driving factors of fire impact at landscape level will be discussed.

The first paragraph describes how fire regime determines total heat exposure and also the relationship between fire regime and species survival.

After that, the spatial heterogeneity of the abiotic environment is related to fire impact.

Concerning the biotic factors, only plants are taken up as animals will be dealt with in other literature reviews within the EUFIRELAB project.

Vegetation is considered both in terms of fuel provider and of being affected by the fire.

7.3 FIRE REGIME AS A DRIVING FACTOR OF FIRE IMPACT

Fire regime influences mainly total **heat exposure** and in some degree also vegetation response.

Total heat exposed by a fire is a function of **fire intensity** and **residence time**, which are greatly determined by the fuel availability, both quantity and arrangement as well as quality (moisture content) together with topography, wind speed, and the structure of the plant community itself, resulting in significant variations in both time and space (BROWN et al., 2000).

Fuel conditions are also influenced the season of the year, and prevailing weather conditions (FOX and FOX, 1987).

Furthermore fuel is influenced by the time since the last fire (the fire frequency).

In general, the higher the **fire frequency**, the lower is the intensity of the fires due to a shorter period for fuel accumulation.

Over long-term scales, the occurrence of fires tends to be self-limiting due to the time needed to accumulate sufficient fuel to carry a new fire.

However, many other factors not related to vegetation age, such as weather, fire policies, fire management or plant invasions, may interact with this self-regulating characteristic of fire.

In this respect, it is important to realise that fire incidence is a spatial phenomenon, that not only depends on structural factors (e.g. type of vegetation, landscape features, climate conditions) but also on spatial factors such as the characteristics of adjacent areas (VÁZQUEZ and MORENO, 2001).

The effect of invasive species on fire frequency and intensity is described by VILÀ et al. (2001) who investigated the effect of Amelodesmos mauritanica, a specie that produces high amount of fuel resulting in a high frequency fire regime with high intensity fires.

Usually the species composition is altered under these conditions.

Furthermore, the total heat transferred to the vegetation also depends on the type of fire.

Fire type is determined by (among others) environmental conditions such as slope and wind and by the fuel type and condition of the fuel.

The occurrence of crown/canopy fires is more likely during high intensity fires.

In general, crown fires seem to have a stronger impact as surface/ground fires, as they usually kill all aboveground vegetation and have a higher rate of spread.

However, both surface and crown fires can cause stand replacement (BROWN and SMITH, 2000) and canopy fires are not always more severe than surface fires.

Canopy fires may destroy more above-ground vegetation, but do not necessarily destroy all the organisms in the duff and underlying mineral soil that are important for recovery (RYAN and NOSTE, 1985).

So, although fire impact of canopy fires is in general higher than for surface fires, this is not a standard.

Vegetation response depends on the type of vegetation, the condition of the vegetation and environmental conditions directly after fire.

For the response of the ecosystem also factors such as ecological conditions and management practices should be taken into account (NAVEH, 1974).

Concerning the fire regime, the impact of fire depends greatly on the **season** in which it occurs.

The degree of damage varies with season, being greatest during the growing season (TRABAUD, 1987).

For example, HARRINGTON (1993) found that the mortality of trees scorched in spring and summer was about 2.5 times greater than that in the autumn for similar crown damage of Pinus ponderosa in Southwest Colorado.

7.4 LANDSCAPE HETEROGENEITY AS A DRIVING FACTOR OF FIRE IMPACT

Landscape includes an abiotic and a biotic component.

The abiotic component considers the physiography, which determines fuel, and weather conditions, which are partly determined by physiography.

The biotic component includes both the fuel (the ability to burn) and the resilience of vegetation (ability to survive).

Abiotic factors considered are the physiography of a site, such as soil type, aspect, slope and position on the hillslope, and the weather (PIÑOL et al., 1998) conditions before, during and after a fire, especially wind, temperature and rainfall.

Physiography has a strong influence on solar radiation and moisture availability, and hence on fuel accumulation.

Aspect is the most important feature of physiography concerning fire impact.

Slopes that receive more sunlight tend to be drier and warmer.

Northward facing slopes and canyons tend to hold more moisture and receive less sunlight; therefore they stay greener longer, which reduces their susceptibility to damaging fires compared to southward facing micro climates.

However, vegetation is also denser, which may lead to high fire intensities and hence higher impact.

South faced slopes suffer higher irradiation causing increased evapo-transpiration and favoured mineralisation of organic matter.

Different fire regimes on Mediterranean soils in opposite slope orientations have an impact on chemical and physical properties of the soil (ANDREU et al., 2001).

The effects of fire on site quality arise from two principal sources: the burning of organic matter above and on the mineral soil (resulting in the release of carbon dioxide, nitrogenous gases, and ash to the atmosphere and the deposit of minerals in the form of ash), and the heating of the surface layers of the soil.

Slope is mainly important for fire spread, due to the fact that hot air rises and preheats uphill fuels, which causes fires to move more rapidly uphill than downhill.

Fuels are naturally elevated by the slope of a hill, which brings them closer to the flames.

Therefore, slope angle is as important as wind speed in determining the spread of a fire.

The spread of fire influences also the duration of fire and thus impact.

The diversity in sunlight and configuration means that each slope has a different micro climate.

Climate is considered the principal determinant of vegetation distribution throughout the world.

Solar radiation, temperature, humidity, precipitation and wind all affect the physiological ecology of plants, thereby affecting their ability to complete life cycles and sustain populations.

The pattern and severity of disturbances such as fire will change with vegetation, and thus with climate.

Weather and fire influence each other.

Several aspects of weather, such as relative humidity, wind, drought, length of fire season, lightning, dry cold fronts and blocking high pressure, influence fire and fire impact.

Fire influences on climate by its contribution to the amount of gasses ($CO2$, CO, $CH4$, NO_X, $NH4$), water vapour, particulates, trace hydrocarbons and trace gasses available in the atmosphere.

A two-way relationship between land cover change and fire exists: the impact of fire largely depends on the ecosystem type and the land cover and fires may induce long-term changes in vegetation cover (BUCINI and LAMBIN, 2002).

Vegetation serves as a fuel, but is also affected by fire and responding to it.

The spatial distribution of quality and quantity of fuel biomass has a strong influence on fire propagation, and conversely, the impact of disturbance on the distribution of species within a landscape is highly dependent upon the spatial and temporal pattern of the disturbance regime.

Characteristics of fire that determine fire impact are significantly related to local variation in species composition.

The local species composition at any one place is dependent on burn history, grazing history, soils, aspect, and topographic position (BROWN et al., 2000).

Complex fuel distribution favours to non-uniform fire behaviour. In general, sites with better growing conditions for vegetation will burn with higher intensity due to the higher fuel accumulation, but also regenerate faster.

The effects of fire are complex because the response of plants varies greatly with:

- the characteristics of fire (fire intensity, rate of spread, energy release rate, residence time, type and timing),
- the intrinsic regeneration capacities of different species (both their inherent resistance to injury and ability to recover) as well as
- the pre-fire status of the vegetation.

Degree of damage varies also with season, being greatest during the growing season.

Furthermore, characteristics of the plant, such as bark thickness, age, composition, chemical content (TRABAUD, 1987) and rooting habit play an important part in survival

Often, the abundance of legumes, species of chaparral and other nitrogen-fixed plants is increased by fire, which may shortly increase the available nitrogen and improve the overall site quality.

Furthermore, the pH is increased due to the burning of organic matter, which increases microbial activity that often gives rise to increased mineralisation and nitrogen availability (BARNES et al. 1998).

However, in many parts of the world, recurrent fires lead to the development of a landscape in which shrubby vegetation dominates, which is low in nutritive value and slow to decompose.

Land use and land cover change affects on fuel accumulation, soil properties (through the direct effect of soil management practices and through the effect of the diverse plant cover involved) but also, in landscape configuration and patch structure and these differences may lead to various responses in case of fire.

Existing land cover change / land use abandonment and forest fire implications studies have mainly been carried out in the Iberian Peninsula (BADIA et al., 2002; CAMMERAAT and IMESON, 1999; GIOVANNINI et al., 2001; LLORET et al., 2002; MOREIRA et al., 2001a; MOREIRA et al., 2001b; PEREZ et al., 2003; ROMERO-CALCERRADA and PERRY, 2002; ROMERO-CALCERRADA and PERRY, 2004).

7.5 EXISTING METHODS TO CHARACTERISE FIRE IMPACTS AT LANDSCAPE LEVEL

The important questions about fire occurrence are: when, where, and of what severity.

Remote Sensing has proved to be efficient, accurate, objective and repeatable for fire scar mapping.

While fire scar mapping is suitable for mapping fire extent, frequency and season, producing accurate maps of other characteristics of fire regimes, particularly fire severity/impact, remains more of a challenge.

Fire severity/impact maps are more dependent on robust field validation (WHITE et al., 1996) than maps of fire presence/absence (HUDAK and BROCKET, 2002).

Computer-aided analysis of remote sensing data may discriminate distinct spectral classess within a burnt area, which may indicate fire severity/impact, the stage of re-vegetation, original cover type, or a combination of these three (HITCHCOCK and Hoffer, 1974).

The difficulty is to correlate or attribute these spectral classes on the image to surface cover types with varying degrees of fire severity/impact (MILNE, 1986).

In some cases, it may be impossible to find a relationship between field observations and remotely sensed imagery.

Existing fire severity/impact studies have mainly been carried out in the USA (KEY and BENSON, 1999a, 1999b, 2000; REDMOND et al., 2001; RIGNOT et al., 1999; RYAN and NOSTE, 1985; TuRner et al., 1999; White et al., 1996; JAKUBAUSKAS, 1989; Jakubauskas et al., 1990; LACHOWSKI and ANDERSON, 1979; BERTOLETTE and SPOTSKEY, 2001; BRUMBY et al., 2001; MEDLER and YOOL, 1997; PATTERSON and YOOL, 1998; ROGAN and YOOL, 2001; HITCHCOCK and HOFFER, 1974), and only a few in the Mediterranean ecosystem (ROGAN and FRANKLIN, 2001; ROGAN et al., 2002; RODRIQUEZ y SILVA et al., 1997; CHUVIECO and CONGALTON, 1988; ESCUIN et al., 2002; RISHMAWI and GITAS, 2001).

Other fire severity/impact studies have been conducted in Australia (MILNE, 1986; BENSON and BRIGGS, 1978; SMITH and WOODGATE, 1985), Indonesia (SIEGERT and NAKAYAMA, 2000; RUECKER and SIEGERT, 2000) and Alaska (KASISCHKE et al., 1994; HALL et al., 1980; MICHALEK et al., 2000).

LANDSAT TM and MSS seem to be the most generally used sensors in fire severity/impact research, thanks to the high spatial, spectral and temporal resolution. Some studies (SIEGERT and NAKAYAMA, 2000; RUECKER and SIEGERT, 2000) use radar imagery for fire impact assessment because of the sensor’s advantages under cloudy conditions.

Just a few studies use the relatively new very high spatial resolution sensors IKONOS and QUICKBIRD for fire severity/impact mapping.

Trends in reflectance are associated with cover type, fire severity/impact (degree of carbonisation), vegetation regeneration (abundance, composition and condition), illumination, change due to slope and the incidence of rainfall, among others.

Wavelengths in the visible spectrum are sensitive for changes in the plant physiology, and will appear brighter in fire-altered areas due to the reduced absorption by leaf chlorophylls.

Near-infrared (NIR) wavelengths are diagnostic of leaf water stress, and will decrease in fire-damaged areas.

Mid-infrared (MIR) wavelengths are sensitive to alterations in ground exposure and soil colour and increase with fire severity.

The following section gives an overview of image analysis techniques that have been used in fire severity/impact research.

The main problem with most studies is that they apply many different techniques to derive features used in classification schemes that may or may not improve results (e.g. band ratios, linear transforms, vegetation indices) without presenting any error analysis, thus making the choice of the best technique difficult (MILLER and YOOL, 2002).

Fortunately, the importance of a proper error analysis has been realised and is applied in most recent studies.

However, accuracy results should be interpreted carefully.

Results of a specific method may change considerably when applied to other fires or other ecosystems.

Moreover, classification accuracies may be affected by several factors such as:

- (a) the constraint nature of ground surveys (e.g. the limited line of sight),
- (b) the undersampling of fire severity classes,
- (c) image registration errors, and
- (d) the human agency.

Vegetation indices have been widely applied in fire severity/impact research.

The simplest type of vegetation index is obtained by dividing the reflectance from the NIR band by the reflectance of the red visible band (MILNE, 1986; JAKUBAUSKAS et al., 1990).

The Normalised Difference Vegetation Index (NDVI) is the most commonly used vegetation index.

It is an indicator of vegetation abundance and may therefore be used in fire severity/impact mapping (CAETANO et al., 1995); RODRIQUEZ Y SILVA et al., 1997; FOX III and STUART, 1994; ESCUIN et al., 2002; ROGAN and YOOL, 2001).

Other vegetation indices applied are the Soil Adjusted Vegetation Index (SAVI) (RODRIQUEZ Y SILVA et al., 1997; ROGAN and YOOL, 2001), the Atmospherically Resistant Vegetation Index (ARVI) (RODRIQUEZ Y SILVA et al., 1997), Modified SAVI and TM7/4 ratio (ROGAN and YOOL, 2001).

In general, indices and ratios based on the wavelengths in the MIR (band 7 of LANDSAT TM) and NIR (TM band 4) spectrum show best results in fire severity/impact mapping.

Research by Clark (2000) has shown that pre- and post-fire differences of mid-infrared to near-infrared ratio (TM band 7/TM band 4) provided the highest contrast fire scar in comparison to TM band 4, PCA, Kauth–Thomas, NDVI, and MSAVI and, therefore provides the best enhancement for classifying changes due to fire among these methods.

A relatively new index is the Normalised Burn Ratio (NBR), a ratio similar to NDVI, but using the reflectance of TM band 7 instead of TM band 4 and sensitive for changes in soil and vegetation moisture.

Results of the NBR are very promising for fire severity/impact research (KEY and BENSON, 1999b, KEY and BENSON, 2000; BERTOLETTE and SPOTSKEY, 2001; MILLER and YOOL, 2002).

Often, the NBR is calculated for both pre-fire an a post-fire satellite scene, then the two pre-/post-fire ratios are differenced and in turn provide a fire severity index that displays low to high burn severity of wildfires.

Low values of the Normalised Burn Index (also being referred to as the Differenced NBR) indicate low burn severity while high values represent high burn severity.

ROOT et al. (2003) investigated the relationship between spectral channels and burn severity by comparing pre and postfire AVIRIS data.

Their results fit well with results for fire severity reported using LANDSAT TM and ETM+ sensors and verified the foundational band-response relationships to burn severity as seen with TM/ETM+, and they confirmed this independently by way of a distinctly different sensor system.

AVIRIS channels not sampled within the TM/ETM+ bandwidths were not more useful for sensing fire effects, although some distinct potential remains for the 913 and 2470 IVIRIS channels to radiometrically enhance the burn signal over TM/ETM+.

The Differenced NBR derived fro AVIRIS was greater in magnitude and had more variation within the burn than the Differenced NBR from TM.

Multitemporal PCTs or multitemporal Kauth-thomas transformation may produce suitable fire severity/impact maps if both pre- and post-burn images are available.

Kauth-Thomas or Tasseled Cap (KT) Transformation is a linear transformation that establishes three new axes in the spectral data of LANDSAT TM.

The first feature, brightness, is related to soil reflectance; the second feature, greenness, is strongly related to the amount of green vegetation present in the scene; and the third feature, wetness, is related to canopy and soil moisture.

Especially KT wetness is sensitive to differences in moisture levels on the landscape and therefore suitable for fire severity/impact mapping.

Studies have been carried out by PATTERSON and YOOL (1998), ROGAN and YOOL (2001) and ROGAN et al. (2002).

A disadvantage of the KT transform is that its coefficient table differs per satellite, making comparison between satellites difficult. It is even doubtful whether the KT transform is appropriate for all satellites.

A KT transform on IKONOS imagery for example results in the features brightness, greenness and yellowness (YUEN, 1998) and misses the for fire severity/impact mapping interesting feature wetness.

Principal Component Analysis (PCA) aims to produce a new dataset, through a linear algebraic expression of the initial variables, in order to minimise the correlation and to associate the variance of the data with the new first components.

Areas associated with permanent landscape features are highlighted in the higher order components, while areas of change are emphasized in the lower ones.

PCA has proved to be useful for fire severity/impact mapping.

However, it considers all data as equally important, while the burnt area usually comprises only a part of the scene (PATTERSON and YOOL, 1998).

Classifications using Principal Component Analysis (PCA) are therefore often outperformed by classifications that are able to extract the most relevant information such as Kauth-Thomas (PATTERSON and YOOL, 1998) and Spectral Mixture Analysis (SMA) (ROGAN et al., 2002).

Furthermore, several variations on PCA, such as (Bendix)-DAS (TANAKA et al., 1983) and Canonical Discriminant Analysis (CDA) (RISHMAWI and GITAS, 2001) have been used in fire severity mapping.

Spectral Mixture Analysis (SMA) is a procedure that extracts sub-pixel information by assuming that the spectrum is a (linear) combination of the pure spectra of the materials located in the pixel area, weighted by their fractional abundance.

SMA has mainly been used in high spectral resolution sensors, but above mentioned advantages in combination with its ability to solve the topographic problem by desegregating the shade component of the spectral signal make SMA also highly effective in burnt area mapping and fire severity/impact assessment (ROGAN et al., 2002; ROGAN and FRANKLIN, 2001; CAETANO et al., 1995; RISHMAWI and GITAS, 2001).

The problem is that results are highly dependent on the input endmembers, and the identification of adequate endmembers might prove difficult in some cases.

Supervised and unsupervised classifications are the two most common classification procedures.

Unsupervised fire severity classifications were applied by MILNE (1986), HITCHCOCK and HOFFER (1974), BERTOLETTE and SPOTSKEY (2001), JAKUBAUSKAS (1989) and JAKUBAUSKAS et al. (1990).

The main problem encountered was the confusion between shaded areas and severely burnt areas.

Unsupervised classification can also be used to determine the number of classes that can be distinguished (BENSON and BRIGGS, 1978).

A disadvantage of unsupervised classification is that it considers the whole scene and does not emphasize on the burnt area only.

Supervised classification was used for fire severity mapping by MICHALEK et al. (2000), ESCUIN et al. (2002), LACHOWSKI and ANDERSON (1979) and CHUVIECO and CONGALTON (1988).

Often image transformations such as PCA and KT are performed prior to supervised classification (ROGAN and YOOL, 2001; ROGAN et al., 2002).

Results may also be improved by including a vegetation index (e.g. NDVI) as an extra band to the image to be classified.

The definition of training statistics appeared to be the main factor influencing the results.

Another classification technique is density slicing: the converting of the full range of data into a series of intervals or slices, each of which expresses a range in the data.

Often, density slicing is applied on a single band (HALL et al., 1980; KEY and BENSON, 1999b), or of a vegetation index (difference) image such as NDVI (FOX III and STUART, 1994) or of a ratio (MILNE, 1986).

Imaging radar can provide high-resolution imagery of burnt areas, independent of cloud cover, smoke and solar illumination.

In addition, radar signals are sensitive to the vegetation volume, structure and moisture content, which provide information complementary to that provided by passive sensors operating at an optical wavelength.

For example, optical sensors cannot penetrate healthy canopy cover and may therefore not detect surface burns in areas with a dense overstorey, while radar is capable of penetrating the canopy.

Radar has been used for fire impact mapping in Indonesia (RUECKER and SIEGERT, 2000; SIEGERT and NAKAYAMA, 2000), Montana USA (RIGNOT et al., 1999) and Alaska (KASISCHKE et al., 1994).

The applicability of satellite SAR imagery for burnt area mapping in Mediterranean regions has been demonstrated by GIMENO et al. (2002).

Rainfall greatly influences the backscatter signal, but it is still unclear in which way.

Some authors advice to use imagery taken under dry conditions (SIEGERT and NAKAYAMA, 2000; RUECKER and SIEGERT, 2000), while others suggest to use images taken under wet conditions (GIMENO et al., 2002).

It seems that the relationship between fire severity/impact and radar backscatter strongly depends upon the ecosystem studied.

In addition to the more widely available image analysis techniques already discussed in the previous paragraphs, a number of other techniques that have also been used for fire severity/impact mapping exist.

Some authors integrate topographic information in the classification process, thus taking into account the irregular illumination of different areas (MEDLER and YOOL, 1997; ROGAN and YOOL, 2001; PATTERSON and Yool, 1998).

Results do not always improve (ESCUIN et al., 2002) but are worth further investigation.

The Decision Tree (DT) classifier has seldom been used in change mapping studies but ROGAN et al. (2002) showed that the Decision Tree classifier outperformed a supervised maximum likelihood classiciaton, because of its ability to cope with normal distributions and intraclass variation found in a variety of spectral data sets.

BRUMBY et al. (2001) applied a machine learning technique implemented in a software package called Genie, suing a genetic algorithm to assemble image processing algorithms from a collection of low-level image processing operators.

REDMOND et al. (2001) used hierarchical machine learning technology and found that results were substantially better than results for image differencing methods and PCA.

High spatial resolution sensors such as IKONOS and QUICKBIRD open a new field of research, with new possibilities such as the detection of smaller objects but also with new problems which need more basic investigations.

The currently available very high resolution imagery has a low spectral depth compared to e.g. LANDSAT TM, especially in the MIR range of the spectrum, causing confusion between burnt areas and non-vegetated areas.

Furthermore, an increased level of detail results in an increased variable illumination partly caused by topography, but also by individual trees.

Typically, pixel-based classifications have difficulties dealing with the rich information content of very high resolution images, which show a very high level of detail and are very strong textured.

They produce a characteristic, inconsistent salt-and-pepper classification, and they are far from being capable of extracting objects of interest (Mansor et al., 2002).

It seems that with the introduction of very high resolution sensors classic classification based on pixel-based approaches became limited and that an object-oriented approach is more appropriate.

MITRI and GITAS (2002) used an object-oriented approach on LANDSAT data to map large burnt area in both Spain and Greece with very good results.

SCHWARZ et al.(2002) showed that an object-oriented classification approach has a significantly higher accuracy in classifying storm losses in Alpine forest areas using IKONOS imagery than the pixel-based method, while the classification of SPOT presented approximately the same results for both methods.

Preliminary studies by GITAS et al. (in preparation) show that object-oriented classification obtains very good results in fire type and fire severity/impact mapping using IKONOS imagery.

Hyperspectral sensors such as Hyperion provide a similar spatial resolution to that of LANDSAT TM but an extremely high spectral resolution in comparison.

Hyperspectral sensors generate a complete spectrum for every pixel of the image, so it is possible to identify materials instead of simply discriminating between materials (as with multispectral sensors).

For fire severity/impact mapping, this might mean that more classes within a burn can be discriminated more accurately.

Remote sensing is very suitable for the detection of temporal (fire frequency) and/or spatial patterns (heterogeneity) of fire impact.

VÁZQUEZ and MORENO (2001) mapped fires in the Province of Avila, Spain, over a period of 30 years and calculated fire-regime parameters, such as fire rotation period, and their relation to topographic features or other characteristics of the terrain by means of a spatially explicit analysis.

Also DIAZ-DELGADO and PONS, (2001) have applied a semiautomatic methodology for fire scars mapping from a time series of Landsat MSS images over the forest and shrubby surface of Catalonia (1975-1993).

Detected fire scars were incorporated into a Geographic Information System in order to characterise the fire regime of the study area.

Fire size distribution and the number of spot fires originated from each fire as well as the maximum distance reached from the main fire are analysed.

Results are a map series of fire history during 21 years as well as a map of the fire recurrence level.

The analysis of landscape change using remotely sensed imagery and landscape pattern metrics is useful to fire sensitivity studies and fire hazard build-up.

The studies (LLORET et al., 2002; MOREIRA et al., 2001a; MOREIRA et al., 2001b; ROMERO-CALCERRADA and PERRY, 2002; ROMERO-CALCERRADA and PERRY, 2004) suggests that systematic changes have occurred since last decades in landscape composition and structure in Mediterranean areas, and that the landscape changes are likely to result in changes in fire risk and fire regime.

NELLIS and BRIGGS (1989) tried to characterise the spatial pattern of a landscape at different scales.

They assessed the utility of textural algorithms applied to remotely sensed datasets of variable scales for measuring the degree of homogeneity and heterogeneity of Northern America tallgrass prairie watersheds experiencing different management treatments.

The dataset used existed out of 5 m resolution aerial photography, 30 LANDSAT TM data and 80 m LANDSAT MSS data.

Values reflecting the degree of textural contrast were derived by applying textural contrast algorithms on band4 to 3 and band 4 to 2 ratios.

They found that the relationship between spatial pattern and ecological processes in tallgrass prairie ecosystems is not restricted to a particular scale, but the results suggest that areas of dense patchiness (e.g. unburnt watersheds) must be analysed at finer scales than areas burnt at least every four years.

7.6 SUMMARY OF PAPERS ENTERED INTO THE DATABASE

An exhaustive search for publications related to the forest fires and landscape yielded so far a total of 856 citations from peer reviewed international publication sources. In regard of the Euromediterranean region, the results from this search indicated that 49 references were related to forest fire perturbations and landscape scale, from which 14 studies was devoted to the impacts of forest fires at this broad scale.

One of the major challenges in disturbance ecology is to predict the role of fire in landscape dynamics.

Understanding landscape-fire interaction result in a complex task due to the multivariate nature of the phenomena and the feedbacks among variables implied.

Several approaches focusing in different aspects of the relationship fire-landscape (i.e. fire hazard, fire spatial patterns and postfire recovery) have been applied in the Mediterranean area.

Usually, to analyse the fire hazard due to progressive fuel build-up, land use changes through time by transition matrices have been used (MOREIRA et al. 2001; ROMERO and PERRY 2002, 2004) applying different indices to estimate the increasing fuel accumulation over time.

The effect of landscape structure on spatial patterns of fires have been studied from the simplest crosstabulations between burned areas and land covers or topography (LLORET et al. 2002; MOUILLOT et al. 2003; VAZQUEZ and MORENO, 2001) to “resource selection functions” (MOREIRA et al. 2001) verified by bootstrap techniques as Montecarlo (ZAVALA et al. 2001).

The studies about the fire effects on landscape structure showed greater diversity of approaches due to the different way to treat the spatial and temporal component.

Almost all works are developed on “burned patches” considering only the fire effects on the area affected (VIEDMA et al. 1998; LLORET et al. 2002) and, few works analysed the effects of fire at regional scale treating “burned and unburned patches” simultaneously (BUCINI and LAMBIN 2002; VIEDMA and MORENO 2004).

On the other hand, we found researches that analysed fire effects at short-term (immediately effects or less than 5 years after fire) (CHUVIECO 1999; RICOTTA et al. 1998; VIEDMA et al. 1999) whereas others worked at long-term scale considering the effects over more than 10 years after (MOUILLOT et al. 2003; TRABAUD and GALTIÉ 1996).

Usually aerial photographs and satellite images have been used to develop such analyses, although few works treat with continuous time series data to deal with temporal dynamics (VIEDMA et al. 1997; DIAZ-DELGADO et al. 2002; VIEDMA and MORENO 2004).

Continuous data by NDVI values, texture indices or semivariograms (CHUVIECO 1999; RICOTTA et al. 1998; VIEDMA et al. 1998, 1999) and nominal data by land cover classes (VIEDMA and MORENO 2004; LLORET et al. 2002; BUCINI and LAMBIN 2002) have been the main landscape variables used to infer the effects of fire on landscapes and, in all the cases, spatially explicit fire perimeters were considered to carry out the analysis.

In general, changes in FRAGSTATS landscape metrics have been used as surrogates of the effects of fire on landscape structure.

The effect of fire frequency on landscape structure has been studied comparing multiple burned areas using either nominal data (LUs) (TRABAUD and GALTIÉ 1996) and continuous data (NDVI) (DIAZ-DELGADO et al. 2002) testing for differences among them.

Finally, the regeneration process has been analysed at landscape scale considering continuous data (NDVI) for multiple burned areas to characterize the pathway of recovery using regression models (VIEDMA et al. 1997; DIAZ-DELGADO and PONS 2001) at middle –term (not more than 10 years after fire).

In other works the effect of topography, lithology and prefire vegetation were considered in regeneration process by means ANOVA or MANOVA analyses (VIEDMA et al. 1997; DIAZ-DELGADO et al. 2002) or by crosstabulation and cluster analysis (VIEDMA and MELIÁ 1996, 1997).

8 BOREAL ECOSYSTEMS

This chapter has been compiled and written by Ilkka VANHA-MAJAMAA FFRI

8.1 INTRODUCTION

Humans have affected the forests in Finland for at least 4000 years, and today almost all forests in Finland have been affected by human utilisation.

Increased population and forest utilisation increased also fire frequency.

However, even fires were more rare before man, their size was on an average higher.

The spread of slash-and-burn cultivation further increased fire frequency compared with natural conditions.

In its core areas the slash-and-burn cultivation was practiced for centuries and it ceased only in the beginning of the 20th century when industrial forest utilisation had already started.

The first fire statistics in Finland are from 1800 century.

At that time, during maximum fire years, 55 000-70 000 ha could be burned, and e.g. between 1865 – 1870 the average size of a fire was 131 ha (SAARI 1923).

Fires were largely caused by human, since slash and burn cultivation was common at those times.

In pristine stands fires were more rare, but likely larger in size on an average.

Effects of fire in Finnish nature remained high until 1950's and 1960's, when prescribed burning was still a common practice in forest regeneration and annually 30 000 ha could be burned.

Since the Second World War, structure and development of forests have strongly been shaped by intensive forest management aiming at efficient timber production.

Forest management, using compartments of 1-10 ha as basic operational units, has aimed at fully utilising the sites' wood production potential by converting naturally heterogeneous stands to homogeneous even-aged single-species stands using clear-cut harvesting and silvicultural treatments such as thinning and planting.

At landscape level the management goal has been a fully regulated even-aged forest, where each stand age class covers an equal area.

This has led to a mosaic of even-aged forest stands with extensive road network for forest utilisation.

Even-aged stands with low levels of fuel, fragmented forests and more developed fire suppression tools has led to a situation where the role of fire and other natural disturbance processes have nearly totally eliminated from Finnish forests (KUULUVAINEN, 2002, Working group...2003).

E. g. dead trees provide habitat for a large number of species, ca. 5000 in Finland (SIITONEN, 2001).

8.2 FOREST MANAGEMENT EFFECTS

In southern Finland's managed forests there is only 2-10 m^3/ha of coarse woody debris (CWD), while in natural forests there is on an average 60-90 m^3/ha of dead wood.

This means that the average volume of CWD has decreased by 90-98%.

Judged from general species-area models this could in the long run mean a disappearance of over 50 % of the species depended on dead wood (saproxylic species) (SIITONEN, 2001).

When a forest fire occurs some trees die immediately, some within a few years and some survive the fire.

The selective effect of fire on different tree species increases the structural complexity of the post-fire stand and enhances conditions for tree regeneration (VANHA-MAJAMAA et al. 1996).

The effects of fire on the soil organic layer enhance tree regeneration and activate the potential soil seed bank of herbaceous species.

Large amounts of dead wood in open sunny conditions, and competition-free substrates created by fire are important habitats for a large number of decomposing fungi and saproxylic insects (PENTTILÄ and KOTIRANTA 1996, WIKARS 1997).

In Finland a large number of species depending on fire have declined during recent decades, due to lack of both natural and prescribed fires (RASSI 2000).

In Finland forest management methods have been modified during the 1990s.

The forestry legislation was reformed in 1997.

The new forestry law sets ecological and social sustainability, preservation of biodiversity and sustainable yield of forests as equally important goals.

Consequently, governmental, industrial and private forestry organisations have all reformed their management guidelines during the past decade.

New management practices that are already widely applied include setting aside habitats of special importance for forest biodiversity (so-called key biotopes), retention of living and dead trees in harvesting, and favouring prescribed burning and deciduous admixture.

All these measures add structural features of natural forests into managed forests, hence the measures can be regarded as restoration in its widest sense.

However, at present we have only limited understanding of the ecological efficiency of these new management practices.

In earlier forest management studies it was found that in mesic spruce forest stands, normally described only as one biotope in stand characteristics, there exists a lot of small-scaled variation in vegetation types (JALONEN AND VANHA-MAJAMAA 2001, VANHA-MAJAMAA AND JALONEN 2001.

In small-sized wet depressions (paludified patches) e.g. species composition, the basal area of aspen (*Populus tremula*, rich in epixylic species and therefore important species for biodiversity) and the amount of CWD are highest (VANHA-MAJAMAA AND JALONEN 2001).

These depressions are common in mesic forests though they are not normally distinguished in stand characteristics.

Since these wet depressions are likely not to burn as easily as dry areas, it is hypothesised that small-scale site type variation in combination with fire play an important role in restoring structures, dynamics and species in these mesic sites.

Based on information from the national forest inventories, over the past 50 years, changes in land use and forest management practices have had a major influence on the forests in Finland.

For centuries, the forests have been utilised by man, and still in the early 1950's signs of earlier slash and burn cultivation, forest grazing, and selective cuttings were still apparent.

After the Second World War, forestry was intensified and new silvicultural methods were introduced.

Consequently, entire ecosystems have been modified and the changes have not always been favourable to all forest species.

Some species are directly threatened by forestry practices, but even species that are relatively common have been affected by altered site conditions and stand structures (REINIKAINEN et al. 2000).

In the beginning of the 20th century, forest fires still played a major role in shaping the landscape pattern, stand structure and species composition.

Currently, fires have only a marginal influence on the forests owing to fire suppression policy.

As a result, species dependent on fire, abundant on burnt sites or during the early stages of post-fire succession, have declined over the last 50 years.

Of the most common plant species e.g. *Calluna vulgaris, Veronica officinalis* and *Pteridium aquilinum* have declined remarkably in Finland (REINIKAINEN et al. 2000) and many species are threatened or endangered (RASSI 2000).

During the period from the 1950's to the 1990's, the number of forest fires has been quite stable.

The total burned area (in hectares) has been declining on an average from the 1950's to the 1990's.

Although the number of forest fires has been increasing in the 1990's, the total burned area is still very small, as the average size of a single forest fire has been decreasing from 11 ha in 1950's to 0,6 ha in 1990's.

Therefore, there is a clear need for actively restore forests with fire in order to maintain and restore biodiversity.

8.3 RESTORATION ACTIVITIES AFTER FIRE

Restoration activities started more than a decade ago, but the annual restoration levels are still very low, especially on mineral soils.

It was not until the mid 1990's that more than 500 ha of forests was restored annually.

Until 1998, burning was the main method of restoration on mineral soils.

Since then, creating dead wood and small gaps have increased as methods of forest restoration.

Until 2003, approximately 1 300 ha of forests have been restored in protected areas.

The most commonly used restoration methods in protected forests have been forming dead wood by tree girdling or felling (46% of all restored forests), burning of stands (with trees) (23%), and imitating gap dynamics by creating small openings in even-aged stands (16%) (Working group… 2003).

Research on the effects of restoration activities is still mainly lacking.

However, preliminary results from the restoration experiments show that prior logging epixylic flora is clearly dependent on biotope, tree species and decay stage of the log.

Species composition of the logs in paludified biotopes differ slightly from the logs in upland biotope type, largely because of the differences in the microclimatic conditions.

Spruce logs have the highest flora of epixylic species, and the more decayed the log was, the richer was the epixylic flora.

Species richness is higher in wet biotopes (RYÖMÄ et al. 2002).

After fire, differences in the fire damages between the biotopes are clear.

Dry biotopes burn quite well, whereas in wet biotopes there exist unburned patches (LILJA et al. 2002).

Even in dry biotope the plant species composition, e.g. in the moss layer, affected the consumption of humus layer in rather similar (40-70%) moisture conditions (Fig. 4) creating a mosaic of burned and unburned patches.

Humus layer thinning is highest with *Pleurozium schreberi* and differed statistically significantly as compared with *Dicranum* sp. and *Hylocomium splendens*, indicating e.g. the morphological differences between the species.

After fire the decrease in humus layer thickness is generally low, but there is a significant decrease in organic matter and carbon.

Relative changes in element concentrations are generally clear.

After fire there are notable differences in understorey vegetation between the biotopes and different fire severity classes.

Succession is generally fast and approximately half of the species present prior the treatments reappear to the sites in two years.

Besides these, several species typical to early successional stages after fire appear in the vegetation community.

Aspen regeneration from seed is higher in exposed mineral soil, burned humus and especially in paludified biotopes, as compared to mature moss-covered forest sites.

In other species groups several fire dependent polyphores and fungi appear to the sites after fire.

Generally preliminary results show that with the use of loggings and prescribed fire it is possible to mimic the effects of natural fires and maintain and restore initial diversity of the stands.

Paludified patches seem to act as refugia for the species and dispersal centres, thereby helping in colonising the sites.

Ecosystem restoration with fire and other means is clearly needed in boreal ecosystems in Finland to accelerate the formation of dead wood and other structural features resembling those of natural forests.

By using fire in a controlled manner it is possible to increase the complexity of stand structures, to increase dead wood and to create open and warm habitats and young successional stages dominated by deciduous trees.

On the landscape level it is necessary to ensure long-term fire continuity to provide habitat for fire-dependent species.

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