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DEVELOPING A PRESCRIBED BURNING EXPERTISE IN ITALY: LEARNING FIRE EXPERIMENTS

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"…cold, and moving carefully on the tracks, fully aware of his entire body, his face, his mouth, his eyes stuffed with blackness, his hears stuffed with sounds,…,he saw the fire ahead.

The fire was gone, then back again, like a winking eye. He stopped, afraid he might blow the fire out with a single breath. But the fire was there and he approached warily, from a long way off…The small motion, the white and red color, a strange fire because it meant a different thing to him.

It was not burning. It was warming.

He saw many hands held to its warmth, hands without arms, hidden in darkness…He hadn't know fire could look this way. He had never thought in his life that it could give as well as take…"

Ray Bradbury, Fahrenheit 451.

Abstract

Prescribed burning is one of the main issue of current fire research in Mediterranean countries. Several benefits are expected: fire hazard abatement, nature conservation management, carbon emission reduction. Nevertheless in Italy very few knowledge is available about prescribed burning applicability. The develop of new expertises is required. To minimize the risk with introducing prescribed burning in fire management practices it is necessary to conduct learning experiments. Manipulative fire experiments which test the effect of different prescribed fire regimes (frequency, seasonality, intensity) have proved to be useful throughout world ecosystems. With the objective to develop and transferring expertises about the design of prescribed fire experiments two studies were carried out. Both studies adopted a microplot scale analysis of fire behaviour for correlation with ecological effects on vegetation. The first study dealt with *Calluna vulgaris* heathlands conservation in North West (NW) Italy. These heathland are under threat because of tree and grass encroachment as a consequence of extensive management. Like Muirburning in Scotland fire could be a suitable management tool in NW Italy. The experiment, started in 2004 at the Managed Nature Reserve of Vauda, Regione Piemonte, was designed to test the effect of fire frequency, backfire and headfire treatments on tree mortality and *Calluna* regeneration. The microplot scale analysis of fire behaviour enabled to cope with fire heterogeneity quantifying fine fuel consumption, rate of spread, flame height, *fireline intensity* and flame pick-temperatures with spatial and temporal resolution. An estimate of temperature-residence time profiles using infra-red termography was attempted. On the basis of short-term monitoring of vegetation responses to fire treatments a set of preliminary prescriptions for heathland conservation management by prescribed fire was established. The second study dealt with coarse woody debris (CWD) consumption by experimental fire in tropical savannas of Northern Australia. The experiment was carried out in the frame of a research project of the Tropical Ecosystem Research Centre (CSIRO) which studies effects of early and late dry season fires on savanna ecosystem. Through the manipulation of fine fuels a wide range of *fireline intensity* and temperature-residence time profiles was obtained. A simplified CWD consumption model was studied. Results were not very substantial, however this experience enabled to test new methodologies which implemented the experiment design at Vauda. Despite constraints this thesis demonstrated that is possible to develop research in the field of prescribed burning in Italy and hopefully will stimulate further discussion in order to improve knowledge and thereby future management.

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1. Introduction

Prescribed burning for fire hazard reduction (McArthur 1969, Fischer 1978) and for ecosystems maintenance (Chandler et al. 1983, De Bano et al. 1998) was introduced in Europe in the early eighties (Botelho et al. 2002). After 25 years, its operational use still remains very limited (Xanthopoulos et al. 2006). Nevertheless dramatic wildfire seasons in 2003, 2005 and 2007, made aware European countries that new fire management policies are required. The “wise use of fire” is considered one of the innovative approaches (Rego et al. 2007). Moreover, a recent study enlightened how prescribed fire could contribute to mitigate CO_2 emissions by wildfires in Mediterranean countries where fire occurrence is high, such as Italy (Narayan et al. 2007). Nevertheless in Italy, prescribed burning for wildfire prevention and conservation management is a contentious issue and very few knowledge is locally available about where, how and with which objectives to use prescribed burning (Leone et al. 1999, Ascoli et al. 2007).

To minimize the risks associated with introducing prescribed fire in Italy, it is important to conduct "learning experiments" (Holling 1978) to improve knowledge and thereby future management. In this sense, the maintenance of experimental plots subjected to various prescribed fire regimes has played an important role in understanding the ecological effects of prescribed fire throughout world ecosystems (Bond and van Wilgen 1996).

In Italy, studies in fire ecology have been conducted mainly studying “natural field experiments” (Hargroove and Pickering 1992) involving the scientific exploitation of un-manipulated wildfire events (Saracino and Leone 1993, Corona et al. 1998, Camia et al. 2002, Marzano 2005, Marzano and Bovio 2006). “Natural experiments” can be very useful for addressing fire issues, particularly in the context of understanding ecosystem responses to particular wildfire events (Anderson et al. 1998). Diversely, a proper understanding of the ecological effects of prescribed fire requires a rigorous experimental approach, incorporating appropriate spatial scale, adequate replication, the collection of extensive baseline (pre treatment) data, and detailed measurements of fire behaviour (Albini 1976, Pyne et al. 1996), that usually cannot be planned in "natural experiments". Moreover, prescribe fire is different from a wildfire. Consequently, to assess the feasibility of prescribed burning in achieving stated objectives manipulative fire experiments are the preferred method.

Up to now few manipulative fire experiments have been conducted in Italy (Buresti and Sulli 1983, Calabri 1988, Giovannini and Lucchesi 1997, Mazzoleni and Pizzolongo 1990, Esposito and Mazzoleni 1993, Bovio et al. 2001, D'Ascoli et al. 2005, De Marco et al. 2005, Ascoli et al. 2006). Consequently, developing a fire experiment expertise in Italy to assess prescribed fire effects it is a subject of scientific interest.

This thesis illustrates the methodology and results of three years of research in the modelling of prescribed fire behaviour and effects on vegetation by manipulative fire experiments. Two case studies are examined. The first study deals with conservation management of *Calluna vulgaris* Hull. dominated heathlands by experimental prescribed fire in North West (NW) Italy. The second study deals with the modelling of coarse woody debris consumption by experimental fire in tropical savannas of Northern Territory (NT), Australia.

1.1. Fire experiments for *Calluna* heathland conservation management in NW Italy

In NW Italy *Calluna* heathlands are relic ecosystems, mainly located on the Po River Plain which border the foot of the Alps. Until the second post-war period heathlands were extended on vast areas (Pavari 1927). These heathlands developed on poor and acidic soils as a result of forest clearance and they have been maintained by anthropogenic disturbance regimes resulting from the combination of burning, grazing and harvesting (Sartori et al. 1988). In the last 50 years, as a consequence of social and economical changes in the area, these traditional practices became marginal resulting in an ecological shift in the structure and composition of *Calluna* heathlands towards grassland or woodlands (Borghesio 2004). Moreover land modifications such as urban development and agriculture growth have highly reduced heathland extension throughout NW Italy (Garbarino et al. 2006, Regione Piemonte 2007).

Nowadays *Calluna* heathland are limited in isolated areas among the extent urban tissue of NW Italy. Consequently remnant heathlands have a high naturalistic value, constituting an important site for bird nesting and a shelter for several endangered plants which find their ecological needs in the heathland matrix. Moreover, *Calluna* heathlands represent a unique cultural landscape (*sensu* Farina 2000) and one of the sights and attractions of NW Italy.

For these reasons heathlands have been included in Managed Nature Reserves (MNR) in order to safeguard them from degradation. Paradoxically this protection is often applied in a passive way rather then adopting suitable management plans. A recent study about landscape dynamics in Parks of NW Italy (Garbarino et al. 2006) showed how the lack of management actions at the MNR of Baragge, which include one of the widest heathland systems of NW Italy, has caused the lost of 50% of the heathland habitat in the last 50 years.

Up to now, heathland conservation management plans based on sound scientific studies do not exist yet. Consequently a long-term experimental project[1], supporting the thesis that prescribed fire, goat grazing and mowing could be suitable management actions for conservation of heathlands in NW Italy, has been started in 2004. The study site is located at the MNR of Vauda, 30 km North of Torino, Regione Piemonte. One of the objectives of the experimental program at the MNR of Vauda (here after Vauda experiment) is to provide at the MNR land managers a technical report which clearly states a set of management actions for conservation of heathlands.

An understanding of the ecological role of fire, cattle and mechanical treatments in restoring and maintaining biodiversity requires multidisciplinary studies that integrate anthropology, pedology, zoology, plant ecology, landscape ecology

[1]The project "*Calluna* heathland conservation management by grazing, mowing and prescribed burning" has been financed by the Regione Piemonte from 2004 to 2007 and involves the following partners: the Ente di Gestione delle Riserve e dei Parchi Naturali del Canavese (http://www.parks.it/parchi.canavese/index.html); the Fire Ecology and Management unit (http://www.agroselviter.unito.it/pianificazione/index.htm) and the Grazing Land-Management unit (http://www.agroselviter.unito.it/range/index.html) of the Department of Agronomy Silviculture and Land Management (Agroselviter) of the University of Torino.

and wildland fire science. In fact the Research Project at the MNR of Vauda involves several research groups with different competences.

The present thesis is meant to deal with the fire research program of the Vauda experiment (Chapter 4). It must be emphasized the fact that the Vauda experiment is the first scientific experience in Italy which studies the use of prescribed fire for conservation management adopting manipulative fire experiments, incorporating appropriate temporal and spatial scales, adequate replications, analysing at a detailed spatial scale the fire behaviour in order to correlate fire characteristics with effects on vegetation and which tries to translate the research results in “unambiguous prescriptions” (Pyne et al. 1996).

1.2. Modelling coarse woody debris consumption by fire experiments in tropical savannas of Northern Australia

At a time of increasing carbon dioxide production, globalisation markets and emerging carbon trade economy, understanding carbon dynamics of terrestrial ecosystems is becoming an increasingly important priority to climate changes assessments (Shulze et al. 2000).

Estimation of the amount of carbon lost to the atmosphere due to the volatilisation during prescribed burning practices must be taken into account when calculating carbon sequestration in forests ecosystems managed by periodic and frequent burning (Pyne et al. 1996, Slijepcevic 2001, Williams et al. 2005, Narayan et al. 2007).

Current fire management practices in tropical savannas of Northern Australia sees a broad use of prescribed fire both for prevention then for conservation management objectives (Andersen 1999, Russell-Smith et al. 2003). Australian savannas are thus a source of carbon greenhouse emissions and a potential carbon sink in relation to prescribed and wildfire regimes (Williams et al. 2005).

Australia, although not yet a party to the Kyoto protocol, has ratified the United Nation Framework Convention on Climate Change (UNFCCC; http://unfccc.int/2860.php), and is committed to greenhouse abatement.

Consequently, to improve fire management practices in savannas of Northern Australia, robust estimates of effects of fire regime on spatial and temporal size, persistence and consumption of any carbon pool in these savannas are required (Williams et al. 2004).

The present thesis (Chapter 5) reports the description and results of a fire experiment for modelling fire behaviour effects on coarse woody debris (CWD) consumption, one of the least studied carbon polls of tropical savannas (Woldendorp et al. 2004). This study has been conducted during a training period spent at the Tropical Ecosystem Research Centre (TERC; web site: http://www.terc.csiro.au/default.htm), NT, Australia, that belong to the Commonwealth Scientific and Industrial Research Organisation (CSIRO; http://www.csiro.au/). The experiment has been carried out In the course of the "Burning for Biodiversity Research Program", a fire experiment started by TERC researchers in 2004 and designed to test the effects of different fire regimes on savannas ecosystems.

1.3. Objectives

This thesis is meant to improve knowledge about prescribed burning applicability in Italy achieving the following objectives:

- contextualize prescribed burning in present fire management scenario at a European and Italian level, with particular attention to Regione Piemonte reality;
- establish a set of prescriptions for *Calluna* heathland conservation management by prescribed fire in NW Italy;
- test methodologies to correlate fire behaviour descriptors to effects on vegetation comparing fire experiments carried out in heathland and savanna ecosystems;
- develop and transferring expertises about fire experiment design for ecological studies which aim at assessing prescribed fire use.

2. Prescribed burning

Prescribed burning is the deliberate application of fire to vegetation in order to attain well-specified management goals by burning in a specific fire environment (the prescription) and following specific operational procedures (the burn plan) (Fisher 1978). The two main and most widespread motivations for the use of prescribed burning in forest management are fire hazard abatement by fuel load reduction (Pyne et al. 1996) and conservation management of fire-dependent ecosystem (Chandler et al. 1983). Nevertheless a wide spectrum of objectives can be accomplished by prescribed fire, including land clearing, site preparation for tree regeneration, silvicultural improvements, range and wildlife habitat management, control of weeds, insect and diseases (Wright and Bailey 1982).

In a sense prescribed burning is simply a new way to name old cultural practices and at the same time represents the use of old cultural practices for new purposes (Pyne et al. 1996).

In this section a brief overview of fire-use history from primitive societies till contemporary fire management policies in Western countries is made in order to contextualize prescribed burning practices under an historical, cultural and operative perspective.

2.1. Fire and civilization

Fire has long been used as a tool in land management (Whelan 1995). Man has probably kept fire for more than 400000 years (Goudsblom 1992), but has learned to use it for ecosystem manipulation only during the last 40000 years (Wright and Bailey 1982, De Bano et al. 1998).

In South Europe fire became the first forceful tool used by man for land modification by the Late Pleistocene (Naveh 1975). Fire was used by the genus *Homo* to open corridors of travel, to protect human societies from wildfire by laying down controlled fields of fire around habitations and to drive and hunt wild animals (Pyne et al. 1996). With increasing human population and advanced hunting and food collecting economies, the extent and intensity of burning probably increased and became more and more linked with grazing by wild and semi-domesticated ungulates, wild goats and cattle. During the Neolithic, fire assisted agriculture to convert forested lands to crops and to provide cleared land for livestock management. Throughout the following phases of intensive agricultural and pastoral land use and especially from the Bronze and Iron Age until the Hellenistic and Roman period, the wide occurrence of fire is well documented in the Bible and Hellenistic and Roman literature (Naveh 1975). The consequences were extensive shifting the composition of European forests, opening fields, and transforming marginal forests to grasslands or heathlands. In the Middle Ages the extent of forests and wild lands diminished significantly and free-burning fires were extirpated in large parts of Europe (Pyne et al. 1996). By the 18th century land reclamation was accomplished and the need for fire uses in agriculture became less and less and gradually these practices were abandoned. Active burning persisted in marginal areas where sedentary agriculture was problematic; fire was used in Finland, Sweden and Russia with the practice of slash-and-burning (Goldammer et al. 2007); in Scotland, England, NW of France and Norway for heathland management (Webb 1998); grassland fires on the Spanish meseta and the Russian steppes (Pyne et al. 1996); pastoral burning was common among the Mediterranean mountains and there are evidences it has been used throughout history till today (Naveh 1975).

There is much written on the past use of fire by people in other parts of the world. Good examples come from various accounts of Australian Aboriginal use of fire for at least 40000 years (Nicholson 1981, Andersen 1999). Deliberate fires were used to clear dense vegetation, to make travel easier, to provide firebreaks around campsites, to hunt (Figura 2-1), to stimulate fruit production, to select vegetation, to drive off mosquitoes and flies and to celebrate spiritual rites (Haynes 1985, Russell-Smith et al. 1997).

A similar fire-use history is documented for North and South America; migration from Asia through the Strait of Bering in Alaska lead human around 9000 years B.C. to spread till the Tierra del Fuego at Cape Horn. Asian immigrants used fire and superimposed a new and extensive fire regime over the existing natural one (Pyne 1982). Native Americans applied fire in a broadcast form for habitat improvement, hunting, pest reduction, warfare and clearing area for home sites (Brockway et al. 2002). Moreover they developed a pattern of slash-and-burn agriculture, with new sites periodically cleared and burned as the old sites declined in fertility (Pyne et al. 1996).

Figura 2-1 Aborigines using fire to hunt kangaroos. Joseph Lycett, 1820.

Similarities between Australian and North America fire use concern also the post-settlement history. European colonists introduced all the fire practices used in the Old World and tried to adopt indigenous ones, often in the face of official criticism (Gill 1981a, Pyne et al. 1982). In North America, frontier farmers seized on fire hunting, reclaimed slash and burned sites, and routinely practiced protective burning around fields and dwellings (Pyne et al. 1996). In Western Australia most farmers favoured the widespread use of fire in adjoining forested land to provide additional land for crops and grazing capacity for stocks until a suitable stage of agricultural development was not reached (Mc Arthur 1969).

These are the two main historical reasons for the burning of forested lands by the agricultural community and they have caused constant friction between rural and forest authorities both in Australia then in North America. The approach by the foresters towards the use of fire was very different from that of the agriculturists. The need for intensive silvicultural management and protection of the young regrowth in order to achieve the necessary balance of age classes and full forest stocking made the use of indiscriminate burning unjustified.

2.2. Forest fire policies in North America and Australia

The need to preserve young regeneration and restock deteriorated logged and burnt-over land, combined with the reaction to many large forest fires in the late 1880s and early 1900s in North America (Pyne 1982) and in 1920s and 1930s in Australia (Mc Arthur 1969), influenced foresters towards a forest fire policy of fire exclusion. Efforts were made to increase the effectiveness of fire suppression by developing better access to reduce response times, mapping vegetation and fuel hazards, and keeping detailed records of any large fires that occurred (Stephens and Ruth 2005).

In U.S.A. Federal fire policy was formally started with establishment of large scale forest reserves during the late 1800s and early 1900s. The U.S. Forest Service was established as a separate agency in 1905 and began systematic fire suppression including the development of an infrastructure of fire control facilities, equipment, fire stations, lookouts, and trails (Stephens and Sugihara 2006). A policy of fire suppression was not universally supported during this period. One of the most vocal groups that argued for the use of fire in forest management was a group of private foresters from the northern Sierra Nevada and southern Cascades (Pyne 1982). In the late 1880s, they promoted the concept of “light burning” modelled after earlier Native American uses of fire. The main objective of this burning was to reduce fuel loads and associated damage when the inevitable wildfire occurred. Federal managers disagreed with this policy because of the damage to small trees and problems with fire escapes. Moreover World War II had a lasting influence on fire suppression. During the war, fire suppression efforts were modest, due to the war effort. However, after the war there was a new, much more intensive fire suppression effort that included the widespread use of the tools that were developed and refined in the war (Stephens and Sugihara 2006). Aerial retardant drops, helitack crews, bulldozers, and smokejumpers became the new tools of choice and this new fire-fighting force was very effective in continuing the policy of full fire suppression (Stephens and Ruth 2005).

The policy of complete fire protection resulted in a rapid build-up of fuels over large areas and the suppression action on fire became increasingly difficult (Pyne et al. 1996). Initial attack began to fail alarmingly and many fires escape and burnt large areas in spite of greatly improved fire control organization, equipment, detection facilities, communications and manpower abilities (Mc Arthur 1969). In the early 1950s both in Australia then in U.S.A. it became obvious to most control fire administrators that a drastic change in fire control policy was required in order to reduce this severe damage to forested lands. In U.S.A., the Leopold Report of 1963 (Leopold et al. 1963), informed the general public that protecting all plant communities from fire could be bad for a range of reasons; excessive fuel build-up, stagnant young pine trees, dense understorey of shrubs and trees in forests that could lead to stand replacing fires, decadent shrub and grassland communities, encroachment of shrubs and trees into grasslands, and ultimately devastating fires that cannot be controlled with any amount of man power. Fuel modification by prescribed burning practices on a scientific and systematic basis appeared to be the most effective and economical solution (Mc Arthur 1969, Stephens and Sugihara 2006).

Forest fire policy in the U.S. Forest Service changed from fire *control* to fire *management* in 1974. Henry De Bruin, Director of Fire and Aviation Management for the Forest Service, at the 14th Tall Timbers fire ecology conference (Missoula, Montana) stated: "we are determined to save the best of the past as we change a basic concept from fire is bad to fire is good and bad" (De Bruin 1974). The Tall Timbers Research Station was a private laboratory in Florida which sponsored a series of annual conferences (1962-1976) on the subject of fire ecology and prescribed burning (Pyne et al. 1996). Fire behaviour research confirmed that fuels were the greatest determinant of fire intensity, and recommended fuel modification by prescribed burning as a basis for wildfire control and a necessary supplement to traditional suppression that had reached its economic and technological limits. Moreover fire ecology studies enlightened the role of fire in shaping global biome distribution and in maintaining the structure and function of several fire-prone communities all around the world from the taiga forests of Siberia to the savannas of Brazil's Cerrado and the eucalyptus forests of North Australia (Wright and Bailey 1982). These ecosystems, because of their long subjection to recurring fire, started to be considered "fire-dependent ecosystems" (Chandler et al. 1983). Many studies concluded that the ecological role of fire in these ecosystems should be restored and suggested that prescribed burning should became an indispensable tool in the conservation management of fire-prone vegetation in nature parks and reserves (Wright and Bailey 1982).

Despite the extensive literature and scientific programs to study prescribed fire effectiveness and sustainability for fire hazard reduction and fire-dependent ecosystems maintenance from 1950s till today, the use of fire in forest ecosystems in Australia and U.S.A. has been, and continues to be controversial; prescribed burning is being increasingly scrutinized and regulated as a source of air pollution, traffic hazards, and escaped wildfire (Haines et al. 1998). Consequently prescribed burning it has been restricted to certain forest types and the degree at which is used changes from State to State (Shea et al. 1981, Von Johnson 1984, Stephens and Sugihara 2006).

2.3. Prescribed burning in Europe

Prescribed burning for fire hazard reduction and for ecosystems maintenance was introduced in Europe in the early eighties (Xanthopoulos et al. 2006). One of the first experimental program to study prescribed burning effects on vegetation was carried out by Luis Trabaud in forests of *Quercus coccifera* L. near Montpellier, France, from 1969 to 1975, and presented at the 14th Tall Timbers fire ecology conference of Missoula (Trabaud 1974). France has been in fact the first country in Europe to introduce prescribed burning (*brulage dirigé*) in its forest fire policy (Botelho et al. 2002). After almost a ten years period for testing at a research level the feasibility, the effectiveness, and the positive and negative effects of prescribed burning on French Mediterranean ecosystems, the first prescribed burning program as management tool appeared in the late 1980s and the number of prescribed burning teams started to increase in the early 1990s (Rigolot 1993). The strategy for wildfire prevention in the French Mediterranean region is mostly based on wild land compartmentalisation through fuel-break networks. These fuel-break enable fire fighters to develop effective strategic actions in good security conditions. To be efficient, these fuel-break are opened and regularly maintained reducing fuel build-up mainly using prescribed burning (Rigolot and Gaulier 2000). The French Mediterranean Region is now divided in 15 administrative districts (départements), each of them including nowadays at least one burning team in activity. About 10000 ha are currently prescribed burnt in France every year (Rigolot and Gaulier 2000).

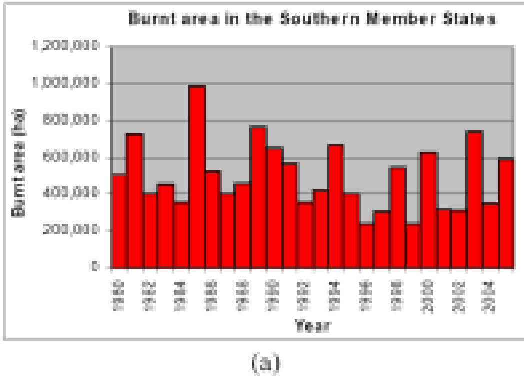
Spain and Portugal are, after France, the European countries in which the use of fire for management purpose has been studied and applied (Botelho et al. 2002). Prescribed burning for fire hazard reduction was the main reason of its introduction in Portugal in the mid 1970s, but benefits to soil properties and the diversity and nutritive value of understorey vegetation were also expected (Fernandes and Botelho 2004). The genesis and history of prescribed burning in north-western Portugal is thoroughly described by Silva (1997). More than adopted, the technique has been adapted to the region specifically, and its operational implementation by the Forestry Services went hand in hand with research studies focused on its ecological effects (Fernandes and Botelho 2003, Moreira et al. 2003). Also in Spain prescribed fire has been introduced mainly for fuel management in marginal lands where the abandoning of agricultural economies, the decrease in the grazing pressure and the shift to sources of energy other than wood, lead to shrubs encroachment. Consequently the landscape, historical fragmentised, has gradually evolved towards a continuous forest and shrubs cover without gaps. This is thought to be one of main causes of the occurrence of large fires. Prescribed burning has thus been considered as one of the management tools to reduce fuel load in these marginal lands (Baeza et al. 2002, Vega et al. 2005).

All the European experiences in fire research converged in International Conferences and Projects founded by the European Commission that put together partners from different countries and different competences on prescribed burning such as the S.T.E.P. Program (Vega et al. 1994). From 1996 to 2000 the Environment and Climate program of the European Commission funded the "FIRE TORCH project: Prescribed Burning as a Tool for the

Mediterranean Region. A Management Approach" (Botelho et al. 2002). The project was involving the main European countries of the Mediterranean Area (France, Spain, Portugal, Italy and Greece). The general objectives of the project were i) to identify and analyse constraints to the use of prescribed burning in Southern Europe; ii) to address current knowledge gaps concerning environmental and operational issues of this technique; iii) to develop decision-support tools for prescribed burning management; and iv) to help diffusion of the technique and training of the practitioners. The outcomes of the project were documented by a series of deliverables dealing with different aspects of prescribed burning management in Europe. One of the most active group of the FIRE TORCH Project was from the "Universidade de Trás-os-Montes e Alto Douro" (UTAD), Vila Real, Portugal, which, on the basis of previous studies on prescribed burning effectiveness for fuel reduction in *Pinus pinaster* forests drafted "A prescribed burning guide for maritime pine stands" (Fernandes et al. 2000a, Fernandes and Rigolot 2007) that represents a valid reference point for designing a prescribed burn plan and even fire experimentations. The main outcomes of the FIRE TORCH Project (Botelho et al. 2002) concluded that there is a general awareness of prescribed burning potentialities in forest management in Europe, mainly as a tool to reduce fuel hazard, and issues of technical nature dominate the motives that are selected as being obstacles to the technique development; moreover it was underlined that the ecological role of fire is poorly perceived by European foresters. Finally the Project's conclusion suggested how future research should pay special attention to the development of evaluation criteria according to management objectives other than wildfire prevention such as ecological sustainability.

Nevertheless, after almost 30 years from the first European experimental program on prescribed burning, the research efforts in Europe have not been translated in active forest fire policies nor at a National level neither by the European Community regulation; consequently prescribed burning operational use in Europe has remained limited (Xanthopoulos et al. 2006).

A succession of disastrous fire seasons in the five southern countries of the Mediterranean Basin Area (Portugal, Spain, France, Italy and Greece) in years 2000, 2003 and 2005, recorded averages in total area burnt (Figure 2-2a) and number of wildfires (Figure 2-2b), well above the average for the last 26 years (European Communities 2006). This dramatic situation, with roughly 700000 ha burnt each year (Fernandes and Botelho 2004), showed how wildfire occurrence is increasing mainly in the Mediterranean forests, despite efforts and new technology in the fire suppression and prevention.



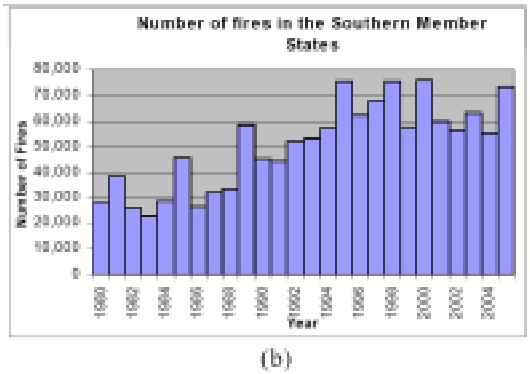


Figure 2-2 Burnt area by wildfires (a) and number of fires (b) in the five Southern Member States for the last 26 years (European Community 2006).

Consequently prescribed burning for fire hazard reduction is attracting a renewed interest of the European scientific community and considerable public funding. At the last two most important European International Conferences on fire issues, the 5th International Conference on Forest Fire Research in November 2006, Portugal, and at the 4th International Wildland Fire Conference in May 2007, Spain, many studies about prescribed burning use for fire hazard abatement and ecosystems maintenance were presented. Moreover from 2006 to 2010 the European Union is funding a Research Project titled “FIRE PARADOX”, that involves 35 partners from 17 countries, including Italy (www.fireparadox.org). The development of a prescribed burning expertise in Europe is one of the main project goals (Rego et al. 2007). The project is undertaking the ambitious task of setting a network of prescribed fire demonstration sites. Included are pilot projects, experimental and demonstration sites for the application of prescribed fire in the abatement of wildland fire hazard, forest succession management and ecosystem restoration.

2.4. Prescribed burning policy and research in Italy

In Italy prescribed burning it has rarely been applied and its use is a contentious issue. The general policy law on wildfire (Legge Quadro 353/2000) does not mention prescribed burning while its draft bill (Progetto di Legge 6195/1999; http://www.senato.it) was trying to introduce prescribed burning practice in the normative; in fact at the Art. 9 of the draft bill it was provided that "…the management objectives of prescribed burning are always due to scientific or economic interests and are limited to: fire hazard reduction, fuelbreak maintenance and fire-dependent ecosystem management…technical procedures and authorization system must be provided by the Regional Fire Management Plan" (Ascoli et al. 2005). As a consequence of this rule gap few Italian Regions (Basilicata, Liguria, Piemonte, Sardegna and Valle d'Aosta) "regulate", or rather "mention" prescribed burning use in their Regional Normative and Fire Management Plan (FMP) and no one actually apply it (Xanthopoulos et al. 2006, Ascoli et al. 2007). The normative inadequacy and its non-use at national and regional level is essentially due to a lack of knowledge about the applicability of prescribed burning in Italy.

The Italian scientific literature concerning prescribed burning is scanty and deals with the subject mainly describing how and why overseas countries use it and the advantages or disadvantages of this technique (Calabri 1984, Bovio 1996, Cesti 1999, Senatore 2000a). Few papers broach prescribed burning from a critical point of view: is it feasible to introduce prescribed burning to Italy?, with which objectives?, in which ecosystems?, which are its ecological effects?, which are the constraints? Some authors (Susmel 1974, Calabri 1981) listed the constraints related to the introduction in Italy of prescribed burning at a large scale: stands characteristics; forests mainly located in mountain territories; breaking up of properties; vast wildland-urban interface; juridical and administrative impediments; public opinion; lack of scientific studies.

Leone et al. (1999) made public the results of a questionnaire survey, conducted in the course of the FIRE TORCH Project, to identify and analyse current knowledge and perception of the prescribed burning technique among the forest researchers and administrators. The authors report that the most important reasons for conducting prescribed burning are considered to be the reduction of hazardous fuels, the management of competing vegetation and the conservation of fire-dependent species; the possible escape of fire and damages to the environment are the main concerns; legal requirements, lack of qualified personnel and poor knowledge of the ecological role of fire in forest ecosystems are the main reasons that hamper the application of prescribed burning in Italy. All the authors stressed the necessity to improve knowledge of prescribed fire effects, for instance with reproducible methodologies and comparable results (Stefani 1985), and promoting long term experimental studies to evaluate fire use sustainability (Calabri 1988).

Up to now few fire experiments have been conducted in Italy, mainly to study fire effects on plants and soil properties, and most of them without a management perspective (Giovannini and Lucchesi 1997, Mazzoleni and Pizzolongo 1990, Mazzoleni and Esposito 1993, Bovio et al. 2001, D'Ascoli et al. 2005, De Marco et al. 2005). Mazzoleni and Esposito (1993) well expressed the difficulty in realizing fire experiments in Italy: "…The setting up of burning experiments in Italy can be a difficult task because lighting a fire, even a very small one, is a criminal activity under current Italian laws. By bending such rules, some modest experimental fires could be set by collaborating with the Government Forest Service, but this situation created a major constraint to the design of the fire experiments, which perforce had to be limited to a single area".

Studies on prescribed burning effectiveness for fuel management and fire hazard reduction have been limited to single fire experiments without a long-term monitoring of the ecological effects and the effectiveness in achieving the objectives; this experiments did not have management implications (Buresti and Sulli 1983, Calabri 1988). Studies on the effectiveness of prescribed burning for conservation management of ecosystems in which recurrent fires play an important ecological role have never been attempted.

Nevertheless, a new interest in prescribed burning application seems to arise in last years: on one side the experience developed in other Mediterranean countries, such as France, Portugal and Spain, demonstrated the feasibility of its application and developed new knowledge adapted to the European contest; on the other side the happening of dramatic wildfire season such as summer 2007 evidenced the lack of an adequate prevention on our territory and the necessity of testing new management tools such as prescribed burning (Narayan et al. 2007).

Figure 2-3 Prescribed fire demonstration at Lido di Metaponto, Basilicata, September 2007 (Surce: Daniel Moya Navarro).

In September 2007, the Regione Basilicata, which has recently introduced prescribed fire in its regional normative (L.R. 13/2005), in collaboration with the University of Basilicata, has realized the first training course in Italy about prescribed burning applicability for an integrated fire management in *Pinus halepensis* Mill. littoral forests of Basilicata (de Ronde 2007). The course has been addressed to the Forest Service personnel, land managers and researchers. It ended with a experimental demonstration (Figure2-3) of prescribed fire application for fuel hazard reduction in a littoral pine forest located at Lido di Metaponto, Basilicata Region. This experience drawn the attention of politics and media.

Moreover, two Italian working teams are joining the FIRE PARADOX project thus constituting and important transfer of knowledge in Italy from European experiences: the first belongs to the Università di Napoli Federico II, the second to the Forest Service of Regione Autonoma della Sardegna (http://www.fireparadox.org/project_consortium.php?PHPSESSID=404cdc376de6c892532e61738890b39f1).

The research activities of the Dip. AGROSELVITER of the University of Torino about prescribed fire for heathland conservation management, exposed in the present thesis, fit in this changing contest and aims at developing new expertises to design fire experiments to asses prescribed fire sustainability under an operative and ecological perspective.

3. Prescribed fire experiment design

3.1. Conservation management by prescribed fire

Probably fire has been a part of natural ecosystems as soon as there was any existing terrestrial vegetation (Harris 1958, Trabaud 1987). Lightening and volcanic eruptions have always provided a natural ignition source of wildfires in specific fuel and weather conditions (Chandler et al. 1983). Bond and Van Wilgen (1996) highlighted "natural fire regimes" (the total pattern of fires over time that is characteristic of a region or ecosystem) as a fundamental disturbance process in the distribution of global vegetation patterns that prevent these ecosystems from achieving the potential height, biomass and dominant functional types expected under the ambient climate (climate potential). Nevertheless, fire use by man for land modification (see Chapter 2) started 40000 years ago and throughout humans societies anthropogenic fire regimes overtook with natural ones worldwide (Naveh 1975, Pyne et al. 1996, De Bano et al. 1998). According to Agee (1996), natural fire regimes are usually defined in an historical sense, typically restricted to the pre-1900s, but they clearly are natural in the sense of incorporating the effects of human activities. Consequently we cannot separate out the human component from "historical" or "natural" fire regimes, above all in Europe.

Many of the world's ecosystems, from maquis in the Mediterranean Regions (Naveh 1975), forests of Sierra Nevada in California (Keeley and Stephenson 2000), open savannas in North Australia (Andersen et al. 1998), fynbos in South Africa (Bond and Keeley 2005), heathland in NW Europe (Hobbs and Gimingham 1987), have evolved with fire, either wildfires or fires set by people (Wright and Bailey 1982). Thus for most ecosystems the "historical" fire regime (Agee 1996) has been, and in some still is, one of the most important "catalysts" for diversity (Pyne et al. 1996). In literature these ecosystems are defined "fire-dependent ecosystems" (Chandler et al. 1983). In these ecosystems the frequent interaction of fire has through time caused its periodic occurrence to become an essential disturbance process vital to sustaining long-term ecosystem health (Bond 2005).

The alteration of the "historical" fire regime as a consequence of the loss of traditional fire use knowledge and fire exclusion policies can bring fire-dependent ecosystems to an ecological shift in the vegetation composition, structure and succession dynamics, modifications of the hydrology, wildlife habitat and consequently to a change in the landscape; these changes risk the loss of the natural, cultural, social and economic heritage inherent in these ecosystems (Clewell and Aronson 2006). Moreover when fire regimes are altered, they can contribute to climate-changing, greenhouse gases, biodiversity erosion and present a direct risk to human habitation (Shlisky et al. 2006).

The Nature Conservancy (http://www.nature.org), one of the most trusted environmental organization (Harris Interactive 2006), in its preliminary global assessment of fire as a conservation threat, identified three broad categories of vegetation responses to fire (Hardesty et al. 2005): fire-dependent / influenced[1], fire-independent[2] and fire-sensitive[3] (Figure 3-1).

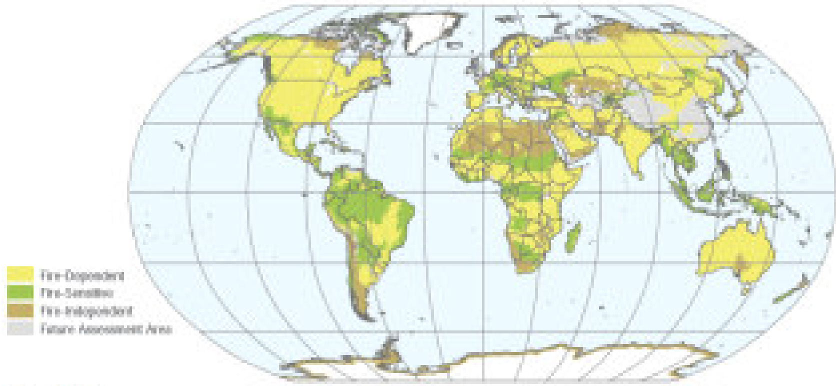


Figure 3-1 Priority ecoregions and dominant fire regions. Of important conservation ecoregions, experts estimated that 46% are predominantly composed of fire-dependent/influenced fire regimes, 36% are fire-sensitive, and 18% are fire-independent (© The Nature Conservancy 2007).

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[1] **Fire-dependent Ecosystems:** Ecosystems in which fire is essential and the species have evolved adaptations to respond positively to fire and to facilitate fire's spread. Often called fire-adapted or fire-maintained ecosystems.

[2] **Fire-sensitive Ecosystems:** Ecosystems that have not evolved with fire as a significant, recurring process. Species lack adaptations to respond to fire and mortality is high even when fire intensity is low. Vegetation structure and composition tend to inhibit ignition and fire spread.

[3] **Fire-independent ecosystems** are those that naturally lack sufficient fuel or ignition sources to support fire as an evolutionary force (e.g., deserts, tundra).

In 2006, The Global Fire Partnership (GFP), that includes The Nature Conservancy, World Conservation Union (IUCN), University of California at Berkeley Centre for Fire Research, and the World Wildlife Fund (WWF), implemented expert workshops to establish scientifically credible data to monitor the integrity of fire regimes at coarse ecoregional levels (Shlisky et al. 2006). Results revealed that 25% of terrestrial area is intact relative to fire regime conditions. Ecoregions with degraded but improving fire regimes cover 53% of global terrestrial area while Ecoregions with degraded and declining fire regimes, such as Mediterranean Regions, cover 8% (Figure 3-2).

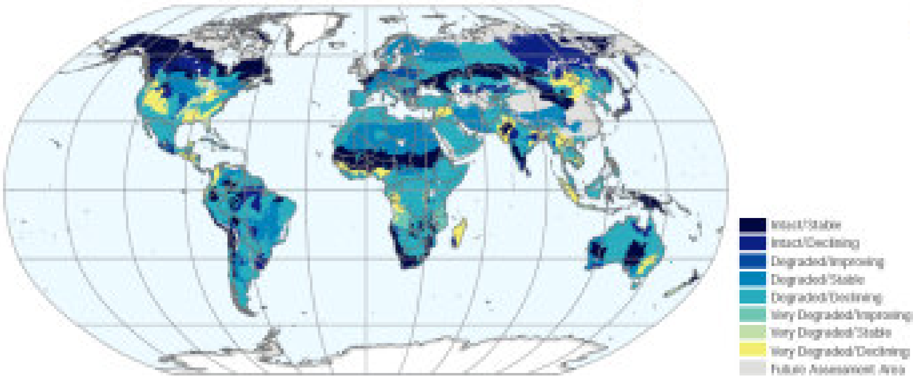


Figure 3-2 Distribution of fire regime conditions globally at the ecoregional level (© The Nature Conservancy 2007).

According to the GFP (Shlisky et al. 2006) the top threats to maintaining an ecologically-acceptable role of fire regimes include ecosystem conversion (livestock ranching, agriculture, urban development), resource extraction (energy production, mining, logging and wood harvesting), and human-caused fires or fire suppression.

The GFP advocates the importance of restoring the historical role of fire regimes worldwide for effective biodiversity conservation implementing an Integrated Fire Management defined as "an approach to addressing the problems and issues posed by both unwanted and desirable fires within the context of the natural environments and socio-economic systems in which they occur, by evaluating and balancing the relative risks posed by fire with the beneficial or necessary ecological and economic roles that it may play in a given conservation area, landscape or region" (Mayers 2006).

The prescribed burning technique is one of the management tools used by land managers worldwide to maintain ecosystems in which fire play a fundamental ecological role (Wright and Bailey 1982, Bond and Van Wilgen 1996, Pyne et al. 1996, Andersen et al. 1998, De Bano et al. 1998).

Proper use of prescribed fire in the maintenance of fire-dependent ecosystems requires a thorough understanding of the fire regime of an area and its effects on plant communities (Bond and van Wilgen 1996). Consequently, fire scientists have tested prescribed burning sustainability designing fire ecology experiments involving whole ecosystem manipulation (Biggs et al. 2003, Williams et al. 2003, Stocks et al 2004, Andersen et al. 2005).

Simulating a fire regime by fire experiments involves more then merely determining the average return interval of fire. Bond and van Wilgen (1996) defined fire regimes as the combination of how often fire occurs (frequency), when it occurs (season) and how fiercely it burns (fire type, intensity, and typical fire sizes). Consequently effective prescribed fire management and development of proper fire prescriptions requires on one side the understanding of the survival strategies of plant species and the succession dynamics of vegetation communities in a specific fire regime (Agee 1996, Bond and van Wilgen 1996); on the other side it requires an understanding of fire processes and heat transfer that explain characteristics of normal fire behaviour (type, intensity, seasonality, ignition patterns) as well as an understanding of how fire behaviour is coupled to specific fire effects (Rothermel and Deeming 1980, Johnson and Miyanishi 1995).

3.2. Fire and plants

Plants to survive fire have developed adaptive traits and survival strategies that allow them to resist to fire disturbance both at an individual and at a community level. Plant architecture, morphological characteristics and life history attributes, reproductive strategies, interact with fire seasonality, frequency and intensity resulting in variable responses to a specific fire regime (Gill 1981b, Bond and Van Wilgen 1996, Ryan 2002).

3.2.1. Adaptive traits of plants

Plants of different species and ages vary in their resistance to fire damage. Larger plants, such as trees, have proportionately more foliage above lethal scorch height in a given fire. The amount of crown damage a tree receives depends on scorch, height, tree height, and crown base height. Species differ in the timing of bud break, and also in the degree to which their buds are shielded from heat. Species that sprout from root crown are more vulnerable that species that sprout farther below the soil surface. Species with large buds and twigs tend to be more resistant to fire damage. Variation in rooting habit between species leads to greater susceptibility to root damage in shallow-rooted species. Perhaps the most dramatic difference in fire resistance between trees of different species and sizes is due to differences in bark thickness (Bond and van Wilgen 1996).

Not all species have characteristics that allow them to resist to fire damage. Fire can provoke injuries to plants in several ways. Crown damage may involve bud kill or foliage mortality. Cambium damage can result in partial or complete girdling of the bole. Bole damage may also involve structural damage to the plant, particularly if it has exposed heart rot. Root damage is perhaps least well documented, as it is hard to measure, but is almost certainly a fact contributing to mortality of shallow-rooted plants. Nevertheless many species once damaged are able to regenerate after fire both by sexual reproduction then by vegetative regeneration. Commonly these species are defined “pyrophytes” (Trbaud 1987).

Some species are stimulated by fire and flower profusely after a burn. These plants are generally Monocotyledons such as Amaryllidaceae, Graminaceae, Liliaceae, Orchidaceae families (Gill 1981b, Trabaud 1987, Bond and van Wilgen 1996). Other species flower regularly between fires but accumulate seeds in seed-banks stored in the canopy, insulated from fire by cones, woody capsules or persistent woody inflorescences. The protective structures only open and release seeds after a fire. This delayed seed release is known as serotiny. Examples of species that present this reproductive strategy are found in all continents: *Pinus halepensis* and *Pinus brutia* in the Mediterranean region (Naveh 1975), *Banksia ornata* and *Eucalyptus regnans* in Australia (Gill 1981b), *Pinus banksiana* and *Pinus contorta* in North America (Wright and Bailey 1982), *Erica sessiflora* in Cape fynbos of South Africa (Bond 1984).

Another strategy is showed by some “hard-seed” genres of Leguminosae family, such as *Acacia*, where dormancy is due to an impermeable seed coat. Dormancy in *Acacia* is broken by the opening of the lens induced by heat so as water can enter in the lens cavity and germination can begin (Bradstock and Auld 1995, Bond and van Wilgen 1996). Germination of seeds is induced not only physically, as it happens for serotiny species or “hard-seed” species, but can also be stimulated by chemical cues. Many fire-annuals have germination stimulated by charred wood or smoke (Keeley et al. 1985).

After seed dispersal, the combination of open space, increased availability of resources and temporary reduction in seed predators is highly favourable for seedling establishment in the post-fire environment. The released seeds often fall onto an excellent seedbed. Ash provides nutrients. The lack of foliage above increases the light reaching the soil, promoting seedlings germination and growth (Trabaud 1987).

Trabaud (1987) critically reviewed the concept and associated terminology of “pyrophytism” stressing the fact that it is often difficult to say which traits are specifically adaptation to fire, as opposed to general adaptations to disturbances and stresses of the environment. Thick bark, crown architecture, above-ground sprouting from epicormic buds protected by bark, below-ground sprouting from buds at the base of the stem, on roots or horizontal rhizomes protected by soil, all contribute to fire survival but are strategies that did not necessarily arise as direct responses to selection by fire as they enable plants to survive to any kind of disturbance such as grazing, browsing, cutting, frost, windrows, avalanches etc. (Trabaud 1987, Bond and van Wilgen 1996). Thus, according to Trabaud (1987), several survival traits, such as the vegetative ones, may best be considered as ‘adaptations’ to any kind of stresses or disturbances, one of these being fire.

3.2.2. Fire and plant succession dynamics

Plant populations respond in diverse way to fire. Some remain stable from one fire to the next. Others may fluctuate from explosive growth to local extinction. According to Bond and van Wilgen (1996), the fire regime (frequency, intensity, season) and the competition dynamics between plants are the main determinant of the population demography in vegetation communities characterized by periodic fires.

Fire frequency determines the length of the interval between two fire events and consequently the state of the population when it is burnt. When a fire occurs it interrupts the normal pattern of growth, mortality and reproduction of plants. The duration of these normal growth phases determines the state of the population when it is burnt and its post-fire response. Thus constant fire intervals should produce constant population growth rates. Nevertheless successional dynamics are regulated by much more complex interactions between different factors.

The fire characteristics such as the fire type, the range of fire intensities showed or the season in which it has occurred affect the 'severity' of a fire event intended as the "impact on the organism, community or ecosystem" (Ryan 2002). Fires in fact vary in their type: ground fires burn underground, in organic layers of the soil; surface fires burn just above the ground surface; crown fires burn in the canopies of trees. Usually the latter are sustained by the surface fires below them (dependent crown fires), but under severe conditions crown fire can race ahead of the surface fires (independent crown fires). Yet another source of variability is weather fires burn with the wind or upslope (headfires) or against the wind or downslope (backfires) and this behaviour have significant effects on the magnitude of the fire intensity and severity (Cheney 1981). Fire intensity may affects the number of seeds that germinates or plants that survive (Brown and De Byle 1987, Moreno and Oechel 1991, Bradstock and Auld 1995, Williams et al. 1999). Moreover the fire interval interacts with the intensity through its effects on fuel load (Andersen et al. 1998).

The fire season has pronounced effects on plant resilience to fire. Fires have the greatest impact on plants during periods of active growth, because carbohydrate and nutrient reserves have been depleted, or during the flowering season, because reproduction is compromised (Bond and van Wilgen 1996). Moreover fire season interacts with fire intensity which increases in dry, hot and windy periods (Andersen et. al. 1998). Consequently constant fire intervals may produce very different effects on population dynamics depending on the fire event characteristics.

Finally between two fire events population size is regulated by density-dependent competition dynamics. Initial seedling or sprout numbers can vary widely after a fire as a consequence of the plant density and age before the fire event. The subsequent mortality can have a density dependent component due to competition for resources such as light or nutrients. Moreover, as density increases, the dimensions of an individual plant and the number of flowers, fruits and seeds decline (Bond and van Wilgen 1996).

The effects that a fire regime can have on vegetation dynamics thus depend on the product of many factors interacting and consequently it is very complex to understand which are the main ones which regulate the existence of a particular vegetation structure and composition at a given time after fire; even more complex is to manipulate these factors using the prescribed burning technique in order to attain the desired ecological effects. One of the most difficult tasks in planning a prescribed fire for conservation management purposes regards the understanding of how specific fire behaviours are coupled with ecological effects.

3.3. Fire behaviour and effects

In the "Introduction to wildland fires" (Pyne et al. 1996), the authors were underlining some problems about methods used in traditional fire ecology experiments: short-term studies, lack of experimental control, no replicates and above all the overlook of the impacts of spatial and temporal variability of fire behaviour.

To plan prescribed fire to achieve stated objectives, to minimize cost of control, and to reduce the risk of escape, a firm basis of fire behaviour estimation must be established (Johnson and Miyanishi 1995). Because specific effects are sought and specific sites are burned under pre-selected conditions to achieve them, in many cases prescribed burning poses the most stringent requirements for fire behaviour prediction models (Albini 1976). In fact fire prescriptions are precise statements of a fire behaviour (intensity, rate of spread, frequency) and the desired ecological effects correlated to this behaviour (Pyne et al. 1996).

3.3.1. Fire behaviour modelling

The behaviour of a fire, whether initiated by humans or stochastic event such as a lightning strike, is dependent on several environmental and atmospheric factors. Factors which influence fire behaviour include fuel characteristics (type, arrangement, load and moisture content) (Brown and Bevins 1986, Catchpole et al. 1998, Fernandes 2001), weather (wind speed and direction, temperature, humidity and precipitation) (Cheney 1981, Molina and Llinares 2001), topography of the fire site (Dupuy 1995, Viegas 2004).

Research in fire physics to describe fire propagation through vegetation fuels and its variability has a long history both in USA (Fons 1946), Australia (Luke and Mc Arthur 1978), Canada (Van Wagner 1977), and Europe (Thomas 1971).

These studies lead fire behaviour specialists to develop fire behaviour models to predict fire spread and heat processes. The models can be grouped in three categories, respectively empirical (or statistical), semi-empirical (semi-physical or laboratory models), and physical (theoretical or analytical) (Morvan et al. 2004). Empirical models predict fire spread and heat processes as a function of different variables measured in field experiments such as fuel load, moisture and arrangement, slope, wind speed, time since fire, that are correlated to fire spread by statistical regression. These models do not have general application but have high performances for the environment in which they have been studied (Cheney and Gould 1993, Marsden-Smedley and Catchpole 1995, Vega et al. 1998, Fernandes 2001, Davies 2005). Semi-empirical models start from physical phenomena but use statistical correlation obtained by laboratory experiments (Rothermel 1972, 1983, Catchpole et al. 1993, Viegas 2004). The physical modelling of fire behaviour try to formalize the complex phenomena of the combustion involved in fire propagation adopting mathematical solutions (Albini 1986). Despite these models can have a general application (Camia 1994), a theoretical formulation capable of a direct, practical, and truly predictive response to the entire fire behaviour variability that can naturally occur is quite distant (Alexander et al. 1998), and will depend on the nature of its inputs as well as on the availability of powerful computing resources. Consequently, fire modelling for operational applications is currently restricted to semi-empirical or empirical models.

According to Albini (1976), potential uses of empirical and semi-empirical fire behaviour models span the spectrum of fire-related decision making from fire-danger rating to prescribed fire design. Fire Danger Rating Systems are management tools based on semi-empirical models of fire behaviour which asses the degree of fire hazard and the risk of fire outbreak on a day-to-day basis (Mc Arthur 1973, Deeming et al. 1977, van Wagner 1987, Marsden-Smedley et al. 1999, Davies et al. 2006). Semi-empirical models constitute also a support to decisions in planning fire prevention at a territorial level (resources distribution, roads, fire breaks and watering location), and, in case of large fires, to predict how wildfires will behave under the prevailing environmental conditions for an optimal organization of fire suppression and extinction. Moreover the development of computer engineering has enabled to create softwares based on complex models that simulates fire spread under prevailing weather conditions (Burgan and Rothermel 1984, Andrews 1986, Finney 1998, Scott 1999, Andrews et al. 2005).

Both in fire-danger rating then in wildfire prevention and control, models outputs need not be highly accurate (Albini 1976). It is important that the system of models (fire behaviour models and fuel models) properly rank the fire behaviour variables estimated and that they respond to changes in weather consistently and with sufficient sensitivity to permit decision boundaries to be established. For these purposes, stylised fuel models are entirely adequate, and indices of relative severity of fire behaviour are sufficient.

Several studies have showed how performances of previously developed fire behaviour semi-empirical can be unsatisfactory and not provide adequate fire behaviour prediction when applied to local fuel type and whether site conditions (Catchpole et al. 1993, Marsden-Smeadley and Catchpole 1995, Davies et al. 2006). In fact, the use of pre-established fuel bed descriptions, such as the fuel model used in the National Fire danger Rating System of U.S.A. (Deeming and Brown 1975, Anderson 1982, Burgan and Rothermel 1984), and occasionally used in Europe (Bovio 1993, Camia 1994, Salas et al. 1994), or the use of “fuel prototypes” to predict fuel characteristics (Sandberg et al. 2001), may be inappropriate for accurate prediction as the specific site being burned may differ substantially from the assumed fuel bed (Albini 1976, Leone et al. 1993).

Consequently effective prescribed fire for conservation management and development of proper fire prescriptions requires the characterization on site of the fuel type and the development of an empirical fire behaviour prediction system for the fuel type being studied using observed fire field data.

3.3.2. Fuel modelling

For any given fire the nature of the fuel-bed plays a key part in determining its behaviour, and an understanding of the fuel environment is a prerequisite for any investigation of fire behaviour. Numerous studies demonstrate the linkages between fuel attributes and rate of spread (Fernandes 2001), peak temperatures and their residence times (Hobbs and Gimingham 1984a, Molina and Llinares 2001) and flame lengths (Bradstock and Gill 1993). Knowledge of fuel loading is thus crucial to understanding fire spread and intensity. However, while higher fuel loadings will in general give hotter fires that spread faster things are unfortunately never that simple, as fuel architecture (distribution and structure) is also critical (Brown 1981).

The density and homogeneity of the fuel bed are important in determining the rate of spread of fire. Bulk density is in fact inversely related to rate of fire spread (Rothermel 1972). This means for a constant fuel height increases in loading will reduce spread rates. This may be as denser fuels lead to a lack of available oxygen for complete combustion (Catchpole and Catchpole 1991). Moreover significant breaks or gaps in vegetation cover will disrupt the flow of the fire due to greater distances over which raised temperatures must be radiated and thus less efficient heat transfer (Brown 1981).

A further layer of complication is added by the effects of fuel partitioning. Deeming et al. (1977) grouped fuel complexes according to size classes that correspond respectively to timelag[1] classes. So called “fine fuels”, with a

[1] The *timelag* is the time required to a fuel particle, in a changed environment such as an heating process, to loose the 63% of the difference between the initial moisture content and the final one of equilibrium. The timelag is directly proportional to the fuel particle dimension. In laboratory analysis is used a standard temperature of 27°C and air humidity of 20% (Pyne et al. 1996). Timelag classes are 1 hr, 10 hr, 100 hr, 1000 hr and usually are respectively made correspondent to fuel particle size of < 0.6 / 0.6 - 2.5 / 2.5 - 7.6 / > 7.6 (values are in centimetres) (Deeming et al. 1977).

diameter inferior to 0.6 cm (Pyne et al. 1996), ignite much more readily than those which are larger, partly because their moisture contents tend to be lower and their greater surface area-to-volume ratio gives them a lower thermal inertia which allows them to dry out and reach combustion temperatures more rapidly (Cesti 2005). Moreover, large surface area-to-volume ratios increase the rates of energy and mass exchange with the gaseous phase, leading to lower ignition delays and higher rates of fire spread (Fernandes and Rego 1998). Consequently, distinguishing the fine fuel fraction from the total fuel load is crucial as it contributes to the zone of continuous combustion and is thereby important in determining the rate of spread (Cheney 1981).

The amount of water held in fuel particles is of vital importance to fire behaviour. The fuel moisture content of both live and dead fuel components affects the duration of pre-heating and amount of energy required to raise fuel to combustion temperature (Pyne et al.. 1996), rates of combustion and the amount of energy released (Byram 1959). High moisture contents reduce fire risk, intensity and spread because the energy produced by a fire must first be used to drive off excess moisture while the production of large quantities of steam may have a smothering effect on the combustion process by reducing the oxygen concentration of surrounding air (Catchpole and Catchpole 1991).

Over the years a wealth of techniques have been developed to assess various aspects of the fuel environment using non destructive methods (Catchpole and Wheeler 1992, Allgöver et al. 2004). These include simple methods based on the assessment of fuel by measurements of variations in plant basal diameter, height and cover that are statistically related to fuel characteristics such as fuel load (Brown 1976). Photographic guides based on calibrated photographs (Anderson 1982) or simplified intercept methods (Étienne and Legrand 1994) are other non destructive methods but usually do not provide the level of accuracy or detail required.

Destructive methods allows to characterise fuel complexes precisely, collecting representative samples (Brown et al. 1982, Fernandes 2001) and subsequently determining fuel load, moisture and fuel partitioning with laboratory measurements (Burgan and Rothermel 1984, Camia 1994). The main drawback of fuel destructive sampling is on one side the fact that is a laborious and long work that comprise both field and laboratory measurements, on the other side the alteration of the fuel bed in small scale fire experiments can affect normal fire behaviour (Fernandes et al. 2000b, Davies 2005).

3.4. Fire behaviour characterization by fire experiments

The main descriptors of fire behaviour that have been used in fire ecology studies to quantify fire characteristics can be grouped as follow: descriptors to quantify fire spread such as the dimensions and orientation of the flames, the rate of spread, the residence time and the acceleration process of the head of the fire front (Rothermel and Deeming 1980, Cheney 1981); descriptors to quantify heat processes such as Byram's *fireline intensity* (Byram 1959) and flame temperatures and their duration at different heights (Hobbs et al. 1984).

3.4.1. Fire spread

Flame height and depth are important descriptors as are related to scorch height on the stem, crown scorch height and consequent probability of mortality of plants (Williams et al. 1998). Flame depth (D; m) and the rate of spread (ROS; m/s) are related as follow (Rothermel and Deeming 1980):

$$Tr = D / ROS \qquad \text{(Eq. 1)}$$

where Tr (sec.) is the residence time of the flame defined as the duration of flaming combustion in a point (Cheney 1981) and can be quantified as the time from initial temperature rise above ambient to the time of 'definitive drop' following attainment of peak temperature (Rothermel and Deeming 1980). The residence time must be distinguished by the "burn-out" time (Tb). This is the time taken for the fuel bed to burn out trough all the combustion phases (e.g. the smouldering phase can last for long time after the fire front is passed). The Tb is correlated to the temperature-residence profile in a point (Figure 3-3), that is the elapsed time from the first rise above ambient, or above a fixed temperature (e.g. 200 °C), to the time the temperature decreases below the fixed temperature value or the ambient temperature (Moore et al. 1995). Under an ecological perspective the Tb is more interesting than the Tr as it is related to the temperatures duration at ground level and consequently to the impact of fire on buds at the stem base, roots and soil properties (Cheney 1981).

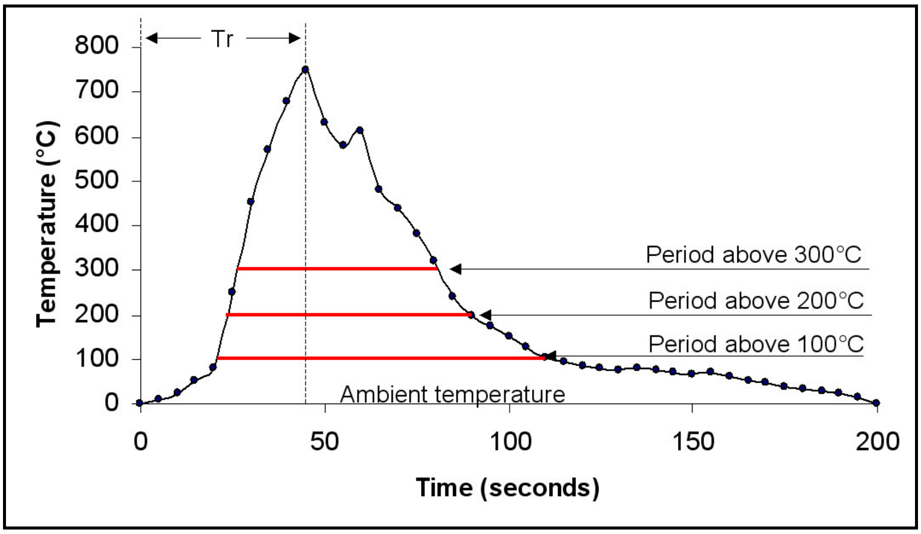


Figure 3-3 Residence time of the flame (Tr) and temperature residence profile for a surface fires (From Moore et al. 1995; modified).

The rate of spread of a fire front is the speed of movement of a fire front generally measured at right angles to the direction of spread. It is one of the main dependent variables predicted by fire behaviour models as it gives information about the fire type, the difficulty of suppression, the maximum area that can be burnt in a given time and flame dimensions (Pyne et al. 1996). Rate of spread has traditionally been estimated by measuring a fire's progress on sequential fire front maps or by timing its advance through a grid of pre-set markers by visual estimation or thermocouples connected to timers (Rothermel 1983, Cheney 1990, Moore et al. 1995); the use of video or infrared cameras can also be useful to map the advancing fire front (Viegas et al. 2002).

In modelling fire front spread the plot size is determinant. In fact Cheney (1981), a fire behaviour modeller of the CSIRO Forestry and Forest Products Division placed in Canberra, Australia, recognised the "importance of acceleration phenomena when planning ecological experiments involving test fires". CSIRO studies about empirical fire behaviour modelling in grassland and eucalypt forests (Cheney and Gould 1993, Cheney and Gould 1995, Cheney and Gould 1997, Cheney et al. 1998, Gould et al. 2004) showed how a fire lit from a point ignition increases its rate of forward spread until a rate of spread, approximating a *quasi* steady state, is reached. The acceleration patterns of fire are very variable but generally the more severe the burning conditions the longer will a fire continue to accelerate. A low intensity prescribed fire (I < 500 kW/m) will reach a steady rate of spread after few minutes. A high intensity fire may continue to accelerate for several hours. To some extent the time required to reach a steady condition can be reduced by igniting a line of fire on a broad front (Cheney and Gould 1993). In fact as the width of the front increases the fire is less affected by minor fuel discontinuities and the fire spread mechanism is more efficient. Moreover the formation of a convection column on a wide front affects the wind field in the fire area; the balance between these two factors will determine the wind speed at the flaming zone and the rate of spread of the fire front.

The Annaburoo fire experiment in 1986 (Cheney and Gould 1995) used plots whose dimensions were up to 60000 m^2 (300 x 200 m) and compared effects of different line ignition length up to 75 m on the acceleration process; the experimental site was in total 2500 ha and demonstrated how fires in open grassland and forests with a grassy understorey develop different *quasi* steady rates of spread, under the same environmental conditions, depending on the effective width of the head fire. The authors suggested that this aspect of fire acceleration should be taken into account when using data from small plots to develop or validate empirical models of fire spread.

3.4.2. Heat processes

Measuring the rate of spread of the fire front (m/s), the fuel consumed (kg/m^2) and the heat of combustion (kJ/g) of the fuel (Alexander 1982) allows to calculate the *fireline intensity* (Equation 1) defined by Byram (1959) as "…the rate of energy released per unit time per unit length of fire front, regardless of his depth", that potentially is an important factor in determining the ease of suppression of a fire, fire-fighter safety as well as its ecological impact. Usually fireline intensities are classified in four groups: low intensity (< 500 kW/m), moderate (500-3000 kW/m), high (3000-7000 kW/m) and very high (7000-70000 kW/m) (Cheney 1981).

$$I = H * W * ROS \qquad \text{(Eq. 2)}$$

Where:

I = *fireline intensity* (kW/m)
H = heat of combustion (kJ/g)
W = mass of consumed fuel (kg/m^2)
ROS = rate of spread (m/s)

The *fireline intensity* has been widely used in fire ecology studies and till today it is a most useful way of describing a fire. Nevertheless its limitations have been recognized. Tangren (1976) critically reviewed the *fireline intensity* definition of being the energy generated from a line of the fire front; he stressed the fact that the energy release is not confined to the leading edge of the fire but is realised over the width of the combustion zone back behind the fire edge. Consequently in fuel beds with a large component of heavy components (e.g. coarse woody debris) more energy will be released by smouldering combustion at considerable distance behind the leading edge of the fire. Alexander (1982) in his review of the *fireline intensity* defined it as the amount of energy released by the “active” zone of the fire front interpreting W as the amount of fuel consumed by the leading edge of the fire front, that is to say the combustion of the fine fuel fraction. Anyway, Cheney (1990) suggested that Byram’s *fireline intensity* should not be used to compare fires in fuel types which are structurally different.

Another important descriptor of heat processes is the temperature reached during a fire. In fact temperatures reached below the soil surface, at ground level or in the shrub and tree’s canopies may have important effects on the vegetative recovery of plants, on the ability of seeds to germinate, on the soil micro organisms and on nutrient losses from the ecosystem (Bond and van Wilgen 1996, Neary et al. 1999). Concerning fire effects on plant tissues, 60°C is the temperature for hydrated-cell death, 100 °C there’s the desiccation of tissues, 300°C the ignition temperature, decomposition of plant materials and charring tissues, 500°C the mineralization of organic matter (Moore et al. 1995). Temperature reached in the combustion of forest fuels can reach up to 1500°C; temperatures decrease with height above the fuel bed (Sullivan et al. 1993).

Maximum temperatures and time temperature-residence profiles have been used as quantitative descriptors of fire intensity in ecological studies. Raised temperatures at ground level may not correlate directly with increased *fireline intensity* but are a product of greater fire residence time (Hobbs and Gimingham 1984a, Bradstock and Auld 1995). Canopy fire temperatures are of less ecological consequence but allow us to compare fires of different type (Alexander 1982). Measurement of fire temperature is thus an important yet difficult part of the study of ecosystems where fire is a natural factor or used in management (Hobbs et al. 1984). There are several approaches to fire temperature measurement in fire experiments.

The first uses a series of heat-sensitive materials which undergo recognizable, irreversible changes at known temperatures. Such methods give no indication of the duration of high temperatures and record only the maximum temperature reached, but they are quick to install, require no recording equipment, and are easy to interpret. The most frequently used, and probably the most convenient materials, have been heat-sensitive paints or pellets of known melting points (Whittaker 1961, Hobbs et al. 1984).

The second uses thermocouples, which provide continuous extremely accurate temperature recordings and allow the duration of high temperatures to be assessed (Kenworthy 1963, Kayll 1966, Moore et al. 1995, Atkins and Hobbs 1995, Giovannini and Lucchesi 1997, Molina and Llinares 2001, De Marco et al. 2005). Nevertheless thermocouples have limits: in a sense they are unpractical because must be connected to the recording equipment (data logger) which has to be installed and protected from the fire and deployed along a grid over the whole area to be burned; moreover, they provide only point measurements (Bradstock and Auld 1995). Finally, significant errors are possible in thermocouples temperature measurements when these devices are used in fire environment, being the magnitude of the errors directly related to the thermocouples diameter (Sullivan et al. 2003, Gujiarro et al. 2006). Several studies have combined different types of measurement to validate temperature measurements by thermocouples and found discrepancies (Hobb et al. 1984, Aranda et al. 2002, Sullivan et al. 2003, Guijarro et al. 2006).

An appealing alternative to the use of thermocouples but that has been relatively scarce in experimental fire studies is the use of infrared images (Martinez De Dios and Ollero 2005). Infrared (IR) thermography is a non-intrusive technique that is able to give temperature measurements with both spatial and temporal resolution. The fire front can be easily visualized through smoke (Figure 3-4) and a calibrated IR camera gives a measurement of the radiance emitted by each point, that can be related to its temperature (Gujiarro et al. 2006). IR cameras provide this information with very good spatial and temporal resolution and, as are well suited for field operation, have been used also to trace the advancing fire front in field experiments (Giroud et al. 1998, Marxer and Conedera 1999, Viegas et al. 2002).

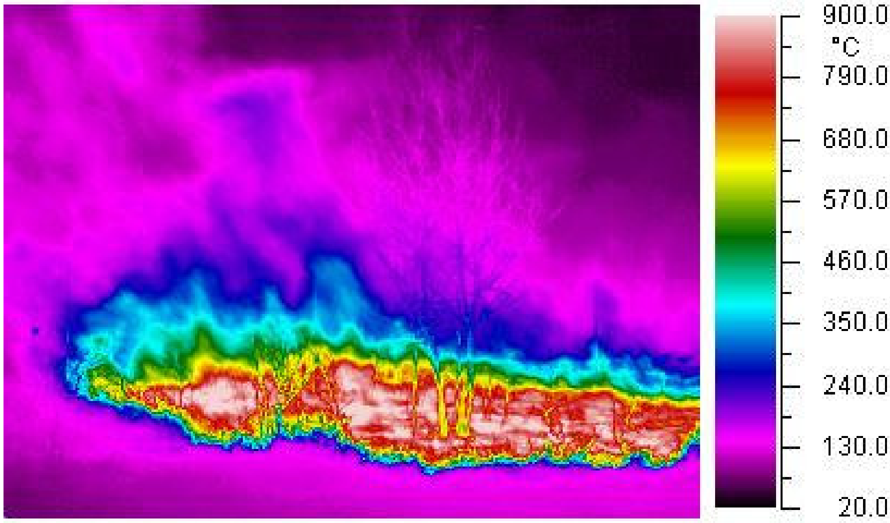


Figure 3-4 Infrared image of an experimental fire front.

However, there are some difficulties when trying to obtain quantitative measurements that have prevented IR imaging from being extensively used. The main drawback of IR cameras when applied to fire studies is related to the spectral structure of the emitted radiation. IR thermography is usually applied to solids, that are regarded as opaque blackbodies (emissivity = 1), whereas a flame is a complex target, in which several different regions (flames, fire front, embers, ashes) may be distinguished. The spatial distribution of each region changes with time, and their spectral profiles of emission may be very different between each other (Meléndez et al. 2004). If the measured spectral radiance of the flame is not corrected for emissivity, an IR camera will assume that the object is a blackbody and the temperature it will provide will be lower than the real one. This uncorrected temperature is called *brightness temperature*. On the other hand, soil and embers behave almost like a blackbody (Aranda et al. 2002). Consequently, in order to obtain reliable temperature measures from radiation measures obtained with the infrared images it is necessary to know the emissivity indexes of the flames (Meléndez et al. 2004). The emissivity of flames is extremely difficult to determine due to the complex nature of the flame environment, but it has been found that the thicker the flames, the higher the emissivity of the flames (Pastor et al. 2002, Sullivan et al. 1993). Hagglund and Persson (1974), as cited in Chandler et al. (1983), determined that 2-m-thick flames had an emissivity in excess of 0.9. However, Sullivan et al. (1993) stressed the fact that using a value for flame of emissivity equal to 1 can be satisfactory when high measurement precision of flame temperature is not required.

3.4.3. Coping with fire heterogeneity

The above mentioned descriptors of fire behaviour characteristics may vary temporally and spatially during a fire (Atkins and Hobbes 1995). Under mild burning conditions (I < 3000 kW/m), fluctuations in wind direction, non controllable and predictable during a fire experiments, are one of the major factors in determining the rate of acceleration and spread (Cheney and Gould 1995). As the wind direction near the ground is highly variable it can be seen how many different rates can be achieved under otherwise similar site conditions (Cheney 1981). For example, a surface fire in heathland ignited against the wind (backfire) is characterised by a *fireline intensity* ranging from 50 to 500 kW/m; a sudden change of wind direction can transform in few seconds the back-fire into a wind-driven fire (headfire) that manifests higher *fireline intensity* up to 4000 kW/m (Hobbs and Gimingham 1984a, Davies 2005).

Moreover, effects of heat on individual plants can vary considerably within a single fire because of localized burning conditions due to the fuel spatial distribution. For example plant mortality may have been caused by consumption of an excessive build up of leaf litter that caused prolonged downward heating around the root crowns (Smith et al. 1993).

The fire behaviour heterogeneity within a single fire experiment affects consequently the spatial pattern of fire severity along the plot (Byram, 1959, Kayll 1966, Rothermel and Deeming 1980, Cheney 1981, Alexander 1982, Atkins and Hobbes 1995); it can be said that a single fire experiment is not a single treatment but rather a range of treatments. Consequently the fire behaviour and fire severity variability within the plot must be assessed when statistical correlation between fire descriptors and effects is attempted (Smith et al. 1993).

Nevertheless in several fire experiments fuel and fire behaviour descriptors have been averaged on whole burns of relatively large areas ("macroplots"), consequently correlation and regression analysis in fire research has been often restricted to studies where numerous fires could be conducted. For example Marsden-Smedley and Catchpole (1995) used 52 experimental burns and wildfires to derive their empirical fire behaviour models for buttongrass moorland. Cheney and gould (1993) measured fire spread on 121 grass fires in order to validate models used in the Australian Fire Danger Rating System (Luke and Mc Arthur 1978).

When regression analysis is used in macroplot studies, each variable is represented by one value per burn; moreover variation within burns may not be measured. This approach has two problems: 1) it requires many burns to establish a reliable regression model; 2) it is unable to exploit measured variability within burns in examining research questions such as effects of a prescribed fire treatment (Smith et al. 1993).

Several studies have tried to cope with fire behaviour heterogeneity by measuring fire behaviour on small microplots within burns (Simard et al. 1984, Smith et al. 1993, Fernandes et al. 2000, Vega et al. 2006). When the microplot, rather than the macroplot or whole burn, is the basic sampling unit, all variables are measured on each microplot within each burn. Microplot data can describe the whole burn and its variability; in addition, they can be used in correlation and regression analyses to quantify relationships among variables. Microplot sampling on more than one burn enables the researcher to treat the burns as replicates and thus test the repeatability of relationships between fire behaviour descriptors and ecological effects indicators (e.g. plant mortality and regeneration success).

4. Fire experiments for *Calluna* heathland conservation management in NW Italy

4.1. Introduction

4.1.1. *Calluna vulgaris* Hull.

Heather (*Calluna vulgaris*, *Ericaceae* Family) is a low, much-branched shrub, hemi-spherical (Figure 4-1) except in dense stands, evergreen with minute leaves. Generally it is less than 0.8 m in height (max. 1.8 m); stems branched from the base, erect or divergent, the lowermost prostrate often rooting adventitiously. Leaves on long shoots widely spaced, about 3-4 mm, active for one season only; those on short shoots about 2 mm, closely spaced or imbricated, active for up to three seasons, usually glabrous, linear, sessile, triangular in section (Gimingham 1960).

Figure 4-1 Heather samples during flowering in August.

In April or May new short shoots begin to appear and shoots elongate. Flowers are produced singly on short stalks, preceded by one pair (or more) of leaves (prophylls) and usually two pairs of bracteoles. Flowering may commence in isolated individuals in late June, but the main flowering period is the latter half of August, continuing through the greater part of September. Seed dissemination begins in September and the bulk of the seed is shed during October and November. *Calluna* reproduction is by seed or vegetatively as a result of prostrate branches rooting adventitiously in a moist substratum.

Rooting depth is determined by soil conditions (especially the moisture regime), direction is usually horizontal in the upper 6-8 cm of the soil and oblique or vertical at lower levels. On wet peats *Calluna* roots are generally confined to depths not exceeding 10 cm, a few sometimes reaching 18 cm; the main rooting zone being limited by the height of the water table in the wettest season.

4.1.2. *Calluna vulgaris* distribution and ecology

Calluna vulgaris is widely tolerant in respect of temperature range and length of growing season, extending from latitude 36° N to 71° 5' N (Gimingham 1960). The range of water content in *Calluna* soils is wide, but growth is best where the soil is at least moderately well drained. Well known as an oligotrophic, calcifuge species it growths on soil with high C/N ratios, usually exceeding 15, and in general deficient in available nitrogen. *Calluna* soils are usually of poor fertility and low base. Approximate limits of pH at the surface for the occurrence of *Calluna* are 3.2-7.0. *Calluna* litter has a pH between 3.4 and 3.9, and its presence lowers the pH of the mineral soil immediately below.

In consequence of its wide ecological amplitude, it occurs in a wide range of plant communities, including forests, in which its role is subordinate to that of other species, and several other types of heath in which it is dominant (Gimingham 1961, 1993).

Calluna dominated heathland are widespread in oceanic or sub oceanic climatic regimes (Figure 4-2) finding their optimum in the United Kingdom, Ireland, along the coasts of NW Europe, in part of Scandinavia, Germany, Czech Republic and Poland (Thompson et al. 1995). Outside their main distribution area, *Calluna* heathlands are present in South, East Europe and Siberia (Gimingham 1960, 1961, Hobbs and Gimingham 1987, Bartolomé et al. 2005).

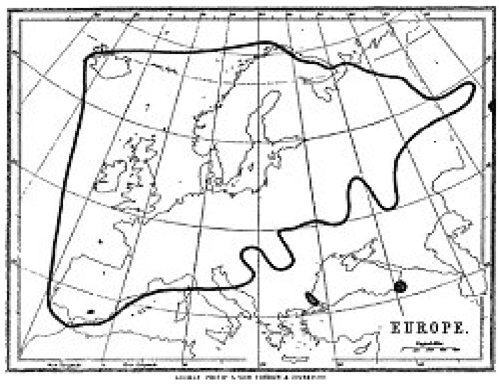


Figure 4-2 Limits of geographical distribution of Calluna vulgaris (from Gimingham 1960).

European *Calluna* heathlands developed 4000 years ago as a result of forest clearance. They have been maintained by anthropogenic disturbances resulting from the combination of traditional management practices such as burning, grazing and harvesting (Gimingham 1972, Webb 1998, Goldammer et al. 2007).

In absence of management *Calluna* will live for at least 40 to 50 years and has been seen to regenerate in a cyclical fashion passing through 4 stages defined by Watt (1955) as: pioneer, building, mature and degenerate (Table 4-1).

Stages	**Description**
Pioneer	Up to an age between about 3 and 6 years. Period of establishment and early growth. After about 2 years the leading shoot loses its identity as basal branches equal it in height, but in the pioneer phase it remains possible to recognize successive growth-zones on the shoots and thus to determine the age of the plant. Usually one winter is passed before first flowering, after which seed is set every year, but probably with periodic fluctuations in quantity.
Building	Up to an age between 15 and 20 years old, normal duration about 10 years. Maximum cover and density of canopy established. Short shoots bright green; freely flowering.
Mature	Up to about 25 years old. Central branches spread apart, permitting enough light to reach ground level for the establishment of other species. At the tips of branches shoots are more clustered and flowering zones are shorter.
Degenerate	25 years and over. Central frame branches die, leaving a gap. Peripheral branches, if more or less horizontal and rooting adventitiously, may continue active growth.

Table 4-1 Cyclical Calluna vulgaris growth stages (From Watt 1955).

In the pioneer phase (Figure 4-3) young seedlings develop or shoots re-grow from stem bases until canopy closure occurs in the building phase. Plants begin to loose vigour in the mature phase and gaps begin to open up. In the degenerate phase significant gaps form in the middle of stumps and older stems become more prostrate.

Figure 4-3 Change in heather stand structure with age: the Calluna cycle, re drawn by Davies (2005) after Watt (1955).

Watt reported that in the degenerate phase *Calluna* seedlings colonize gaps and a mixed age stand results with individual bushes showing different life phases (Watt 1955). Nevertheless Gimingham (1987), in his critical review about *Calluna* heathland ecology, documented that seedlings survival rates in gaps formed by degenerates bushes are low. In fact *Calluna* regeneration from seed has been reported to be negatively affected by its own litter accumulated along the stand growth (Mallik and Gimingham 1985); the negative feedback between plant and soil thus explain the cyclical degenerative-regenerative phases of natural *Calluna* stands in absence of management. Moreover Gimingham (1987) stressed the fact that the '*Calluna* cycle' is relevant only in the absence of any more permanent and long-lived invader, such as a tree specie, which might take other the patch. Consequently the gaps are occupied by invasive species and other species of the community for varying periods of time, thus leading to a succession towards a different structure and composition of vegetation.

4.1.3. *Calluna* heathland conservation issues in Europe

At present, heathlands are under threat because of a range of impacts: the abandonment of traditional management practices which results in a reforestation process (Thompson et al. 1995, Webb 1998); moreover, in the absence of intensified management, increased atmospheric nitrogen and sulphur deposition causes early replacement of *Calluna* by grasses (Heil and Diemont 1983, Terry et al. 2004, Britton and Fisher 2007) such as *Molinia spp.* (Marrs et al. 2004, Brys et al. 2005) and ferns (i.e. *Pteridium aquilinum*) dominated communities (Watt 1955, Snow and Marrs 1997). Changes has resulted in the considerable loss in area of these communities in all Europe. For example, Sweden and Denmark lost 70% between 1860 and 1960, the Netherlands has lost 95% of its maximum area in a similar period, England and Wales 27% between 1947 and 1980 and Scotland

18% between 1940 and 1970 (Rose et al. 2000, Pakeman et al. 2003). Because of the reduction in their overall distribution, European heathland have been classified as greatly endangered and are considered to have a high conservation value within the European Union. This value is recognised by a range of local designations or communitarian ones such as EC Habitats Directive (92/43/EEC).

Considering their economic, nature conservation, landscape, aesthetic and tourism-related values, *Calluna* heathlands are consequently one of the most intensively studied heath system in the world and their conservation management is a critical issue (Hobbs and Gimingham 1987). The restoration of traditional management practices including prescribed fire, grazing at different stocking rates, and mechanical cutting have been applied to halt the loss of this habitat throughout Central and NW Europe (Sedláková and Chytrý 1999, Niemeyer et al. 2005, Vandvik et al. 2005, Britton and Fisher 2007, Goldammer et al. 2007). *Calluna* heathlands management by prescribed fire has been studied above all in Britain and Scotland (Whittaker and Gimingham 1962, Hobbs and Gimingham 1984b, Hester et al. 1991, Pakeman et al. 2003) and has been codified in 'prescriptions' and 'burn plans' such as the 'Muirburn' Code (SEERAD 2001).

4.1.4. Moorlands management by fire in Scotland: 'Muirburning'

In Scotland 'Muirburning' (SEERAD 2001, Scotland's Moorland Forum 2003) has been the traditional form of land management for 300 years till today. Areas of *Calluna* heathlands (moorlands) are burnt in rotation in order to provide areas for nesting and forage for game species such as the red grouse (*Lagopus lagopus scoticus* Latham) and red deer (*Cervus elaphus* L.), as well as for grazing sheep (*Ovis aries* L.) (Gimingham 1960, 1993, Davies 2005). The aim of traditional Muirburning is to promote the growth of young shoots from the protected stem bases and to increase the forage value for a number of years following a fire (Gimingham 1972). In fact there is a decline in the nutritional value of the plant as it ages whilst increasing proportions of biomass are allocated to the formation of woody stems further decreasing its palatability (Gimingham 1987).

Moreover periodical burning has been traditionally used to regenerate *Calluna* heathlands thus preventing the evolution towards woodlands or grasslands (Hobbs and Gimingham 1987, Goldammer 2007).

In fact fire provides numerous benefits important to the regeneration of *Calluna* heathlands communities. On a macroscale level periodic burning favour heather by reducing woody species (Wright and Bailey 1982, Gimingham 1972, 1993). On a microscale level fire eliminates allelophatic compounds present in the litter and leaves of *Calluna* and enable germination of its own seeds otherwise inhibited (Bonanomi et al. 2005). Moreover germination of the small seed of *Calluna* is stimulated by brief exposure to light and fluctuating temperatures, both characteristic of the post-burn environment but also of other-gap creating disturbances such as mowing (Bond and van Wilgen 1996). Several studies have found that in the light, when temperatures are kept constant, the optimum for germination lies about 20 °C. However, even better results were obtained when seeds were subjected to alternation between 30° (8 hours) and 20° (16 hours). Rather few seeds germinate in the dark at constant temperatures, more in fluctuating temperatures (Gimingham 1960).

Calluna germination is also slightly stimulated by the fire heat pulse (Whittaker and Gimingham 1962). A few hours heat pre-treatment at 70-80° or shorter periods at temperatures up to about 100° may accelerate and increase germination, although adverse effects on total germination are reported with longer treatments. At 120° exposure for 30 seconds increases germination, although 60 seconds slightly depress it, while 60 seconds at 160° are lethal (Whittaker and Gimingham 1962). Thus, while a normal heath fire may kill a number of seeds at the soil surface, the germination of those which are partially protected may be stimulated (Gimingham 1960).

The temperatures reached and their duration may have also a direct bearing on the survival or destruction of the stem bases (Kayll 1965, 1966, Hobbs and Gimingham 1984a). Whittaker (1961) and Kenworthy (1963) founded that prevailing weather conditions at the time of burning are more important than age of stand in determining the temperatures produced. Slight changes in wind speed and direction, or the moisture content of plants and soil can have pronounced effects.

The structure of the stand may influence the subsequent pattern of regeneration. Even-aged stands of *Calluna*, 10-15 years old, form a dense and continuous canopy, 25-35 cm high, supported by a relatively large number of small stems per unit area. When burnt, the fires run quickly, removing the crowns and most of the stems, leaving a clear burn (Whittaker 1961). The large number of stems per unit area provide a large number of loci for sprouting, and other factors being favourable, good generation results. With increasing age, *Calluna* develops a different growth-form. In the late mature (*sensu* Watt 1955) and degenerate phases (from about 20-25 years) the central branches of a plant spread apart and at the branch tips shoots are more clustered (Gimingham 1960). Thus the spatial distribution of the fuel for a fire is discontinuous and the burn is irregular and uneven. Surface temperatures produced by fires in such a community will be low (Kayll 1966). Stems are frequently only superficially charred and litter may not be burnt, especially on wet ground. Under these conditions vegetative regeneration, if it occurs, tends to be clumped at old plant centres rather than developing from a denser and more even distribution of buried stem bases.

Gimingham (1960) suggested that the age of the plants before burning is also important as their capacity to regenerate vegetatively tends to decline beyond the age of about 15 years. Younger individuals normally regenerate abundantly from the stem bases, especially when there has been extensive branching below and the lower parts of the stems lie in the soil surface, or buried in moist litter. Such regeneration gives rise to a densely bushy growth form, in contrast to the taller, more tree-like form of plants grown from seed. Under favourable conditions, cover may be restored in 2-3 years.

Most authors agree that the chances of successful recovery are greatest when plants are aged less than 15 years (Kayll and Gimingham 1965, Gimingham 1993, Davies 2005). Consequently current advice for dry moorland is that heather should be burnt on a roughly 15 year rotation or when about 30 cm tall (Scotland's Moorland Forum 2003) though this should vary according to productivity, habitat type and exploitation level (Gimingham 1993). Research suggests that on many moors the rotation may, in fact, be significantly longer (Hester and Sydes 1992). Burning is most often carried out in early spring (March-April), but autumn burning is also practised although plants may be more liable to be killed at this time of year. *Calluna* will recolonize from seed when killed out by fire, but full renewal of cover may require 6 or more years, and may be preceded by the temporary dominance of other species.

The burning practice consists in a series of small fires on long thin strips distributed across the land to form a mosaic of adjacent areas burnt in different years. Moreover some portion of land must be left unburnt in order to maintain the habitat for other plant species not adapted to fire thus preserving higher biodiversity (Gimingham 1993). Consequently Muirburning maintains a patchwork of all *Calluna* stand stages (Figure 4-4). The reasons behind this burn pattern lies in the territorial behaviour of red grouse which are favoured by a mixture of heather ages within each 1 ha territory (Palmer and Bacon 2001): while it grazes on young nutritious shoots of the pioneer and early building stages it also requires taller heather in which to nest and hide from predators.

Figure 4-4 Traditional muirburning is to maintain a patchwork of different stand stages of heather by burning each year a series of long thin strips (SEERAD 2001).

The burning technique consists in lighting a fire front along a strip, no more than 30 m across, by using a paraffin oil burner and a 'broom' to control its edge (Figure 4-5). Each year 5-10% of the *Calluna* area of a moor is burnt between 1 October and 15 April (Kayll 1966, Gimingham 1993).

Figure 4-5 Muirburning practice in Scotland using brooms to control the fire front and flanks on strips no more than 30 m across (from SEERAD 2001 and Davies 20005).

4.1.5. *Calluna* conservation management in NW Italy

In Italy *Calluna* heathlands are relic ecosystems, mainly located in isolated areas on the Po River Plain which border the foot of the Alps, where forest removal combined with local climatic conditions and acidic soils, favoured the establishment of heathlands belonging to the *Calluno-Ulicetea* phytosociological class (Sindaco et al. 2003, Mugion 1996). Locally heathlands are known with different names: Vaude and Baragge in Piemonte Region and Brughiere and Groane in Lombardia Region (Mugion 1996). In this thesis we refer to Piemonte *Calluna* heathlands (Vaude).

Fire and grazing have been an important evolutionary factor in the development of Vaude in NW Italy. In the past, fire was one of the common management techniques applied by shepherds to reduce the proliferation of trees and to maintain pastures for livestock management. Nowadays, as a consequence of social and economical changes in the area, both practices are marginal (Borghesio 2004). Modifications in land use and management have changed the fire and grazing regimes resulting in a process of transformation into new communities with a lower inherent biodiversity. These changes are similar to those described in the Atlantic area with trees and grass encroachment (Borghesio 2004).

Despite the extensive literature concerning *Calluna* heathlands conservation management in their main oceanic distribution area, very little work has been done in South Europe (Valbuena et al. 2000, Calvo et al. 2002, Bartolomé et al. 2005) and even less in Italy. Information about their sensitivity to fire regimes, grazing pressure, land-use and climate changes is very scarce.

In NW Italy remnants heathlands have been included in Managed Nature Reserves (MNR) in order to safeguard them from deterioration and preserve their values. Nevertheless, suitable conservation management plans based on sound scientific studies do not exist yet (Garbarino et al. 2005).

4.1.6. *Calluna* conservation management at the MNR of Vauda

The MNR of Vauda (7°41’ 17” E, 45°13’ 13” N), 2635 ha in area, 20 km N-NE of Torino (Figure 4-6), has been established in 1993 to preserve the naturalistic and landscape characteristics of one of the most important and extended *Calluna* heathlands of NW Italy.

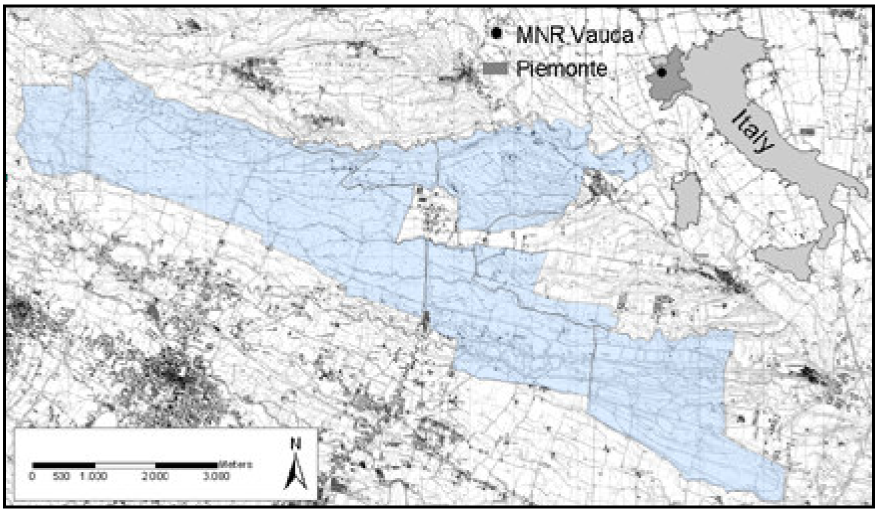


Figure 4-6 The Managed Nature Reserve of Vauda boundaries.

The Vauda boundaries coincide with a military firing ground of the Italian Army that spared the MNR from urban and agriculture development for at least two centuries. Despite military rights, shepherds were allowed to mow during the haymaking time and lead cattle to pasture. Similarly to NW Europe heathland management (Gimingham 1972, Webb 1998, Goldammer et al. 2007) fire was applied by shepherds to reduce the proliferation of trees and to maintain pastures. Periodical burning in winter and subsequent grazing and mowing have selected and maintained heathlands structure and composition.

Nowadays, modifications in land use and management have changed the traditional fire regime while grazing and mowing practices in the last 40 years have been extensive or absent. The loss of the traditional fire use knowledge, lead to uncontrolled late winter fires (February-March), with some sites burnt every year under extreme fire weather and others where fire has been completely excluded (Mugion 1996). From data archives (Data Source: Regione Piemonte) there are evidences of wildfires which interested the MNR from early 90s till today for a total burnt area of 1867 ha (71% of total Reserve area). Figure 4-7 shows the perimeter of the wildfire events with an area superior to 10 ha which occurred inside the MNR of Vauda boundaries from 2001 to 2006. The data archives document a high frequency fire regime in the West side of the Reserve (S. Carlo Canavese municipality) while other parts have not been interested by fire in the last 6 years.

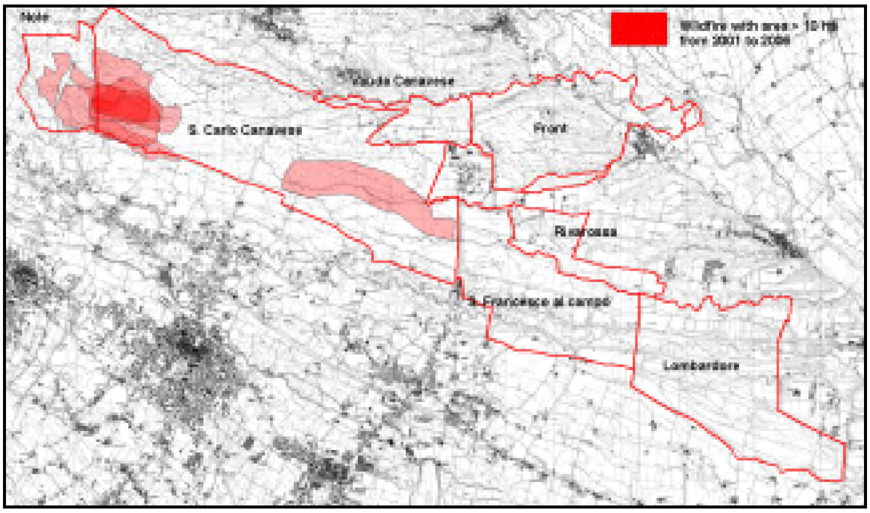


Figure 4-7 The MNR of Vauda boundaries divided per municipality and perimeters of all wildfires (>10 ha) from 2001-2006 as reported in official documents of the Regione Piemonte.

The changed fire and grazing regimes at the MNR of Vauda is resulting in an ecological shift adverse to *Calluna* heathlands (Borghesio 2004). Less frequent disturbances enable species like European aspen (*Populus tremula* L.) and Birch (*Betula pendula* Roth) to form large, dense stands of juvenile trees that secondarily develop into woodlands with the establishment of English oak (*Quercus robur* L.) and other species of the climate-limited potential vegetation (*sensu* Bond 2005). Repeated burning causes the transformation of heathlands into grasslands dominated by Tall moor grass (*Molinia arundinacea* Shrank) (Mugion 1996). Consequently, *Calluna* heathlands are naturally developing into woodlands or grasslands with the risk of losing the high biodiversity of these ecosystems (Mugion and Martinetto 1995, Cattaneo and Biddau 2000, Borghesio 2004, Regione Piemonte 2004). In fact, among the dense agricultural fabric and the urban development of Torino and Milano cities, two of the major metropolitan areas of NW Italy, remnants heathlands constitute an important shelter for several endangered plant and animal species (Mugion and Martinetto 1995, Cattaneo and Biddau 2000, Borghesio 2004, Regione Piemonte 2004). Moreover, Vaude represent a cultural landscape and one of the sights and attractions of Piemonte Region.

A recent study promoted by the MNR land managers in the course of the European project Interreg IIIA (Regione Piemonte 2004) defined the species richness by habitat and censed more than 600 plant species, enlightening the naturalistic importance of the Reserve in housing several national endangered species (*Gentiana pneumonanthe, Salix rosmarinifolia, Ranunculus flammula, Eleocharis carniolica, Juncus bulbosus, Achillea ptarmica, Scutellaria minor, Rhynchospora fusca, Carex hartmanii*) (Figure 4-8) and rare species connected to wet sites located in the heathland matrix (*Campanula bertolae, Diphasium tristachyum, Eleocharis carniolica, Juncus tenageja, Lythrum portula, Veronica scutellata*).

Figure 4-8 Salix rosmarinifolia (left) and Ranunculus flammula (Right).

Moreover the Reserve represents one of the few strips of land for birds nesting. Cattaneo (1990) registered 123 birds species. A more recent study of the same author (Cattaneo and Biddau 2000) reported a decrease in the number of individual for species adapted to heathland (*Alauda arvensis, Saxicola torquata, Hippolais polyglotta*) while others disappeared (*Luscinia megarhynchos*, *Emberiza citrinella*, *Lanius coriullo*).

The loss of animal and plant species is a serious threat to a protected area. In continuity with the previous Interreg project (Regione Piemonte 2004) a long-term and multidisciplinary experiment has been established to determine how prescribed fire, grazing and mowing can be restored to provide the greatest benefit for *Calluna* heathlands. The objective of the research study is to provide the land users with a suitable management plan for conservation management of *Calluna* heathlands inside the boundaries of the MNR of Vauda.

This Chapter deals with the fire research program carried out at Vauda from 2004 to 2007 in collaboration with the Research Group in Grazing and Land Management Unit of Dip. Agroselviter of the University of Torino (www.unito.agraria/pastralismo.it), the Italian Forest Service (Corpo Forestale dello Stato - here after CFS; www.corpoforestale.it) and the Volunteer Fire Brigades Corp of the Piemonte Region (Corpo Volontari Antincendi Boschivi della Regione Piemonte - here after AIB; www.corpoaibpiemonte.it).

4.1.7. Fire research issues and objectives at Vauda

The fire research program at Vauda studies the effects of fire behaviour and fire frequency on heathland vegetation in order to identify the suitable prescribed fire regime to regenerate *Calluna* stands and to control tree encroachment by fire. Moreover, it aims at characterizing the fire behaviour in heath and grass fuels in order to establishes the operative limits in applying prescribed fire treatments. In this section the fire research issues that drive the experimental design at Vauda are outlined.

1) *Fire season*: prescribed fire treatments at Vauda are applied only in late-winter season. This choice is due to a range of grounds. Evidences show that shepherds used fire in winter to eliminate dry biomass and stimulate plant production during growing season (Mugion and Martinetto 1995, Borghesio 2004). Moreover, climate constraints do not enable to test early-winter (December-January) versus late-winter fires (February-March) as the beginning of winter is characterized by higher rain and snow precipitations. Finally prescriptions provided by the Fire Management Plan in Piemonte Region allow to operate only in the dormant season in winter (Regione Piemonte 2007).

2) *Fire frequency*: two different fire return intervals are studied at Vauda. The first is a yearly fire treatment which enable to study the effect of a high frequency fire regime which is characteristic of some parts of the Reserve. The second fire treatment is studied following an “adaptive” approach (Holling 1978): it does not have a fixed fire return interval; in fact it is not yet know how many years are needed for a *Calluna* stand, which experienced a fire event, to reach a structure and composition that satisfy the management objectives at Vauda and that are resilient to a new fire treatment. In Scotland current practice of

Muirburning in heathland is that *Calluna* is burnt on a roughly 15-20 years rotation (Scotland's Moorland Forum 2003, Davies 2005). Prescriptions say that no fire should be used before the *Calluna* reaches a mature stand (*sensu* Watt 1955) with an average height of 20-30 cm. This may take 8-10 years in the most productive situations and up to 20-25 years in the least productive areas (SEERAD 2001). No references are available to compare *Calluna vulgaris* growing capacity in NW Italy with the one showed in Scotland or in other European sites, nevertheless these information can be a useful reference point.

3) *Fire behaviour*: fire experiments at Vauda have been designed to study at a microplot scale fire characteristics and their effects on vegetation (Smith et al. 1993, Fernandes et al. 2000). This approach is required to cope with fire behaviour heterogeneity. In fact during a fire experiments the fire behaviour varies because of wind speed and direction changes, micro-relief, fuel distribution and moisture; this variability affects the spatial pattern of fire severity (Kayll 1966, Cheney 1981, Hobbs and Gimingham 1984a, Atkins and Hobbs 1995). Moreover, choosing different ignition patterns it is possible to have contrasting fire behaviours: low *fireline intensity* fires in heathlands, ranging from 50 to 500 kW/m, can be attained by igniting fire against the wind (backfire); instead a wind-driven fire (headfire) manifests higher *fireline intensity* up to 4000 kW/m (Davies 2005) and temperatures at ground level may reach 800 °C (Whittaker 1961, Kayll 1966, Hobbs and Gimingham 1984a). This contrasting fire behaviours could have different effects on *Populus* and *Betula* mortality on one side and *Calluna* resprouting capability on the other and consequently must be studied carefully, in order to asses their ecological effects, their management implications and finally set specific 'prescriptions' for *Calluna* conservation by prescribed fire (Pyne et al. 1996).

4.2. Material and methods

4.2.1. Study site

The MNR of Vauda is located at an altitude from 240 to 480 m on a stream terrace plateau formed in the Late Pleistocene by the Stura di Lanzo River, and dissected by the major creeks which form the western (Torrente Fisca) and the eastern (Torrente Malone) boundaries (Mugion and Martinetto 1995). On the plateau soils are acidic, rich in silt and clay and classified as Typic Fragiudalf (Soil Taxonomy), with a loamy subsurface horizon of high bulk density (fragipan) that makes this soil poorly drained and frequently water-logged until water is eliminated by evaporation and evapotranspiration.

The major vegetation associations are characterized by different stages of the succession towards oak-hornbeam forest (*Carpinion* Issl. 31 em. Oberd. 53), ranging from open heathland dominated by *Calluna vulgaris* and *Molinia arundinacea* Schrank (tall moor-grass) to aspen-birch thicket stands, on the stream plateau and woodland on the creek slopes.

The climate of the area is continental, with about 81% of the mean annual rainfall (1000-1100 mm) falling between April and November (Figure 4-9). The driest month is March with 35 mm of rain and 0.3 days of snow. Mean annual temperature is 11.8 °C, with monthly means ranging from 1.6 °C in January to 21.9 °C in August (Nimbus 1993-2004; http://www.nimbus.it).

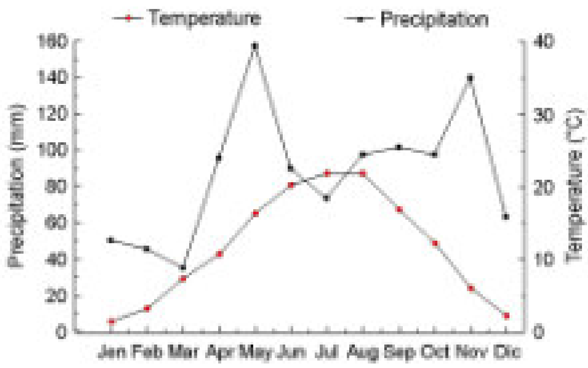


Figure 4-9 Monthly temperatures and precipitation at Vauda (Nimbus 1993-2004).

4.2.2. Experimental design

The experimental design of the Vauda project aims at testing the effects on vegetation of several treatments which apply the sole prescribed fire, grazing or mowing and their factorial combinations. All treatments are repeated in four sites. A control area (no management) is also present in each site.

The 4 experimental sites, 1 ha each (Figure 4-10), have been placed in the South East side of the MNR of Vauda, within the administrative boundaries of the Lombardore municipality (Figure 4-7). Sites were characterized by homogeneous *Calluna vulgaris* stands in the building phase (*sensu* Watt 1955) with *Molinia arundinacea* commonly occurring and different stages of tree encroachment. The choise of the 4 experimental sites was also conditioned by the existence of roads to facilitate control and mop-up operations during fire treatments.

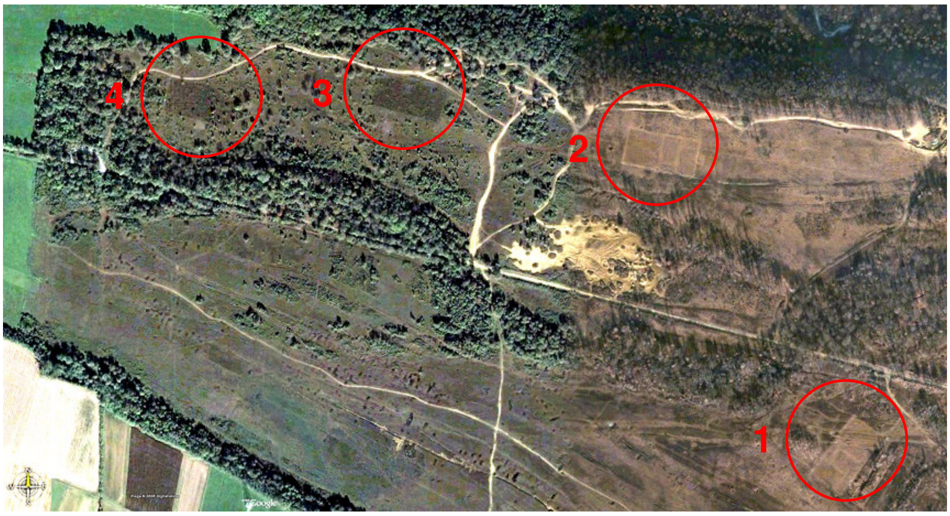


Figure 4-10 Aerial image showing the 4 experimental sites at Vauda (Google Earth)

According to the sole fire research program, each site was partitioned into several experimental units ranging from 625 to 4000 m^2 in which the following treatments have been applied from winter 2005: 1) backfire each N. years; 2) headfire each N. years; 3) headfire each year; 4) no treatment (control).

Treatment 1 and 2 test the effect of two contrasting fire behaviours: back versus headfires in order to assess their management implications; treatment 3 tests the effects of high frequent fires repeated each year which approximates the wildfire regime documented in some parts of the MNR of Vauda; treatment 4 enables to monitor the evolution of vegetation in absence of management actions.

4.2.3. Experimental sites fire history

The fire history of the experimental sites is controversial. In all sites there are evidences of past wildfire occurrence such as dead trees, scorched barks on survived trees or stump resprouting. The time since fire has been supposed to be equal to sprouts age determined by tree ring counting. In each site 10 *Populus tremula* and *Betula pendula* sprouts with the largest diameter were selected. Sprouts in site 1, 3 and 4 were 15 years old in 2004, thus leading to believe that a wildfire occurred in 1989-1990. Sprouts in site 2 were 6 years old, meaning a fire event in 1998. Nevertheless wildfires data archives do not gives evidence of such fire events within the administrative boundaries of Lombardore municipality. In the same years large wildfires occurred inside the MNR of Vauda boundaries in areas belonging to other municipalities surrounding the experimental sites. For example from 1997 to 1999 a total area of 262 ha (10% of the entire Reserve Area) were burnt in the neighbouring municipality of S. Francesco al Campo. Consequently these wildfire could have affected also the experimental sites.

4.2.4. Fuel sampling

The importance of modelling fuel characteristics for fire ecology studies, which aims at correlating fire behaviour effects on vegetation, it was evidenced in Chapter 3. Given the lack of previous studies on fuel characterization in *Calluna* heathland of NW Italy, it was chosen the destructive sampling. The sampling and fuel characterization methodology used that allow an accurate analysis of the fuel complex found its reference in several studies (Brown et al. 1982, Burgan and Rothermel 1984, Andrews et al. 1986, Camia 1994). Fuel load, moisture and arrangement were estimated harvesting fuel components inside the plot by: i) one week before each fire; ii) on the dates of the fire experiments; iii) the day after each fire event.

i) Sampling one week prior to burning allowed to estimate fuel load and to evaluate the variability of fuel moisture between samples in order to plan a quick assessment of fuel moisture on the burning date. In each plot several cluster of sampling areas (Figure 4-11), were positioned randomly in a variable number according to plot size (6 samples in 4000 m^2 plots; 5 samples in 1250 m^2 plots; 3 samples in 625 m^2 plots). They were used to estimate fuel components such as coarse woody debris (CWD), live fuel with a diameter major to 0.6 cm and the fine fuel component (Ø < 0.6 cm).

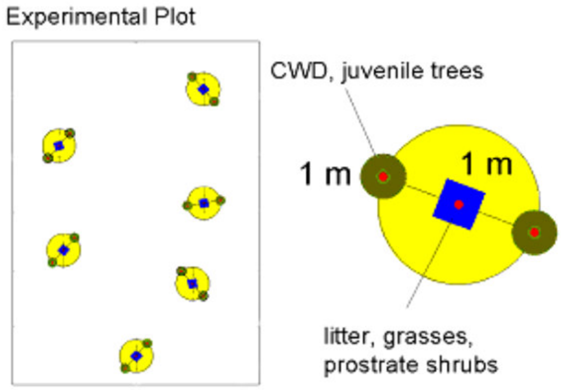


Figure 4-11 Sample description: the centre of the yellow circle is randomly selected along the plot. In the green circles, CWD and juvenile trees (Ø > 0.6 cm) are harvested; in the blue quadrate (1 x 1 m) litter, grasses and prostate shrubs (Ø < 0.6 cm) are clipped.

For each sample the fuel complex was divided into 3 components:

a) Litter, Grasses (*Molinia*) and prostrate shrubs (*Calluna*) were harvested in a square (1 x 1 m) (Figure 4-12). Harvested fuel was subsequently divided in dead fuel (litter, cured *Molinia*) and live fuel (*Calluna* stems and shoots). A further separation in size classes of the dead and live fuels was not necessary as only a few amount had dimensions higher then 0.6 cm; consequently all the fuel was included in the fine fuel fraction. The duff layer was not sampled.
b) Coarse woody debris (CWD), with a diameter superior to 0.6 cm, was picked in two circles (1 m in diameter). This method was preferred to the line intersect method (Van Wagner 1968, Brown 1976) as the diameter of the woody material was lower then 10 cm.
c) Live juvenile trees, with a root collar diameter between 0.6 cm and to 2.5 cm, were harvested in the same circles.

Figure 4-12 Collection of litter, grasses (Molinia), prostrate shrubs (Calluna).

All the dead and live fuels collected were stored in plastic bags, weighed the same day at the laboratory, oven-dried at 100°C for 48 hours and then reweighed to calculate percent fuel moisture (%) variability and fuel load (tons/ha) on an oven-dry weight basis.

Calluna stands and *Molinia* tussocks height were estimated per each plot with a linear intercept method (1 measurement each 20 cm) (Wilson 1963) along a transect placed in the middle section of the plot. Average height of the total fine fuel component was calculated as the mean of the heights of the dead and live fine fuel weighed on the basis of its relative contribution to the total dry fine fuel load (Equation 3) (Burgan and Rothermel 1984, Camia 1994):

$$\delta = \sum_{i=1}^{4} \delta_i \frac{w_i}{w_o} \tag{3}$$

Where:

δ = height of the fine fuel component
δ_i = height of the *i* component (dead or live)
w_i = dry fuel load of the *i* component
w_o = total dry fuel load of the fine fuel component

Bulk density of the fine fuel component was calculated dividing the dry fuel load per unit area of the dead and live component per its height (Equation 4). Average bulk density of the fine fuel has been calculated averaging the bulk densities of the dead and live components weighed on the basis of their relative contribution to the total dry fine fuel load (Burgan and Rothermel 1984).

$$\rho_b = \frac{w_o}{\delta} \tag{4}$$

Where:

ρ_b = bulk density (kg/m3)
w_o = dry fuel load of the fuel component (kg/m2)
δ = height of the fuel

ii) On the date of each fire experiment, just before fire ignition, only fine fuels were sampled by harvesting 3 (25 x 25 cm) squares. Fuel samples were subsequently divided in live and dead fuel, weighted with a field balance as soon as collected, stored in plastic bags and oven-dried the same day at 100°C for 48 hours and then reweighed to obtain fuel moisture.

iii) The day after fire experiments the residual or un-combusted fine fuel was collected in 6 (1 x 1 m) squares positioned close to the areas for before burning sampling. All the samples were stored in plastic bags, weighed the same day in the laboratory, oven-dried at 100°C for 48 hours and then reweighed to calculate fuel load on an oven-dry weight basis. Fuel consumption (t/ha) was determined by subtracting from before-fire total dry fine fuel load and residual total dry fine fuel load sampled the day after fire.

4.2.5. Burning procedures

Plots were line ignited along the shorter side of each plot. Fireline ranged from 25 to 50 m; some were ignited along the windward side and others along the downwind side, so as to perform headfire and backfire, and therefore cause contrasting behaviour of the fire front and effects. Nevertheless there was a variability of wind direction resulting in frequent changes in fire behaviour within each fire experiment.

Precautionary measures against wildfire included a 3 meters firebreak around the perimeter of each plot; moreover, plot location took into account the presence of natural firebreaks such as roads, wet soils or discontinuities in the fuel cover. The experiments were performed with the support of Corpo Forestale della Stato and Squadre Volontari Antincendi Boschivi della Regione Piemonte teams.

4.2.6. Weather data collection

During the fire experiments relative humidity, air temperature and wind speed and direction at an height of 4 meters were recorded during each burning using a mobile weather station of CFS placed near the edge of the plots. Measurements were registered each 5 seconds from the start of the experiment. Surface wind speed was recorded at 2 meters height with two digital cup-type anemometer. The anemometers were positioned upwind, close to the ignition line, but away from indraft influences of the fire. Measurements were registered each 5 seconds by 2 operators and their spatial and temporal variability was measured.

Sullivan et al. (2001) stressed the fact that the actual wind affecting the fire front is immeasurable because its value must be estimated from remote anemometry. In the Vauda experiments it was assumed that the wind speed and direction measured per each time interval by the remote anemometers were correlated with the spread of the leading edge of the fire. It is assumed also that during the time lag that is necessary to the wind to move from the anemometer to the fire front the wind field remain stationary. This assumptions may work well for grasslands, shrublands and open woodlands (Cheney and Gould 1993) but their applicability to forests is misleading (Sullivan et al. 2001).

4.2.7. Fire spread estimation

Rate of fire spread (ROS; m/s) and its variability within the plot were estimated at a microplot scale by timing the time of arrival (T1) of the fire front to 2 m high metal rods positioned within the fuel bed at the intersection points of a regular grid. The number of rods and grid spacing varied according to plot size. For example, in 4000 m^2 plots, 4 x 7 rods were placed so as to have 4 lines (15 meters one from each other) of 7 rods parallel to the longest side of the plot (Figure 4-13). Along each parallel line the 7 rods define 6 segments each 12 m long. As the fire front was lit at the shortest side of the plot and it was assumed to move parallel to the longest side, the length of each segment divided by the time took by the fire front to cover the distance of that segment ($T1_{i+1} - T1_i$) was equal to the average ROS that the fire front had along the segment. Consequently this method enabled to measure 24 values of ROS per each plot. Moreover it enabled to distinguish between ROS of head (central segments) and flanks (lateral segments) of the fire front. In 1250 m^2 and 625 m^2 plots 3 x 6 rods (15 segments) and 3 x 3 rods (6 segments) were placed respectively.

The residence time were estimated by timing the moment at which the active flame zone of the fire front passed the marked rod (T2). The difference between $T1_i$ and $T2_i$ was assumed to be equal to the residence time.

Three observers moving freely around the advancing fire front took with a stopwatch T1 of the fire to each marked rod; the three estimates of T1 per rod (if available) were then averaged to have a reliable value for each rod. Other 3 observes took the T2, taking care to distinguish the active fire front and burning of larger pieces of fuel (Rothermel and Deeming 1980). The three estimates of T2 for each rod (if available) were then averaged.

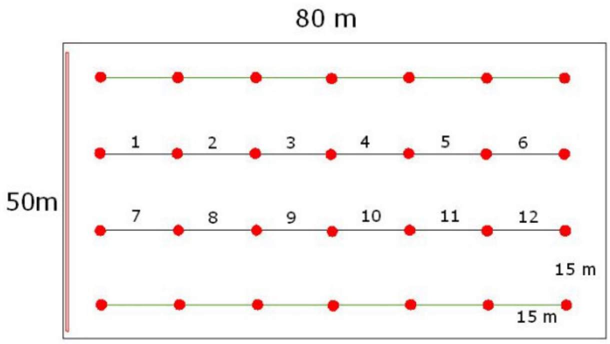


Figure 4-13 Regular grid of marked rods along plots of 4000 m^2 (50 x 80 m); central segments (black) are enumerated (1-12); flank segments are evidenced in green; ignition lines is evidenced in red.

As the rods had height increment markers positioned each 0.5 meters up to an height of 2 meters, visual estimation of flame geometry was possible all along the plot using the rods as a reference: 3 observers visually estimated height from the ground surface to the top of the main flame, and angle between the frontal edge of the flame and the horizontal. Height was estimated to the nearest 0.5 metre. Angles were visually estimated to the nearest 15°, with vertical flames being assigned 90° (Figure 4-14).

Figure 4-14 Back fire (left) and head fire (right) passing a marked rod.

4.2.8. Heat processes characterization

Fireline intensity (I; kW/m), was calculated at a microplot scale (one value per each segment) applying the Byram's (1959) equation (Equation 6):

$$I = H \times W \times ROS \qquad (6)$$

Where:

H = low heat of combustion (kj/kg)

W = fuel consumption (kg/m2)

ROS = rate of spread (m/s).

High heat of combustion (Byram 1959, Alexander 1982) of *Calluna vulgaris* and cured grasses (mainly *Molinia arundinacea*) was determined by Gillon et al. (1997) equal to 22.940 kJ/kg and 17.760 kJ/kg respectively. The low heat of combustion of each fuel component (H) was obtained by initially reducing the high heat of combustion of 1263 kJ/kg, for the latent heat absorbed when the water of reaction is vaporised (Byram 1959), and then a second reduction of 24 kJ/kg per moisture content percentage point (Alexander 1982).

Fuel consumption (W; kg/m^2) was determined by subtracting average residual fuel load from before-fire fuel load. Residual fuel load was estimated the day after fire in each plot with 6 destructive samples (1 x 1 m) per plot. As fire was mainly sustained by the dry fine fuel component, glowing combustion and smouldering were almost absent. Consequently it was possible to estimate accurately the fuel consumed in the active combustion zone necessary to measure the *fireline intensity* (Byram 1959, Cheney 1981, Alexander 1982).

Fire pick temperatures were measured with pellets[1] made with materials melting at temperatures ranging from 38°C to 954°C. Sets of 20 pellets each (Figure 4-15), 10 cm long, were positioned close to the rods of the central line at an height of 10 cm and at 1 m height, to have peak temperature between 0-10 cm and 90-100 cm respectively. Moreover, thermo labels which change colour when temperatures, ranging from 38°C to 84°C, are reached, were buried 1 cm below soil surface in correspondence of each rod of the central line to measure peak temperatures which could affect the root collar meristems and buds on surface roots of heather and trees.

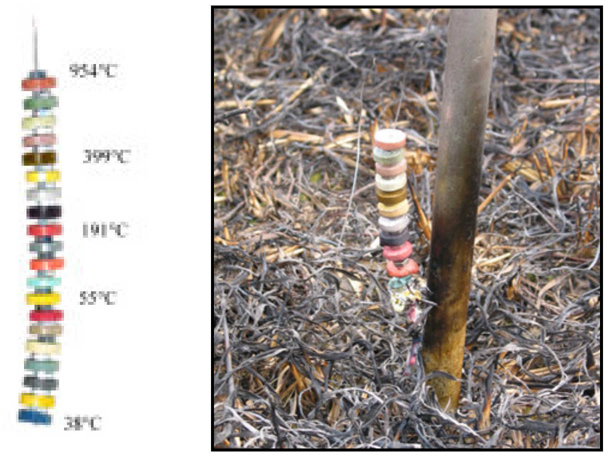


Figure 4-15 A set of 20 pellets melting at different temperatures ranging from 38°C to 954°C. Effects of a light fire in wet fuel: peak temperature (0-10 cm) up to 190°C.

[1] Omega Pellets:; http://www.omega.com/Temperature/pdf/STK_PLT_LAQ.pdf

Several experiments were filmed with an Infra Red Thermo Camera set with an emissivity (ε) equal to 0.98, placed at an height of 2.5 meters (Figure 4-16), in order to estimate average temperature residence time profile (TRP) of head and back fires. Once the camera was activated, it was able to film by itself, at 10 seconds fixed intervals, the moving fire front passing trough the visual field of the camera. The thermo photos sequences were processed with the Thermography Explorer 4.5 of GORATEC Technology GmbH & Co KG (http://www.goratec-engineering.de). This software enables to visualize thermo photos and to obtain data and graphs about the spatial distribution of temperatures at fixed point on the photos.

Figure 4-16 Infra Red Camera filming a head fire temperature residence profile.

4.2.9. Photo and video documentation

All the photos showed in this thesis were taken by a photographer charged with documenting field measurements and fire behaviour during each fire experiment. Moreover, experiments from 1 to 8 were filmed both by a video camera positioned at an height of 3.5 m on the top of the meteorological station of CFS, then by two professional cameramen instructed about relevant operative and scientific issues to film. A number of 9 hours of filming led to the editing of two film documents: a 1 hour documentary with all the experimental burns showed and additional interviews to researchers, CFS and AIB teams, to be used for didactical purposes; a 5 minutes short video, given in attachment

(Att.1) that synthesizes fire research issues such as fire behaviour heterogeneity and fire control, to be used to divulgate the research project at seminars and conferences and to link to the research group web site (http://www.agroselviter.unito.it/pianificazione/vaudavideoENG.htm).

4.2.10. Vegetation survey

In order to correlate fire behaviour and effects on vegetation, pre and post-fire monitoring of vegetation structure and composition was carried out along fixed belt trasects (2 x 10 m long), distributed randomly within each plot in a number ranging from 2 to 10 according to plot size.

To compare heathland pre and post-fire sward spatial structure and species frequency the Vertical Point Quadrat method (Wilson 1963) was used. It consisted in counting along the line transect all the species that touch a wire positioned each 10 cm inside the vegetation. The specific contribution (CS%) of each specie to the sward composition was then computed (Equation 6).

$$CS = \frac{FS_i}{\sum_{i=1}^{i=n} FS_i} \times 100 \qquad (6)$$

Where:

FS = number of times a specie touch the wire

n = number of species found along the transect

The CS% is directly related to the percentage cover of each specie, consequently was applied to monitor and compare its competitiveness in a pre and post fire environment.

To study tree responses to treatments, pre and post fire structure and composition of the stand were also assessed along the belt trasects. Per each individual tree have been measured: relative coordinates along the transect, root collar diameter (Dc), diameter at breast height (dbh), height (h), crown insertion height (hc) and incidence of the crown area (Ca). These measures enable to calculate structural indices such as Basal Area (G), symmetry (Pearson symmetry) of tree distribution per diametric classes (S) and total crown area incidence (C). During the growing seasons after treatments crown and stem mortality, sprouting capability (sprout number and dimensions) and number of dead plant (i.e. top-killing and failure in resprouting) were assessed.

4.2.11. Statistical analysis

Different statistical analysis have been used according to the nature of data using SPSS 13.0 software.

i) The t-test analysis has been used any time it has been necessary to test if significant differences ($p<0.05$*; $p<0.01$**; $p<0.001$***) existed between 2 groups of independent samples (i.e. Backfire versus Headfire effects; burnt 1 time versus burnt 3 times).

ii) The Analysis of Variance and the Tukey-Kramer HSD post-hoc test has been used any time it has been necessary to test if significant differences ($p<0.05$*; $p<0.01$**; $p<0.001$***) existed between more then 2 groups of independent samples (i.e. Differences in species structure and composition between sites).

iii) Regression and correlation analysis has been used to study ROS empirical models as a function of fuel and weather variables. This has enabled to characterize contrasting fire behaviours and identify *fireline intensity* classes. For all models a value of $R^2 > 0.6$ has been considered satisfactory.

iv) Logistic regression (Quinn and Keough 2002) have been used to study stem mortality as a function of root collar diameter and *fireline intensity* classes. In this regression model the stem mortality is considered a binary variable (1 = dead; 0 = live) and the regression will model the probability that Y (probability of plant mortality) is equal to 1 (i.e. 100% probability of depth) for a given value of X (Root collar diameter) both for back then headfires.

4.3. Results and discussion

The results are here exposed according to the following logic steps: firstly the vegetation structure and composition, how appeared in 2004 before starting the experimentation, are characterized for each one of the 4 experimental sites. A second step illustrates the fuel modelling and fire behaviour microplot analysis of all the fire experiments carried out at Vauda from 2005 to 2007, with few anticipations about their effects on vegetation as this is required for a clearest exposition. Finally a third step reports a deeper analysis of the ecological effects of backfire versus headfires and high frequent fires (burnt 3 times) versus low frequent fires (burnt 1 time) on tree mortality and *Calluna* regeneration.

4.3.1. Vegetation structure and composition before burning

Before burning, vegetation in the 4 experimental sites was characterized by a *Calluna* heathland dominated by *Calluna vulgaris* stands in the building phase (*sensu* Watt 1955) and *Molinia arundinacea*. Vegetation composition is reported in Table 4-2. The analysis of variance and the Tukey post-hoc test evidenced that the average specific contribution (CS%) of *Calluna* and *Molinia* in site 2 was significantly different ($p<0.05^*$) from the other 3 sites, with a lower presence of heather. Other grass, herbaceous and shrub species where also present but with a CS% significantly lower comparing to the two dominant species in all sites.

Site 1		Site 2		Site 3		Site 4	
Specie	**CS**	**Specie**	**CS**	**Specie**	**CS**	**Specie**	**CS**
Calluna vulgaris	49	*Calluna vulgaris*	22	*Calluna vulgaris*	43	*Calluna vulgaris*	36
Molinia arundinacea	34	*Molinia arundinacea*	59	*Molinia arundinacea*	38	*Molinia arundinacea*	44
Carex panicea	5	*Salix rosmarinifolia*	5	*Carex tumidicarpa*	3	*Peucedanum sp.*	8
Genista tinctoria	4	*Potentilla erecta*	3	*Graticola officinalis*	3	*Carex pallescens*	1
Leontodon hispidus	2	*Nardus stricta*	1	*Nardus stricta*	3	*Carex panicea*	1
Potentilla erecta	2	*Peucedanum sp.*	1	*Salix rosmarinifolia*	3	*Altre*	10
Danthonia decumbens	1	*Altre*	9	*Potentilla erecta*	1		
Genista germanica	1			*Carex panicea*	1		
Altre	2			*Altre*	5		

Table 4-2 Specific contribution (CS%) of Calluna vulgaris, Molinia arundinacea and other species by site before fire treatments.

With regard to the tree layer component the 4 sites were characterized by different stages of tree encroachment which increased from site 1 to 4 as evidenced by the distribution of tree stems frequency by size classes of root collar diameter (Dc) (Figure 4-17). In all sites *Populus tremula* was the main specie followed by *Betula pendula*; few individuals of *Quercus robur* were founded, representing a first stage of the succession toward oak-hornbeam forest. All trees observed had Dc lower then 7 cm even if few individuals with larger Dc were found outside the transects.

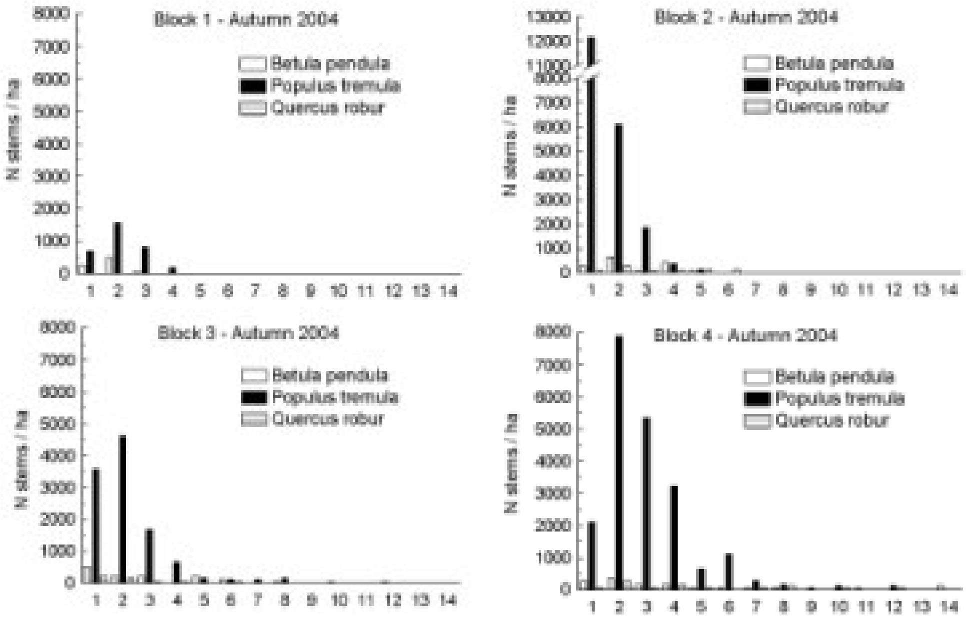


Figure 4-17 Tree frequency (N stems/ha) distribution by size classes (0.5 cm) of root collar diameter (Dc) in each site before burning in Autumn 2004.

Tree stem distribution in site 2 was characterized by a higher number of stems in the first Dc class (< 0.5 cm) which could mean a recent resprout as a consequence of a disturbance event. Disturbance in site 2, such as a recent wildfire, could also explain the lower CS% of *Calluna* and the higher one of *Molinia*. Despite data archives do not report a recent wildfire in this area, the site fire history determined by tree ring counting (par 4.2.3) evidenced a fire event which occurred 6 years before. Consequently the vegetation structure before burning in site 2 represents an interesting evidence of vegetation dynamics after fire.

4.3.2. Fire experiments description

A number of 22 fire experiments (exp.) followed each other for three winter seasons from 2005 to 2007, during February and March (Table 4-3). In winter 2005 a number of 8 exp. were carried out in long unburnt plots: exp. from 1 to 4 were ignited along the upwind or downwind side of rectangular plots of about 4000 m^2 (80 x 50 m) in order to perform back and head fires. During this period exp. from 5 to 8 were ignited along the downwind side of plots of about 1250 m^2 (25 x 50 m) which were subsequently burnt both in winter 2006 (exp. from number 9 to 16) and in winter 2007 (exp. from number 17 to 22) in order to test the effect of frequent headfires on vegetation.

Exp. N°	Date	Site	T_Y (n° years)	I_T (h.m.s)	A_B (m^2)	Bf vs Hf	Veg. Type
1	21-feb-05	4	15	10.38.00	4000	Bf	1
2	21-feb-05	3	15	13.13.00	4000	Hf	1
3	10-mar-05	1	15	10.28.00	4000	Hf	1
4	10-mar-05	2	6	11.56.00	4000	Bf	1
5	17-mar-05	1	15	9.42.00	1250	Hf	1
6	17-mar-05	2	6	10.56.00	1250	Hf	1
7	17-mar-05	3	15	11.40.00	1250	Hf	1
8	17-mar-05	4	15	12.31.00	1250	Hf	1
9	16-mar-06	4	1	10.20.00	625	HF	2
10	16-mar-06	4	1	10.55.00	625	HF	2
11	16-mar-06	3	1	11.59.00	625	HF	2
12	16-mar-06	3	1	12.32.00	625	HF	2
13	16-mar-06	2	1	13.07.00	625	HF	2
14	16-mar-06	2	1	13.08.00	625	HF	2
15	16-mar-06	1	1	14.12.00	625	HF	2
16	16-mar-06	1	1	14.28.00	625	HF	2
17	2-feb-07	3	1	12.27.00	1250	HF	2
18	16-feb-07	1	1	10.47.00	1250	HF	2
19	16-feb-07	2	1	12.06.00	625	HF	2
20	16-feb-07	2	1	12.22.00	625	HF	2
21	16-feb-07	4	1	13.08.00	625	HF	2
22	16-feb-07	4	1	13.23.00	625	HF	2

Table 4-3 Experiments ordered per date and characterized by site, time since fire (T_Y; years), ignition time (I_T; h.m.s), area burnt (A_B; m^2); planned ignition pattern (Bf = backfire; Hf = headfire) and dominant vegetation type (1 = Calluna dominated heathland; 2 = Molinia dominated grassland).

Experiments from number 1 to 8, were carried out in long unburnt plots where *Calluna vulgaris* has almost reached the mature phase; consequently the vegetation structure in winter time was characterized by a continuous layer of *Calluna* green canopies interlaced with cured tussocks of *Molinia arundinacea*. In experiments from number 9 to 22, as a consequence of the effect of previous fire treatments which rejuvenated *Calluna* stands to a pioneer stage, the vegetation structure was completely different with the dominance in the cover of cured *Molinia* and other grass species which entered in the sward after the *Calluna* canopy cover reduction by fire.

The differences between these two main vegetation covers which characterizes the 22 fire exp. means differences in load and arrangement of the live and dead fine fuel component of the fuel complex which are one of main determinants of fire behaviour (Cheney 1981). Consequently the following analysis about the fuel modelling and fire behaviour characterization were done separately for the two vegetation types distinguishing two groups of experiments: the first group include exp. from 1 to 8 and it will be named “Heath fires”; the second one, which include exp. from 9 to 22, it will be named “Grass fires”.

4.3.3. Fuel characterization

As it has been argued in Chapter 3, the fire behaviour is mainly determined by the dead and live fine fuel components of the fuel complex; consequently, for a clearest exposition of the fuel modelling and the fire behaviour analysis, the results concerning the coarse woody debris (CWD) and the live fuel component with diameters above 0.6 cm are showed separately from fine fuels.

According to the fine fuel component the 22 fire experiments can be divided into 2 groups: exp. from 1 to 8 (Heath fires), present an average total dry fine fuel load (W_{TF}) equal to 11.31 t/ha (Figure 4-18a) and an average coefficient of variation (CV%) lower then 10% (Figure 4-18b). In comparison exp. from 9 to 22 (Grass fires), present a lower average W_{TF} of 3.21 tons/ha with average CV% higher then 10%. Moreover in Heath fires the ratio between the live fine fuel (W_{LF}) and the dead one (W_{DF}), was respectively of 1.22 in Heath fires and 0.14 in Grass fires. A t-test showed significant differences in W_{TF} and W_{LF}/W_{DF} between Heath and Grass fires (two-tailed t-test; $p < 0.001$***).

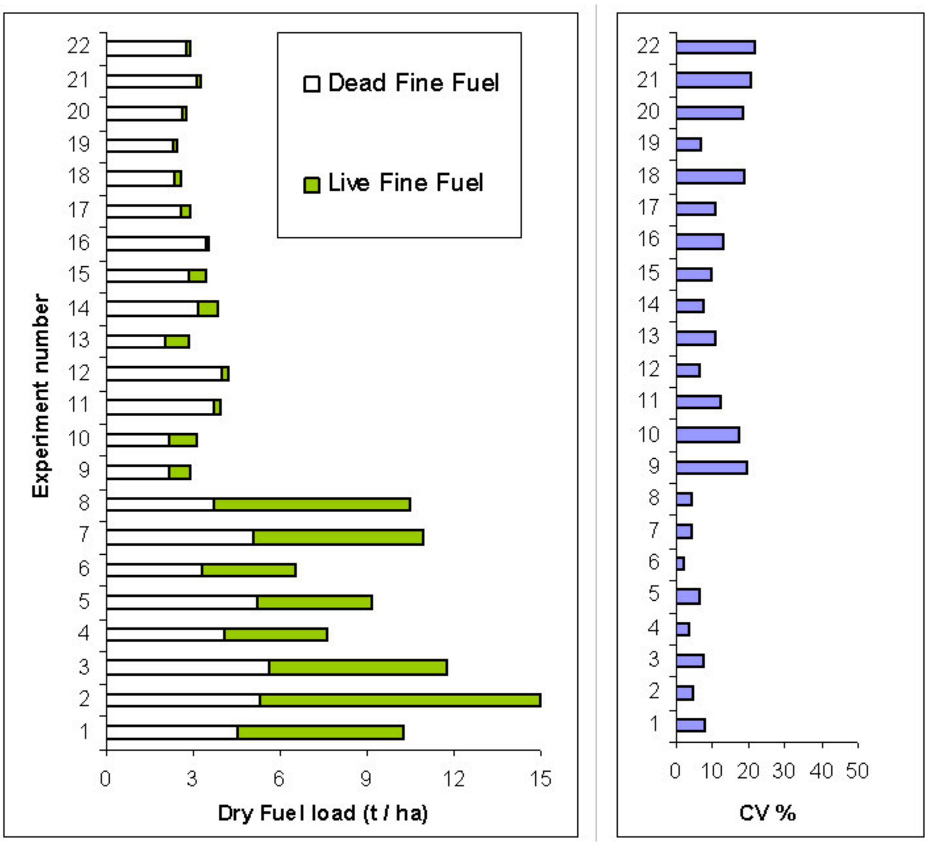


Figure 4-18 a) average dead and live dry fine fuel load (t/ha) by fire experiment; b) Coefficient of Variation (CV%) of the average total dry fine fuel load.

The second step in characterizing fine fuels regarded the moisture content. Before starting the first season of experiments in 2005, a series of test samples were done in order to assess the moisture variability within samples and consequently plan a representative but quick sampling (i.e. less numerous and smaller samples) the day of each fire experiment when a high efficiency is required. Moreover this analysis may give important information about the fuel moisture variability within plots which could influences the fire behaviour during the experiment. The sampling during the weeks before exp. 1-8 showed how the fuel moisture variability was relatively low within plots with CV% < 10% (narrow bars in Figure 4-19). The CV% of the average fuel moisture of 6 samples per plot (1 x 1 m) presented a mean of 6.2% $\pm$ 0.9% for the dead fine fuel (Md) and 1.5% $\pm$ 0.3% for the live fine fuel (Ml); the average of the total fine fuel moisture (Mt) showed the maximum CV% of 10.1% in plot 6.

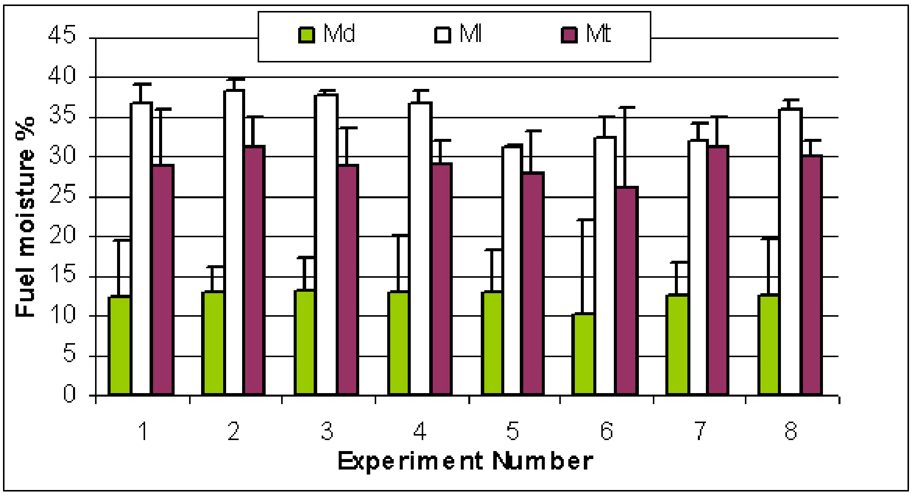


Figure 4-19 Average values of dead (Md), live (Ml) and total (Mt) fine fuel moisture for Calluna heathland fuel in plots from 1 to 8 (6 samples per plot of 1 m^2) one week before fire experiments; the coefficient of variation of the mean is represented by narrow bars.

As a consequence of the low variability in fuel moisture between fuel samples (CV% < 10%) fuel moisture content could be assessed quickly and reliably on the date of each fire experiments by dipping vegetation in 3 samples of small size (0.25 x 0.25 m). Results concerning fine fuel moisture content at each date are showed in Figure 4-20 and 4.21.

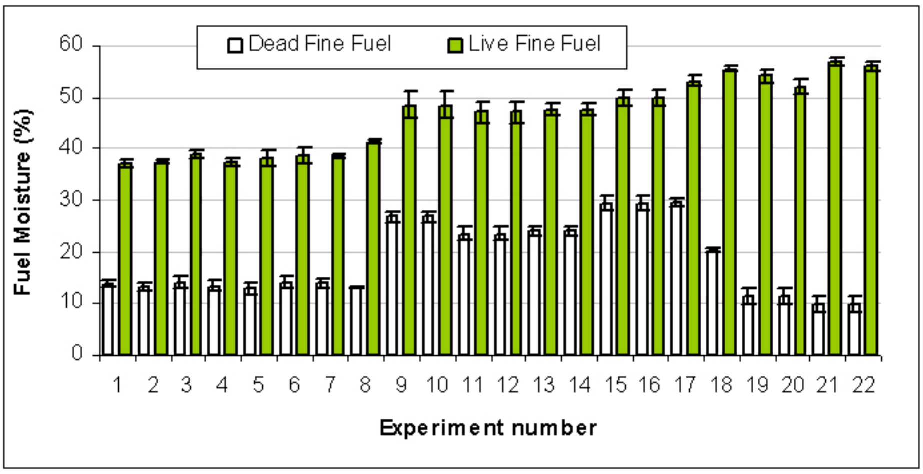


Figure 4-20 Average Dead Fine fuel moisture (Md) and Live Fine fuel moisture (Ml) the day of each fire experiment with the SE of the mean represented by the narrow bars.

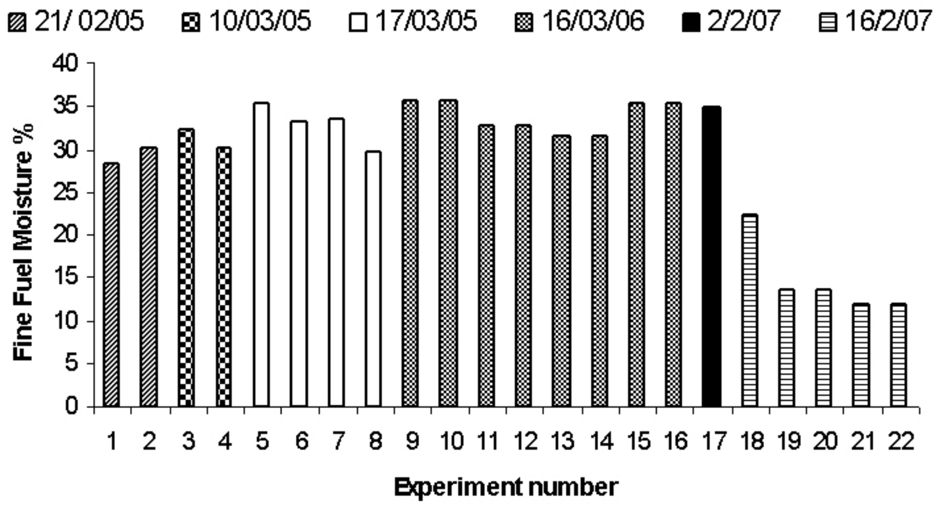


Figure 4-21 Total fine fuel moisture (Mt) the day of the experiment grouped by burning date

The Md for exp. from 1 to 8 (Figure 4-20) was similar in all the dates of burn of 2005, despite few rain occurred between dates (Table 4-4), and presented an average of 14% ± 0.14%; this value is lower than the one showed by the dead fine fuel for experiments of 2006 with an average of 26% ± 0.88%. These differences can be explained by the climate condition as in 2006 a higher amount of rain occurred in the month before fire experiment dates (Table 4-4). A correlation analysis between Md on the date of burn of each fire experiment and the climatic variables displayed in Table 4-4 showed the highest and significant values with the cumulative sum of the rain occurred in the 30 days before (R_{30}) the experiment (Pearson Correlation, two-tailed test, Md vs R_{30}: 0.869**). Moreover, exp. 17 failed, as a consequence of the high Md (30%) due to the rain of the day before (Table 4-4). Nevertheless it gave interesting information about the operative limit according to fuel moisture, day since rain and RH%. Exp. from number 18 to 22 showed the lowest values of Mt (Figure 4.21) despite few rain occurred 6 days before; nevertheless the fuel, which was mainly constituted by cured grasses, dry out rapidly as a consequence of warm temperatures and windy days. The high fuel dryness, as it will be described in the fire behaviour analysis, was one of the main determinants of high rate of spread of the fire front in these experiments, well above the operational limits.

Date	Day since rain	R_{30} (g)	T_{30} (°C)
21/02/05 Exp. 1-2	30	0,2	34
10/03/05 Exp. 3-4	6	4,2	65
17/03/05 Exp. 5-8	13	3,8	61
16/03/06 Exp. 9-16	10	57,9	117
02/02/07 Exp. 17	0	29,8	133
16/02/07 Exp. 18-22	6	10,8	210

Table 4-4 Experiments grouped per date of burn and characterized by day since last rain and cumulative values of average daily rain (R_{30}; g) and average daily temperature (T_{30}; °C) on a period of 30 days before the fire experiment (Data source: Regione Piemonte).

After the fuel load and moisture characterization of the fine fuel component, two different fuel models for *Calluna* heathlands and *Molinia* grasslands at Vauda in winter time could be distinguished (Table 4-5).

a) *Calluna* heathland fuel model (Heath fires)

Exp. 1 - 8	W_{DF} (t/ha)	W_{LF} (t/ha)	W_{TF} (t/ha)	Md (%)	Ml (%)	H_{DF} (m)	H_{LF} (m)	ρ_{DF} (kg/m^3)	ρ_{FF} (kg/m^3)	ρ_{TF} (kg/m^3)
Average	4.64	5.60	11.35	14	38	0.56	0.18	841	3117	2115
Min	3.37	3.17	7.62	13	37	0.47	0.13	716	1937	1368
Max	5.65	9.66	14.98	14	41	0.65	0.20	1008	4901	3492

b) *Molinia* grassland fuel model (Grass fires)

Exp. 9 - 22	W_{DF} (t/ha)	W_{LF} (t/ha)	W_{TF} (t/ha)	Md (%)	Ml (%)	H_{DF} (m)	H_{LF} (m)	ρ_{DF} (kg/m^3)	ρ_{FF} (kg/m^3)	ρ_{TF} (kg/m^3)
Average	2.82	0.39	3.21	21	50	0.36	0.05	789	880	879
Min	2.05	0.08	2.43	24	47	0.3	0.03	489	160	765
Max	4.02	0.97	4.24	30	57	0.42	0.08	1004	2626	1081

Table 4-5 Average and range of values grouped per vegetation types (a – b) for following fuel variables; W_{DF}: dead fine fuel load; W_{LF}: live fine fuel load; W_{TF}: total fine fuel load; Md: fine dead fuel moisture content; Ml: live fine fuel moisture content; H_{DF}: dead fine fuel height; H_{LF}: live fine fuel height; ρ_{DF}: dead fine fuel bulk density; ρ_{LF}: live fine fuel bulk density; ρ_{TF}: total fine fuel bulk density.

For a complete characterization of the fuel complex at Vauda, in Figure 4-22 the fuel load of CWD and live fuels (> 0.6 cm) is given for the 22 experiments. According to these two fuel components the variability between plots is higher as a consequence of different stages of tree encroachment in the 4 sites as showed in Figure 4-17. For example experiments 1 and 8, both placed in site 4 (Table 4-2), have the highest fuel load of the live fuel component and consequently also of CWD because of higher production of dead woody material by tree self-pruning and tree mortality. After fire occurrence in 2005, as it will be better discussed in next paragraphs, followed a high tree stem mortality with a further production of CWD in the following seasons. This is showed in figure 4.22 were exp. from 9 to 22 do not have the live fuel component as a consequence of tree stem mortality. Moreover exp. 9, 10, 21 and 22, all carried out in site 4 respectively in 2006 and 2007 (Table 4-2), have the highest values of CWD as a consequence of tree stem mortality in 2005. This relation between live fuel before burning and CWD after fire can be also observed for experiments carried out in site 3 were tree encroachment was also present (experiments 2, 7, 11, 12, 17). Inversely for experiments in sites 1 and 2, characterized by early stages of tree encroachment, the few live fuel before burning (i.e. experiments number 3, 4, 5, 6) means low amounts of CWD after burning (i.e. experiments 13,14, 15, 16, 18, 19, 20).

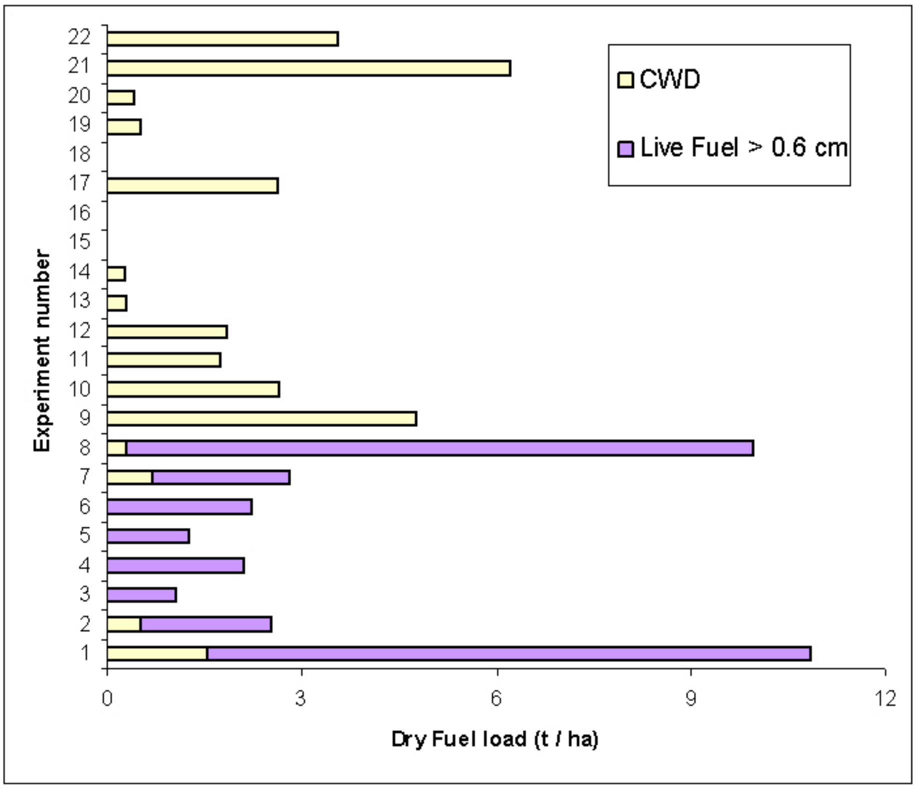


Figura 4-22 Average values of coarse woody debris (CWD) and dry live fuel load with diameters superior to 0.6 cm per each fire experiment.

4.3.4. Fire behaviour analysis

The fire behaviour analysis has been done separately for the two fuel models and vegetation types: the modelling of behaviour in *Calluna* heathland uses the data recorded in exp. from 1 to 8 (Heath fires) while for *Molinia* grasslands the analysis uses data recorded in exp. from 9 to 22 (Grass fires).

In order to give an idea of the range of fire behaviour at Vauda, in Table 4-6 the observed variations in fire behaviour descriptors and environmental variables for the 2 groups of experiments are showed. Average values of fire behaviour descriptors and environmental variables for the 22 fire exp. are given in (Table 4-7).

Heath fires Exp. 1 - 8	**ROS (m/min)**	**W (%)**	**I (kW/m)**	**H (m)**	**U (km/h)**	**T (°C)**	**RH (%)**
Average	5.49	74	1858	1.1	7	12	46
Min	0.5	54	125	0.3	1	2	22
Max	12	90	6642	2	9	19	91

Grass fires Exp. 9 - 22	**ROS (m/min)**	**W (%)**	**I (kW/m)**	**H (m)**	**U (km/h)**	**T (°C)**	**RH (%)**
Average	6.42	70	559	0.8	8	12	52
Min	0.3	50	20	0.1	1	7	41
Max	30	100	1939	1.5	10	17	91

Table 4-6 Average and range of values of fire behaviour descriptors and environmental variables grouped per vegetation types for following variables; ROS: rate of fire spread of the fire front head; W: percentage of fuel consumption; I: fireline intensity*; H: flame height; U: windspeed at 2 meters height; T: air temperature; RH: air relative humidity.*

The values displayed in Table 4-6 clearly show the wide range of fire behaviour descriptors both in Heath and Grass fires. This variability could be partially related to the differences in ignition pattern between fire experiments in order to perform back or headfires as shown by differences in ROS and I between fire experiments (Table 4-7). Nevertheless, it has also been determined by a high variability of weather conditions within fire experiments. In order to give a visual perception of the fire behaviour heterogeneity within plots burnt in 2005, exp. from 1 to 8 have been described drawing a Fire Behaviour Map (FBM). In Figure 4-23 are showed and described the FBM for exp. 1 and 2. FBM of exp. from 1 to 8 are given in attachment (Att.2) which reports also the average and standard error values of fire behaviour descriptors and environmental variables per each fire experiment.

N°	Data	W_{TF} (t/ha)	Mt (%)	ROS (m/min)	I (kW/m)	U (km/h)	RH (%)	T (°C)	HF vs BF
1	21-feb-05	10.3	28	1.92	434	6,7	85	2	BF
2	21-feb-05	14.9	30	5.99	2138	6,5	65	2	HF
3	10-mar-05	13.4	32	11.76	3556	7	25	14	HF
4	10-mar-05	7.6	30	2.48	514	5,1	22	13	BF
5	17-mar-05	11.9	35	2.59	437	1,7	58	14	HF
6	17-mar-05	7.7	33	5.44	654	3,1	48	16	HF
7	17-mar-05	14.4	34	5.49	2376	6,6	44	19	BF
8	17-mar-05	10.3	30	8.36	1955	5,2	42	19	HF
9	16-mar-06	2.9	36	1.41	23	3,8	57	7	BF
10	16-mar-06	3.1	36	2.27	163	3,8	56	8	HF
11	16-mar-06	4.0	33	0.51	67	5,6	48	11	BF
12	16-mar-06	4.2	33	2.91	282	5,6	48	10	HF
13	16-mar-06	2.8	31	0.97	72	8,1	47	11	BF
14	16-mar-06	3.9	31	2.99	196	7,9	47	11	HF
15	16-mar-06	3.5	35	0.66	36	6,0	41	13	BF
16	16-mar-06	3.5	35	5.42	312	6,1	41	13	HF
17	2-feb-07	2.9	35	--	--	3,7	91	10	HF
18	16-feb-07	2.6	22	7.83	506	8,0	54	11	HF
19	16-feb-07	2.4	13	16.54	1473	10	42	17	HF
20	16-feb-07	2.8	13	8.11	1153	7,5	56	13	HF
21	16-feb-07	3.2	12	17.38	1753	7,5	50	15	HF
22	16-feb-07	3.0	12	16.43	1625	6	48	15	HF

Table 4-7 Average values of fire behaviour descriptors and environmental variables grouped per fire experiment; Date; W_{TF}: Total fine fuel load; Md: total fine fuel moisture content; ROS: rate of fire spread; I: fireline intensity*; U: wind speed at 2 meters height; RH: air relative humidity; T: air temperature; Ignition pattern (back vs head fires).*

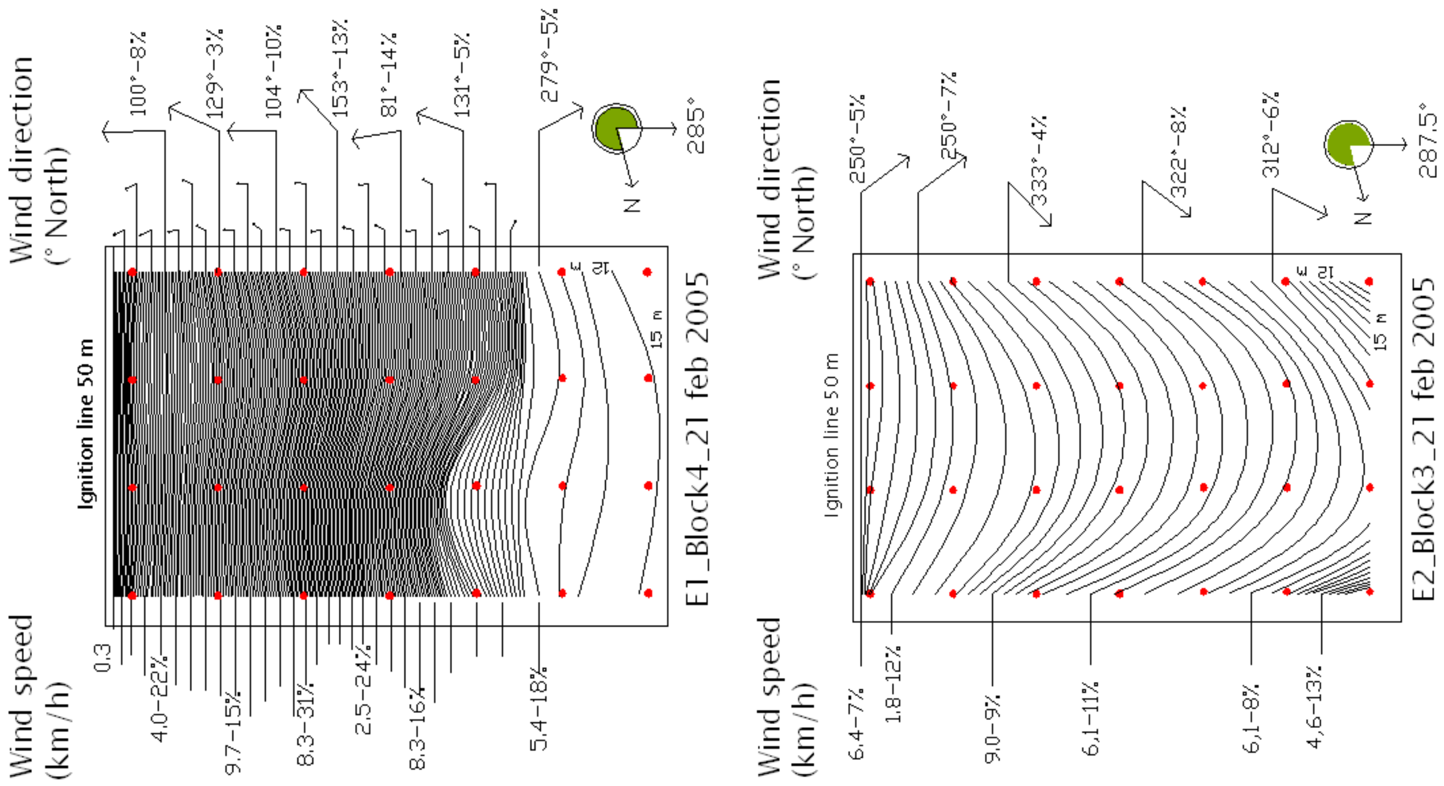


Figure 4-23 Fire Behaviour Maps for experiment 1 and 2 in February 2005: plot size is 80 x 50 m; curves represent the advancing fire front at 30 second intervals from the ignition line. Wind direction (degree from North) and speed (km / hr) are reported on the right and left sides of each contour of fire front. The average value and CV% of wind direction and speed each 12 minutes (Exp.1) and each 3 minutes (Exp.2) are given. Red dots represent the rows position along the plots. Images were produced with a CAD software.

The two contrasting FBM in Figure 4-23 clearly show how wind direction affects the ROS of a fire front between and within fire experiments: in fact to run the same length (80 m) the back fire (exp.1), took 1 h 25' whilst the head fire (exp.2) took only 0 h 15'. Moreover exp.1, initially a backfire, showed an acceleration phase as consequence of a sudden change in wind direction.

In order to study the effect of wind speed and direction and their interaction in determining the ROS, a scale variable, named Wind Vector (Vv), was studied. The Vv is calculated multiplying the value of wind speed (U; km/h) and the wind direction index (Vd) whose values range from –1 to +1 (Figure 4-24).

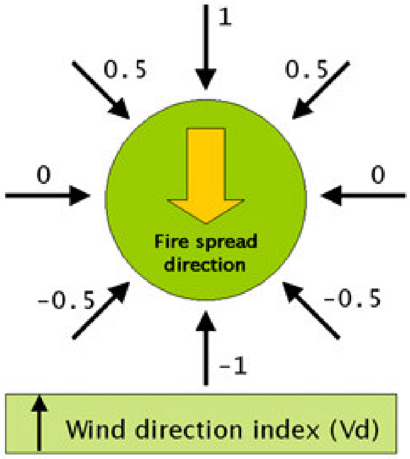


Figure 4-24 Values of wind direction index (Vd): if the wind direction is opposite to the fire front spread direction (back fire), the index assumes continuous values from –1 to 0, while in a wind driven fire Vd from 0 to 1; values change according to the angle between wind direction and the axis of to fire spread direction (i.e. Vd=-1 if wind is opposite to fire spread but parallel to spread axis; Vd= 0 if wind is perpendicular to spread axis; Vd= 1 if wind is parallel to fire spread axis and blow in the same direction).

In Table 4-8, fire behaviour and fuel characteristics are described for exp. 1 and 2 in order to show how, in similar fuel conditions, wind speed and direction, characterized by the average value of the variable Wind Vector (Vv), are the main factors affecting the ROS and consequently the *fireline intensity*. It is interesting to notice how in exp. 1 the average ROS, I, H show a higher standard error as a consequence of wind direction variability (higher Vv standard error), as it is evidenced by the FBM (Figure 4-23) and by video recording in attachment (Att.1).

Esp.	**ROS** (m/min)	**I** (kW/m)	**H** (m)	W_{TF} (t/ha)	**Mt** (%)	**Vv**
1	1.91 ± 1.18	489 ± 301	0.8 ± 0.1	10.26	28	-3.23 ± 1.58
2	5.99 ± 0.78	2347 ± 304	1.4 ± 0.1	14.97	30	3.89 ± 0.20

Table 4-8 Comparison between experiment 1 and 2; ROS: rate of spread; I: fireline intensity; H: flame height; W_{TF}: total fine fuel load; Mt: total fine fuel moisture content; Vv: wind vector.

The ROS measured along each segment of the poles grid in all burned plots (except 17), was then studied as a function of the average value of Vv measured for the laps of time the fire took to run that segment. The fire behaviour microplot analysis enabled to have a consistent number of observations (n = 46 observations both for Heath then for Grass fires) to fit empirical models. Regression models which study the ROS at the head and flanks of the fire front as a function of Vv both for Heath fires then from Grass fires are showed in Figure 4-25a and Figure 4.25b respectively.

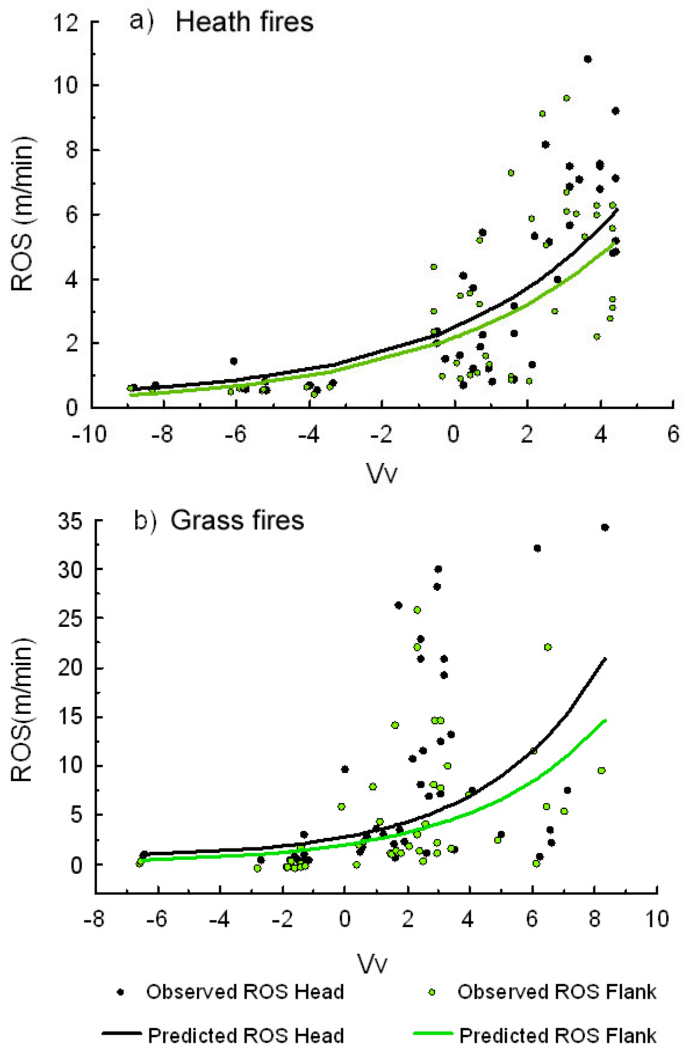


Figure 4-25a
ROS vs Vv models for Experiment 1-8 (n=46 observations)

- ROS Head Model:
$ROS = 2.28+0.22*e^{Vv}$
*$R^2 = 0.676$; sign.****
- ROS Flank Model:
$ROS = 2.13+0.20*e^{Vv}$
*$R^2 = 0.596$; sign.****

Figure 4.25b
ROS vs Vv models for Experiment 9-22 (n=46 observations)

- ROS Head Model:
$ROS = 2.19+0.27*e^{Vv}$
*$R^2 = 0.322$; sign.****
- ROS Flank Model:
$ROS = 2.01+0.24*e^{Vv}$
*$R^2 = 0.394$; sign.****

For a better fit of the models all the observations made on segments where fire had an acceleration, such as within 5 meters distance from the ignition line or segments interested by sudden changes in wind direction which determined an acceleration phase (Cheney and Gould 1993, 1995), were excluded; moreover, in several experiments frequent gusts and changes in wind speed and directions made anemometers measurements not representative of the wind affecting the fire front active zone (Sullivan et al. 2001) and consequently their observation were not included in regression models.

Both for Heath fires (exp. 1-8) and for Grass fires (exp. 9-22) the predicted values by models for the flank fire (green lines) are lower then the ones predicted for the head of the fire front (black lines). Despite experimental fires may not reach steady state as they are tightly controlled at the flanks, so they advance as a strip rather than growing as an uncontrolled wildfire, nevertheless an initial acceleration in the middle section of the fire front was observed. This phenomena was also visualized in the FBM (Figure 4-23; Att.2) when the wind driven fire front contours showed a higher acceleration and ROS in the middle of the plot.

The significance of regression coefficient was always high ($p < 0.001$***). Heath fires models fitted the observed data better then the Grass fires models: in Heath models R^2 reached values of 0.6 meaning that the 60% of ROS variance was explained by changes in wind speed and direction; the Grass fires models showed lower R^2 ranging from 0.3 to 0.4. Consequently both for Heath and Grass fires other factors rather then wind, such as the fuel characteristics (moisture content, arrangement, load), played also an important role in determining ROS variability.

A clear example of how the ROS can be affected by fine fuel moisture is given by exp. 19-22. In fact the ROS at the head of the fire front in Grass fires reached very high values up to 30-35 m/min, with an average ROS of about 17 m/min (Table 4-7), well above the operational limits of a direct fire fighting intervention. These extreme values, higher then the one expressed in Heath fires, are partially due to a wider range of Vv but can also be explained by their dead fine fuel moisture content which was very low (10%) (Figure 4-20). Also fuel load was recognized to have a positive effect on ROS (Fernandes 2000, Davies 2005) but differences in fuel load at Vauda, both within Heath and Grass fires (Figure 4-18), are too small to determine significant differences in fire behaviour.

Unfortunately, both for Heath and Grass fires, the high correlation between the variable Vv and the others fuel variables with a significant effect on rate of spread (Table 4-9) did not allowed to plot them against the rate of spread residuals from first models and consequently to examine their abilities to explain the remaining variation in ROS in order to improve the models.

a) Heath fires - Experiment 1 - 8

	Vv	W_{DF}	W_{FF}	Md	Mt	δ	ρt
Vv	1	0.25*	0.34**	-0.31**	0.24*	-0.33**	0.43**
W_{DF}		1	0.86**	0.13	0.26*	0.20	0.34**
W_{FF}			1	0.01	0.22	0.12	0.64**
Md				1	0.06	-0.05	-0.29**
Mt					1	-0.68**	-0.39**
δ						1	0.22*
ρt							1

b) Grass fires - Experiment 9 - 22

	Vv	W_{DF}	W_{FF}	Md	Mt	δ	ρt
Vv	1	0.04	-0.17	-0.36**	1.00**	-0.60**	-0.55**
W_{DF}		1	0.87**	0.10	0.17	0.31*	0.18
W_{FF}			1	0.40**	0.51**	0.63**	0.39**
Md				1	0.98**	0.72**	0.15
Mt					1	0.82**	0.29*
δ						1	0.62**
ρt							1

*Table 4-9 Correlation matrix (Pearson correlation) between the variables with a significant effect on rate of fire spread both for Heath fires then Grass fires (a – b). Correlations significant at the 5, 1 and 0.1% levels are expressed by *, ** and ***, respectively. See Tables 4.4 for the explanation of the symbols for the variables.*

Nevertheless, both in Heath and in Grass fires, the behaviour analysis at a microplot scale enabled to characterize back versus headfires; in fact all the observations of ROS made per each one of the segments of the grid were divided into two groups according to the positive (headfire) or negative (backfire) value of Vv. A spatial distribution of ROS and I within the plot for the two contrasting fire behaviours was obtained consequently.

In Table 4-10 average values of fire behaviour descriptors grouped per vegetation type and positive or negative values of Vv are displayed.

Heath fires Esp. 1-8	**ROS (m/min ± SE)**	**Tr (sec. ± SE)**	**I (kW/m ± SE)**	**H (m)**
Vv < 0 (Backfire)	1.65 ± 0.67	49 ± 6	307 ± 51	0.6 ± 0.1
Vv > 0 (Headfire)	7.12 ± 0.82	31 ± 5	2052 ± 257	1.5 ± 0.3

Grass fires Esp. 9-22	**ROS (m/min ± SE)**	**Tr (sec. ± SE)**	**I (kW/m ± SE)**	**H (m)**
Vv < 0 (Backfire)	0.82 ± 0.22	35 ± 2	42 ± 12	0.3 ± 0.1
Vv > 0 (Headfire)	10.92 ± 1.87	23 ± 2	785 ± 152	1.6 ± 0.2

Table 4-10 Average values (±SE) of fire behaviour descriptors grouped per positive or negative value of Vv both for Heath (exp. 1-8) then for Grass fires (exp. 9-22). ROS: rate of fire spread (m/min); Tr: residence time (s); I : fireline intensity *(kW/m); H: flame height.*

Heath headfires, despite lower ROS values when compared to Grass headfires, reached higher *fireline intensity* values as a consequence of higher fuel load and higher low heat content of heath fuels. The microplot analysis evidenced significant differences in fire behaviour between back fires, characterized by ROS < 2 m/min, residence times up to 50 seconds and I < 500 kW/m, and head fires, characterized by ROS > 2m/min, shorter residence times and I > 500 kW/m, both for Heath and Grass fires. Consequently effects on tree mortality and *Calluna* regeneration of these two contrasting behaviours were compared to understand which are ecological and operative implications subsequently in using back versus head fires in heathlands management.

4.3.5. Back fire versus Head Fire: effects on vegetation

To study the effects of fire behaviour on *Populus tremula* and *Betula pendula* mortality and *Calluna vulgaris* regeneration only the effects on plots burnt in 2005 (exp. 1-8) were considered because in plots burnt 3 times the effect of fire frequency and the one of fire behaviour cannot be distinguished and the constraint of the experiment design at Vauda do not allow to test their interaction on vegetation.

a) Effects of fire behaviour on *Populus tremula* and *Betula pendula* mortality

Pre and Post-fire tree stem distribution by Dc classes in sites 3 and 4 is showed in Figure 4-26. Compared to pre-fire stand distribution in 2004, the structure of the population was strongly affected by fire: juvenile tree stems were almost completely top-killed and only individual with higher Dc escaped fire. Nevertheless, most of the individuals of *Populus* and *Betula* showed a high resprouts capability in next growing season. Consequently the tree stems distribution lost the right tail showing an exceptional increase in stem density in the first Dc class (up to 50000 stems/ha), due to tree sprouting response. Few resprouts mortality occurred between 2005 and 2007 so that a shift to higher diameter classes was observed (up to 3 cm).

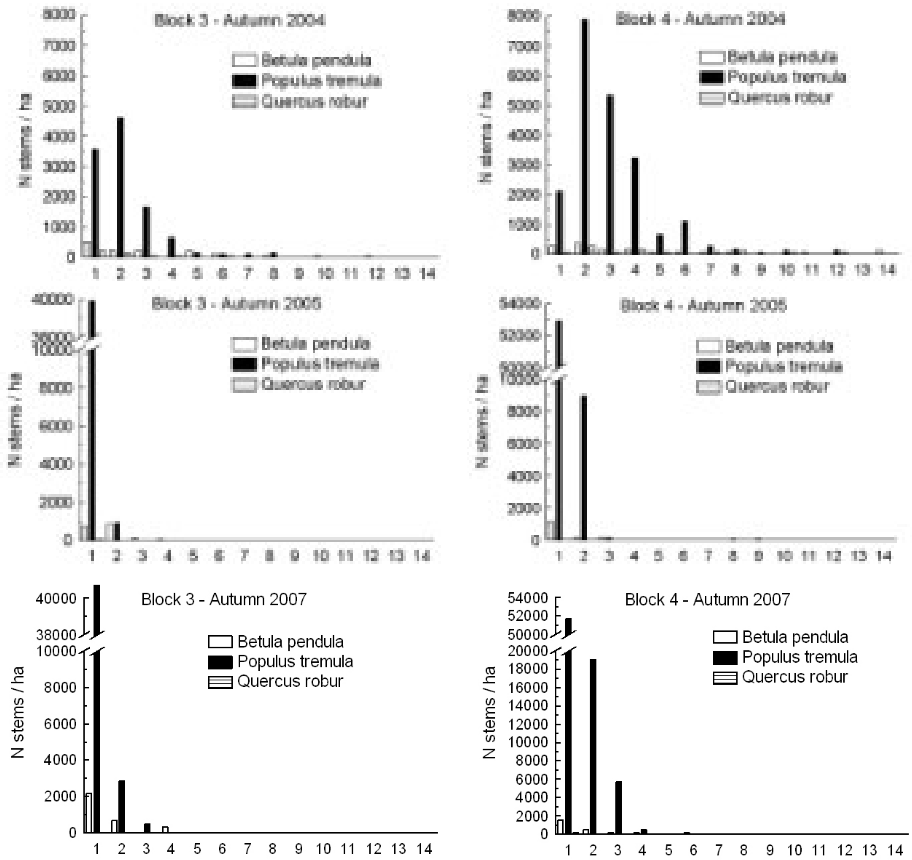


Figure 4-26 Tree species frequency Dc size classes (0.5 cm) in sites 3 (left) and 4 (right) before fire treatments (2004) one year (2005) and three years (2007) after fire.

The microplot analysis enabled to distinguished if trees[1] in site 3 and 4 experienced a backfire (I < 500 kW/m) or a headfire (I > 500 kW/m). Consequently differences in stem mortality, stump mortality and resprouts capability, that occurred as a consequence of the two *fireline intensity* classes, could be assessed. In all the following analysis differences between sites 3 and 4 were not studied.

In Figure 4-27 and 4-28 logistic regressions, which test the binary variable stem mortality after fire (dead or live) of *Betula* and *Populus* against Dc, are showed. In each graph, two curves, showing how the probability of *Betula* and *Populus* stem mortality decreases with increasing Dc both for back and head fires, are compared. In Table 4-11 and 4-12 statistics of the regression models are displayed.

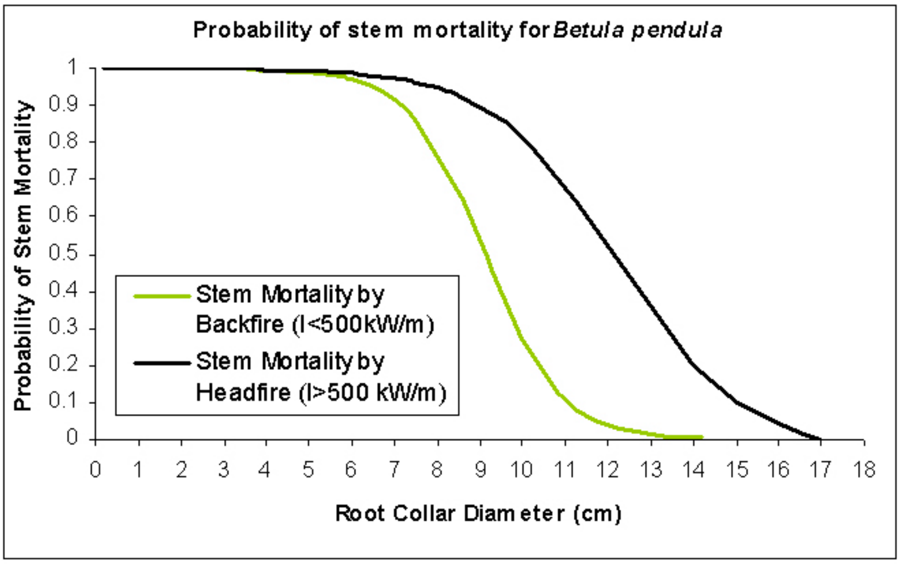


Figure 4-27 Probability of stem mortality of Betula pendula after back and headfire at varying dimension of Root Collar Diameter (classes are 1 cm).

Backfire	**B**	**S.E.**	**Wald**	**df**	**Sig.**	**R^2**
Root Collar Diameter	-1.136	0.388	8.587	1	0.003	0.825
Constant	10.362	3.503	8.752	1	0.003	
Headfire	**B**	**S.E.**	**Wald**	**df**	**Sig.**	**R^2**
Root Collar Diameter	-0.717	0.347	4.282	1	0.039	0.652
Constant	8.663	3.654	5.620	1	0.018	

Table 4-11 Maximum likelihood estimates of the logistic regression model parameters of Betula *stem mortality against Root Collar Diameter (Dc) both after backfire than headfire.*

[1] To fit better models an additional number of larger trees have been measured outside transects.

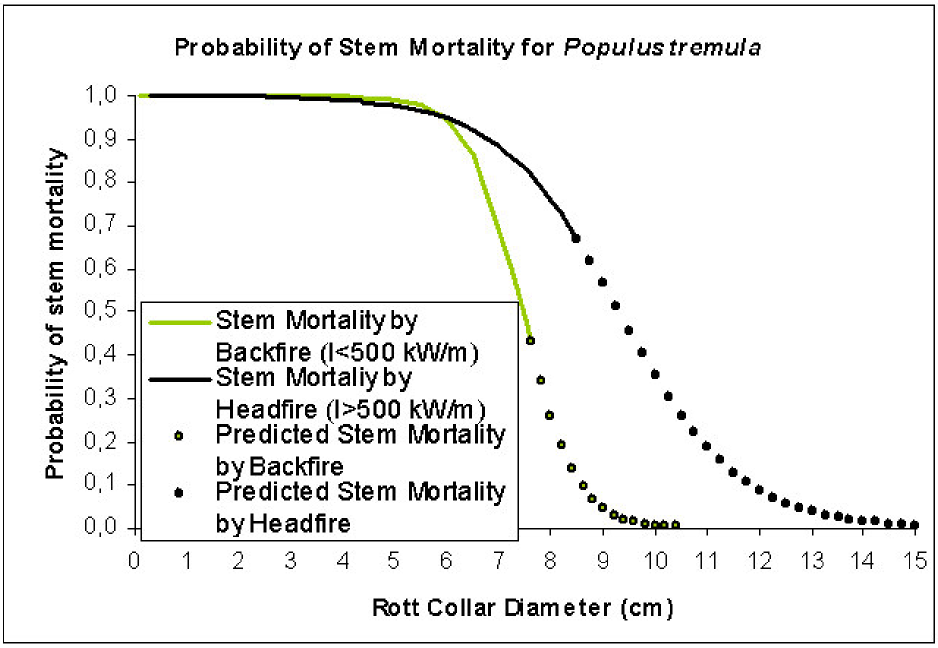


Figure 4-28 Probability of stem mortality of Populus tremula after back and headfire at varying dimension of Root Collar Diameter (classes are 1 cm).

Backfire	B	S.E.	Wald	df	Sig.	R^2
Root Collar Diameter	-1.942	1.034	3.526	1	0.060	0.689
Constant	14.478	7.101	4.157	1	0.041	
Headfire	**B**	**S.E.**	**Wald**	**df**	**Sig.**	**R^2**
Root Collar Diameter	-0.875	0.411	4.536	1	0.033	0.456
Constant	8.147	2.939	7.685	1	0.006	

Table 4-12 Maximum likelihood estimates of the logistic regression model parameters of Populus *stem mortality against Root Collar Diameter both after backfire than headfire.*

All the models ran were significant ($p < 0.05$*; 0.01**). The fitness of the models is satisfactory but *Populus* models showed lower R^2, probably because of the lower range in diameter of trees monitored. Both for *Betula* and for *Populus* the probability of stem mortality is higher after headfires (at equal root collar diameters). This effect is probably due to the higher scorched surface along the stem and crown scorched percentage due to the higher flame height and volume in head fires. Nevertheless, up to 7 cm of Dc, both for *Betula* and for *Populus*, back and head fire models show the same probability of stem mortality (between 90% and 100%).

According to the resprouting capability of individuals, two different analysis were performed for *Betula* and *Populus* as they showed two different resprout strategies: *Betula* resprouted from basal buds at the root collar while *Populus* mainly by suckering from roots, as it is showed by the spatial distribution of stems along a 2 x 10 m transect before and after burning (Figure 4-29).

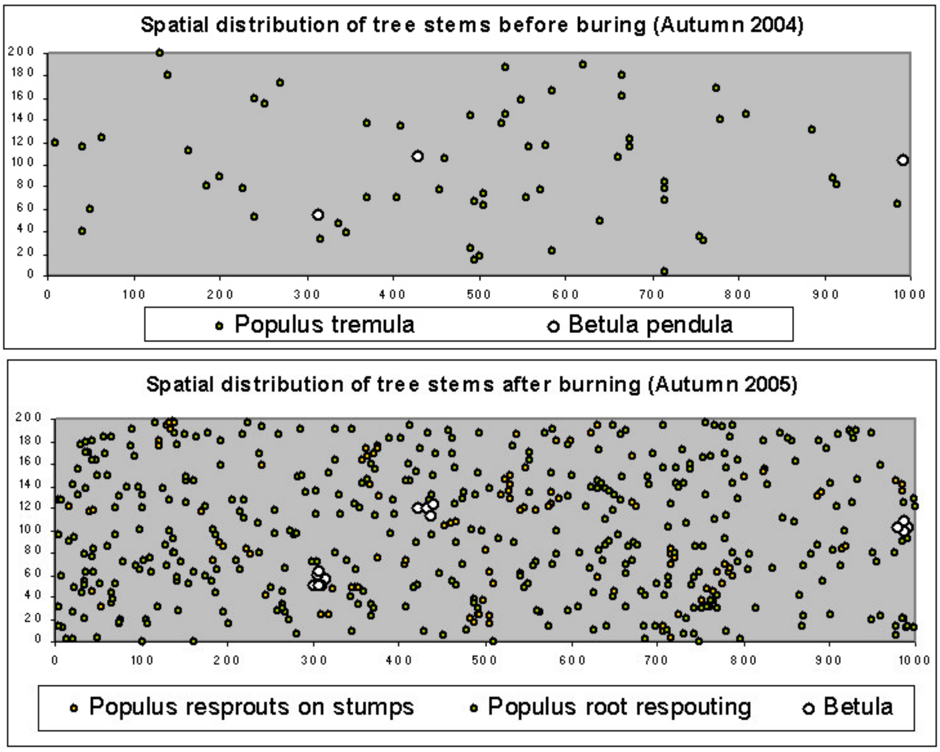


Figure 4-29 Spatial distribution of Populus *and* Betula *stems along a 2 x 10 m transect before and after burning:* Betula *resprouts are clustered while* Populus *ones spread around stumps.*

For *Betula pendula*, a linear regression analysis evidenced how stump resprouts capability is positively influenced by the Dc of the larger individual on the stump before burning (Figure 4-30). Residuals of first models were subsequently plot against *fireline intensity* classes to examine their abilities to explain the remaining variance in suckers number. Nevertheless, *fireline intensity* had no significant effect on the number of resprouts per stump. A non significant logistic regression evidenced that also stump mortality (i.e. failure in resprouting vs resprouting) is not determined by *fireline intensity*.

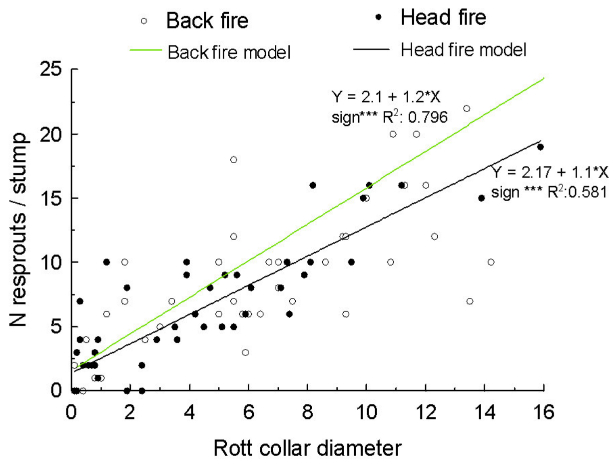


Figure 4-30 Number of resprouts per stump of Betula pendula by root collar diameter of the larger individual on the stump before fire, both for back then for headfires. Fitted linear regression models and equations (sign; R^2) are showed.

For *Populus tremula*, the study of resprouts capability on individual stumps was not possible as trees response after fire was mainly expressed by root resprouts, with suckers developing all around the transects, making the counting per stump impossible (Figure 4-31).

Figure 4-31 Root resprouting of Populus tremula *the first growing season after fire*

Consequently, in order to compare fire behaviour effect on *Populus* resprouts capability, all the transects were split in two groups of *fireline intensity* (Backfire versus Headfire). For each transect two indices were studied: total basal area of all the suckers (G) and cumulative canopy cover of all the suckers (C). This two indices well expressed the density of resprouts and their competition effect on the grass and shrub layers.

Tree density before burning varied between transects, having probably a covariate effect on resprouts density. In fact the resprouts density after fire is also determined by the dimension of individuals, as showed in Figure 4-30, and by the number of stumps per unit area before burning, as it is showed by differences in sucker density 3 years after fire between site 3 and 4 in figure 4-26. In order to remove the covariance effect of tree dimension and stump density before burning the indexes G and C were standardized dividing the index value, measured on the transect after burning, by the value that the index had before burning on the same transect (Table 4-13).

Backfire	**G (m^2/ha)**	**C (m^2/ha)**	**Stand. G**	**Stand. C**
Beforefire Autumn 2004	3.7 ±2.2	4474 ±2152	0.4 ±0.3	0.4 ±0.1
After fire Autumn 2007	1.6 ±0.4	1560 ±293		
Headfire	**G (m^2/ha)**	**C (m^2/ha)**	**Stand. G**	**Stand. C**
Beforefire Autumn 2004	4.4 ±2.6	4776 ±2389	0.7 ±0.2	0.5 ±0.1
After fire Autumn 2007	1.9 ±0.3	1806 ±315		

Table 4-13 Average values (±SE) of total basal area (G) and total canopy cover (Cc) of Populus tremula before and after fire treatments both for Back then Headfires. Standardized averages (after / before) of G and Cc are also displayed.

Once standardized, the two groups of independent samples for back and headfires were compared with a t-test. No significant differences were found in standardized averages of total basal area and canopy cover of *Populus* resprouts after burning i.e. a weak effect of fire behaviour on the resilience of individuals.

Such results, according with the ones obtained for *Betula*, could be explained by the relatively low transfer through the soil of the heat generated by fire fronts. It must be noticed that experiments have been carried out in winter 2005; consequently soil was frozen, which factor enhanced the insulating properties of soil in protecting buried buds at the root collar or on roots, thus determining a weak impact of fire on tree meristems.

Moreover both *Populus* and *Betula* are characterized by high resilience to fire and the results obtained at Vauda were in accordance with previous studies on species with similar ecology (Brown and De Byle 1987, 1989). Although there are no specific comparative studies, the ecology of *Populus tremula* may be probably related to that of *Populus tremuloides* Michx. (Quaking Aspen) (Chantal et al.

2005, Latva-Karjanmaa et al. 2005), which has been widely studied in North America (Brown and DeByle 1987). Brown and DeByle studies (1987, 1989) on prescribed fire effects in *Populus tremuloides* stands document that high *fireline intensity* (wind-driven fire fronts) top-kills larger stems than low-intensity fires (backfires), as it has been described also at Vauda. Moreover, Brown and DeByle (1989) documented that after fire *Populus tremuloides* show a dramatic resprouting capability depending on high proportion of aspen stems mortality. When apical dominance is reduced, sucker-stimulating cytokinins increase in roots where buds are protected from heat by the soil stratum resulting in high sprout capability. At Vauda fire enhanced resprouts thus generating a thick layer of small canopies, 20 cm in diameter with an height ranging from 10 to 50 cm (Figure 4-31), which competed with the shrub and grass layer anyway, despite high stem mortality.

Nevertheless, average G and C of *Populus tremula* significantly decreased after fire treatment both for Back and Head fires (Table 4-13) affecting tree cover and changing the competition with grasses and shrubs to advantage of *Calluna* regeneration.

b) Effects of fire behaviour on *Calluna vulgaris* regeneration dynamics

Both back and headfires left very little of the *Calluna* foliage or stems. In fact, despite *Calluna* is an evergreen specie and consequently has a higher water content compared to dead fine fuels, nevertheless moisture content was always lower then 40% (exp. 1-8 in Figure 4-20). Moreover, inside the *Calluna* canopy the dead fine fuel component, constituted by cured grasses and dead branches of *Calluna*, was densely interlaced with living braches and leaves. Finally, the little imbricate leaves had a high surface to volume ratio thus determining a high flammability of the canopy and an efficient combustion.

Molinia arundinacea, similarly to *Molinia caerulea* in NW European heathlands (Marrs et al. 2004, Brys et al. 2005), was strongly enhanced by fire. Aerts (1989) has demonstrated that *Molinia* species show a high phenotypic plasticity with respect to nutrient turnover and productivity. Increased soil nutrient levels after fire may, therefore, lead to vigorous re-growth of *Molinia* plants. Moreover, burning removes all litter, thus reducing the allelophatyc effect of *Calluna* litter (Bonanomi et al. 2005), and may also result in increased growth of *Molinia* due to higher midday soil temperatures. Brys et al. (2005) founded that fire significantly increases *Molinia* aboveground biomass, seed set and germination. The increased aboveground biomass and associated litter production may, in turn, result in a competitive advantage to *Molinia* and, finally, in competitive exclusion of other heathland species. Consequently the sward spatial structure and composition of *Calluna* dominated heathland of all Sites in 2004 was completely changed by fire towards *Molinia* dominated grasslands in 2005 (Figure 4-32).

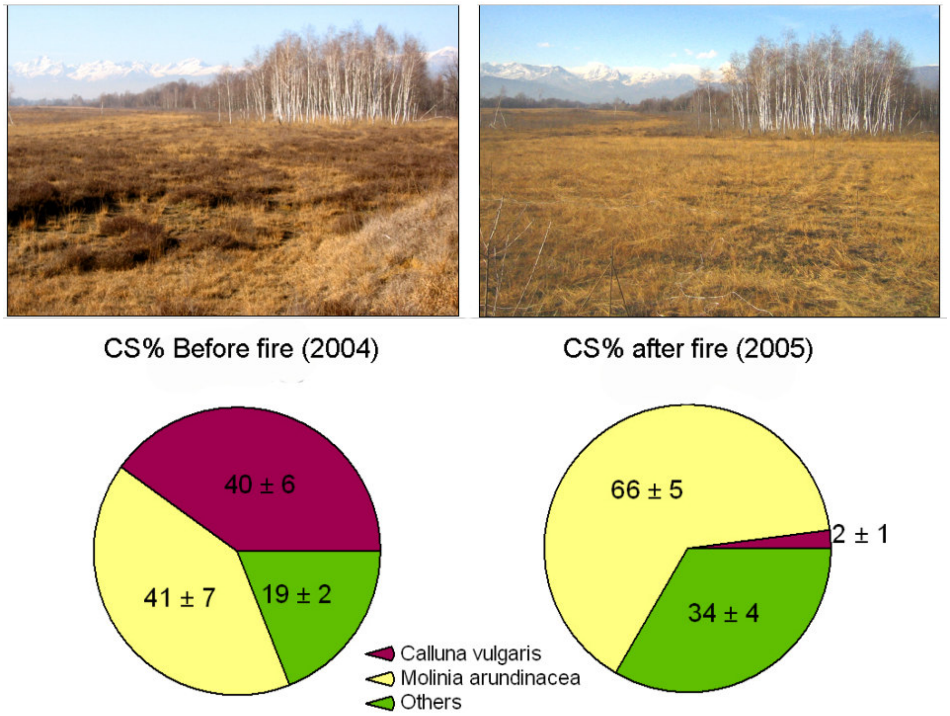


Figure 4-32 Average (±SE) specific contribution (CS%) of Calluna vulgaris, Molinia arundinacea and other species before (2004) and after fire treatments (2005). Averages are obtained using values of CS% of all transects (n. 36) in all Sites.

Despite crown scorch and the competition of suckers and *Molinia*, young green shoots of *Calluna vulgaris* started to resprouts from basal meristems of burned stumps since the first growing season after fire. Three years after fire many *Calluna* individuals showed a high capability of recovering (Figure 4-33).

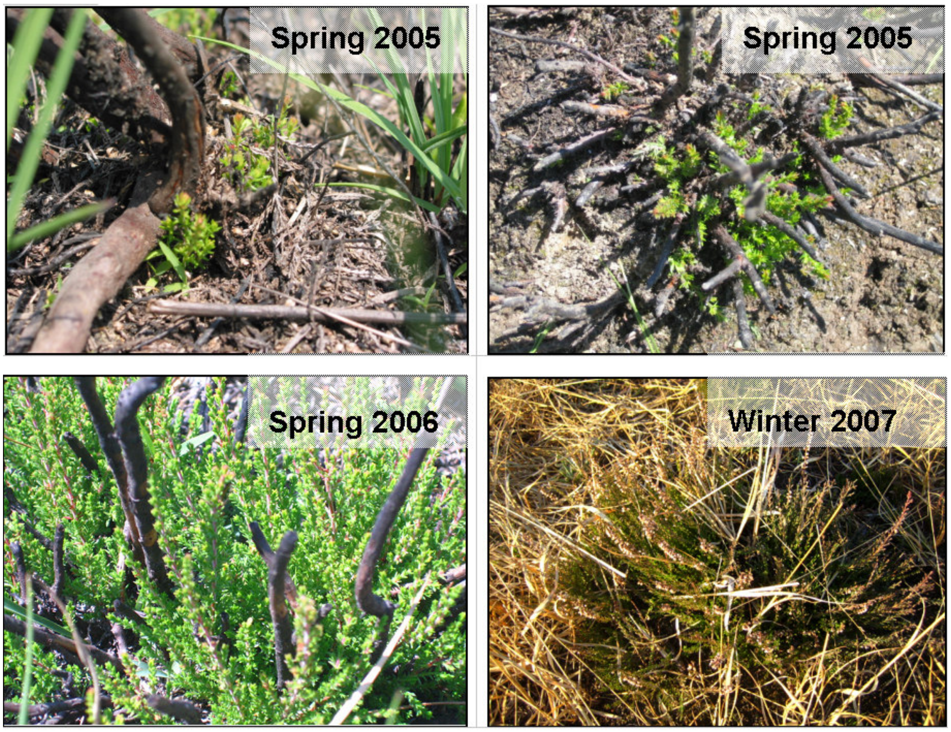


Figure 4-33 Calluna resprouts the following seasons after fire.

Nevertheless the resprouts of *Calluna* was unequal within plots showing a patchiness pattern of regeneration success. These could be determined by the interaction of many factors such as the density of *Calluna* before fire treatments, age of individuals, difference in soil properties and different behaviour of experimental fire fronts (Gimingham 1987). The study of how soil properties affect *Calluna* regeneration is one of the research issues of Vauda project but it will not discussed in this thesis.

The fire behaviour microplot analysis enabled to compare back fire versus head fire effects on sward composition and *Calluna* regeneration in site 1-3-4. In Figure 4-34 the sward specific contribution before fire treatments (2004), the first growing season after fire (spring 2005) and the third growing season after fire (spring 2007), is showed. The average CS% of *Calluna* in transects which experienced head fires was significantly higher (11%) in comparison with the one in transects which experienced back fires (3%).

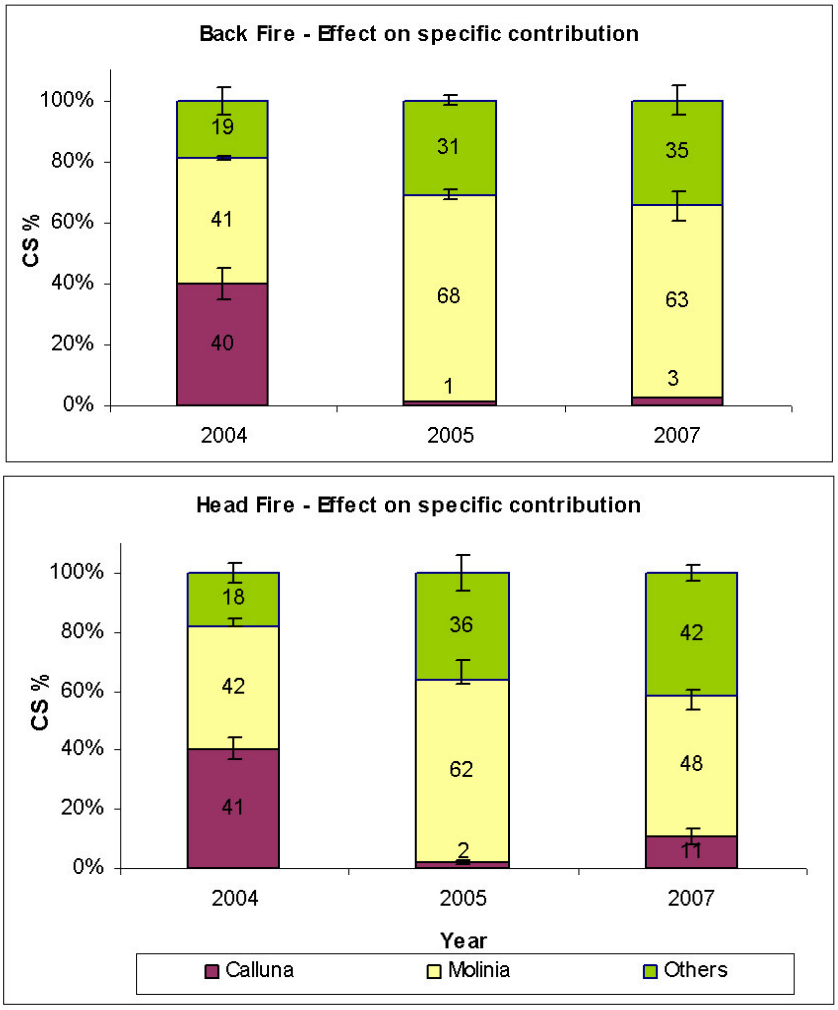


Figure 4-34 Average specific contribution (CS% ± SE; narrow bars) of Calluna, Molinia and Other species in 2004, 2005, 2007, both for back and head fires.

In order to eliminate the covariate effect of the density of *Calluna* individuals before burning, which could have determined higher values in CS% after fire, for each transect the CS% of sward species in 2007 was standardized dividing it for the CS% value that the same specie showed before burning in 2004 in that transect. The t-test analysis evidenced differences between the CS% ratio of back versus headfires showing significant higher values for head fires ($p < 0.01$**).

Consequently, according to fire behaviour effects on *Calluna* regeneration dynamics, the short-term results indicated a minor impact of headfires on

Calluna meristems which results in higher resprouts capability and consequent competitiveness of this specie in filling the gaps within *Molinia* cover.

The ecological explanation of this result must be found in the different physical processes which characterize backfire and headfires and especially in the pick temperatures and temperatures residence profiles they manifest (Whittaker 1961, Kenworthy 1963, Kayll 1965, 1966, Hobbs and Gimingham 1984a).

4.3.6. Infrared analysis and time temperature residence profiles effects

The measure of pick flame temperatures by pellets, melting at different temperatures ranging from 30°c to 954°C, did not give the expected precision and efficiency in the results. In fact in many cases the set of 20 pellets was not affected by flames as a consequence of fire behaviour heterogeneity and subsequent patchiness of the burning. Moreover the pellets showed an inertia, requiring several seconds to start the melting process. Consequently they are not a suitable device to have precise estimates of pick temperatures. Nevertheless, both in back and in head fires pellets burnt up to 800°C. None of the 40 thermo labels, ranging from 34°C to 80°C, buried 1 cm below the soil surface in different experiments changed in colour. This result could be determined by the weak heat transfer trough the soil as mentioned above. Nevertheless, the precision and reliability of this labels in estimating pick temperatures once buried it is questionable and at the moment it has not been tested by comparisons with other devices such as thermocouples.

A number of 480 infrared thermo photos, divided in 16 sequences of 30 frames at 10 seconds intervals, were recorded, both during back and headfires. In Figure 4-35 a sequence of 8 thermo photos (10 seconds interval one from each other), processed with the Thermography Explorer 4.5, shows a moving headfire. The ruler in the central portion of the frame enables to individuate fixed points (pixels) where the temperature is read with an apposite tool of the software. Consequently, positioning the ruler in each frame of the sequence and processing the thermo photo it is possible to have a sequence (profile) of temperatures at several fixed point for each fire front studied.

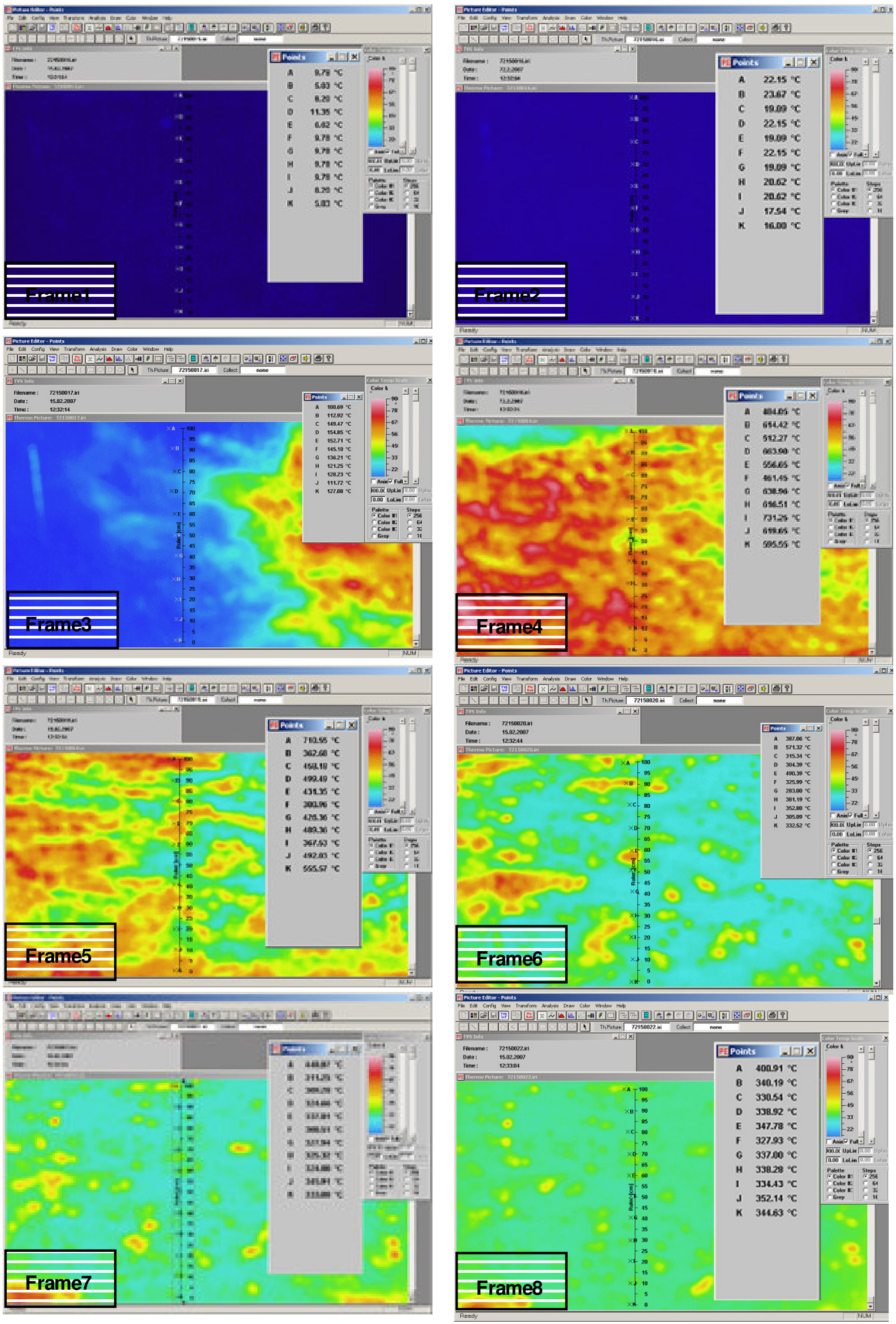


Figure 4-35 Sequence of 8 thermo photos at 10 seconds intervals filming an headfire (coming from the left side).

The thermo photos sequence showed how temperatures, measured at fixed points in the field camera, starting from an environmental temperature (Frame 1), reached values up to 800 °C in 10 seconds after the fire front arrival (Frame 3-4) in the camera field. Once the head of the fire front is passed temperatures quickly slowed down to 300 °C in 20 seconds (Frame 5-6). Processing several thermo photos sequences (Exp. 5-8) an average time temperature profiles (TTP) for back and for head Heath fires was drawn (Figure 4-36).

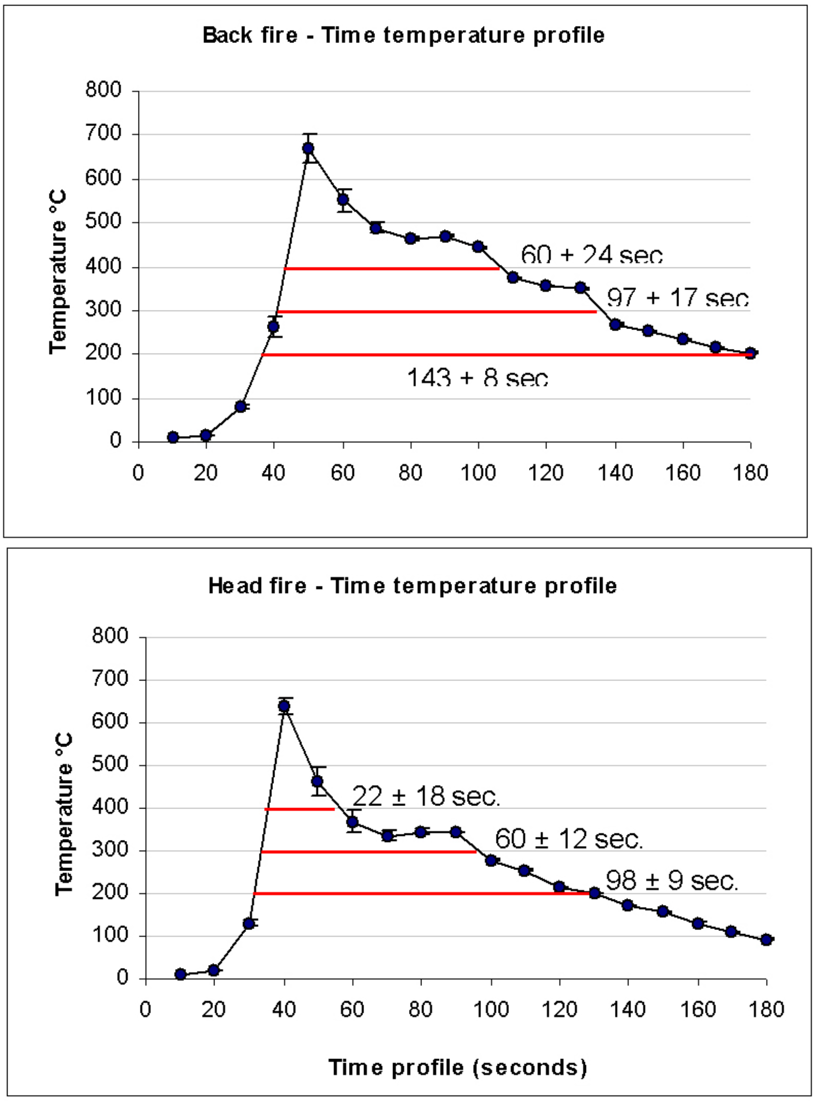


Figure 4-36 Average Time temperatures profiles both for back then for head Heath fires. Average elapsed times (± SE) above critical temperatures (200°C; 300°C) are displayed.

A t-test analysis evidenced significant differences ($p < 0.01^{**}$) between back and headfires in TTP above critical temperatures (400°C, 300°C, 200°C). Consequently, despite similar pick temperatures were reached both in back then headfires (up to 800°c), the results show how slow moving backfires in heath fuels were characterized by longer intervals in which critical high temperatures were exceeded: more then 25 seconds difference above 400°C, 30 seconds difference above 300°C and 60 seconds difference above 200°C in comparison with headfire. This phenomena could explain the more successful *Calluna* regeneration after headfires which has been evidenced before.

These results are in accordance with previous studies about pick temperatures and their profiles manifested by heath fires. Fires with higher duration have been proved to have a deleterious effect on *Calluna* resprouting potential (Kayll and Gimingham 1965): in fact newly developed buds at damaged scars are unlikely, and the residence of temperatures may affect below-ground buds. Moreover resprouts developed from dormant adventitious buds or their associated hormones could be damaged by heat (Mohammed and Gimingham 1970).

In Scotland, 'Muirburn' practice is characterised by fast moving headfires, with wind speed below 10 Km/hr (SEERAD 2001, Davies 2005). Studies on 'Muirburn' fire characteristic recorded pick temperatures at ground level ranging from 60 to 840 °C (Whittaker 1961, Kenworthy 1963, Kayll 1966, Hobbs and Gimingham 1984a). All studies measured temperatures at ground level higher than 400 °C for a time interval lower than 60 seconds. Consequently quick fires, that is to say wind driven fires, should be suitable to regenerate *Calluna* stands. In previous literature, no data were available for back fires average TTP with which to compare data obtained at Vauda. Moreover, these results should be furthermore tested with more experimental data and by comparison with TTP measured by using thermocouples.

4.3.7. Fire frequency effects on vegetation

To study the effects of fire frequency, a comparison between the effects on sward composition and tree stand structure of plot burnt only once in 2005 and plot burnt 3 times in 2005, 2006 and 2007, was carried out.

In Table 4-14 the CS% of grass and shrubs species as observed in Spring 2007 in plots burnt 1 time (in 2005) and plots burnt 3 times (in 2005, 2006 and 2007) is displayed.

Frequent fires seems to greatly enhance *Molinia arundinacea* cover, as it has already been stressed before about the marked resilience of this specie after fire (par. 4.3.5). In accordance with previous studies (Hobbs and Gimingham 1987, Gimingham 1993), burning each year had a negative impact on heather regeneration which is almost absent in plots burnt 3 times (CS% = 1%), while after a single burn it started gradually to recover under the grass layer (CS% = 8%).

Burnt 1 time		**Burnt 3 times**	
Specie	**CS %**	**Specie**	**CS %**
Calluna vulgaris	8 ± 2.2	*Calluna vulgaris*	1 ± 0.5
Molinia arundinacea	53 ± 1.9	*Molinia arundinacea*	61 ± 1.8
Agrostis tenuis	1 ± 0.5	*Agrostis tenuis*	1 ± 0.5
Carex panicea	7 ± 1.3	*Carex panicea*	9 ± 1.8
Carex tumidicarpa	4 ± 0.6	*Carex tumidicarpa*	1 ± 0.2
Panicum acuminatum	8 ± 2.9	*Panicum acuminatum*	14 ± 4.4
Potentilla erecta	3 ± 1.1	*Potentilla erecta*	2 ± 0.8
Salix rosmarinifolia	4 ± 1.7	*Salix rosmarinifolia*	1 ± 0.9
Others	12	*Others*	10

Table 4-14 Specific contribution (CS%; ±SE) of shrub and grass species in Spring 2007 as observed in plots burnt 1 time in 2005, and plots burnt 3 times (2005, 2006, 2007).

It is interesting to notice the high specific contribution of *Panicum acuminatum* Swartz (new name *Dichanthelium acuminatum*, common name Wolly panicum) both in plot burnt 1 (CS% = 8%) or 3 times (CS% = 14%). Woolly panicum is a perennial exotic grass native of California which usually appears in early secondary succession reproducing by seeds. Moreover it shows a great adaptation to frequent fires regenerating vegetatively by basal buds which may sprout after aerial portions are burned. Woolly panicum presence in response to fire varied with frequency and severity of fire. It occurs with greater frequency following annual fires than following less frequent fires. However, if the fire is severe enough to damage buds and roots, woolly panicum presence declines (DeSelm et al. 1974).

According to fire frequency effect on tree population, in Figure 4-37 the tree stem distribution by size classes of root collar diameter as observed in 2004 before burning, and in 2007 after burning in plots burnt 1 time or 3 times is showed. Differences between site 3 and 4 were not studied. Consequently frequency values showed are averages measured grouping data from both sites.

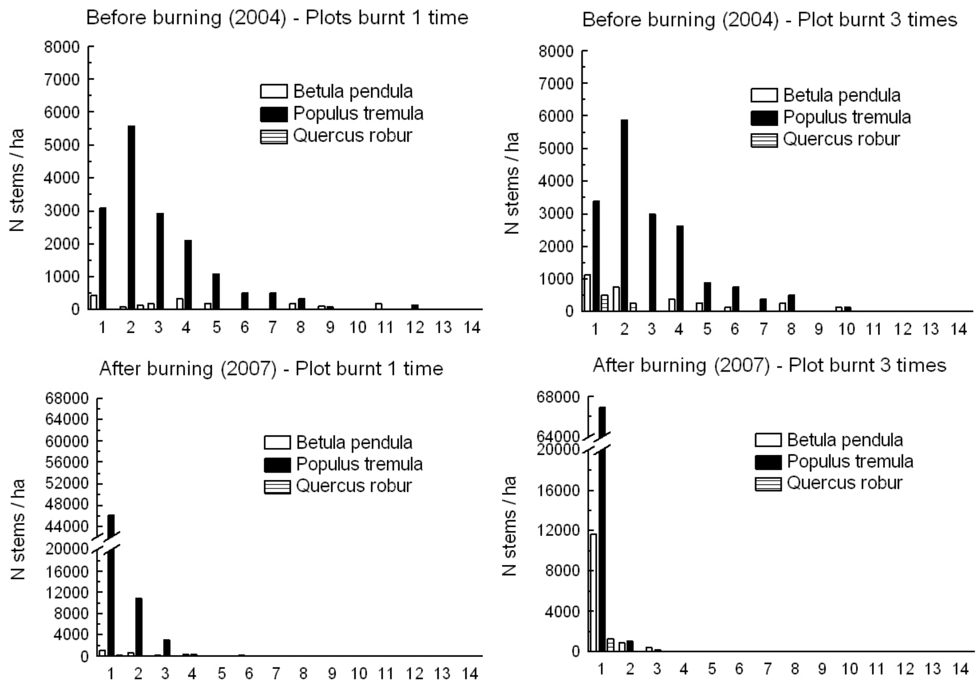


Figure 4-37 Tree frequency (N stems/ha) distribution by root collar diameter classes (0.5 cm) before (2004) and after burning (2007) in plots burnt 1 time (left) and plots burnt 3 times (right).

Tree distribution 3 years after fire in plots burnt 1 time showed a longer right tail both for *Betula* then *Populus*, with suckers resprouted the first growing season after fire in Spring 2005 up to 3 cm in Dc. The tree stem distribution is similar to the one showed by *Betula* and *Populus* in site 2 before starting the experimentation (Figure 4-17) thus supporting the thesis of a recent wildfire event which occurred in site 2. The C, around 3600 m^2 before burning in 2004, decreased to 1039 m^2 in 2007, with a reduction of about the 72% of canopy cover.

Tree distribution in plots burnt 3 times, as a consequence of stem mortality which followed each year after the fire treatment, shows few trees with a diameter higher then 1 cm. Nevertheless, both *Betula* and *Populus* stumps were able to produce each year a high number of resprouts, as it is showed by the high number of stems in the first diametric class (up to 67000 stem/ha, 30% higher then in plots burnt 1 time). The C, around 3800 m^2 before burning, decreased to 229 m^2, with a reduction of about the 94% of canopy cover. A t-test evidenced significant differences in standardized values of C and G between plots burnt 1 time and plots burnt 3 times.

4.4. Conclusion

4.4.1. Fire, effects and management implications

Fire experiments at Vauda showed that a fire front in a *Calluna* stand is not an homogenous fire treatment. Micro-relief, fluctuations in the speed and direction in the wind, variation in the moisture contents of plants and variation in the kind and spatial distribution of available fuel can all have marked effects on the behaviour, temperatures, *fireline intensity* and consequences on vegetation responses.

All fire experiments at the MNR of Vauda were carried out on a flat terrain and consequently slope was not a determinant factor of fire behaviour. Moreover this land characteristic facilitated the application of prescribed fire enabling the CFS and AIB teams to reach the experimental plots with all the necessary devices to carry out successfully the experiments. In three years of experimentations no escaped fires or injuries occurred.

On the contrary fuel characteristics highly determined fire behaviour. Variation in fine fuel moisture content within the plot was relatively low and in most cases the coefficient of variation was inferior to 10%. The implications of this variation in fuel moisture for fire behaviour are, however, not necessarily great if such differences occur only on the micro-scale creating a fine matrix. Nevertheless, spatial variation will become more important where large areas have lower fuel moisture content than surrounding zones, such as an area with higher cured *Molinia* cover, as this will allow an increase in rate of spread when a fire moves into this drier region. This aspect must be carefully taken into account in planning and conducting a prescribed fire because could determine sudden changes in fire behaviour. Fire experiments at Vauda showed how Grass fires reach rates of spread significantly higher then Heath fires despite similar weather conditions. The reason of this behaviour it is due to the lower live fuel rates which characterizes grassland fuels in winter, mainly constituted by cured grasses. In fact, the lack of a mechanism for active uptake of water gives to cured grasses a crucial role in fire behaviour as they not only dry out more rapidly but also reach ignition temperatures with lower energy inputs than corresponding live fuel particles, thus determining a higher rate of spread of the fire front (Cheney 1981, Cesti 2005, Davies 2005).

Variation in fine fuel moisture between fire experiments were more significant. Grass fires behaviour variability demonstrates the importance of dead fuel moisture content in driving fire behaviour and suggests that relatively small variations in the quantity may greatly affect fire behaviour. This happened in exp. 17, which failed as a consequence of Md up to 30%, and in exp. 19-22 in which a Md lower then 10% determined ROS up to 30 m/min. Such a fire front requires the intervention of many operators and means to control it and this is not consistent with the rationale of a prescribed burning plan.

Moreover experiments revealed a number of interesting trends which firstly confirmed that dead fuel moisture contents are much lower than those of live fuels. Dead fuels behave differently from live fuel components and their moisture status is primarily determined by time since last precipitation. Consequently, the planning of a prescribed fire treatment requires the set up a dead and live fine fuel moisture prediction systems based on correlation with precipitation, temperatures and drying conditions (i.e. windy days) over the preceding period the burning dates.

If the fine fuel moisture content has determined different fire behaviours between experiments, the wind speed and direction, the less predictable and most variable factor (Rothermel 1972), have been very variable within fire experiments determining dramatic changes in fire behaviour. Changes in wind direction have in fact transformed backfires with a *fireline intensity* lower then 100 kW/m in headfire of 4000 kW/m within few seconds and meters.

Despite the high variability within fire experiments, which made vain the experimental design of lighting fire fronts along the upwind or downside of the plots so as to perform a balanced number of back and head fires, the fire behaviour microplot analysis has enabled to describe quantitatively backfires and headfires in *Calluna* heathland fuels:

- Backfires are characterized by ROS < 5 m/min, residence times of active flame zone up to 50 seconds, flame height inferior to 1 m and *fireline intensities* inferior to 500 kW/m;
- Headfires are characterized by ROS > 5 m/min, residence times of active flame zone up to 30 seconds, flame height superior to 1 m and *fireline intensities* superior to 500 kW/m;

Consequently the resistance and resilience of *Populus tremula* and *Betula pendula* and the regeneration success of *Calluna vulgaris* after fire have been studied comparing backfires versus headfires effects.

Both for *Betula* and for *Populus*, a high stem mortality followed fire treatments in all burned plots. Head fires have a greater impact in scorching tree stems and crowns determining a higher probability of stem mortality as a consequence of higher scorched surfaces in comparison with backfires. Nevertheless, up to 7 cm in root collar diameter, both back and headfire determined a mortality between 90 and 100%. Low diameter individuals of *Betula* and *Populus* have thin bark and consequently both in back and headfire lethal residence time, flame height and temperature for plants' tissues are reached. Tree stem survival increases with larger diameters and tree height. These results are in accordance with previous studies about fire effects on plants which documented how tree survival increases with stem size (Brown and De Byle 1987, 1989, Bond and van Wilgen 1996, Williams et al. 1999). Nevertheless, considered the fact that large parts of the MNR of Vauda have an initial state of encroachment of juvenile trees, with Dc < 7 cm, the management implications of using back versus headfires seems to be the same as both determine the almost complete stem mortality of stands.

Moreover *Betula* and *Populus* showed a dramatic resprouts capability after fire. Results showed that there was little effect of differences in *fireline intensity* on resilience of trees. No differences were found comparing the effects of *fireline* *intensity* on stump mortality and average sprouts per stump; if the heating process is not transmitted downward in the soil, 100 or 3000 kW/m has the same severity on dormant buds below grounds.

Consequently the main reason in using headfires instead of backfires has to be found in the higher *Calluna* regeneration success that follows more intense but quick moving headfires. The vegetative regeneration of *Calluna* in areas where the microplot fire behaviour analysis evidenced the occurrence of headfires was in fact significantly higher comparing to the regeneration in backfire areas. Most of regeneration was from protected lateral meristems on the stem bases. Such a response after headfires has been observed also by Gimingham (1960) and Kayll (1966). The reason of this difference in the resilience of *Calluna* affected by back or head fires seems to be mainly due to the lower temperature time profile of head fires comparing to the back ones. The results obtained at Vauda with an infrared thermo camera confirm the ones reported by previous studies on 'Muirburn' fire characteristic which recorded pick temperatures at ground level ranging from 60 to 840 °C and measured temperatures at ground level higher than 400 °C for a time interval lower than 60 seconds using thermo sensitive material or thermocouples (Whittaker 1961, Kenworthy 1963, Kayll 1966, Hobbs and Gimingham 1984a). Nevertheless, as it has been pointed out in Chapter 3, to obtain reliable temperature estimates from radiation measures it is necessary to know the emissivity indexes of the flames (Meléndez et al. 2004). Sullivan et al. (1993) stressed the fact that using a value for flame of emissivity equal to 1 can be satisfactory when high measurement precision of flame temperature is not required. Consequently these previous results observed at Vauda must be accurately tested with a more consistent number of observations and compared with measurements obtained with other devices to validate infrared thermo analysis.

Another promising result about *Calluna* regeneration success after fire was also obtained describing the vegetation structure and composition before burning in site 2. Evidences of a recent wildfire, which occurred at least 6 years before the starting of the project in 2004 were present in site 2. The vegetation survey before fire experiments described a vegetation cover dominated by *Molinia arundinacea*, with a CS% equal to 59%, but *Calluna vulgaris* seemed to regenerate successfully showing a CS% up to the 22% of the cover, thus confirming its well documented resilience to fire.

Despite the *Calluna* regeneration success after prescribed or wildfires looks promising on a short-term period, several concurrent post-fire vegetation dynamics must be carefully assessed on a longer period. On one side fire induced a high resprouts reaction of tree individual which quickly re-established a canopy cover thus wielding a competition effect on *Calluna* young shoots. In absence of other disturbances the tree stand seemed to evolve towards a thicket stand of juvenile trees. On the other side fire greatly enhances *Molinia arundinacea* cover shifting heathlands towards grassland with the appearance of fire adapted exotic grass species such as *Panicum acuminatum*.

It is thus evident that a sole fire treatment is not a suitable action for heathland conservation management despite its effectiveness in determining a high tree stem mortality. Consequently, after an initial fire treatment to regenerate *Calluna* stands and induce tree stem mortality, it is thus necessary to control suckers and depress *Molinia* with other management actions rather then fire. In fact frequent fires can control the resprouts population, determining high rates of mortality of young suckers thus reducing the tree competition for lights (decreased canopy cover) with the shrub and grass species. Nevertheless, frequent fires had a negative impact on *Calluna vulgaris* regeneration and highly advantage *Molinia arundinacea* which after 3 years of repeated burning shows a complete dominance. There are evidences that grazing and mowing after a single fire will be effective in limiting *Populus* suckers density (Bartos and Mueggler 1981, Bailey et al. 1990, Dockrill et al. 2004). The Vauda project has thus been designed to test also the applicability of goat grazing and mowing following fire and a long term monitoring of the effect of different treatments will highlight the suitable management tool for conservation management of *Calluna* heathlands.

4.4.2. Prescriptions for heathland management by prescribed fire

Despite it has been demonstrates that the sole fire treatment cannot be a suitable tool for heathland conservation management, the results obtained with the Vauda fire experiment give important information, whether under an ecological or an operative perspective, that enable to drawn conclusion on the ecology of *Calluna* heathlands and tree encroachment in relation to fire in NW Italy. Moreover they enable to elaborate a preliminary set of “prescriptions” for prescribed burning treatments which should be followed by grazing or mowing to control tree resprouts and *Molinia arundinacea* cover.

- Prescribed fire treatment must be applied during the dormant season in a period between the second half of January and the end of March.
- The operative window of available days for burning after a rainy day goes from 5 to 20 days since last rain; exceeding these period will lead to operate in too dry environment. Windy period, snow falls and fog banks have to be reckoned with.
- The desired dead fine fuel moisture content should range from 15% to 25% while the total fine fuel moisture between 30% and 40%.
- Wind speed has to range between 1 to 10 km/h; air humidity between 30% and 60%; air temperature between –2 to 15 °C.
- Fire treatment has to be applied on an area ranging from 300 to 600 m^2 along strips 10-20 m wide and 30 meters long. The perimeter of the area to be burnt have to be previously sprayed with water in order contain the fire front inside the boundaries of the established area.
- The fire front has to be ignited using the ‘strip head fire’ technique (Pyne et al. 1996): this ignition pattern adapted to heathland management consists in preparing a buffer zone (if does not exist yet) by burning or spraying with water a strip of land 10-20 m wide and 2 m deep; subsequently an operator, using a drip torch, will ignite the fire front along a strip 10-20 m wide at distance of 2 m downwind the buffer and parallel to the buffer line so as to perform a wind driver fire which quickly will burnt out into the buffer zone. Subsequent 10-20 m wide strips should then be ignited downwind parallel to the first strip at a distance of 5-10 m one from each other in order to have short headfires which burnt out in the ‘black’ area already burnt by the upwind strip head fire. This technique will enable 4 operators, equipped with 3 drip torches ad a 1000 litres water-truck, to cope with sudden changes in wind direction or speed and control safely the fire front. A well trained team in suitable fire weather conditions should be able to burn up to 20000 m^2 per work day.

- The above mentioned prescriptions should thus determine headfires with a ROS between 5-10 m/min and fireline intensity ranging from 500 kW/m up to 2000 kW/m suitable to regenerate *Calluna vulgaris* stands and scorch juvenile trees of *Populus tremula* and *Betula pendula*; higher fireline intensity could be necessary to induce tree stem mortality in areas where an advanced state of encroachment is present with trees larger then 7 cm at the root collar.
- Each year should be burnt the 5% of total *Calluna* heathland area inside the MNR of Vauda. Areas burnt in the same year have not to be adjacent in order to create a patchwork of different *Calluna* stand ages. Part of the land should be left untreated to enhance biodiversity.
- The fire return interval of the fire treatment has to be comprised between 10-20 years according to the *Calluna* regeneration success after the previous burning. A successful burn should determine a *Calluna* regeneration up to 30% of CS% in 10 years.
- Fire must be used when *Calluna* stands reaches an average canopy cover (CS%) of 50-60% and an average height of 30 cm which, according to present knowledges, should be reached in 15-20 years after a fire treatment.
- Prescribed fire treatment should be followed by grazing or mowing.

4.4.3. Prescribed fire for *Calluna* heathland conservation management in Piemonte: is this possible?

The Piemonte Region has been one of the first Italian Region to insert prescribed burning in the Regional normative (L.R. 16/1994). It is interesting to notice how the Art. 9 of the L.R. 16/1994 names this technique “controlled burning” (*fuoco controllato*) instead of “prescribed burning” (*fuoco prescritto*). This is a fine distinction to underline the fact the fire is intentionally applied under control on a limited area, but in a sense it belittles the meaning of using fire in a well defined fire environment, adopting specific technical procedures to achieve pre-determined management objectives (Pyne et al. 1996).

The Regional Law provides general objectives, such as "fire prevention, fuel break management and fire dependent ecosystem maintenance" but does not mention specific objectives to achieve in order to work out well defined management issues. The Art 9, comma 2, of L.R. 16/1994, refers to the FMP to provide for the technical procedures in applying prescribed burning; nevertheless the current FMP (Regione Piemonte 2007) barely regulate it. Prescribed burning objectives and prescriptions indicated in the FMP (Table 4-15) are general rules derived by foreign bibliography and not the result of scientific studies about local prescribed fire assessment of its sustainability. In fact the "prescription" does not concern only the environmental "windows" in which is possible to apply prescribed burning but also the management objectives to achieve and the performance indicators to monitor in order to assess its management effectiveness (Andersen 1999).

General prescribed burning prescriptions in Piemonte	
It is possible to apply prescribed burning	in the dormant season
	on slopes inferior to 20%, with head fire
	with *fireline intensities* lower than 120 kcal/m/s (500 kW/m)
	air humidity ranging from 30% to 50%
	fuel moisture content ranging from 7% to 20%
	air temperature ranging from –6 °C and +10 °C
	wind speed ranging from 3 Km/h to 15 Km/h

Table 4-15 Prescribed fire prescriptions by the FMP (Regione Piemonte 2007).

Moreover the authorization system in Piemonte is quite complex (Art. 9, comma 4, L.R. 16/1994): CFS proposes the use of prescribed burning and apply for the authorization at the competent councillorship, that assesses the occurrence of the fire environment and that is responsible of the control, suppression and mop-up phases during burning. In the execution of the burn the CFS can ask the collaboration of AIB Teams. The day of the burn the CFS must advise the territorial authorities, the Prefecture and the local Fire Brigades (Ascoli et al. 2005). Nevertheless at the moment few CFS functionaries have the required expertise to plan a prescribed burn. It is thus evident that nowadays the application of prescribed burning in Piemonte it is hampered by several constraints: on one side there's very little knowledge about *why*, *where*, *when* and *how* to use prescribed fire as a management tool; on the other side the complex authorization system lacks of the flexibility that is required to apply the prescribed burning technique..

Despite these constraints, in Regione Piemonte, the actual normative allows fire experimentations, and thanks to the scientific interest of the land administrators, to the collaboration of the CFS and AIB teams, and to a series of favourable conditions, it has been possible to establish a long-term experimental study to monitor prescribed burning effects for conservation management of *Calluna* heathland of NW Italy.

We must be aware of the difficulties in translating research results into prescriptions that are applicable by the MNR of Vauda land managers. All treatments have been applied from winter 2005 and at least 6 more years of vegetation monitoring are required to understand long-term *Calluna* heathlands responses to management. Like all experiments, the Vauda fire experiment is affected by logistic and resource constraints. Despite these constraints this experience hopefully will stimulate further discussion about the use of prescribed burning and will serve as a starting expertise to drive future work on prescribed fire studies in Piemonte.

5. Modelling coarse woody debris consumption by fire experiments in tropical savannas of Northern Australia

5.1. Introduction

Savannas are the predominant vegetation in the northern region of Australia, they cover approximately 20% of the country (almost 2 million km^2), with an arc-like distribution over Queensland, the Northern Territory (NT) and the northern part of Western Australia (Figure 5-1) (Walker and Gillison 1982).

Figure 5-1 Distribution of tropical Savannas in North Australia (left; Source: Tropical Savannas Management CRC; modified); savanna in Darwin (NT) area (right).

Although there are a range of savanna types most of them are characterised by a woody layer dominated by species of *Eucalyptus spp*. and *Acacia spp.* which form an open overstorey canopy (Figure 5-1). Approximately 80% of the 503 described eucalyptus species occurs in Australian savannas. In Darwin area, NT, two major species are present: *Eucalyptus tetrodonta* (F. Muell) and *Eucalyptus miniata* (Cunn. Ex Schauer), either single or together (Chen et al. 2003). Other important woody perennial taxa include *Melaleuca* (Myrtaceae), *Banksia* and *Hakea* (Proteacea), *Albizia* (Mimosaceae), *Bauhinia* (Fabaceae) and *Terminalia* (Combretaceae). The understorey is dominated by a variety of annual and perennial C4 grasses such as *Sorghum* spp., *Heteropogon* spp. and *Chrysopogon* spp. (Andersen et al. 2005).

4.4.4. Climate and fire in tropical savannas

The main features of the tropical savanna climate are the overall rainfall and length of the dry season. In Northern Australia, rainfall varies from < 400 mm/yr to >1800 mm/yr, and over 90% falls in the wet season between late October and early April (Figure 5-2). There is a dry season of about 7-8 months, but a considerable interannual variability in the duration and timing of the onset and end of the wet season is showed (Williams et al. 1998).

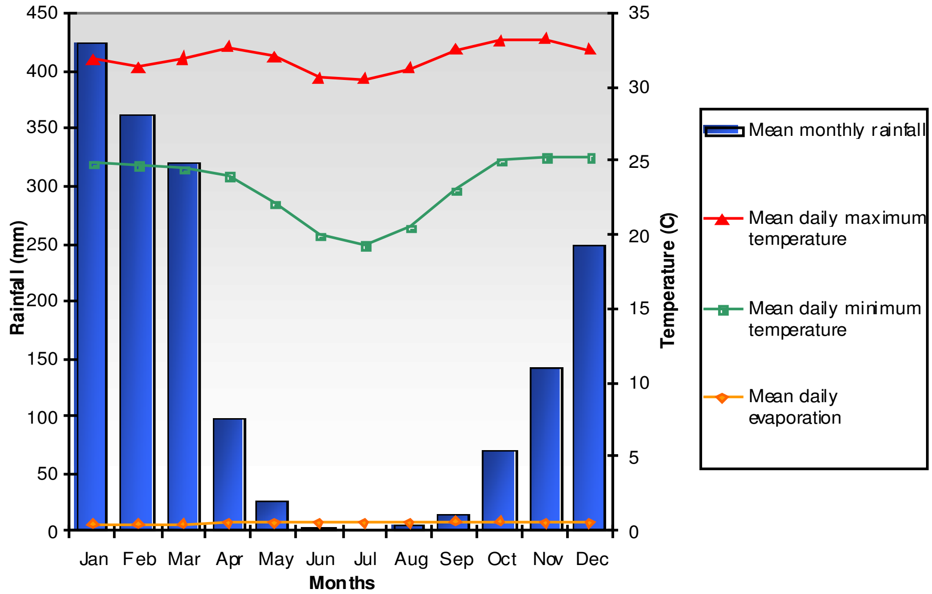


Figure 5-2 Mean monthly rainfall measured at the Darwin airport meteorological station on a time series from 1941 to 2004 (Source: Australian Government, Bureau of Meteorology of Northern Australia).

The alternation of wet and dry season is strongly linked with vegetation growth and sensitivity to fire due to the long dry out time (Gill et al. 1996). Therefore fire is a routine part of the ecological functioning of savannas as well as an inevitable part of the cycle of the vegetation growth, and is linked with its evolutionary history (Andersen et al. 1998).

Before the arrival of people, lightning was the ignition source for savannas. Pollen core and charcoal dating evidence indicates a correlation between an increase in the use of fire since the arrival of the first humans to Australia and the expansion of sclerophyllous vegetation and contraction of fire-sensitive communities (Hopkins et al. 1993).

Thus fire has been and arguably is the greatest natural and anthropogenic environmental disturbance in this region. Vast tracts are burnt each year by pastoralists, Aboriginal landholders and conservation managers (Williams et al. 2002). These fires started through human ignition, constitute the vast majority, although fires ignited by lightning associated with the onset of monsoonal conditions (typically November-December) may be a significant source in some locations (Russell-Smith et al. 2003).

To understand the seasonality of fire throughout the year, in Figure 5-3 an average repartition of the Forest Fire Danger Index (FFDI) (Gill et al. 1996), which values above 50 are considered extreme, is showed. Despite the rapid onset of the dry season, average daily FFDI in the early dry season remains below 20. They reach around 25 in the September-October period, with an average maximum FFDI during this month around 40 and maximum values that rarely exceed 60. FFDI declines again with the onset of the wet season but fire is still possible as a consequence of lightning and prescribes burning (Williams et al. 2002).

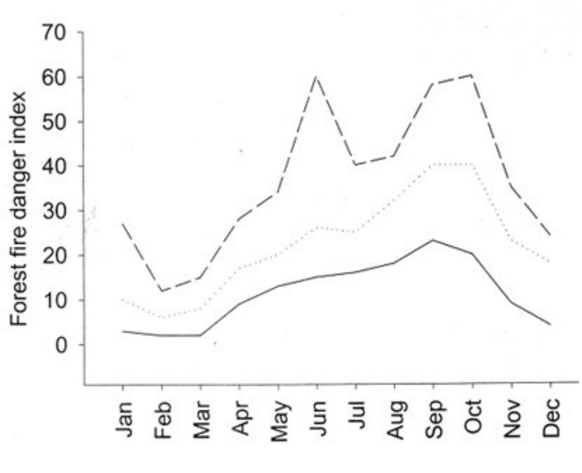


Figure 5-3 Average monthly values of the Australian Fire Danger Index measured at the Jabiru station, NT, from 1999 to 2006. Scattered line represents maximum absolute values; point line represents the average of maximum values recorded each year; continuos line represents the average of all the values recorded each day for all the years (source: Gill et al. 1996).

The reason of the FFDI distribution is due to the onset of the dry season which influences the *Sorghum* spp. grasses water content (e.g. *Sorghum intrans* F. Muell). These grasses dominate the understorey across most of Australia's tropical savannas and can grow as high as 3 metres in the wet season. They dry out throughout the dry season from the end of May until late October and reach their most flammable state when the next wet season is about to begin. In addition to the heavy growth of other grasses and herbs during the wet season, supplemented by litter from the woodland trees that are continually dropping leaf litter throughout the dry season, all this provides the fuel accumulation for savanna fires (Williams et al. 1998).

In the absence of fire, dry fine fuel load reaches a maximum (they may attain 10 t/ha) within about three to five years after a fire event because of the high decomposition rates in the tropics, however over much of the region average fuel load is 4.6 t/ha (Williams et al. 2004).

Fire intensity is seasonal, with early dry season fires (June-July) of low *fireline intensity* (e.g. 2000 kW/m) increasing as the dry season progresses and fuel load accumulates and cures (up to 8000 kW/m). However, by the late dry season and pre-monsoonal period (August-October), fire intensity can be an order of magnitude greater and these fires tend to burn over very large fronts and are damaging with crown scorch of > 90%. But generally, fires are of low to moderate intensity, typically being restricted to the grass-layer, and rarely, if ever, entering the canopy (Williams et al. 1998).

Fire is thus a routine part of the annual wet-dry cycle of tropical Australia, with up to half or more of some regions being burnt each year (Russell-Smith et al. 1997, Edwards et al. 2001). Consequently Fire management of eucalypt forests and woodlands in the wet-dry tropics of Australia has historical importance in relation to Aboriginal burning practices (Haynes 1985); ecological importance in relation to tourism and the conservation of flora and fauna (Andersen et al. 2005); contemporary economic importance in relation to the production of cattle; and a potential global importance in relation to the production of emissions which affect atmospheric composition and climate (Williams et al. 2005).

4.4.5. Savannas fires and carbon emission

As savannas are known to contain 30% of the terrestrial carbon, the close relation with wildfires, traditional and prescribed fire uses is assuming to have a particular effect on carbon fluxes and storage capacities (Williams et al. 2005).

Savannas in Australia are calculated to contain about one third of continental carbon stocks. For Northern savannas (Chen et al. 2003), it corresponds to an average of 204 t C/ha, of which 84% are stored below-ground (soil and roots) and 74% are specifically stored in soil minerals. Then, carbon stored in the different tree components represents the next major pool, with 24% of the total amount of carbon, followed by the inferior herbaceous stratum (0.5%), litter (0.5%) and dead branches (0.5%).

Open forest savannas systems are characterised by a highly dynamic carbon cycle, strongly dependent on seasonal variation and moisture. Hence, carbon turnover is rapid in this ecosystem and residence time are low (< 20 years) in comparison to other ecosystems (Chen et al. 2003). Vegetal growth is prolific during the wet season and most of carbon is fixed during this time. However, as the dominant trees are evergreen, reproduction, transpiration and photosynthesis occur in the trees during the dry season too (Williams et al. 2005). Thus the carbon fixation is still efficient with a Gross Primary Production (GPP) defined as the total carbon assimilated over area and time by photosynthesis, minus photorespiration, of 20.8 t C/ha/yr, of which 76% occurs in the wet season, and 24% in the dry season. Net Primary Production (NPP), defined as the difference between GPP and autotrophic respiration, representing the net result of CO_2 fixation by photosynthesis and CO_2 loss via plant respiration, is high, with up to 11 t C/ha/yr in non fire years (Chen et al. 2003).

There are two main measures (Williams et al. 2004) of the strength of landscape sequestration of carbon in various forests around the world: Net Ecosystem Productivity (NEP) and Net Biome Production (NBP) (Schulze et al. 2000).

NEP is defined as the carbon fixed by ecosystems due to NPP, less carbon emissions back to the atmosphere due to heterotrophic respiration. It is thus the net carbon balance of an ecosystem over a time period. NBP is thus NEP less carbon loss due to disturbances (fire, timber harvesting, insect plagues etc…). As NEP and NBP represent the annual change in C stored at an ecosystem scale, they indicate whether the ecosystem is a carbon "sink" or "source" for CO_2 relative to the atmosphere (Williams et al. 2004).

In savannas the NEP shows a value of 3.8 t C/ha/yr but is strongly linked to the seasonal fluxes, particularly dominated by wet season; in fact during the dry season the ecosystem is a weak carbon sink with 0.2 t C/ha/yr fixed. Thus, during the dry season, NBP will depend directly on the impact of fire on the ecosystem. Given the size of this ecosystem and the extent of burning, it is likely that savannas fires have a major impact on continental-scale carbon balance (Chen et al. 2003).

In 2005 the Australian National Greenhouse Gas Inventory (Australian Greenhouse Office 2005) has estimated an average emission of CO_2 by savanna burning over a period from 1990 till 2005 of about 8.7 Mt C/yr with an increase of 30.9% (Figure 5-4).

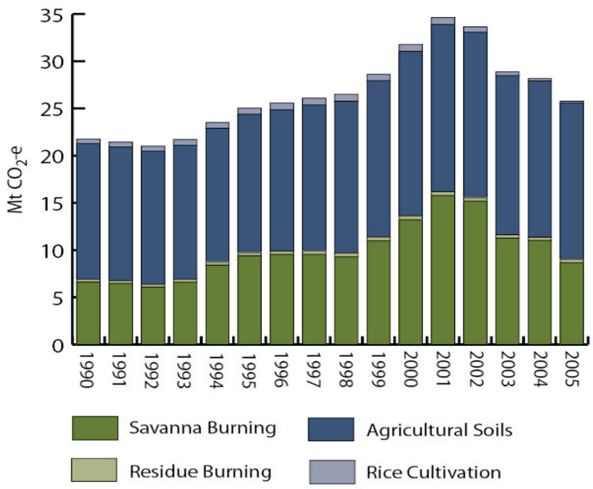


Figure 5-4 Trends in CO_2 emissions from savanna burning (Source: Australian Greenhouse Office 2005).

As savannas are undoubtedly a source of carbon greenhouse emissions and also a potential carbon sink, sequestration potential can be increased by managing for reduced fire frequency and extent over vast landscapes (Williams et al. 2005). Furthermore, in managing for carbon outcomes, via reduced fire frequency there are likely to be direct collateral benefits. For example, an appropriate conservation management of the broad range of plant and species requires a reduction in fire extent and frequency (Williams et al. 2005).

The main issue of the current research in Northern Australia is thus to know how to improve carbon storage in this specific ecosystem. To achieve this goal it is necessary to measure the components of the carbon stocks and fluxes in Australia's savannas in order to asse ss accurately the amount of carbon store or loss in all parts of the savannas.

Coarse Woody Debris (CWD) component of the fuel complex is significantly affected by prescribed burning (Knapp et al. 2005) and in general by wildfires (Williams et al. 2004), nevertheless is the least studied of the forest carbon pools (Woldendorp et al. 2004).

4.4.6. Coarse Woody Debris carbon pool

Coarse woody debris (CWD) is defined as dead woody material (including standing dead trees, stumps, dead branches, whole fallen trees, coarse roots, and wood pieces that have resulted from the fragmentation of larger stags and logs), in various stages of decomposition, located above the soil (Woldendorp et al. 2004).

The size distinction usually depends upon the ecosystem under examination and the data requirements. From studies that have just quantified CWD without litter, the minimum size has varied considerably, from 1 cm diameter (McGee et al. 1999) to ≥ 25 cm diameter (Bingham et al.1988).

CWD is an important structural component of many forest ecosystems and has a role in a number of aspects of ecosystem functioning, including maintaining forest productivity, nutrient cycling, sites for seedling establishment, providing habitat for vertebrates and invertebrates, contributing to soil and slope stability, and providing long term carbon storage (Woldendorp et al. 2005).

Although CWD is dead organic material that will eventually decompose, it can remain in an ecosystem for hundreds of years. In the main studies concerning carbon emission by fire, coarse woody fuels are not taken into account although they may significantly contribute to emissions. There are no data at present on the consumption of such fuels in Australian savanna fires (Williams et al.2004), although it is likely some coarse fuels will be consumed during dry season fires (Figure 5-5). Furthermore, this combustion in combination with a reduction of photosynthetic capacity (e.g. via reduction of the leaf area) may indeed make savannas subject to annual early or late season fires a source of carbon rather than a weak sink (Chen et al. 2003).

Figure 5-5 Burned logs after a wildfire; they are technically named 'Ghost'

The decay of coarse woody debris already have been studied through different decomposition processes such as respiration processes, biological transformation, fragmentation, leaching, weathering, and few studies have examined the relationship between CWD and fire (Knapp et al. 2005, Wright et al. 2002). Clearly further research is needed on the consumption of coarse in relation to fire (Williams et al. 2004).

4.4.7. Objectives

In the course of the 'Burning for Biodiversity Research Program' (http://www.terc.csiro.au/burningforbiodiversity/), designed by the Tropical Ecosystem Research Center (TERC), CSIRO, to test different fire regimes effect on savanna ecosystem in Darwin area, a fire experiment to study CWD consumption by fire was carried out.

The experiment objective was to elaborate a simplified model to assess how CWD consumption is affected by fire behaviour and its interaction with CWD size. Knowing the distribution of CWD by diameter classes and the rate of consumption per each diametrical class for a given *fireline intensity* (I) and residence time (Tr) allows to create models for predicting the CWD consumption due to a specific fire regime, either prescribed then wildfire, on an area.

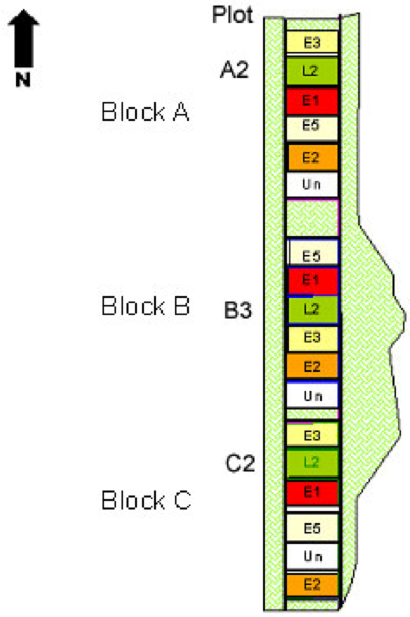
Nevertheless, *fireline intensity* varies significantly within a fire experiment along the plot both for weather variables and for heterogeneous fuel complex distribution, thus affecting the spatial pattern of fire severity (Byram 1959, Rothermel and Deeming 1980, Cheney 1981, Alexander 1982, Atkins and Hobbes 1995, Pyne et al. 1996). This variation in fire treatment has important effects on CWD dynamics and increase the unexplained variability in modelling its consumption. To cope with fire heterogeneity in experimental plots for modelling CWD consumption a detailed spatial analysis of fine fuel load, CWD consumption and rate of spread has been tried at a microplot scale in collaboration with the TERC, CSIRO, in the course of a study on carbon sequestration in tropical savannas in the Northern Territory, Australia.

5.2. Methodology

4.4.8. Study site

The study site was located 40 km South from Darwin (12° 42’ S, 131° 0’ E), NT, Australia, in Territory Wildlife Park (TWP), a 400 ha vegetation reserve in which fire has been excluded for at least 20 years. The experiment here documented was realized inside the plots of the ‘Burning for Biodiversity Research Program’ started by TERC in 2004 which has been designed to test effects of fire season, frequency and intensity on tropical savannas by experimental fire regimes.

The randomised block design, set up by TERC along the West boundary of the park, consists in an eighteen plots line made up with 3 replicates sites of 6 plots, each one treated with a particular experimental fire regime (Figure 5-6). Treatments from E1 to E5 simulate the common prescribed burning practices for fuel reduction, carried out from late May to middle July (early dry season fire) with different return intervals, from 1 to 5 years. Treatment L2 simulates high frequent wildfires commonly occurring in the area from August to late October (late dry season fire). A control plot (unburnt) is repeated in each site.



E1	Early burnt every year
E2	Early burnt every 2 years
E3	Early burnt every 3 years
E5	Early burnt every 5 years
L2	Late burnt every 2 years
Un	Unburnt

Figure 5-6 Experimental sites and treatments at Territory Wildlife Park (Source: TERC).

The vegetation at the site is *Eucalypt* open-forest savanna with an overstorey dominated by *Eucalyptus tetrodonta* F. Muell and *Eucalyptus miniata* Cunn. Ex Schauer. These two species contribute to the 70% of the overstorey leaf area index (LAI) and standing biomass (O'Grady et al. 2000). Sub-dominant tree species include *Erythrophyleum chlorostachys* Muell, *Terminalia ferdinandiana* Muell, *Eucalyptus porrecta* Blak and *Eucalyptus bleeseri* Blakely. The understorey is comprised of semi-deciduous and deciduous small trees and shrubs with a seasonally continuous cover of annual grasses such as *Sorghum* spp. Cured grasses in dry season (from late May to late October) constitute the main fine fuel which carries fire in these savannas.

Eucalyptus tetrodonta and *E. miniata* open-forest savanna are commonly associated with well drained lateritic red soils, which tend to have A horizons of well drained, highly weathered sands (clay content typically < 5%) (Chen et al. 2003) of low nutrient status, with a massive and earthy structure. Transition at 15-30 cm to a sandy loam B horizon is gradational and can extend up to 1-2 m, where ferricrete boulders occur in a matrix of mottled, heavy clays forming a duricrust of low permeability and variable depth. These soils are generally acidic (pH approximately 5.5) and low in available N and P (Calder and Day 1982).

4.4.9. Experiment design

In the frame of TWP late dry seasons fire treatments (treatment L2), carried out the 17th of October 2006 by CSIRO researchers, an experiment was replicated within the plots A2, B3, C2, already burnt in 2004 (Figure 5-6) to study CWD consumption as a function of *fireline intensity*, CWD diameter and density.

In order to have a consistent number of observations and replicates to fit a model, 405 wood logs of *Eucalyptus tetrodonta* and *E. miniata* were collected outside the plots in long unburnt areas. All logs collected were solid pieces with bark firmly attached; 1st -2rd decay classes according to Pyle and Brown classification (1999).

Logs were cut at a length of 30 cm and grouped in 4 diameter classes: 1) 132 logs from 0,6 to 1 cm; 2) 134 logs from 1,1 to 2,5 cm; 3) 89 logs from 2,51 to 5 cm; 4) 45 logs from 5,1 to 10 cm. As this attempt was a pilot experiment it has been decided not to use higher diameter classes. Logs were weighed, labelled and then grouped in sets of 9 logs constituted by a sequence of 3, 3, 2, 1 logs belonging to the 1st, 2nd, 3rd and 4th diametric classes respectively. A sample for each diameter class was dried to measure the moisture content.

Any single replicate of the experiment design was constituted by 3 sets of logs (3 observations per each replicate) placed inside an equilateral triangle (5 x 5 x 5 m) (Figure 5-7) evidenced on the terrain by three rods. Logs were disposed all in the same direction in order to be impacted by the fire front with the same angle within each replicate as this was supposed to be a possible factor affecting consumption.

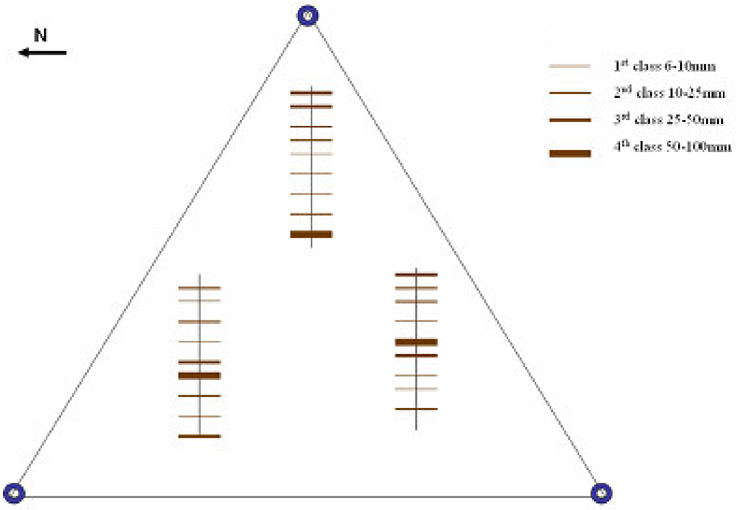


Figure 5-7 Sets of CWD inside triangle (Source: Bernard S.)

A number of 15 triangles (15 replicates) were distributed inside the 3 TWP plots (A2, B3, C2). In order to have an homogenous and continuous layer of fine fuel in each replicate, gaps in fuel distribution were filled with native cured grasses and litter collected outside the plot to reduce fire behaviour variability within the triangle one day before burning. Moreover to generate a range of *fireline intensities* between replicates fine fuel load was enhanced in some triangles (Figure 5-8). After having enhanced fuel, 4 samples per replicate were collected to estimate fine fuel load at a microplot scale, harvesting fine fuel in quadrates of 25 x 25 cm^2, weighing it in situ, and replacing it in the same spot in order to not affect fuel distribution with a destructive method. For each replicate a sample of 100 g was taken in order to measure fuel moisture and estimate dry fine fuels load on an oven dry weight basis.

Figure 5-8 Fuel modification inside 4 trianlges (side: 5 m).

4.4.10. Fire spread estimation

Rate of spread (ROS; m/s) was estimated using the Simard et al. (1984) methodology. This method calculates a value of rate of spread within a triangle knowing the time of arrival of the fire front to the vertexes of the triangle. The method have been developed for field fire experiments in order to solve a problem in ROS estimates; in fact the authors noted that timing the spread between a linear array of points assumes that the fire spreads parallel to a line drawn between the points, but in general this assumption is violated. They studied an equation (Equation 7 ;8) that measure ROS passing trough a triangle independently from fire spread direction.

$$\Theta = \tan^{-1}\left[\left(\frac{t_3-t_1}{t_2-t_1}\right)\left(\frac{b}{c\sin A}\right)-\frac{1}{\tan A}\right] \tag{7}$$

$$ROS = \frac{b\cos\theta}{t_2-t_1} \tag{8}$$

Where:

t_1, t_2, t_3 = times that the fire arrive at the first, second and third vertices of the triangle (Figure 5-9);

Θ = angle of spread relative to a base line between t_1 and t_2;

b = length of the base side from which Θ is measured;

c = length of the side which for with b the A angle;

A =angle formed by b and c.

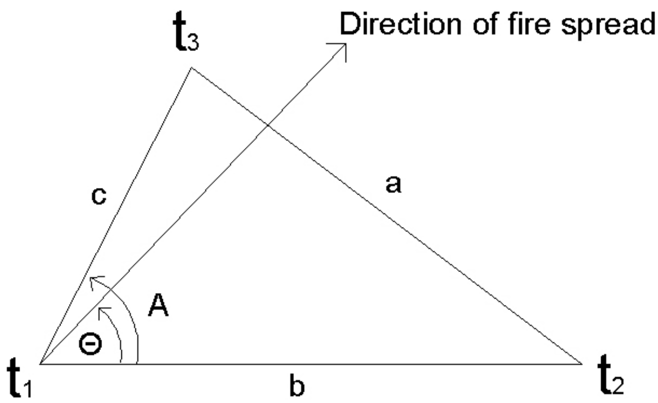


Figure 5-9 Fire spreading across a triangle (from Simard et al. 1984).

Measuring the time of arrival and departure of the fire front at the vertexes of each one of the 15 triangles was then possible to estimate the ROS and Tr of the fire front at a microplot scale thus having an average value for each triangle. Moreover, the time of arrival of the fire front to the vertexes of larger equilateral triangles (20 x 20 x 20 m), in a number of 2 per each plot, was also measured to estimate an average ROS, not affected by fuel modification, at a plot scale.

The fire front time of arrival and residence time was measured using a temperature-residence-time meter (TRTM) (Moore et al. 1995), buried before starting the fire experiments at the vertexes of each triangle according to the scheme showed in (Figure 5-10). The TRTM is a small electronic device that measures the time registered by a thermocouple above a certain defined temperature and consequently enables to record the time of fire arrival at, and departure from, the sensor. It is a instrument that is buried in soil with a thermocouple sensor exposed above ground in the path of a fire. The fire meter is comprised of a small plastic box incorporating a digital stopwatch and electronic circuits, a connector for a detectable thermocouple, and external electronic contacts (for batch battery charging of meters in a 'holding box' and for synchronizing the starting and the resetting of all meters) (Figure 5-11). The detachable thermocouple consists of 1 m thermocouple wire on a 60 cm lead of thermocouple extension wire. The meter is triggered (on or off) by a thermocouple-generated voltage equivalent to an approximate thermocouple temperature of 200 °C. This thermocouple temperature was chosen as being high enough to avoid triggering the meter in air and appropriate to indicate the presence of flames when placed 10 cm above the ground.

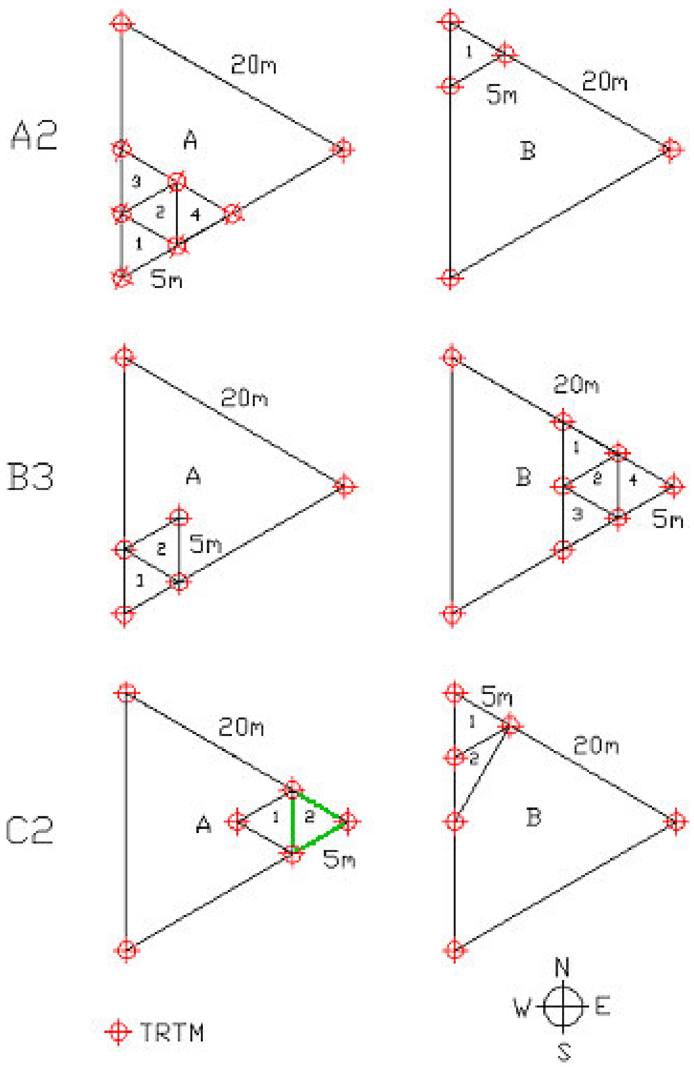


Figure 5-10 Triangle distribution inside plots A2, B3, C2. In each plot 2 triangles (20x20x20 m) are disposed in order to measure ROS at a plot scale. Smaller triangles (5x5x5 m) are placed inside the larger triangles having in common several vertexes in order to optimise the number of TRTM timers (red crosses). Replicates are identified by a code: i.e. C2A1 (evidenced in green colour) where C2 refers to the plot, A to the larger triangle and 2 to the smaller triangle.

Figure 5-11 Temperature-residence-time meter

4.4.11. Heat processes characterization

Fireline intensity (I; kW/m), was calculated at a microplot scale (one value per each triangle) applying Byram's equation (1959). Fine fuel load and moisture were estimated at a plot scale the day of each fire experiment harvesting 10 squares (25 cm x 25 cm) per plot. The consumption of fine fuels both at a plot then within triangles was equivalent to the entire amount of the dry fine fuel load as 100% of fine fuel was consumed as it is showed in Figure 5-12. The low heat of combustion of cured grasses of *Sorghum* spp. was assumed to be 20000 kJ/kg (Williams et al. 1998).

Figure 5-12 Replicates and log sets before (left) and after (right) burning

The intensity of fire front was also quantified estimating pick temperatures of flames by using sets of 20 thermo sensible crayons that melt in a range of 38 to 950 °C. At least 2 sets of crayons were placed in each triangle within the first 10 cm from the ground (Figure 5-13).

Figure 5-13 Sets of thermo sensible crayons placed on a log

4.4.12. Coarse woody debris consumption estimate

The CWD consumption within triangles was estimated for each log, recognizable thanks to the attached label, by subtracting from before-fire log weight the weight measured after fire. The moisture content of CWD was assumed to be equal before and after burning. The variable percentage of consumption was Log10 transformed to reduce variability within data. The effects of CWD diameter, residence time and fireline intensity on the dependent variable were tested by a full-factorial Analysis of Variance (ANOVA) and the Tukey Post-hoc test (Quinn and Keough 2002).

5.3. Results and discussion

4.4.13. Fire behaviour at a plot scale

The 17th of October 2006 three fire experiments were carried out by the TERC researchers in TWP reserve in collaboration with the fire fighter teams of the NT Bushfire Council (www.nt.gov.au/nreta/natres/bushfires/index.html). Plot A2, B3, C2 (100 x 100 m) were burnt according to the L2 treatment planned by the 'Burning for Biodiversity Program' experiment design (Figure 5-6). Fire was lit first along the up-wind side in order to create a black area of few meters to control the headfire lit along the down wind side immediately after.

In Table 5-1 the ignition time, average fuel load, moisture content, rate of spread, residence time above 200 °C and *fireline intensity* at a plot scale are given for each one of the three experiments. Experiments started in late morning when fuel have completed to dry out after night. Air temperature ranged between 25 and 30 °C, and wind speed ranged between 5 and 15 km/hr; these conditions were supposed to approximate fire weather conditions which occur in late dry season fires (even if much higher wind speed and temperatures can occur). An average fuel load of 3.6 t/ha was homogeneous between and within plots. The lower values of fuel load in respect of average fuel load in tropical savannas of 4.6 t/ha (Williams et al. 2004) is probably due to recent fire experiments carried out in the same plots in 2004. Fuel moisture ranged from 7 to 10% decreasing from late morning to midday hours with CV% < 10% within plots.

Exp.	Plot	Time	Wt (t/ha)	Mt (%)	ROS (m/min)	Tr (s)	I (kW/m)
1	A2	11:30	3.38 ± 0.32	10 ± 0.4	18 ± 1.2	47 ± 5	1994 ± 220
2	B3	12:05	4.04 ± 0.43	7 ± 0.2	22 ± 3.6	52 ± 7	2925 ± 395
3	C2	12:35	3.86 ± 0.62	6 ± 0.3	2 ± 0.6	79 ± 11	139 ± 70

Table 5-1 Experiments carried out in plot A2, B3, C2 described by ignition time, average (± SE) fuel load (Wt) and moisture content (Mt), rate of spread (ROS) and fireline intensity (I).

In exp. 1 and 2, as a consequence of constant wind direction, wind driven fire fronts (Figure 5-14) burnt most of the plot area showing a rate of spread and a *fireline intensity* higher comparing to exp. 3 where changes in wind direction determined a backfire behaviour of the main front (Figure 5-14) and a higher variability of *fireline intensity* (CV%=50%). Nevertheless ROS and I in exp. 1-2 where equal to the lowest range of values for late dry season fires as estimated experimentally by Williams et al. (1998). At the Kapalga fire experiment (Andersen et al. 1998) the authors measured an average *fireline intensity* of 7700 kW/m for late dry season fires (Williams et al. 1998). These results can be due to differences in plot size and ignition line length and their positive effect on acceleration of the fire front (Cheney and Gould 1993); at Kapalga plots were 20 km^2 in area and fire was allowed to reach a *quasi* steady ROS (Cheney 1981). Differences were also due to fuel load which at Kapalga ranged between 3 and 13 t/ha (Williams et al. 1998).

Figure 5-14 Headfire (left) and backfire (right) during fire experiments at TWP

4.4.14. Fire behaviour at a microplot scale

In 5 m side triangles, where fuel load had been manipulated, a wider range of fire behaviour descriptors was observed (Table 5-2) in comparison with plot scale values. Fuel load ranged from 2.13 t/ha, where no manipulation was done, up to 7.75 t/ha in triangles where cured grasses and dry leaves were scattered to increase fuel load. Fuel moisture ranged from 3% to 13% with an average of 8% and a CV%=12%.

Rate of spread between triangles ranged from 0.6 to 42.7 m/min as a consequence of different wind speed between experiments and variability in wind direction and speed within experiments. Residence time above 200 °C flame temperatures ranged between 42 and 101 seconds, showing higher values in backfires; nevertheless the Pearson Correlation analysis (Two-tailed test) showed a non significant correlation between Tr and ROS, and Tr and I.

Fireline intensity ranged between 74 kW/m and 8909 kW/m as a consequence of different rate of spread, fuel load and moisture between triangles reaching in several triangles values that approximate average values for late dry season fires as estimated by Williams et al. (1998). Finally pick temperatures estimated with pellets ranged between 800 °C and 900 °C; nevertheless in many triangles fire did not affected the set of pellets thus confirming the fact that this device is not suitable for a reliable estimate of fire pick temperatures in fire experiments.

Repl.	Exp. N°	Plot	Code	Wt (t/ha)	Mt (%)	ROS	Tr (s)	I (kW/m)
1	1	A2	A2A1	4.28	6	20.0	42	2850
2	1	A2	A2A2	2.67	8	15.7	45	1401
3	1	A2	A2A3	7.62	4	15.7	46	3994
4	1	A2	A2A4	7.58	10	26.9	40	6803
5	1	A2	A2B1	5.70	13	14.8	63	2817
6	2	B3	B3A1	3.25	4	9.1	51	985
7	2	B3	B3A2	6.63	12	8.2	56	1816
8	2	B3	B3B1	5.62	10	17.2	52	3216
9	2	B3	B3B2	7.20	9	37.1	60	8909
10	2	B3	B3B3	4.58	9	28.3	52	4329
11	2	B3	B3B4	3.80	6	42.7	74	5409
12	3	C2	C2A1	2.13	3	1.5	85	110
13	3	C2	C2A2	5.32	3	1.4	75	242
14	3	C2	C2B1	7.75	5	0.6	101	148
15	3	C2	C2B2	3.31	13	0.7	80	74

Table 5-2 Average values of 15 replicates identified by code, average fine fuel load (Wt), fine fuel moisture content (Mt), rate of spread (ROS), residence time (Tr), fireline intensity (I).

4.4.15. CWD consumption

Experimental fire fronts poorly affected CWD placed inside triangles in all diameter classes. Despite wood density and moisture content of logs were homogenous across diameter classes, average percentage of consumption in weight (% Cons.) was higher in logs belonging to low diameter classes, comparing to the consumption of logs with diameter higher then 25 mm (Table 5-3).

Diam. Class (mm)	N. logs	D (kg/m^3)	% M	% Cons.
1 (5 < Ø < 10)	132	857 ± 230	8 ± 0.6	22 ± 2
2 (10 < Ø < 25)	134	836 ± 80	10 ± 0.4	14 ± 2
3 (25 < Ø < 50)	89	834 ± 40	12 ± 0.3	6 ± 1
4 (50 < Ø < 80)	45	834 ± 30	10 ± 0.3	5 ± 1

Table 5-3 Number of logs, average (±SE) density (D), moisture content (% M) and percentage consumption in weight (% Cons.) for each diameter class.

CWD consumption variability was studied with an ANOVA test which factors were: diameter of logs, residence time and *fireline intensity* classes (Table 5-4). The dependent variable 'consumption' (% Cons.) was $\log_{10}$ transformed ($\log_{10}$%Cons.) to decrease the range of variability. Values of % Cons. equal or inferior to 1% were excluded to further reduce variability across groups. Before running the ANOVA, the null hypothesis that the error variance of the dependent variable is equal across groups was studied with the Cochrans's test. No significant differences were found in the variability of data across groups.

Factor Classes		**N.**
Diameter Classes (mm) (Diam. Cl.)	1 **(5 < Ø < 10)**	123
	2 **(10 < Ø < 25)**	124
	3 **(25 < Ø < 50)**	81
	4 **(50 < Ø < 80)**	34
Tr Classes (sec.) (Tr. Cl.)	1 **(50 < Tr < 60)**	96
	2 **(60 < Tr < 70)**	95
	3 **(70 < Tr < 80)**	51
	4 **(90 < Tr)**	120
Intensity Classes (kW/m) (Int. Cl.)	1 **(0 < I < 500)**	91
	2 **(501 < I < 2000)**	71
	3 **(2001 < I < 4000)**	98
	4 **(4001 < I)**	102

Table 5-4 Number of data divided in Factor classes used in the ANOVA.

The ANOVA test results (Table 5-5) evidenced significant differences in log10%Cons. between diameter classes of CWD. The Tukey post-hoc test (Table 5-6) distinguished between 3 subsets: 1, 2 and 3-4 diameter classes, evidencing significant differences ($p < 0.05$*) between logs with smaller diameters and logs larger then 25 mm. Residence time, *fireline intensity* classes and their interaction effect did not have a significant effect on the dependent variable meaning a week effect of fire behaviour on CWD consumption. The fit of the model was low ($R^2 = 0.341$), consequently most of the consumption variability was not explained by factors tested.

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	21,907(a)	45	,487	2,097	,000
Intercept	79,262	1	79,262	341,444	,000
Diam. Cl.	**8,561**	**3**	**2,854**	**12,292**	**,000**
Tr. Cl.	,547	3	,182	,785	,503
Int. Cl.	,397	3	,132	,569	,636
Diam Cl. * Tr Cl.	1,394	9	,155	,667	,739
Diam Cl. * Int. Cl.	1,005	9	,112	,481	,887
Tr Cl. * Int. Cl.	2,996	8	,374	1,613	,120
Diam Cl. * Tr Cl. * I Cl.	,375	10	,038	,162	,998
Error	73,356	316	,232		
Total	362,838	362			
Corrected Total	95,262	361			

Table 5-5 ANOVA test results; dependent variable: log_{10}%Cons.; factors: CWD diameter classes (Diam Cl.), Residence time classes (Tr. Cl.), fireline intensity classes (Int. Cl.)

Diameter Classes	N	Subset (log_{10}%Con.)			Subset (%Cons.)		
		1	2	3	1	2	3
4	34	,6010			3.99		
3	81	,6371			4.34		
2	124		,8611			7.26	
1	123			1,0765			11.93

Table 5-6 Tukey post-hoc test results

5.4. Conclusions

An experiment to study a simplified model of CWD consumption of different sizes by fire behaviour was attempted in the frame of 3 late dry season fire experiments carried out by the Tropical Ecosystems Research Centre (CSIRO), according to the 'Burning for Biodiversity Program' in Territory Wildlife Park, Darwin (NT).

Fire behaviour observed and quantified during the experiments at a plot scale showed a *fireline intensity* ranging between 139 and 2925 kW/m; these values are lower comparing to average value of 7700 kW/m measured experimentally for late dry season fires in previous fire experiments (Williams et al. 1998). Nevertheless, enhancing fuel load inside 15 equilateral triangles (5 m side), placed along burned plots, resulted in as many fire behaviours at a microplot scale within the same experiments. As a consequence of differences in fuel load, wind speed and direction 15 different values of *fireline intensities,* ranging between 74 up to 8909 kW/m, were recorded. These values span the spectrum of *fireline intensity* both of early and late dry season fires in tropical savannas of Northern Australia (Williams et al. 1998). Consequently, the manipulative method used enabled to better simulate dry season fires and provided a wider range of values which is a prerequisite for fitting models (Quinn and Keough 2002).

The CWD placed inside triangles was poorly affected by fire behaviour with an average consumption of 5% for logs larger then 25 mm. The ANOVA test showed significant differences in consumption between diameter classes with a higher consumption of smaller logs. These result was expected as the log surface to volume ratio, which decrease with increasing diameter, is one of the main fuel characteristics which is inversely related to fuel ignitability (Allgöver et al. 2001). Consequently estimates of CWD consumption at a broad scale should take into account the distribution of coarse fuels among diameter classes and the probability of consumption in relation to wood size.

No significant differences in log consumption by *fireline intensity* and residence time classes was found, meaning a weak effect of fire behaviour. These results are not in accordance with previous studies on CWD dynamics which observed a higher consumption, which ranged between 50% and 80%, and showed a difference of 30% in consumption between early and more intense and severe late dry season prescribed burning (Brown et al. 1985, Knapp et al. 2005). The reason of this could be due to the different nature of coarse wood used in this study and to the constraints of manipulative fire experiments in simulating natural processes (Andersen et al. 1998).

The CWD placed in triangles was collected in a long unburnt area and had never been affected by a previous fire. Logs belonged to the 1^{st}-2^{nd} decay classes of Pyle and Brown classification (1999) and consequently were less likely to be consumed by fire than decayed logs (Brown et al. 1985, Knapp et al. 2005).

Logs were cut 30 cm in length and then placed inside triangles where manipulated fuel was added. These modifications, consisting also in a deterioration and compaction of fuel inside and around the triangle, caused by the stepping on of several operators which prepared the experiment, could have determined a different behaviour and efficiency of flames in burning logs.

In conclusion, the experiment here exposed developed an interesting methodology in quantifying fire behaviour descriptors for correlation with fire effects which could be used in other ecological studies. Nevertheless, the results obtained evidenced also the limits of manipulating ecological processes which complexity must be accurately analysed before designing the experiment.

6. General conclusions

6.1. Heath and savanna fire experiments: a comparison

In this thesis two fire experiments, which studied short-term effects of experimental fire on vegetation dynamics, were exposed. The first study dealt with *Calluna* heathland conservation management by experimental prescribed fire in NW Italy. The second one dealt with modelling CWD consumption by experimental fire which simulate dry season fires in tropical savannas of Northern Australia. Despite the specific environment, research issues, objectives and relevance of results obtained, which clearly distinguish these two studies, the common issue of three year of research has been to develop an expertise in designing fire experiments to asse ss fire effects on ecosystem dynamics.

In both studies a rigorous experimental approach, incorporating appropriate spatial scales, adequate replication, the collection of extensive baseline (pre-treatment) data and detailed measurements of fire behaviour were adopted. Fire behaviour was analysed at a microplot scale (Smith et al. 1993, Fernandes et al. 2000) in order to cope with fire heterogeneity within a fire experiment and have a precise estimate of fire descriptors for correlation with specific effects. This analysis has been proved to be required as in both studies significant differences between average values measured at a plot scale and values measured at a microplot scale were observed. Moreover this methodology enabled to cope with typical constraints which characterize the modelling of experimental fire effects at a plot scale: the inadequate number of repetitions and the small range of observations (Smith et al. 1993, Andersen et al. 1998). The microplot scale analysis in fact provided a consistent number of repetitions and a wider range of observations with a relative few number of fire experiments.

Nevertheless, the microplot scale analysis of fire behaviour was carried out adopting two different methodologies. In heath fire experiments it was used a grid of rods to estimate rate of spread by timing the time of arrival of the fire front to rods. This methodology assumes that the fire front spread direction is parallel with the grid axis. In savanna fire experiments ROS was estimated timing the arrival of the fire front to vertexes of equilateral triangles and then calculating an average value for each triangle using the Simard's formula (Simard et al. 1984). This method enables to estimate ROS independently form fire front direction. Thus, considering the high variability of fire spread direction as a consequence of wind changes, topography and spatial pattern of fuel distribution (Cheney 1981), the second methodology must be preferred to the first one. Moreover the single triangle can work as a survey area in which effects can be monitored and correlated to fire behaviour descriptors.

Another difference between heath and savanna fire experiments regarded the timing of arrival and departure of the fire front at fixed point. In heath experiments they were visually estimated by three operators; in savanna fires they were measured using temperature-residence-time meters (Moore et al. 1995). Undoubtedly the second method is more precise, unbiased and reproducible as the TRTM measures the fire front time of arrival when a pre-determined flame temperature is exceeded; nevertheless, it requires also a visual estimate to cope with the malfunctioning of the electronic device.

The TRTM used in savanna fires demonstrated to be also a useful device in measuring the fire front residence time above a fixed flame temperature. Diversely in heath fires the temperature-residence time was estimated using an infra-red thermo camera. Processing thermo images with apposite softwares enabled to measure an adequate number of temperatures with spatial and temporal resolution; consequently temperature-residence time profiles were derived. Nevertheless the use of a thermo camera in field experiments has several constraints; the cost of the technology, up to 30000 euro, the need of a professional operator and the risk in exposing the camera to flames must be taken into account; moreover, as it has been observed in conclusions of Chapter 4, the emissivity of flames, required for a reliable estimate of IR fire temperatures, is extremely difficult to determine due to the complex nature of the flame environment (Chandler et al. 1983, Pastor et al. 2002). Thus, using a value for flame emissivity equal to 1 can be satisfactory when high measurement precision of flame temperature is not required (Sullivan et al. 1993).

Despite constraints in measuring the temperature-residence time, the experiments here exposed demonstrated the importance of determining this parameter in understanding the effects of fire behaviour on ecological processes. In heath fire experiments, *Calluna* regeneration was surprisingly more successful where high *fireline intensity* fire fronts occurred (I > 500 kW/m). This regeneration pattern could be explained by the lower values of temperature-residence time showed by fast moving wind driven headfires. This relation was observed also in savanna fires. Consequently, in accordance with previous studies (Whittaker 1961, Kayll 1966, Cheney 1990, Moore et al. 1995), estimating the temperature residence time is more relevant then measuring fire pick-temperatures which showed little differences between low and high intensity fires (Bond and van Wilgen 1996). In connection with this, in both studies the use of pellets melting at different temperatures to estimate fire pick-temperatures did not turn out in a reliable device.

Further considerations can be expressed about plot size. Firstly a square plot size, as it was in savanna fires (100 x 100 m), must be preferred as it enables to light the fire front from the preferred side according to wind direction and to the desired fire treatment (back, head, flank fire). Secondly, plot area, higher in savanna fires (roughly 1 ha) comparing to heath fires (up to 0.4 ha), can be of relative importance. In fact it must be distinguished between fire experiments designed to simulate wildfire behaviour at a landscape level and fire experiments projected to study prescribed fire behaviour effects. In the first case small experimental plots preclude a valid assessment of the spatial patterning common in low-intensity fires, and may not allow for the development of high-intensity fires (Marsden-Smedley and Catchpole 1995). In fact larger the ignition line faster the fire front will reach a *quasi* steady rate of spread and will express the potential intensity (Cheney and Gould 1993, 1995). Further, the securing of small plots with fire-breaks disrupts ecological processes such as surface hydrology and soil erosion, and spatial scales are inappropriate for landscape-scale issues such as faunal movements, grazing impacts, stream hydrology (Carpenter 1996, Andersen et al. 1998). Consequently small plot limits the applicability of the results.

In the second case, when prescribed fire effect must be assessed, plot size can be secondary. In fact prescribed fire is applied on limited areas and fire behaviour is constrained anyway by specific prescriptions (weather, topography, ignition pattern). The largeness of the burned area depends from the specific prescribed fire treatment required to achieve stated objectives. In the case of *Calluna* heathland management strips 30 m long and 10 m wide are required (SEERAD 2001, Davies 2005), consequently experimental plot area, ranging from 625 to 4000 m^2, resulted appropriate.

In conclusion, the methodologies here presented can be useful in research studies on prescribed fire effects which aims to translate the research findings into prescriptions and burn plans. Despite results about modelling CWD consumption were not very substantial, as probably biased by excessive manipulation of experimental plots, the methodologies learnt implemented the Vauda fire experiment design. Finally previous results at Vauda enabled to state a set of prescriptions for *Calluna* heathland conservation management in NW Italy. Nevertheless, we must be aware of the difficulties in translating research results into prescriptions that are applicable by land managers.

6.2. Translating research result into prescriptions

Which is the suitable prescribed fire regime to maximize utilities?, which are the ecological effect on a long-term period?, which are the costs of a prescribed burn plan? It must be recognized that when selecting a prescribed fire regime to achieve management objectives (i.e. fire hazard abatement), there may be no 'optimum' regime for maximizing diversity, minimizing impact on ecosystem and abating management costs at the same time (Bond and van Wilgen 1996). Consequently in evaluating all these issues we must be careful not to jump to too far reaching-conclusions. Too frequently the decision to use fire has been based on as little information as it is the decision to suppress all fires, both being based on what appeared at the time to be self-evident reasoning (Johnson and Miyanishi 1995).

Moreover the introduction of prescribed fire in Europe and even more in Italy must be adapted to local land characteristics and societies. As there are few documented experiences of prescribe burning applicability in Europe comparing to overseas countries, European researchers find their scientific and management references in systems different from the one in which they work. Firstly it must be taken into account that land characteristics in Europe such as land use history, extension of the urban-forest interface, vegetation associations are different comparing to U.S.A., Australia or South Africa. Moreover must be recognized that societies are also really different. Consequently there's the risk, as in other branches of wildfire science, to study and apply prescribed burning under a misleading perspective.

For example, in addressing the ecological role of fire for conservation management in an European context, and even more in an Italian one, it is necessary to distinguish two different philosophies and perspectives. In fact, as it has been well expressed by Chandler et al. (1983), fire-dependent ecosystem maintenance can be viewed in two different contexts.

In the one, the aim is to keep the land in a "natural" state of evolution. Under this philosophy the ecosystem is to be allowed to follow its natural line of succession and fire should be allowed to play its natural role in the modification of that succession. Consequently in some natural parks or remote areas, when a wildfire occurs under a specific fire weather, it is let to burn until this "prescription" is satisfied and it is intended as a "natural prescribed fire" (Pyne et al. 1996). This philosophy developed in that countries in which still have sense to talk about "natural landscape" and "natural fire regimes", such as large part of North and South America, Africa, Australia and Asia, where wide areas are inhabited and man manipulation has been extensive or indirect.

In an European context it is even more difficult to distinguish a "natural" role of fire in an environment which has been intensely modified by human activities throughout history; moreover the use of "natural prescribed fire" seems to be inapplicable, considering the population density and the wide urban-forest interface and consequently the necessity to suppress any kind of fire for public security.

In the second context however, ecosystem maintenance has also been taken to mean preserving the "cultural landscape" as it exists or existed at a particular moment in human history: "Freezing the landscape, as it were, so that our grandchildren may enjoy the view as we ourselves saw it as children" (Chandler et al. 1983). In landscape ecology the concept of "cultural landscape" it refers to geographic areas in which the relationships between human activity and the environment have created ecological, socio-economic, and cultural patterns and feedback mechanisms that govern the presence, distribution, and abundance of species assemblages. There are many types of cultural landscapes, but all are historically dependent on initial landscape conditions and on the culture of a given time (Farina 2000). In Europe it is probably more appropriate to talk about “cultural landscapes” rather than “natural” ones; fire induced by man has been and still is one of the main human activities and disturbances that shapes the landscape in many parts of the Old Continent (Naveh 1975, Marzano 2005); nevertheless a past history of fires in an ecosystem is not necessarily justification for, nor does it provide enough understanding of the use of prescribed burning (Johnson and Miyanishi 1995).

6.3. …concluding

In conclusion it must be accepted on one side the relevance of transferring and developing knowledge about prescribed burning applicability in Italy in this moment, as fire management policies need new tools for preventing increasing wildfire occurrence. On the other side it must be recognized that the effect of prescribed fire on ecosystem dynamics is a complex subject that requires long-term and multidisciplinary "learning experiments" (Holling 1978). Despite constraints this study demonstrated that is possible to develop research in the field of prescribed burning in Italy and I hope it will stimulate further discussion in order to improve knowledge and thereby future management.

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8. Attachments

8.1. Documentary video

The documentary video here presented documents the Experiment number 1, realized the 21 of February 2005 at the MNR of Vauda. in site 4 on a Plot of 50 x 80 m. The fire behaviour of experiment 1 is also described by the Fire Behaviour Map showed in Attachment 2.

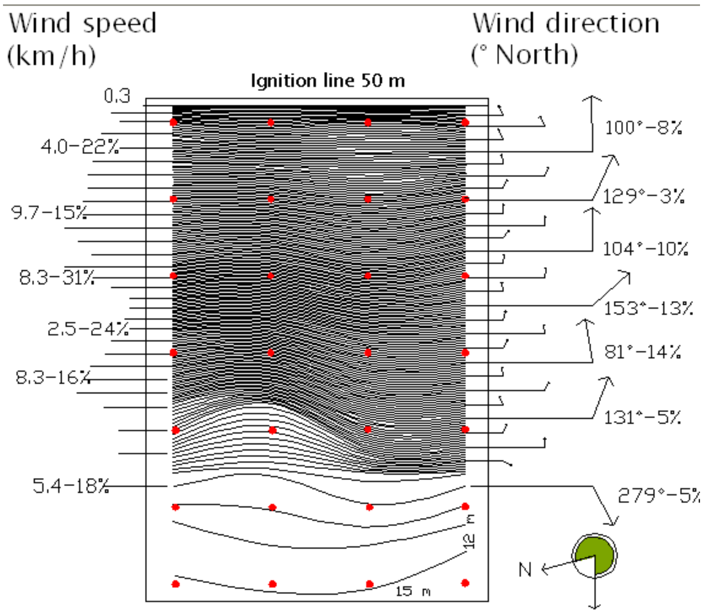
The video has no audio content but try to illustrates simply with images the ignition, control and mop-up procedures under an operative perspective. All the fire fighting teams which appear during the video belong to the CFS corps and to the AIB Teams of the Regione Piemonte. All the person filmed have been informed of the video content.

Moreover the video illustrates the operational procedures and the devices used to described the fuel characteristics and the fire behaviour of the fire front such as the regular grid of marked poles. the infra-red camera or the whether station; researchers are filmed in the intent of collecting data.

Finally this video clearly shows the dramatic differences in fire behaviour between a back fire and a headfire in *Calluna* heathland. In fact, the experiment, programmed as a backfire after 1 h 20’ characterized by constant wind speed and direction suddenly turned to a headfire for a change in wind direction and quickly burnt out in 5 minutes the remnant part of the Plot.

Two cameramen instructed about all the noteworthy issues filmed 9 hours of film which will be edited in future for further didactical or promotional proposes. The documentary video has been recorded on a DVD which is contained in the envelope placed at the back cover of the thesis binding. The video is also available at the following web site of the Dep. Agroselviter of the University of Torino: http://www.agroselviter.unito.it/pianificazione/vaudavideoENG.htm.

8.2. Fire Behaviour Maps



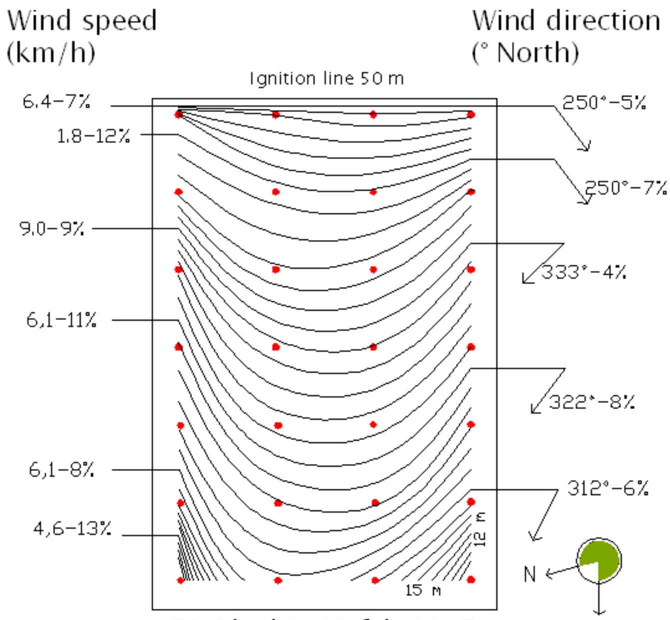
E1_Block4_21 feb 2005

Experiment 1
Site: 4

Date: 21/02/05
Time: 10:38am
Area: 4000 m^2

W_{TF} (t/ha): 10.3±1.2
Mt(%): 28 ± 2.2
U(km/h): 6.7 ± 0.9
T(°C): 2 ± 0.1
RH(%): 85%

ROS(m/min): 1.9 ± 1.2
I (kW/h): 434 ± 187



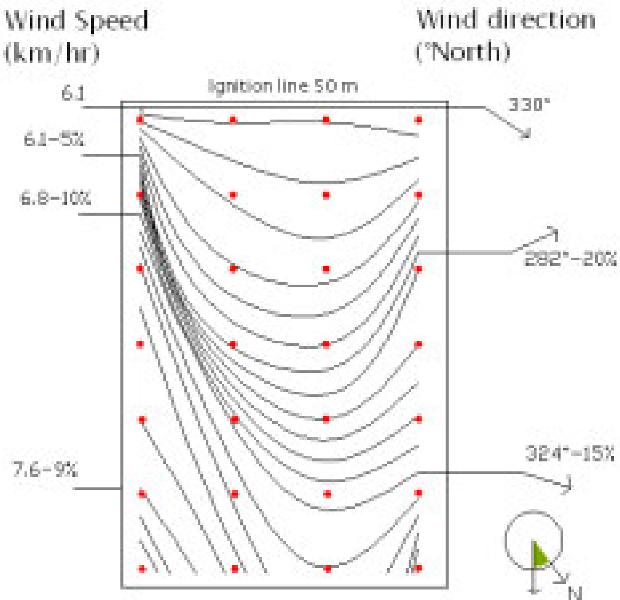
E2_Block3_21 feb 2005

Experiment 2
Site: 3

Date: 21/02/05
Time: 13:30pm
Area: 4000 m^2

W_{TF} (t/ha): 14.9 ± 0.8
Mt(%): 30 ± 1.2
U(km/h): 6.5 ± 0.6
T(°C): 2.3 ± 0.3
RH (%): 65%

ROS(m/min): 6.0 ± 0.8
I (kW/h): 2138 ± 201



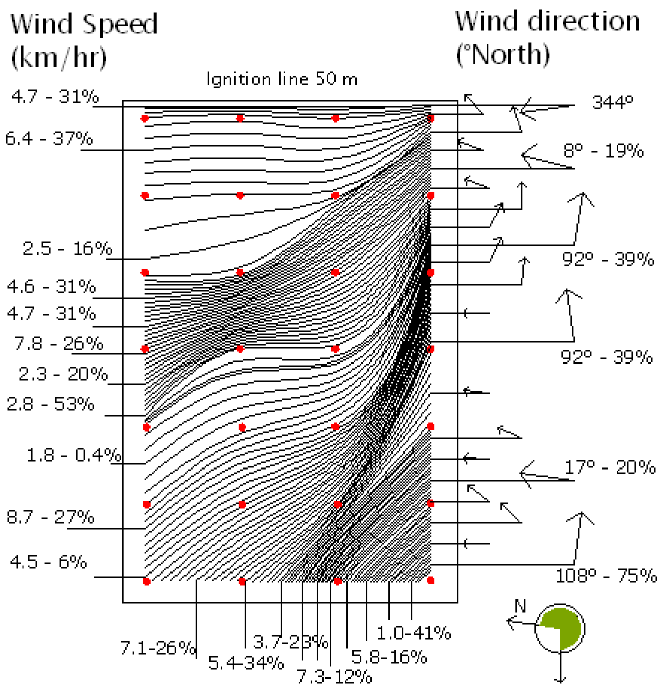
E3_Block 1_10 Mar 05

Experiment 3
Site: 1

Date: 10/03/05
Time: 10:28am
Area: 4000 m^2

W_{TF} (t/ha): 13.4±1.0
Mt(%): 32 ± 3.8
U(km/h): 7 ± 0.4
T (°C): 14 ± 3
RH (%): 25%

ROS(m/min): 11.8±1.6
I (kW/h): 3556 ± 496



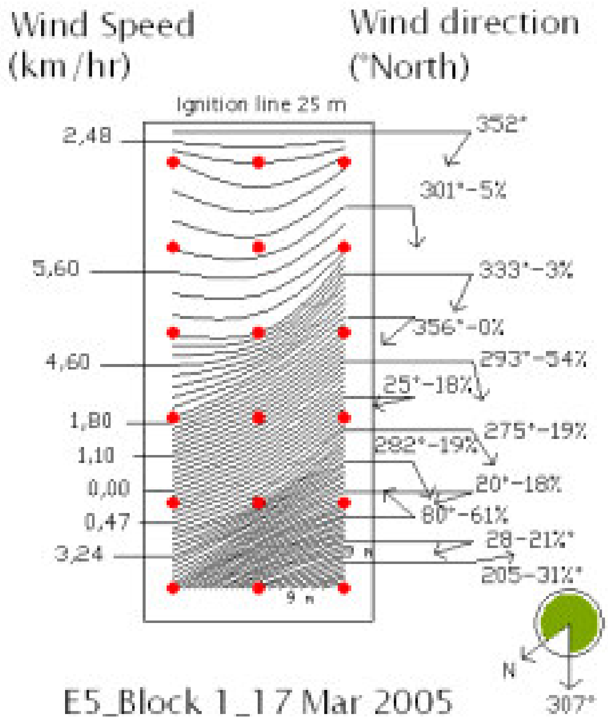
E4_Block 2_10 Mar 05

Experiment 4
Site: 2

Date: 10/03/05
Time: 11:56am
Area: 4000 m^2

W_{TF} (t/ha): 7.6 ± 0.3
Mt(%): 30 ± 2.5
U(km/h): 5.1 ± 0.4
T (°C): 12.5 ± 0.1
RH (%): 22%

ROS(m/min): 2.5 ± 0.5
I (kW/h): 514 ± 107



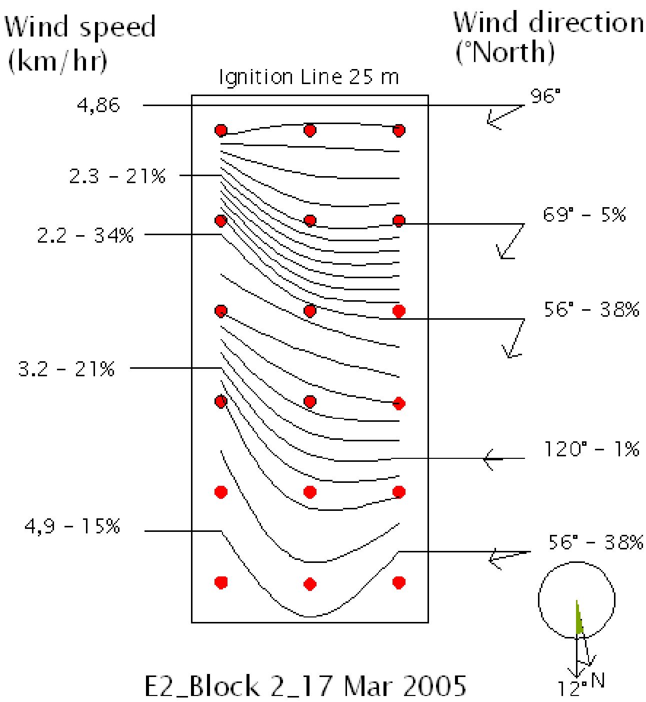
E5_Block 1_17 Mar 2005

Experiment 5
Site: 1

Date: 17/03/05
Time: 9:42am
Area: 1250 m^2

W_{TF} (t/ha): 11.9 ± 0.8
Mt(%): 35 ± 2
U(km/h): 1.7 ± 0.2
T (°C): 13.6 ± 0.1
RH (%): 58%

ROS(m/min): 2.6 ± 0.7
I (kW/h): 437 ± 105



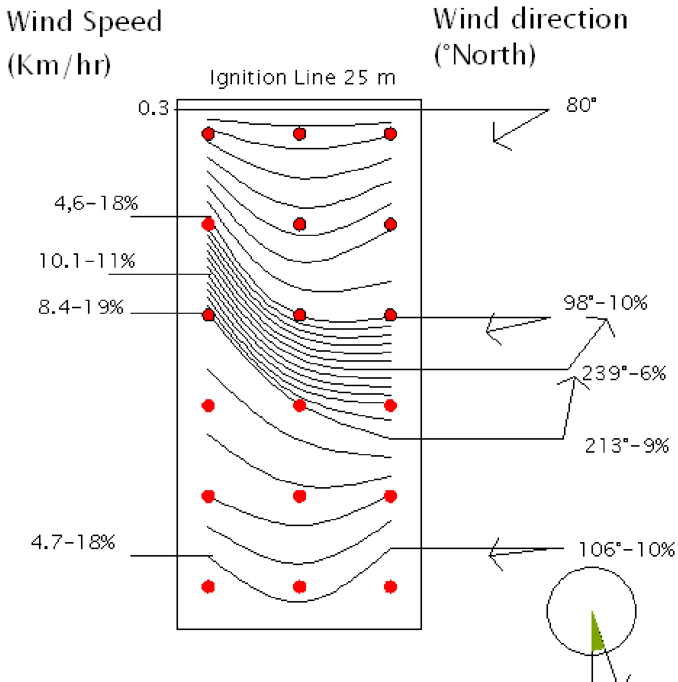
E2_Block 2_17 Mar 2005

Experiment 6
Site: 2

Date: 17/03/05
Time: 10:56am
Area: 1250 m^2

W_{TF} (t/ha): 7.7 ± 0.2
Mt(%): 33 ± 4.5
U(km/h): 3.1 ± 0.4
T (°C): 16.3 ± 0.1
RH (%): 48%

ROS(m/min): 5.4 ± 1.2
I (kW/h): 654 ± 145



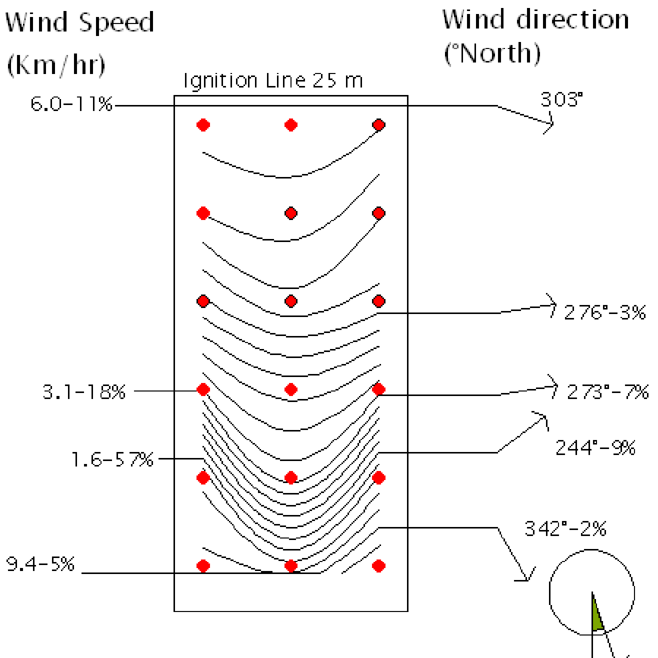
E7_Block 3_17_03_2005

Experiment 7
Site: 3

Date: 17/03/05
Time: 11:40am
Area: 1250 m^2

W_{TF} (t/ha): 14.4 ± 0.6
Mt(%): 34 ± 2
U(km/h): 6.6 ± 1
T (°C): 19 ± 0.3
RH (%): 44%

ROS(m/min): 5.5 ± 0.1
I (kW/h): 2376 ± 339



E8_Block 4_17_03_2005

Experiment 8
Site: 4

Date: 17/03/05
Time: 12:31pm
Area: 1250 m^2

W_{TF} (t/ha): 10.3±1.2
Mt(%): 30 ± 0.6
U(km/h): 5.2 ± 0.7
T (°C): 19 ± 0.1
RH (%): 42%

ROS(m/min): 8.4 ± 2.2
I (kW/h): 1955 ± 507