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Chapter 9

Forest Fires and Air Quality Issues in Southern Europe

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Abstract

Each summer forest fires in southern Europe emit large quantities of pollutants to the atmosphere. These fires can generate a number of air pollution episodes as measured by air quality monitoring networks. We analyzed the impact of forest fires on air quality of specific regions of southern Europe. Data from several summer seasons were studied with the aim of identifying air pollution episodes related to the occurrence of forest fires. Emissions from forest fires were estimated on the basis of vegetation burned and fire characteristics. Emissions from aircrafts used to fight forest fires were also calculated. It was possible to identify and quantify several particulate and photochemical air pollution episodes caused or enhanced by smoke from forest fires. A case study is described and a mesoscale air quality modelling system, specifically adapted to forest fires, was applied to simulate the fire impact on air quality. Results from the modeling exercise were considered to be reasonable when compared to air concentration values from monitoring networks, taking into account all the uncertainties inherent to this kind of simulation.

9.1. Introduction

Due to frequency of occurrence and the magnitude of effects on the environment, health, economy, and security, forest fires have increasingly become a major subject of concern for decision makers, firefighters, researchers, and citizens in general. According to the European Commission (2005), more than 12 million ha of forest have burned in

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the past 25 years in the five southern European Union (EU) member states alone, with an average value of nearly 500,000 ha/year. Moreover, approximately 60 million ha of forest, representing more than 50% of the former 15 EU member states forests, have been classified as high or medium fire risk areas (European Commission, 2004).

One of the consequences of forest fires is the atmospheric emissions of various environmentally significant gases and solid particulates that contribute to local, regional, and global phenomena in the biosphere. Pollutants emitted include atmospheric particulate matter (PM) and gaseous compounds, such as carbon dioxide (CO_2), carbon monoxide (CO), methane (CH_4), nonmethane hydrocarbons (NMHC), nitrogen oxides, (NO_X), and nitrous oxide (N_2O). Smoke pollution due to forest fire events can represent an important public health issue to the community, particularly for personnel involved in firefighting operations (Brustet et al., 1991; Miranda et al., 1994a, 2005a; Reinhardt et al., 2001; Valente et al., 2007; Ward et al., 1993). In addition, high levels of tropospheric ozone can occur at great distances from emission sources (Crutzen & Andreae, 1990; Crutzen & Carmichael, 1993). The environmental effects of these emissions are related to the transport and deposition processes involved.

Severe air pollution episodes caused by fires in Amazonia (Brazil), Indonesia, and Philippines in 1997–1998 and, more recently, in Australia and Russia have drawn worldwide attention to the problem of air quality due to forest fires. Increasingly, smoke pollution caused by wildland fires is considered an important health issue with major risks for the population and the environment. The World Health Organization (WHO) has even provided guidelines for forest fire episodic events to protect the public from adverse health effects (World Health Organization, 1999). This concern also applies to prescribed fires, especially in Australia and North America where this land management technique is frequently used.

The main purpose of this chapter is to provide an overview of the effects of forest fires on the air quality in southern European through two main approaches: the cross-analysis of air quality and forest fire behavior data, and the application of an air quality modelling system to a specific case study.

9.2. Forest fires and air quality in Europe

The link between forest fires and air quality is not commonly made. From the point of view of the community dealing with the fire, the main concerns are the direct effects of fire, such as human fatalities and property damage. In addition, air quality problems are usually analyzed in terms of the main anthropogenic sources, particularly the classic industrial and road transport sectors. However, forest fires can be a significant source of air pollutants, and air quality management strategies should consider these effects.

The extreme fire events that occurred in the summer 2003 in southern Europe highlighted the need to better analyze this phenomenon. In Portugal, for example, the firemen involved in these episodes regarded the summer of 2003 as the most important operational challenge of the past 30 years (Liga Bombeiros Portugueses, 2003). The decrease in the number of reported fires compared to the 1993–2002 period was not accompanied by a decrease in the area burned. On the contrary, fire area in 2003 was approximately four times higher than the annual average for the same period (Direcção Geral das Florestas, 2003). As a consequence, unusual air pollutant concentrations were registered at several monitoring stations in the national air quality network during 2003, most of them located inside urban areas (Monteiro et al., 2005). Moreover, increases in the number of hospital admissions and deaths due to respiratory and cardiovascular diseases were also reported either as a direct consequence of the fires or in association with the concomitant high air temperatures (Direcção Geral das Florestas, 2003). In Spain during the same year, deaths related to forest fires were caused not by the fire itself, but by smoke inhalation (Caballero, 2003).

9.2.1. Forest fires in southern Europe

Fire is the most important natural threat to forests and wooded areas of southern Europe. This area can be defined to include Spain, Portugal, Italy, Greece, and Mediterranean France (Corse, Provence–Alpes–Cote d'Azur, Languedoc–Roussillon, Rhone–Alpes). In the past decade (1996–2005), the average annual number of forest fires throughout southern Europe exceeded 61,000, which is 34% more than recorded during the previous decade (1986–1995) (Table 9.1). In particular, there has been an increase in the average number of annual fires in Portugal (94%), Spain (49%), and Greece (22%). Only in Italy was there a reduction (−26%) in the number of forest fires. On average, there were about 54 fires/100 km^2, with a maximum in Portugal (308 fires/100 km^2) and a minimum in Greece (14 fires/100 km^2). The average area burnt annually during the period 1996–2005 was approximately 423,000 ha, which was 19% less than in the previous decade (1986–1995) (Table 9.2). The mean total area burnt decreased in Spain (by 45%), Italy (by 35%), Mediterranean

Table 9.1. Number of forest fires in southern Europe (1996–2005)

Year	Forest fire (number)					
	Med. France	Greece	Italy	Portugal	Spain	Total
1996	1789	1508	9093	28,626	16,771	57,787
1997	2784	2273	11,612	23,497	22,320	62,486
1998	2586	1842	9540	34,676	22,448	71,092
1999	2995	1486	6932	25,477	18,237	55,127
2000	2430	2581	8595	34,109	24,118	71,833
2001	2788	2535	7134	26,533	19,099	58,089
2002	1677	1141	4601	26,488	19,929	53,836
2003	3490	1452	9697	26,195	18,616	59,450
2004	2028	1748	6428	21,870	21,396	53,470
2005	1871	1544	7951	35,697	26,269[a]	73,332
Total (1996–2005)	24,438	18,110	81,583	283,168	209,203	616,502
%	4.0	2.9	13.2	45.9	33.9	100.0
Total (1986–1995)	23,880	14,817	109,690	145,958	140,481	460,569
Difference	558	3293	−28,107	137,210	68,722	155,933
%	2.3	22.2	−25.6	94.0	48.9	33.9
Total national area km^2 [b]	113,249	131,957	301,333	92,040	505,960	1,144,539
No. fire/100 km^2	21.6	13.7	27.1	308.1	41.4	53.9

Sources: Promethee Database, France; Corpo Forestale dello Stato, Italy; Direcçao Geral dos Recursos Florestais, Portugal; Dirección General para la Biodiversidad, Ministerio de Medio Ambiente, Spain; Greece, European Commission (2006).

[a]Provisional data—lacking some data from Andalucia and Extremadura.

[b]Only Mediterranean area of France (Corse, Provence–Alpes–Cote d'Azur, Languedoc–Roussillon, Rhone–Alpes).

France (by 28%), and Greece (by 24%). Only Portugal experienced an increase of about 66% during the same period, mainly due to the 2003 and 2005 forest fire seasons.

9.2.2. Emissions

Air pollutant and precursor emissions in Europe are mainly due to combustion and energy transformation industries (SO_X), road transport (CO, NO_X, NMHC), solvents and other similar products (NMHC), and agriculture (NH_3) (Ritters, 1998). The major air pollutants emitted in southern Europe (France, Greece, Italy, Spain, and Portugal) over the period 1990–2003 are reported in Fig. 9.1. These data are based on a compilation of National Emission Inventories carried out according to

Table 9.2. Burnt area (ha) in southern Europe (1996–2005)

Year	Burnt area (ha)					
	Med. France	Greece	Italy	Portugal	Spain	Total
1996	3119	25,310	57,988	88,867	59,814	235,098
1997	12,250	52,373	111,230	30,535	98,503	304,891
1998	11,243	92,901	155,553	158,369	133,643	551,709
1999	12,782	8289	71,117	70,613	82,217	245,018
2000	18,860	145,033	114,648	159,605	188,586	626,731
2001	17,965	18,221	76,427	111,850	92,386	316,849
2002	6298	6013	40,791	124,411	107,464	284,977
2003	61,507	3517	91,805	425,716	148,173	730,718
2004	10,596	10,267	60,176	129,539	134,193	344,771
2005	17,356	6437	47,575	338,262	179,851[a]	589,481
Total (1996–2005)	171,976	368,361	827,310	1,637,767	1,224,829	4,230,242
%	4.1	8.7	19.6	38.7	29.0	100.0
Total (1986–1995)	237,681	485,902	1,278,521	988,107	2,214,973	5,205,184
Difference	−65,705	−117,541	−451,211	649,659	−990,144	−974,942
%	−27.6	−24.2	−35.3	65.8	−44.7	−18.7
Total national area (ha '000)[b]	11,325	13,196	30,133	9204	50,596	114,454
Total (1996–2005) burnt area/Total national area (%)	1.5	2.8	2.7	17.8	2.4	3.7

Sources: Promethee Database, France; Corpo Forestale dello Stato, Italy; Direcçao Geral dos Recursos Florestais, Portugal; Dirección General para la Biodiversidad, Ministerio de Medio Ambiente, Spain; Greece, European Commission (2006).

[a]Provisional data—lacking some data from Andalucia and Extremadura.

[b]Only Mediterranean area of France (Corse, Provence–Alpes–Cote d'Azur, Languedoc–Roussillon, Rhone–Alpes).

the Coordination of Information on Air Emissions (CORINAIR) methodology (European Environmental Agency, 2005a). In the year 2003, emissions amounted to 14,518 Gg (CO), 4614 Gg (NO_x), 4406 Gg (NMHC), and 3098 Gg (SO_X). Due to incomplete datasets, no estimate was available for NH_3 in 2003; in 2002, the estimate was 1786 Gg. There is clearly a decreasing trend in the emissions of NMHC, SO_X, and CO (Fig. 9.1). The trend is less pronounced for NO_X, while NH_3 emissions have remained fairly unchanged. Figure 9.2 shows the relative contribution of southern European emissions to the total emissions in the EU 15 member states.

It is worth noting that, despite the decreasing trend in emissions overall, the contribution from southern European countries has increased over the period 1990–2003. On average, this increase has been $11 \pm 6.5\%$, and driven primarily by SO_X in particular, a class of pollutant that has

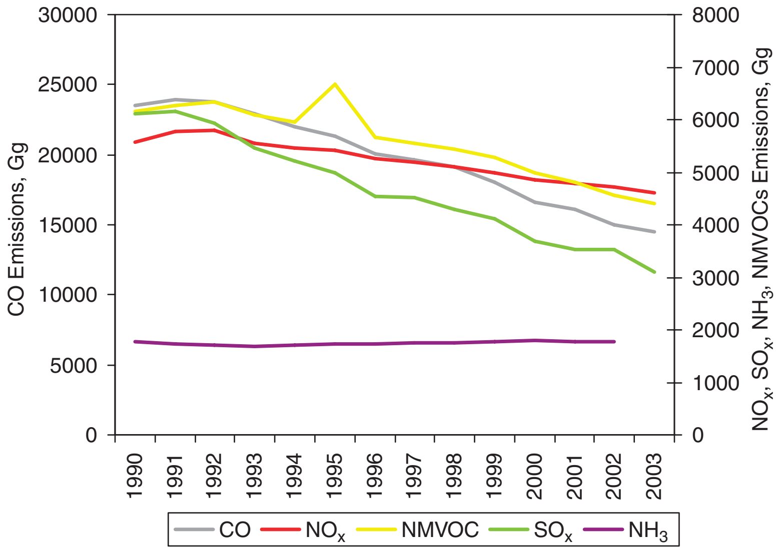

Figure 9.1. Emissions of CO (left *y* axis), NO_X, NMHC, SO_X, and NH_3 (right *y* axis) in southern Europe (France, Greece, Italy, Portugal, and Spain) for the period 1990–2003 (European Environmental Agency, 2005a).

had particularly effective reductions in countries that experienced very high emission rates in the past. For example, SO_X emissions in Germany were three times higher than in Italy in 1990.

Notwithstanding this trend, it is important to understand the contribution by forest fires to these total values. Of the pollutants analyzed, forest fires are major contributors of CO, NO_x, NMHC, and NH_3. In the European context and according to data from the European emission inventory CORINAIR (European Environmental Agency, 2004), forest fire emissions represent 0.2% of NO_2, 0.5% of NMHC, 1.9% of CO, and 0.1% of NH_3. For Portugal the contribution of forest fire emissions in 2003 to the total value (for the 1994 base year) equates to 14.1% CO, 5.2% NO_2, 2.7% NMHC, 2.2% CH_4, 1.3% NH_3, and 0.6% SO_2. These emissions were estimated using the national emission inventory for non-forest fire emissions and a model (EMISPREAD; Miranda et al., 2005b), which takes into account type of fuel and combustion phase, to estimate forest fire emissions from southern Europe.

Globally, fires are a significant contributor of CO_2 and other greenhouse gases (GHG) to the atmosphere. Fires account for approximately one-fifth of the total global emission of CO_2 (Sandberg

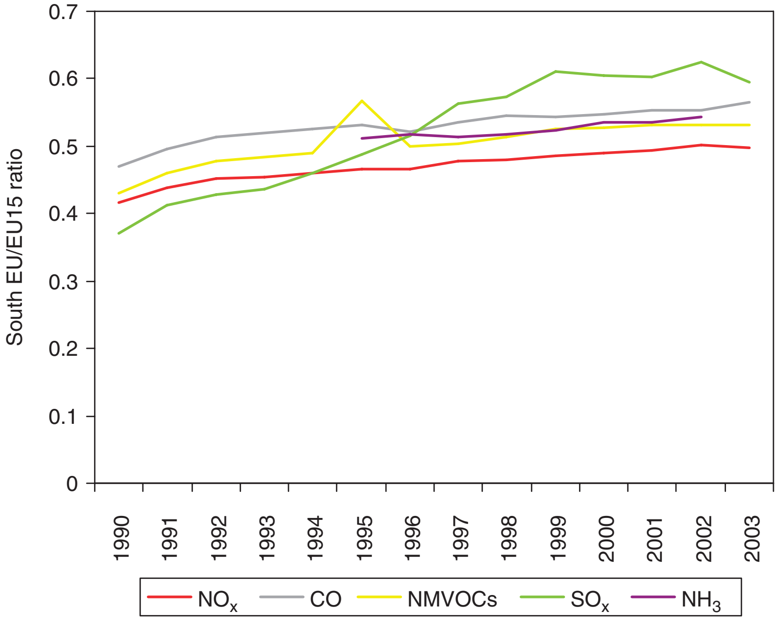

Figure 9.2. The ratio between the emissions in southern Europe and the EU 15 member states (European Environmental Agency, 2005a).

et al., 2002). Even taking into account the notion that fires in temperate ecosystems are minor contributors compared to biomass burning in savannas, boreal, and tropical forests, the contribution to total CO_2 equivalent emissions produced during forest fires can reach 7% if the annual area burned and exceeds 100,000 ha (Miranda et al., 1994b).

When assessing forest fire emissions, pollutants emitted during ground and air-fighting activities should also be evaluated. As with many vehicles, firefighting aircraft engines produce water vapor, unburned or partially combusted hydrocarbons, CO_2, CO, NO_X, SO_X, particulates (sulfur- and carbon-based), and other trace compounds. These emissions concentrate in the lower troposphere, as firefighting aircrafts do not exceed 3000 m.

Aircraft emissions are characterized in terms of emission index (EI; units of gram of emissions per kilogram of fuel burned), which is representative of a particular engine state, along with the time spent in that state performing the various operative modes (e.g., idling, taking off, climbing out, approaching, reverse thrust). The time that firefighting aircraft operates in each mode depends on a variety of factors including distance from airport to fire, distance from fire to water supply point and terrain morphology; however, no data are currently available to determine the time spent in the different operative modes.

Emission levels of H_2O, CO_2, and SO_2 are determined by the fraction of hydrogen (H), carbon (C), and sulfur (S) compounds contained in the fuel. In contrast, emissions of NO_X, CO, and HC vary according to the combustor conditions (Sutkus et al., 2003). It is likely that aircraft emissions are also modified by the presence of other atmospheric pollutants, including those from forest fires.

A preliminary analysis of the firefighting aircraft emission during the period 2000–2003 was carried out in Italy. Flight data were collected by the National Unified Aerial Operational Center (COAU) and included take off, in zone, off zone, and take on times. Travel (time to go and back from aircraft base to forest fire) and operative (time spent in active firefighting activities) times were then calculated. Since specific EIs for the engines used in the firefighting aircraft were not available, the average EI for the main pollutants were used according to Schumann (2002) and Penner et al. (1999). These EIs included ($g\,kg^{-1}$) 3150 for CO_2, 1260 for H_2O, 14 for NO_X, 4 for CO, 1 for SO_2, and 0.6 for HC. For NO_x, the emission index ($EI(NO_x)$) is given as gram equivalent NO_2.

On the basis of average fuel consumption, flight time, and EI, emissions from Italian firefighting aircrafts over the period 2000–2003 were calculated according to Draper et al. (1997).

The average annual emission was about 61 t NO_X, 3 t HC, 4 t SO_2, 18 t CO, 14,000 t CO_2, and 6000 t H_2O. According to the flight time distribution, over 60% of the emissions were concentrated close to the fire areas. In addition, forest fire emissions were estimated using the EMISPREAD model for Italy in 2001. Emissions from firefighting aircraft were compared with these values (Table 9.3), and it is obvious

Table 9.3. Forest fire and aircraft emissions in Italy in 2001

Source		Emissions (t)			
		CO_2	CO	HC	NO_X
Forest fires		950,769	46,417	5861	2632
	In zone	7906	10	2	35
Aircraft	Travel	5536	7	1	25
	Total	13,442	17	3	60
Total		964,211	46,434	5864	2692
Aircraft/forest fires	%	1.41	0.04	0.05	2.28

that aircraft emissions for some pollutants such as CO_2 and NO_X have to be considered when total emission values associated with forest fires are analyzed.

9.2.3. Air quality

The contribution of forest fires to total atmospheric emissions makes it clear that some relationship between forest fires and air quality is to be expected. However, it is not always easy to identify air pollution episodes caused or exacerbated by forest fires. Pollutants emitted from forest fires are transported and dispersed in the atmosphere, and their effects on air quality can occur far from the emitting source. Although major wildfires are limited to some hundreds of hectares, their impacts, with no natural or political boundaries, can be felt and reported far beyond the physical limits of the fire spread. Depending on meteorological conditions, smoke plumes and haze layers can persist in the atmosphere for long periods of time, and prevailing conditions will influence the chemical and optical characteristics of the plume. There is emerging evidence that smoke from widespread wildfires in Portugal in summer 2003 contributed to the high ozone levels measured at the air quality monitoring stations in Paris, France (Hodzic et al., 2006).

Air quality issues have changed with time in Europe. Early concern about sulfur compounds and acidification of ecosystems (freshwaters and forests) was particularly relevant in the 1970s and 1980s in central Europe. Today, concerns mostly include tropospheric ozone and PM_{10} (particles with mean aerodynamic diameter smaller than $10\,\mu m$) levels impinging on populations in urban environments (European Environmental Agency, 2005b; Lövblad et al., 2004; Percy & Ferretti, 2004). According to a recent report (European Environmental Agency, 2005b), the proportion of the population exposed to values exceeding EU protection values for PM_{10}, SO_2, and NO_2 or target values for O_3 is 25–55% (PM_{10}), 30% (O_3 and NO_2), and $<1\%$ (SO_2). These data have some limitations because the spatial coverage varies from year to year, and only populations in urban areas equipped with monitoring networks are considered. However, they are useful for making a qualitative estimate of the potential impact of the different air pollutants. In southern Europe, ozone and PM_{10} have been claimed as the most important air pollution problems.

Ozone, in particular, shows distinctly higher levels in southern Europe, which is also identified as the most critical area in terms of burnt area and forest fire risk, than in other regions of Europe (Beck et al., 1999). Ozone is a secondary photochemical pollutant, and its production depends on

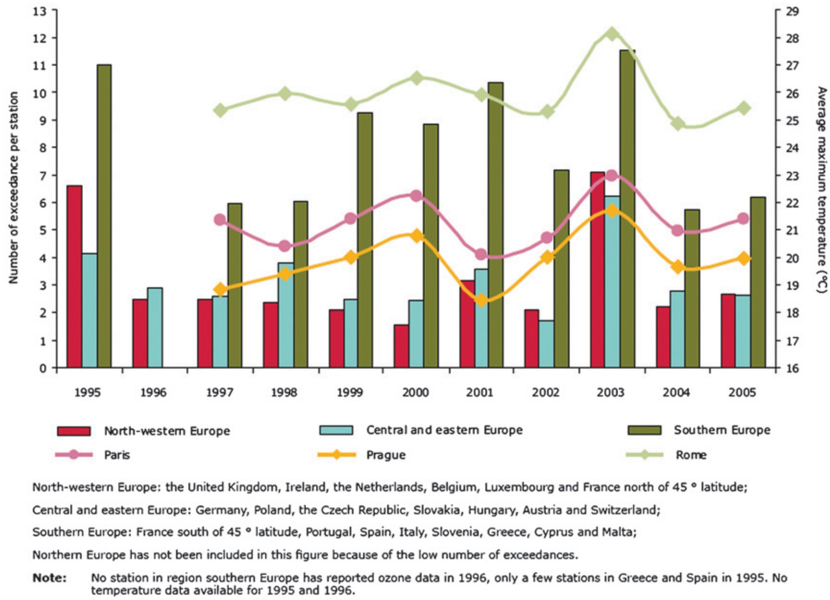

Figure 9.3. Average number of ozone exceedances per station for three European regions observed during each year for the period 1995–2005. The summer average maximum daily temperature in selected cities are also presented (European Environmental Agency, 2006).

levels of solar radiation and high temperatures. Figure 9.3 (European Environmental Agency, 2006) illustrates this relation between meteorological conditions and number of exceedances of permissible ozone levels in different regions of Europe—northwestern, central and eastern, and southern Europe. Daily maximum temperatures observed in three capital cities in these regions (Paris—northwestern region; Prague—central and eastern region; and Rome—southern region) averaged for the period April–September of a particular year are shown in Fig. 9.3. The ozone number of exceedances is based on data from the monitoring networks in Europe. Frequent occurrence of ozone exceedance was quite common in southern Europe, where the highest temperature values were also reported (city of Rome), compared to other temperature values measured in cities from northwestern, central, and eastern Europe.

Emissions from forest fires are also very dependent on weather conditions. An analysis of fire occurrence and burnt area per month in Italy, Greece, Mediterranean France, and Portugal can be used to evaluate the yearly distribution of forest fire emissions to the atmosphere. Two peaks are usually recorded: in spring (March/April) and in summer

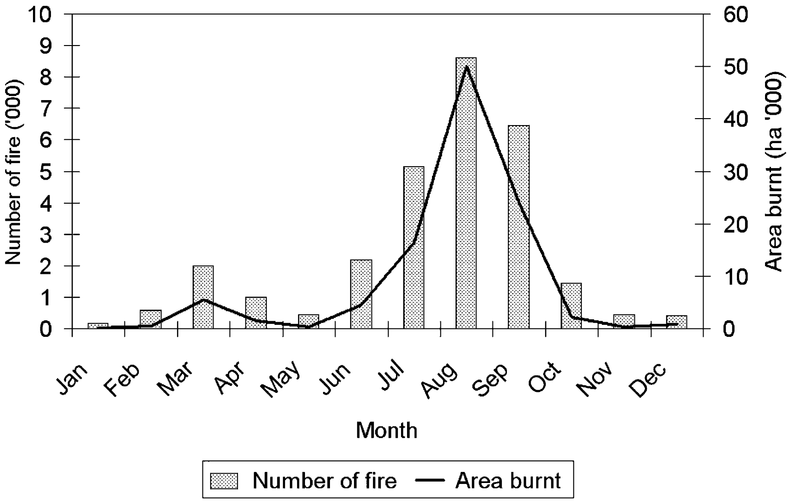

Figure 9.4. Fire number and burnt area per month (Portugal—average values 1997–2001). (Source: Direcção Geral dos Recursos Florestais, Portugal.)

(July, August, and September). The second peak is higher, with about 50–70% of the fires occurring annually, depending on year and country, and covering 70–90% of the annual area burnt. As an example, Fig. 9.4 shows the situation in Portugal.

The presence of ozone precursors emitted by forest fires, namely NO_X and NMHC, is also important. Southern European countries have good conditions for photochemical production (European Commission, 1999) and forest fire occurrence, namely very hot, dry, and sunny summers, and forested areas, both of which contribute to higher emissions of ozone precursors.

Weather conditions are also another possible link between forest fires and episodes of particulate air pollution in southern Europe, particularly where PM_{10} is a critical pollutant. During summer, low humidity and high wind speed values increase the risk of occurrence of large forest fires, while being related to high values of PM with the transport and resuspension of dust and pollen (Coutinho et al., 2005).

9.3. Air quality management and forest fires

Air quality is one of the areas in which Europe has been most active in recent years, namely focusing on the definition and implementation of air quality strategies, the improvement of air quality monitoring networks, and the development of scientific knowledge to better characterize and understand European air quality. The European Commission (EC) has aimed to develop an overall strategy through setting long-term air quality objectives. A series of directives has been introduced to control levels of certain pollutants and to monitor their concentration in air. In 1996, the Environmental Council adopted the Air Quality Framework Directive 96/62/EC (AQFD) on ambient air quality assessment and management. This Framework Directive, followed by its subsequent directives, presented several innovative aspects and concepts regarding air quality management strategies, such as the need to establish plans and programs for zones and urban agglomerations where the air quality thresholds are not met and the possibility of using simulation models as a tool to evaluate air quality. These two new aspects of the AQFD can be directly applied to forest fires, which are responsible for air pollution episodes.

9.3.1. Plans and programs

The AQFD establishes that if certain air quality standards are surpassed, member states have the obligation to prepare plans and programs (PPs) that must indicate measures to improve air quality and to comply with specified limit values.

Several member states have already delivered their plans and programs to the European Commission regarding air quality exceedances in 2001–2003, and stating how they intend to improve the air quality in a near future. These plans and programs mainly concern urban agglomerations, densely populated areas, and PM_{10} and NO_2 pollution.

In Portugal, Porto and Lisbon agglomerations are the only ones that submitted plans and programs to implement measures to reduce PM_{10} air pollution. According to EU Framework Directive PM_{10}, daily averaged values should not exceed $50\,\mu g\,m^{-3}$ more than 35 times/year (as of January 2005). The number of PM_{10} exceedances for the monitoring stations in the Porto agglomeration air quality monitoring network was analyzed (Coutinho et al., 2005). For the 3 years under study (2001–2003), all stations registered some exceedance of the annual average PM_{10} threshold value, and many stations did so more times than the 35 daily exceedances allowed by the directive.

The identification of PM_{10} sources was fundamental for the definition of plans and programs for the Porto agglomeration that has to ensure reaching the threshold value in 2005. Moreover, according to the European Commission, plans and programs should focus on anthropogenic sources, but forest fires are considered natural ones. For the analysis of the need to develop plans and programs, PM_{10} exceeding days due to natural sources (e.g., forest fires) can be deducted from the total exceeding days. However, due to the number of sources involved, the relation between forest fire emissions and PM_{10} values measured is not clear-cut. Therefore, a specific approach was developed to identify episodes of particulate pollution in the Porto agglomeration related to forest fires. To do this, days for which values were higher than the daily limit value for PM_{10} ($50\,\mu g\,m^{-3}$) at three or more monitoring stations simultaneously, plus a margin of tolerance for the specific year under analysis, were considered to be episodic days that could potentially be influenced by a forest fire smoke plume. For these days, the hybrid single-particle Lagrangian integrated trajectory model (HYSPLIT) was applied to estimate backward trajectories from the forest fires to the Porto agglomeration.

In very simple models, the trajectory and dispersion of the smoke plume can be simulated through straight-line trajectories. Other models assume that the atmosphere is neutrally buoyant to compute trajectories of air parcels that are transported by a tridimensional wind field, which is calculated by a numerical weather prediction model. This so-called ‘‘trajectory technique’’ is considered to be simple in its concept and only requires modest computer resources (Borrego et al., 2004). Trajectories can be run forward in time to determine receptor areas or backward in order to determine the pollutant source areas. In general, multiple trajectories are required due to instability of the atmospheric flow.

The HYSPLIT model, developed by the U.S. National Oceanic and Atmospheric Administration, is a system that can estimate trajectories and dispersion and deposition fields of gaseous and particulate pollutants (Draxler et al., 2005). It uses meteorological grid data from re-analysis of weather forecasting models. Figure 9.5 shows the estimated trajectories for August 3–4, 2003, when several forest fires were spreading through the center of Portugal. HYSPLIT was applied for air mass altitudes of 250, 500, and 750 m and for a period 1–2 days. Satellite photos for these days are also shown in Fig. 9.5.

This methodology, which was used for all the fires with a burned area larger than 100 ha, was applied for 3 years of air quality monitoring in the Porto agglomeration (Borrego et al., 2005). In this way, it was possible to calculate the contribution of forest fires to PM_{10} pollution episodes as 35%, 8%, and 18%, for 2001, 2002, and 2003, respectively. This information was used to differentiate between anthropogenic and forest fire contributions to particulate matter pollution episodes. After subtracting episodic days due to forest fires from the total exceeding days, the need to develop PPs still persisted.

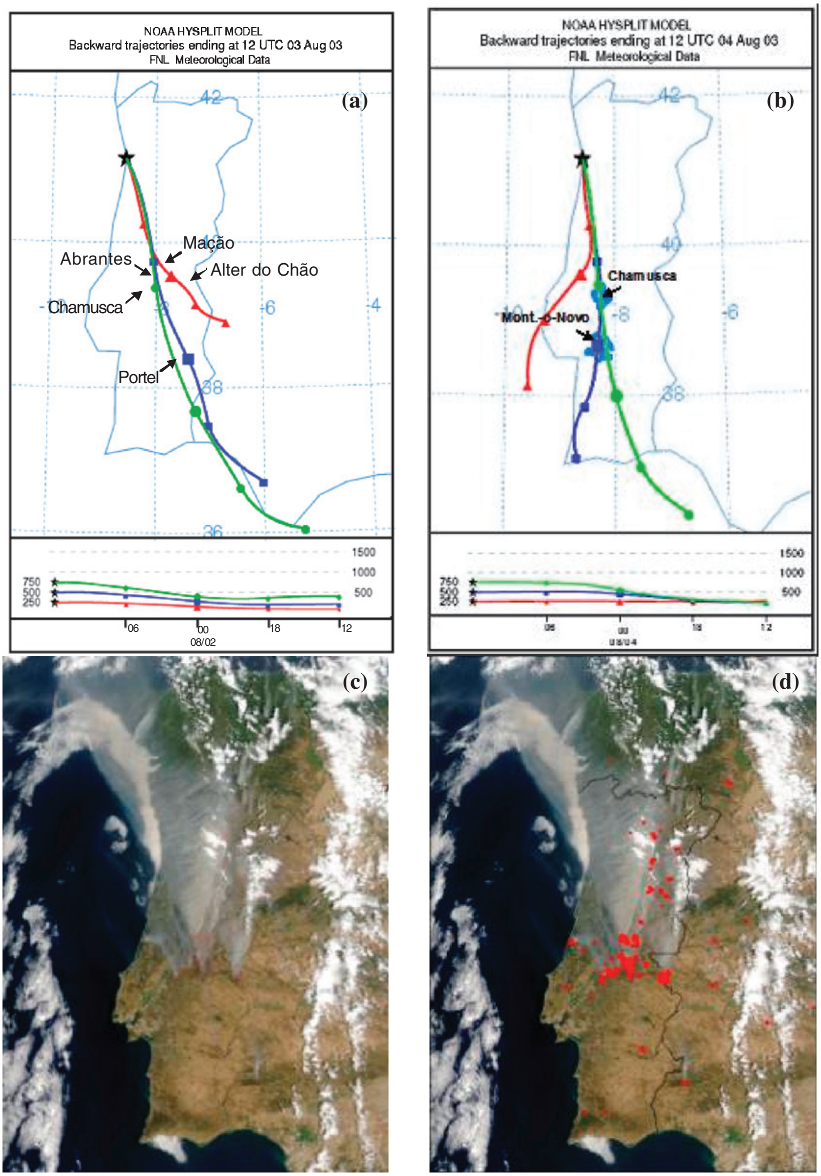

Figure 9.5. Backward trajectories (a, b) and satellite images (c, d) for August 3 (a, c) and August 4 (b, d), 2003, in Portugal. Blue areas correspond to burned areas, stars indicate the Porto agglomeration, and different colored lines indicate trajectories at different heights.

9.3.2. Modeling smoke plume impact on urban air quality in Lisbon

Smoke from forest fires can be a serious problem in cities and urban areas. In Portugal, forest fires in summer 2003 were considered the most devastating, and this is clearly reflected in the values measured by the air quality monitoring networks (Miranda et al., 2005c). Several air quality stations registered extremely high pollutant concentrations due to fire emissions and transport from surrounding areas. Ash from nearby fires reached many urban areas, reducing visibility, and depositing in urban areas.

Lisbon suffered the effects of smoke from forest fires north of its urban area in September 2003. This situation represents a very interesting case study for smoke dispersion and air pollutant concentration estimation because of the high population density involved and hence higher risk of human exposure to smoke. The Lisbon airshed, with a population of 3.5 million inhabitants, is the most important urban center in Portugal. It was built in a very complex topographic region, dominated by a large estuary and multiple hills and surrounded by small mountain ranges with elevations over 400 m above sea level.

The urban area of Lisbon has an air quality monitoring networking that includes several stations with different typologies (urban background or urban traffic), according to location and environmental criteria. Air quality data analysis enabled the identification of September 13, 2003 as the most critical day for high CO and PM_{10} concentration values in the Lisbon urban area. On this day, 33 fires were active in the Lisbon district, burning an area of about 400 ha of forest stands and shrubs. Figure 9.6 shows the location of the fires and the air quality monitoring stations in the Lisbon airshed. Figure 9.7 presents the hourly averaged PM_{10} (Fig. 9.7a) and CO (Fig. 9.7b) concentrations measured on September 13 at the monitoring stations indicated.

A dramatic increase in the concentration of PM_{10} measured at the end of the day was evident. The day was very warm, with temperatures reaching 35°C in Lisbon airport. Winds mainly blew from the east quadrant, changing towards southeast at the end of the afternoon, when the highest concentration values were measured. PM_{10} concentration values were generally high, but due to the fires, the average $50\,\mu g\,m^{-3}$ value was breached at all measuring stations except two (Quinta do Marquês and Mem-Martins), which are located west of Lisbon and were not affected by the smoke plumes. Avenida da Liberdade registered the highest daily averaged value of $150\,\mu g\,m^{-3}$.

The European Directive has established $10{,}000\,\mu g\,m^{-3}$ CO as the maximum eight-hour average allowable in a day. Even with a large

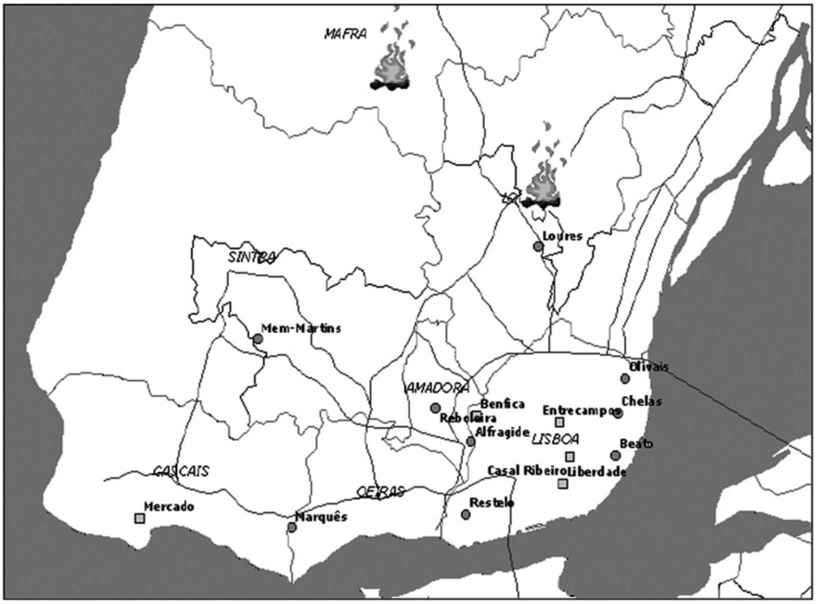

Figure 9.6. Location of the September 13, 2003, main forest fires and air quality monitoring stations in the Lisbon airshed (circles represent urban background stations, squares are urban traffic stations, and flame symbol represents forest fires).

increase in CO values at the end of the day (Fig. 9.7b), the values measured were not significant in terms of air pollution.

Numerical modeling of smoke dispersion allows us to understand how pollutants emitted by a forest fire are transported and dispersed in the atmosphere by estimating the resulting air pollutant concentration fields. AIRFIRE (Miranda, 2004) was developed to take into account the possible impact of forest fires on photochemical production. It integrates a modified version of the non-hydrostatic meteorological model (MEMO) (Moussiopoulos et al., 1993), the Rothermel fire progression model (Rothermel, 1972), and the atmospheric dispersion model for reactive species (MARS; Moussiopoulos et al., 1995). Using data available for September 13, 2003, AIRFIRE was applied to a modeling domain of $200\,\text{km} \times 200\,\text{km}$ with a horizontal resolution of $4\,\text{km} \times 4\,\text{km}$. This domain was chosen in order to consider mesoscale circulations, such as sea breezes in the Lisbon area.

Carbon monoxide and PM_{10} emissions from forest fires were estimated using data relating to the area burnt and the type of vegetation. Average emission factors for the two main types of fuel, shrub, and forest were

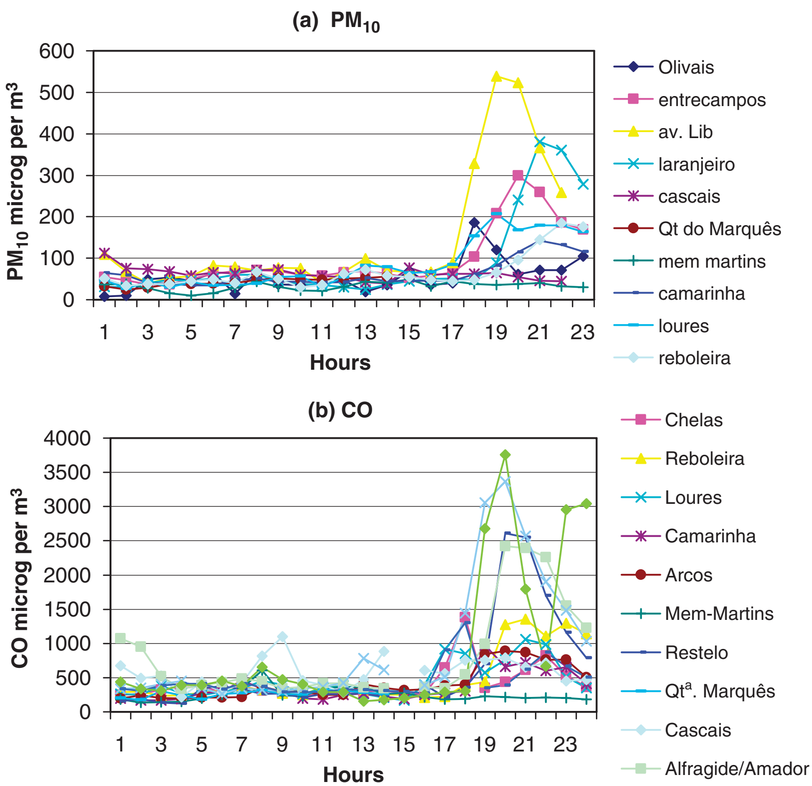

Figure 9.7. Measured PM_{10} (a) and CO (b) concentration values ($\mu g\,m^{-3}$) for September 13, 2003, at the different monitoring stations in Lisbon.

used, but these did not take into account differences in emissions due to smouldering and flaming combustion. These emission factors came from a selection of values for southern European forest fires (Miranda, 2004). The system was also applied considering only anthropogenic emissions from the national emission inventory (Instituto do Ambiente, 2006).

For meteorological conditions, MEMO used upper air data from the Lisbon airport as initial and boundary conditions. Aerological data, including temperature and wind speed and direction, obtained at 00.00, 12.00, and 24.00 LST (local standard time) were used as input data. Twenty vertical layers above the topography were considered, the first one (the surface layer) at a depth of 20 m. The top layer of the model was set at 6000 m.

The temporal evolution of the hourly averaged PM_{10} concentration fields during September 13 was estimated. Figure 9.8 shows the surface hourly averaged PM_{10} concentration patterns for 18.00 (Fig. 9.8a and b) and 22.00 LST (Fig. 9.8c and d) as estimated by AIRFIRE, considering all the sources in the simulation domain (Fig. 9.8b and d), which include forest fire emissions, and only considering the anthropogenic ones (Fig. 9.8a and c). Surface patterns of wind as estimated by AIRFIRE are also presented in Fig. 9.8. There is an obvious contribution by forest fires to the particularly high levels of PM_{10} measured in the urban area of Lisbon

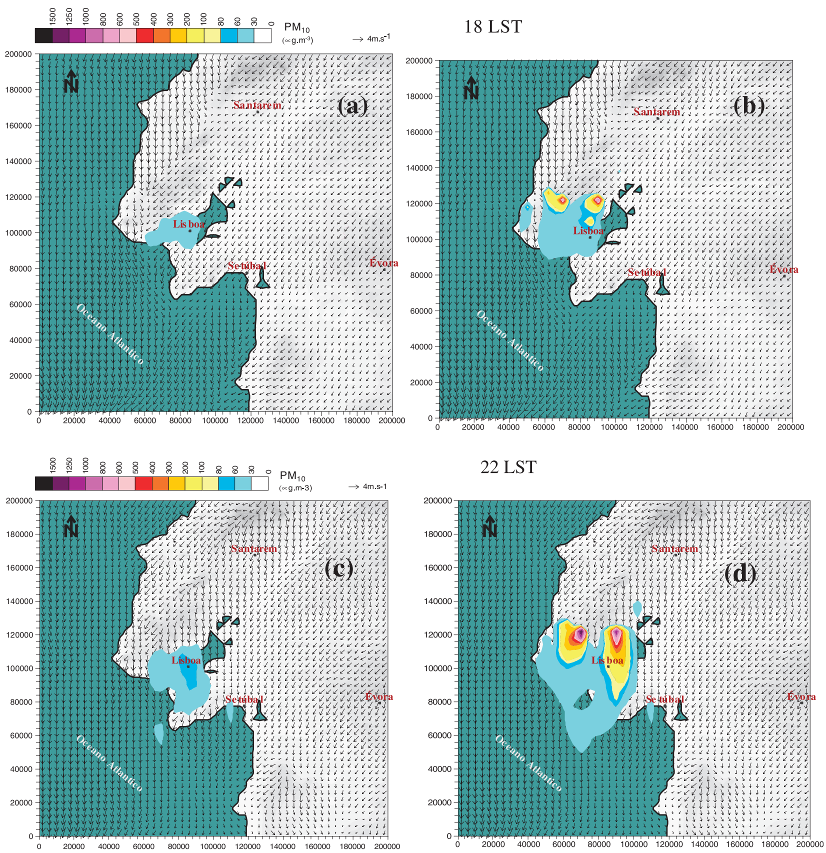

Figure 9.8. Hourly averaged surface PM_{10} concentration ($\mu g\,m^{-3}$) and wind values at 18.00 (a, b) and 22.00 (c, d) LST, both excluding forest fire emissions (a, c) and including forest fire emissions (b, d) in Lisbon.

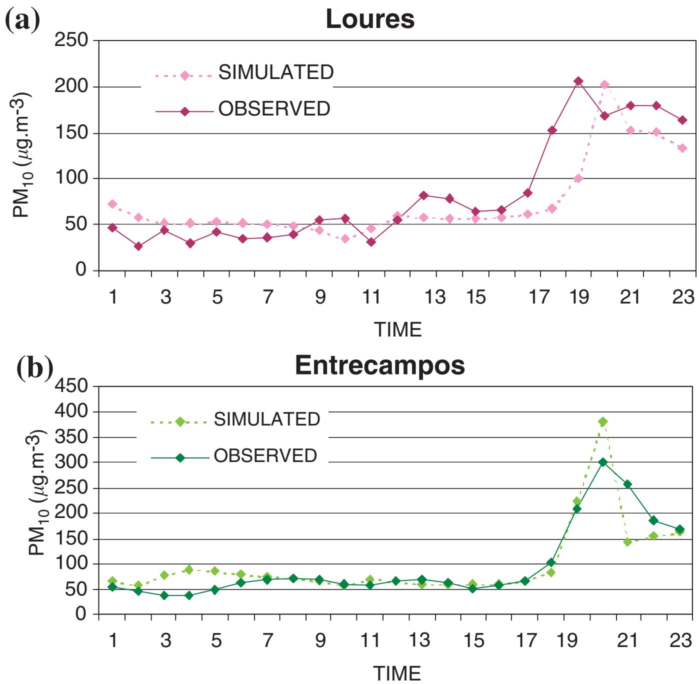

Figure 9.9. Qualitative comparison between measured and simulated hourly averaged concentration values of PM_{10} ($\mu g\,m^{-3}$) at Loures (a) and Entrecampos (b).

at night. More details regarding this Lisbon case study simulation can be found in Miranda et al. (2005c).

The accuracy of the modelling system was evaluated by directly comparing the simulated results with hourly averaged PM_{10} measurements from the Lisbon air quality monitoring network with Loures (Fig. 9.9a) and Entrecampos (Fig. 9.9b). There was reasonable agreement between simulated and observed values for most of the air quality monitoring stations (Miranda et al., 2005c). Taking into account the errors associated with this type of simulation, including those related to fuel load or fuel consumption, the simulation results can be considered an important source of information for managers of air quality. Currently, it is not a common procedure in Europe to model air quality by including forest fire emissions, but the use of this modelling tool can be extended to other European study cases.

9.4. Conclusions and future directions

The effect of forest fires on air quality is an issue of concern in many regions of the world, including the southern European countries. While pollution emissions are generally decreasing, the role of forest fires is becoming increasingly important. Emissions from forest fires may cause substantial exceedances of the air quality threshold, particularly in agglomerations with the high population densities. There is a strong need to take into account the role of forest fires when determining management strategies for air quality. To protect people, environmental policies must integrate the traditional pollution-oriented and land management issues into a unique system that can integrate both problems. Better land management can help to reduce both the number of air pollution episodes and the risk of unwanted fires.

This chapter illustrates how forest fires and air quality issues can be linked as we have shown by describing two case studies within different scopes and purposes. The quantification of forest fires' contribution to air pollution episodes is a fundamental stage of the development of plans and programs, particularly in southern European member states.

The application of numerical air quality modeling systems is also an added value when evaluating and assessing air quality levels in areas affected by forest fires. Fortunately, European researchers and managers are realizing the importance of forest fires to the degradation of air quality, and some modelling research has been developed and applied (e.g., Hodzic et al., 2007; Miranda et al., 2007). These studies are simulating larger areas—Europe and a European country—and longer periods (at least 1 month) than the one presented here. This different modelling scale implies the simulation of a larger number of fires and the use of a coarser grid resolution. It is a broader approach instead of the more specific approach presented in our case study. Both modelling approaches can be useful. The specific approach can be applied to simulate air pollution episodes related to the occurrence of forest fires. In this case the information from the numerical system, which can be applied almost in real time, can help to identify critical areas where people are exposed to high levels of air pollutants. The general approach is quite useful for the characterization and evaluation of air quality by each European member state or, at a higher level, by the EC.

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