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MODELLING SMOKE EFFECTS ON LISBON AIR QUALITY

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1. INTRODUCTION

Each summer season wildland forest fires burn a considerable area of south European landscape. Summer 2003 was one of the most severe fire seasons experienced during the last decades in Southern Europe and, due to persistent extreme fire conditions, Portugal suffered the worst forest fire season that the country has faced in the last 23 years, with a total burnt area of almost 5 times the average (EC, 2004).

Smoke has to be considered as one of the several disturbing effects of forest fires. Its impacts on air quality and human health can be significant since large amounts of pollutants, like particulate matter (PM_{10}), carbon monoxide (CO), volatile organic compounds (VOC) and nitrogen oxides (NO_x), are emitted to the atmosphere. The effects of these emissions are felt at different levels: from the contribution to global climate change to the occurrence of local atmospheric pollution episodes (Miranda, 2004). Therefore, the smoke released from forest fires is currently considered an important public health issue (Schwela *et al.*, 1999), particularly near or inside urban areas due to the highest risk of human exposure.

During summer 2003 there were reports of more than one thousand people (mainly civilian) needing medical assistance due to smoke intoxications, burns and wounds from forest fires in Portugal (EC, 2004). Satellite images, like the one shown in Figure 1 (URL1), and air pollutants concentration values measured on the Portuguese air quality monitoring network highlighted the impact of forest fire atmospheric emissions.

Forest fires spreading north of Lisbon city at September 2003 impacted the urban air quality. The Lisbon air shed, with a population of 3.5 million inhabitants, is the most important urban center in Portugal. Because of its urban/wildland characteristics, high population density, and hence higher risk of human exposure to smoke, and the high levels of pollutants registered, this is a very interesting case for the study of the influence of forest fires emissions on air quality.

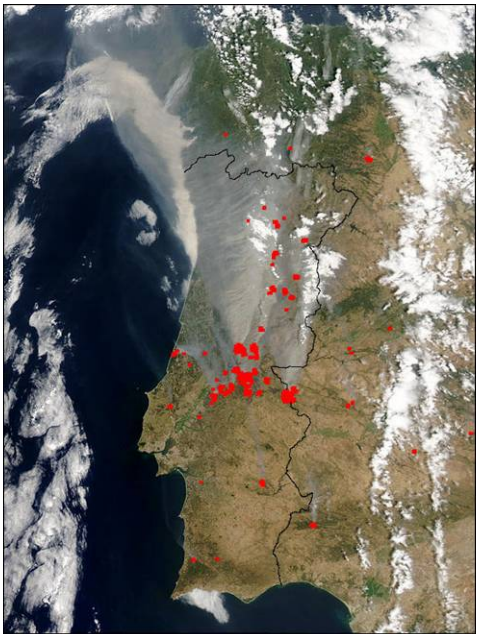

Figure 1. Portuguese territory satellite image, 4th August 2003 (URL1).

A first analysis of measured values of CO and PM_{10} during September was carried out aiming to identify one period where Lisbon was clearly affected by smoke from forest fires. It was possible to identify the 13th of September as one of the most critical days regarding CO and PM_{10} concentration values in Lisbon urban area. In this day, 32 forest fires were registered in the Lisbon Metropolitan Area (LMA), burning an area of about 400 ha of forest stands and shrubs. Table I presents some information concerning the main fires characteristics, like beginning and ending hours, burnt area and type of consumed vegetation. The city suffered particularly the effects of smoke from forest fires spreading north of the urban area at yhe municipalities of Mafra and Loures.

In Figure 2 the location of the main September 13th fires, as well as the LMA air quality monitoring network is presented. The urban area has an air quality monitoring network, which includes several stations, with different typology (urban background or urban traffic), according to location and environmental criteria.

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Table I. Main fire occurrences in Lisbon Metropolitan Area at September 13th.

Municipality	Begining	Ending	Burnt area (ha)	Vegetation type
Mafra	00:00 AM	11:59 PM	379.41	forest
Sintra	00:00 AM	11:59 PM	0.50	shrub
Loures	03:46 AM	06:30 AM	0.50	forest
Lourinhã	05:37 AM	07:36 AM	6.00	forest
Alenquer	2:00 PM	5:15 PM	2.00	shrub
S.M. Agraço	3:10 PM	5:30 PM	0.50	shrub
Alenquer	4:05 PM	8:20 PM	4.00	shrub
Loures	5:20 PM	9:00 PM	1.00	shrub
Sintra	5:35 PM	9:10 PM	2.00	forest
Azambuja	5:40 PM	11:59 PM	1.18	forest
V.F. Xira	5:55 PM	11:50 PM	3.00	shrub
Mafra	8:35 PM	11:59 PM	6.43	shrub
Total	-	-	406.53	-

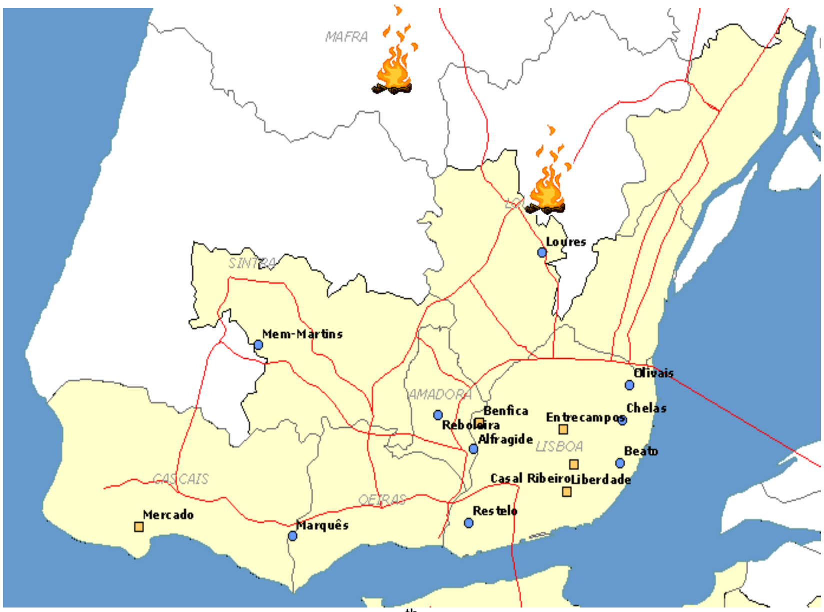


Figure 2. Location of the main September 13th fires and air quality stations in the Lisbon airshed. (blue circles are urban background stations and red circles ares urban traffic stations) (URL2)

Smoke dispersion numerical modeling allows the understanding of how pollutants emitted by a forest fire will be transported and dispersed in the atmosphere, estimating the out coming air pollutants concentration fields. The main purpose of this paper is to estimate the effects of the September 13th 2003 fires on Lisbon air quality through the application of a mesoscale photochemical modeling system, AIRFIRE (Miranda, 2004), and to evaluate its performance comparing results with air pollutants concentration measured values.

2. METHODOLOGY

The methodology applied for the estimation of the effects of the smoke from forest fires in Lisbon's air quality covers two main topics: the modeling system and the emission estimation.

2.1 AIRFIRE numerical system

AIRFIRE was developed taking into account the possible impact of forest fires on photochemical production. It integrates 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). Figure 3 presents a scheme of AIRFIRE. Estimation of the atmospheric flow is needed for both fire progression and smoke dispersion calculations. Smoke dispersion and photochemical production depend on the emission of pollutants, which is highly dependent on the fire progression and characteristics. The models integrating the system and the links established between them are described below.

The meteorological model

MEMO (Moussiopoulos *et al.*, 1993) is a three-dimensional Eulerian non-hydrostatic prognostic mesoscale model, which describes the atmospheric boundary layer for unsaturated air. The atmospheric physical phenomena are simulated by numerically approximating a set of equations in terrain-following co-ordinates. These include mass continuity, the momentum and transport equations for scalar quantities, such as potential temperature, turbulent kinetic energy and specific humidity. The conservation of mass equation is solved in its anelastic form. Turbulence and radiative transfer are the most important physical processes that have to be parameterized in a prognostic mesoscale model. For turbulent parameterization K-theory is applied. A zero-, one- or two-equation turbulence model can compute the exchange coefficients for momentum and heat.

High-resolution wind flow is an important prerequisite for fire behavior and smoke dispersion predictions. The wind flow resulting from high-resolution simulations is, however, highly affected by larger scale influences, i.e. by synoptic or regional scale phenomena. It is therefore important to take into account such influences properly. In order to predict smaller-scale flow in the fire region, as well as the atmospheric processes over the general geographic region of the fire, multiple-level grid nesting simulations are possible. MEMO allows horizontal resolution of 500 m up to 10 km and the model top height is between 4 and 10 km.

MEMO has been successfully applied and verified in various European air sheds including Athens (Moussiopoulos *et al.*, 1993), Barcelona (Baldasano *et al.*, 1993), and Lisbon (Coutinho *et al.*, 1993). It has also been applied and tested in an inland region of Portugal, which is critical in terms of fire risk (Miranda and Borrego, 1996). More details on the model can be found in Flassak and Moussiopoulos (1987).

A PM_{10} dispersion and deposition module was included in MEMO model. Surface deposition of particles occurs via diffusion, impactation, and/or gravitational settling. Particle size is the dominant variable controlling these processes. The approach suggested by Venkatram and Pleim (1999) has been adopted in MEMO.

The fire model

Fire spread with time is an important aspect of the modelling system, because it produces heat and releases gaseous and solid products to the atmosphere. In spite of its simplicity, the basis to simulate fire progression in AIRFIRE is the Rothermel model, which still is the most complete and practical spread model of low intensity, surface forest fires. It is an empirical fire spread model essentially based on observations from laboratory experiments and in an equation expressing an energy balance within a unit volume of the fuel ahead of the flame.

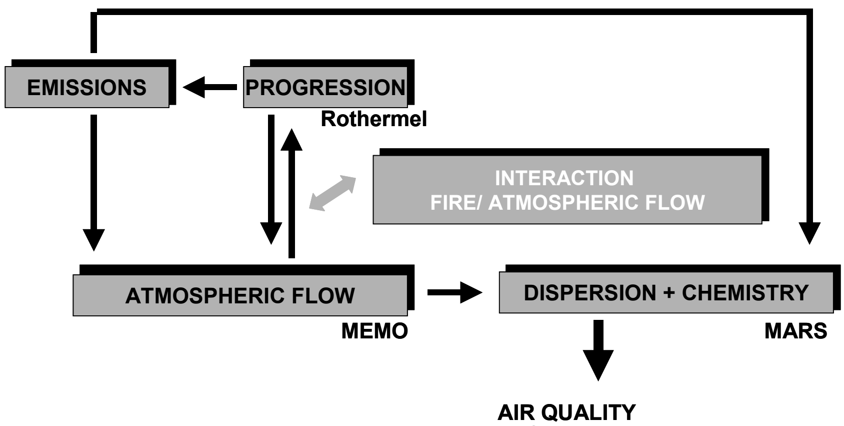


Figure 3. AIRFIRE modeling system (adapted from Borrego *et al.*, 1997).

The Rothermel model requires a priori knowledge of the mid-flame wind speed and gives as output the spread rate in the maximum spread direction, but does not provide any information about fire size and shape. Information concerning these two variables is, however, of vital importance for the estimation of fire emissions to the atmosphere. Within AIRFIRE fire growth simulation is performed using the deterministic model, based on Huygen's principle, which uses an elliptical spread at each point of the fire front (Anderson *et al.*, 1982).

In order to simulate the effect of a heat source like fire in the atmospheric flow, MEMO model has been modified, chiefly in the calculation of energy fluxes and atmospheric heating rate. The interaction between the fire spread model and flow field calculations is undertaken through a process of sequential computations, in which the velocity field obtained in a certain time step is used as input to the fire spread model, and the heat released by the fire is included in the atmospheric flow calculation, as a linear function of the burning cell area of the domain. More details on the model can be found in Miranda (2004).

The photochemical model

Since the final purpose of the integrated system is the calculation of air pollutant photochemical production and dispersion, the inclusion of the MARS model in the described system is a fundamental component. MARS, a 3D Eulerian model, numerically simulates photo-oxidants formation considering the chemical transformation process of pollutants together with its transport in the atmospheric boundary layer (Moussiopoulos *et al.*, 1995). The model solves the differential concentration transport equation system in terrain-following co-ordinates, with the meteorological variables calculated by MEMO. Two chemical mechanisms are included in MARS: EMEP, which describes the tropospheric gas-phase chemistry with 66 species and 139 photochemical reactions; and the KOREM mechanism, which is a simpler one, including 39 chemical reactions and 20 reactive pollutants. In KOREM mechanism, used in the present study, VOC are lumped in five classes: methane, alkanes, alkenes, aromatics and aldehyds.

MARS is a well-tested model in Europe (Baldasano *et al.*, 1993; San José *et al.*, 1997; Borrego *et al.*, 2000).

2.2 Emission estimation

Wildfire emissions depend on a number of factors, related with the fuel characteristics, the efficiency and the combustion phase, the type of fire, meteorological factors, the topography effects and the geographic location.

In characterizing emissions, two distinct combustion phases have to be considered: flaming and smoldering. In the flaming phase, the carbon dioxide and water vapor are the main emitted compounds, with fewer emissions of NO_x, sulphur dioxide and carbonated particulate matter. In this phase, oxidized emissions predominate due to the high burning efficiency. On the other hand, in the beginning of the smoldering phase reduced or partially oxidized emissions prevail, mostly CO, non-methane hydrocarbons (NMHC), ammonia (NH_3) and low carbon content particulate matter.

Emissions from wildfires can be determined from emission factors, expressed as a function of consumed biomass ($g.kg^{-1}$) or burnt area ($kg.ha^{-1}$). A wide range of emission factors is available from bibliography, according to the type of fire, combustion phase and type of fuel. Generically, emissions can be estimated from the emission factors through:

$$E_i = A \times B \times \beta \times EF_i$$

with:

E_i – compound *i* emissions (g)
A – burnt area (m^2)
B – fuel load or fuel mass ($kg.m^{-2}$)
β – global burning efficiency
EF_i – compound *i* emission factor ($g.kg^{-1}$)

The fuel load and the combustion efficiency are two of the main factors for the determination of emissions, changing with the geographic location, topography, fuel type and season. Seiler and Crutzen (1980) suggest 0.25 as a typical efficiency combustion factor for the Mediterranean forest.

Meteorological factors such as air temperature and relative humidity, precipitation and wind, directly affect the emissions. In Portugal, the presence of high temperatures, low relative humidity and strong East winds, favors the dryness of the fuel, increasing the fire risk and the fire spreading speed.

This forest fire emission approach was included as a specific module in AIRFIRE (Miranda, 2004).

3. AIR FIRE APPLICATION TO LISBON

Lisbon is a coastal city built in a complex topographic region, dominated by a large estuary and multiple hills, surrounded by small mountain ranges reaching heights over 400 m above sea level. The coastline is irregular through all the west side of the study region. The entire coastline present in the domain is subjected to atmospheric circulations resulting from local and synoptic forcing, with influence on pollutants transport, dispersion and formation.

The domain in analysis is heavily inhabited, with around 3.5 million inhabitants, representing around 35% of Portugal's population. In this area the industrial activity, mainly in Lisbon's surrounding cities, and the intense road traffic activity are responsible for one of the highest emission rates in Portugal.

3.1 Simulation conditions

The modeling system was applied to a domain of 200 km x 200 km, with a horizontal grid spacing of 4 km. In the vertical direction the grid consists of 20 layers non-equidistant with a minimum grid spacing of 20 m near the ground and a vertical extension of 6000 m.

MEMO input data includes: topography and land use, non-reactive species emissions and meteorology. Wind direction and speed and temperature data from four meteorological stations (Lisbon-Airport, Evora, Beja and Sines) were used. Radiosonde data, namely temperature and wind speed and direction, obtained at the Lisbon airport at 00.00 and 12:00 local standard time (LST) of September 13th (URL3) were used as initial and boundary conditions to the simulations, representing the synoptic forcing.

MARS model input includes topography and land use (the same from MEMO model), MEMO estimated meteorological variables (wind components, temperature, turbulent kinetic energy, Monin-Obukhov length, among others) and emissions. Initial and boundary conditions for the chemical species considered in the model were defined according to Moussiopoulos *et al.* (1995).

The emissions from the wildfires of September 13th were estimated. Table II presents the fuel load, emission factors and combustion efficiency for CO, CH_4, PM_{10}, NMHC and NOx.

Hydrocarbons (HC) speciation, presented in Table III, was selected according to the emission profile for biomass burning (EC, 1994), corrected for the use of KOREM photochemical mechanism.

Table III. Hydrocarbons speciation according to MARS.

Class	% of HC total mass
Methane	11.5
Alkanes	3.5
Alkenes	38.5
Aromatics	22.5
Aldehydes	8

Besides wildfire emissions the contribution of other pollution sources to air quality degradation was also taken into account. Emission data from the national emission database, that includes emissions associated to large point sources, traffic, industry, residential and commercial heating and natural sources. With the use of a geographic information system, emissions were re-calculated for each grid cell of the simulation domain. The pollutants considered were CO, NOx and VOC, the last one disaggregated in the groups considered by KOREM mechanism.

PM_{10} emissions were calculated through a per capita emission factor recommended by Pulles and Visschedijk (2003) for Europe (5.5 kg.inhabitant^{-1}.day^{-1}).

Table II. Fuel load, emission factors and combustion efficiency.

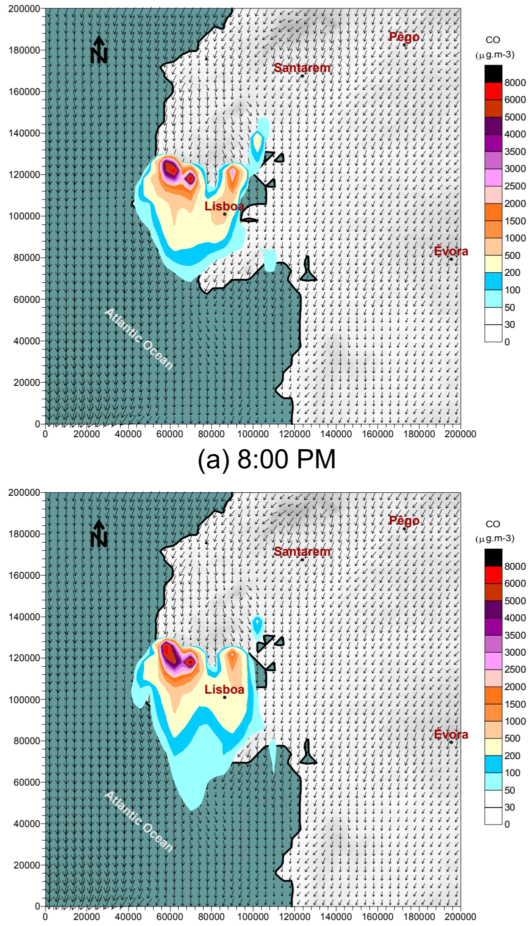
Fuel	Fuel load (kg.m^{-2})	Combustion efficiency	Emission factor (g.kg^{-1})				
			CO	CH4	NMHC	PM_{10}	NOx
Shrub	13 – 24	0.85	82	4	9	10	7
Pines	30 – 175	0.25	49 - 100	5 - 6	5 – 7	10	3 - 5
Eucalyptus	90 – 122	0.25	105 - 128	5 - 6	6 - 8	12 - 13	3 - 5

3.2 Results

Hourly wind and concentration fields, as well as fire progression lines, were estimated by AIRFIRE. In this paper results are presented for CO (concentration fields), O_3 (hourly evolution and statistical analysis) and PM_{10} (concentration fields, hourly evolution and statistical analysis).

Carbon monoxide

An analysis of atmospheric flow as well as of CO concentration spatial distribution along the day was performed, considering LMA emissions and emissions from the forest fires. Figure 44 presents the wind fields and horizontal ground level of CO concentrations at 8:00 and 10:00 PM. Maximum concentrations are reached at the end of the day (10:00 PM), with 9000 $\mu g.m^{-3}$ in the forest fires proximity and 1500 $\mu g.m^{-3}$ over Lisbon's center.



(a) 8:00 PM

(b) 10:00 PM

Figure 4. Wind fields and horizontal ground level of CO hourly concentrations.

Ozone

Ozone, when produced in the low atmosphere is the predominant element in the group of pollutants that constitute photochemical "smog", being used as an indicator of this phenomenon. The effect of the fires in the ozone measured in the monitoring stations is not so clear as the effect from primary pollutants (PM_{10} and CO) due to proximity of the fires in relation to the city.

Figure 5 presents the comparison between simulated and observed O_3 hourly concentrations for 4 Lisbon's air quality stations (Entrecampos, Reboleira, Olivais and Loures).

In general, simulated concentrations are inferior to the observed ones, however the daily evolution is well translated by the model. At the end of the day simulated concentrations diminish significantly, while observed ones remain high.

To evaluate quantitatively the obtained results a statistical analysis was performed based on three parameters: bias, correlation factor (CF) and root mean square error (RMSE). Table IV presents the obtained statistical results. The negative bias in all stations indicates under-estimation of the concentrations. The correlation is high, but the deviations significant, with similar results for all stations.

Table IV. Statistical analysis of O_3 simulated hourly concentrations.

Air quality station	Bias	CF	RMSE
Entrecampos	-32.8	0.94	35.9
Reboleira	-43.6	0.95	46.9
Loures	-39.2	0.69	55.8
Olivais	-22.7	0.70	30

Particulate matter

According to European Air Quality Legislation PM_{10} daily averaged values should not exceed 50 $\mu g.m^{-3}$ more than 35 times per year. PM_{10} concentration values are in general high, but due to forest fires influence this averaged value was exceeded at all the measuring stations, with hourly-averaged values reached 500 $\mu g.m^{-3}$.

With the objective of evaluating the influence of the smoke released by the fires in Lisbon's air quality, in the case of PM_{10}, AIRFIRE was applied to two distinct situations: i) considering LMA emissions for a normal week day; ii) considering LMA emissions and emissions from the forest fires. Figures 6 and 7 present the horizontal ground level of PM_{10} concentration at 6:00 PM and 10:00 PM, for the two simulated situations. The influence from the fires on PM_{10} concentration is clearly visible, with larger pollutant clouds and significantly higher concentrations.

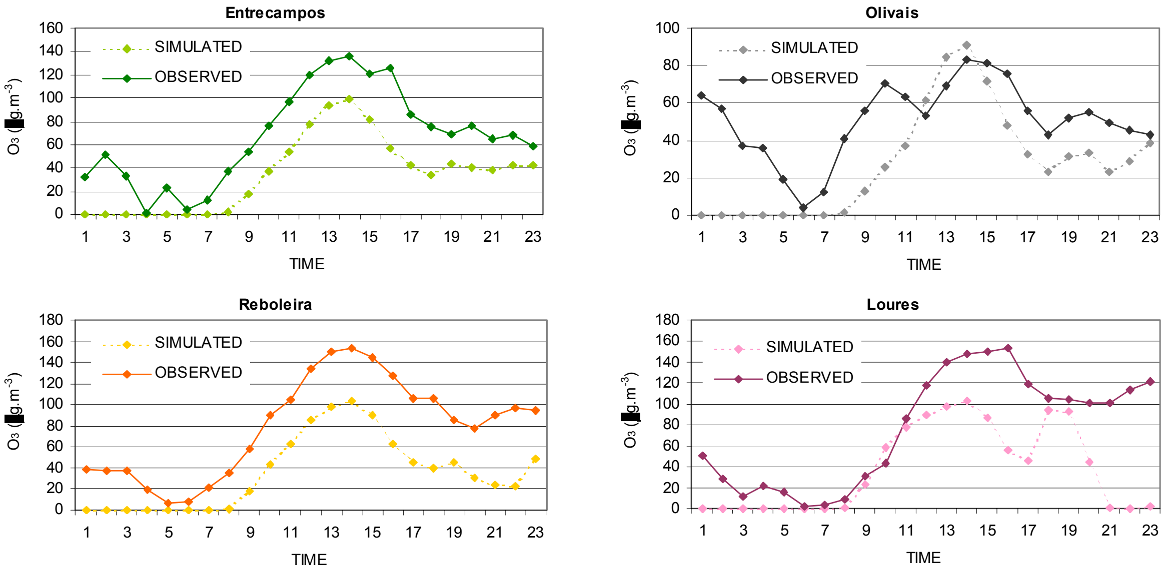
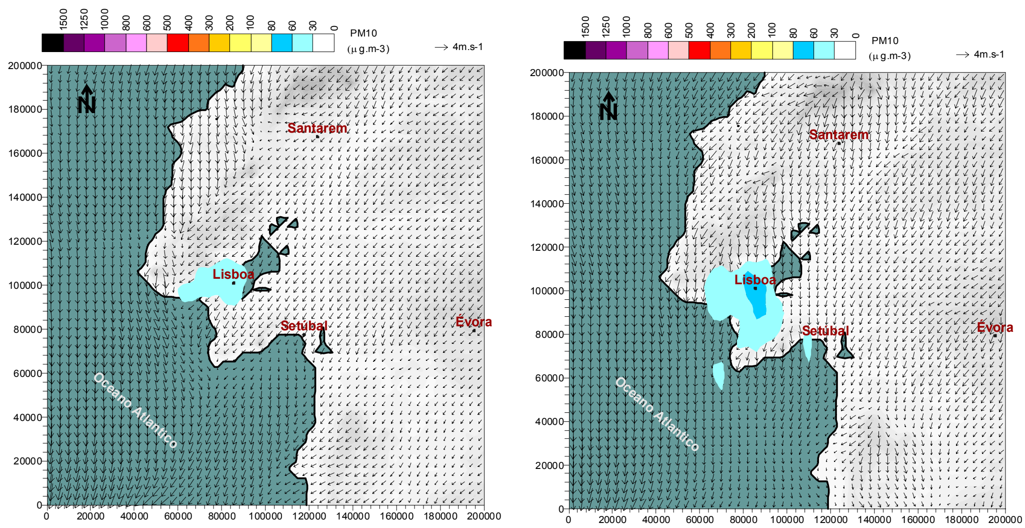
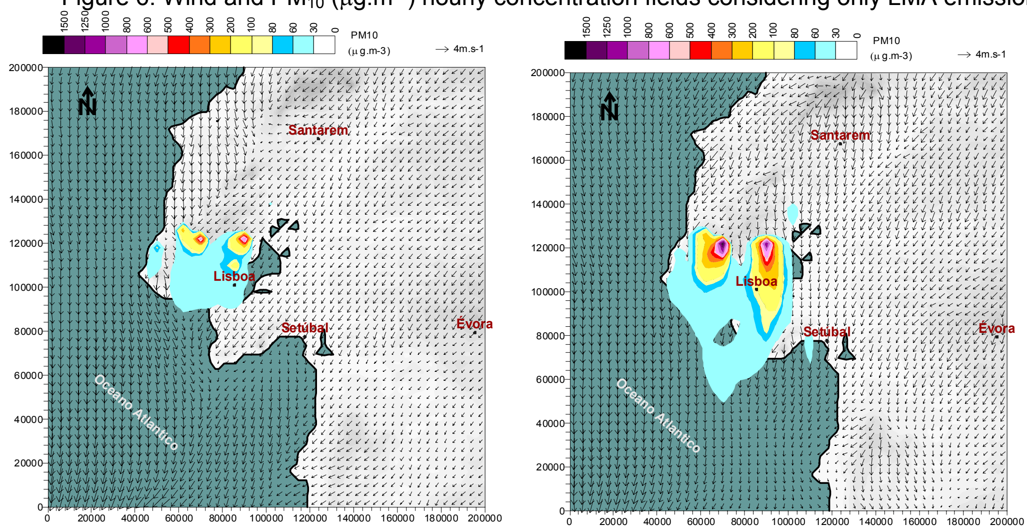


Figure 5. O_3 ($\mu g.m^{-3}$) simulated and observed hourly concentration evolution.



(a) 6:00 PM b) 10:00 PM

Figure 6. Wind and PM_{10} ($\mu g.m^{-3}$) hourly concentration fields considering only LMA emissions.



(a) 6:00 PM (b) 10:00 PM

Figure 7. Wind and PM_{10} ($\mu g.m^{-3}$) hourly concentration fields considering LMA and fire emissions.

Figure 8 presents the comparison between simulated and observed PM_{10} hourly concentrations, for some of Lisbon's air quality stations. A reasonable agreement between simulated and observed values was found. However, in Av. da Liberdade and Laranjeiro stations simulated values were significantly bellow observed ones, probably due to under-estimation of fire emissions associated to uncertainties in the factors involved in its calculation, namely fuel load and emission factors. The PM_{10} peak concentration registered in Olivais station doesn't coincide nor in time nor in intensity with the simulated peak. Due to its proximity with Tejo river, Olivais area may not be well simulated with the used numerical resolution (4 km x 4 km).

The obtained results were again quantitatively evaluated using a statistical analysis. Table V presents the results.

Table V. Statistical analysis of PM_{10} simulated hourly concentrations.

Air quality station	Bias	CF	RMSE
Av. da Liberdade	-56.6	0.77	126.9
Entrecampos	4.7	0.88	35.3
Cascais-Mercado	-6.8	0.19	23.5
Laranjeiro	-26.4	0.82	86.0
Loures	-8.5	0.81	34.6
Olivais	101.3	0.50	129.8

The negative bias in all stations, except Entrecampos and Olivais, indicates an under-estimation of PM_{10} concentrations. The values obtained are quite satisfactory, with Entrecampos station presenting the best results. Cascais and Olivais stations present the worst results with very low correlations. Av. da Liberdade present high RMSE due to high values registered at the end of the afternoon. Since Av. da Liberdade is a traffic station, highly influenced by local emissions, looking at the model resolution, the not so good performance of the model is understandable.

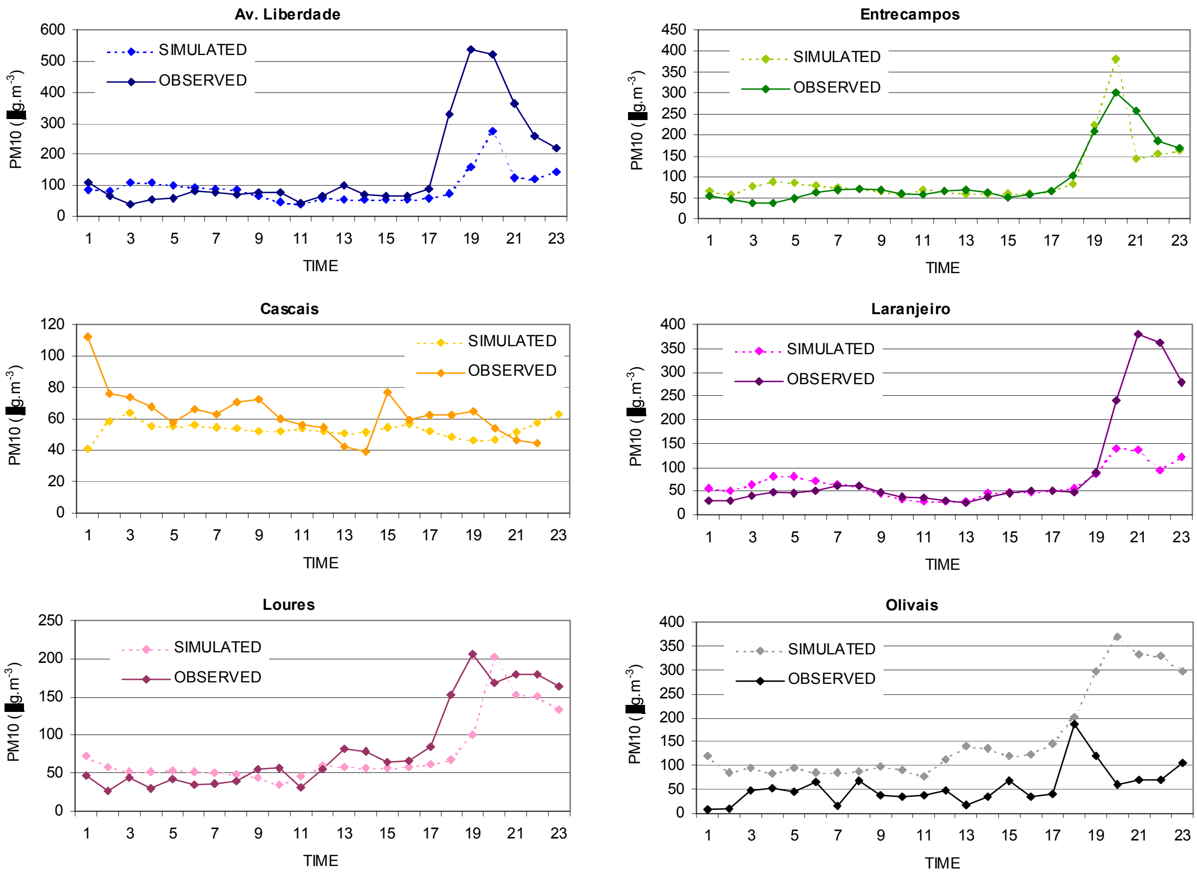


Figure 8. PM_{10} ($\mu g.m^{-3}$) simulated and observed hourly concentration evolution in Av. da Liberdade, Entrecampos, Cascais, Laranjeiro, Loures and Olivais air quality stations.

4. FINAL REMARKS

AIRFIRE numerical system was applied in the simulation and dispersion of the smoke resulting from various fires occurring north of Lisbon in September 13th 2003. The obtained pollutant fields allowed verifying the decisive influence of the smoke released in Lisbon's city air quality.

AIRFIRE simulations permitted the estimation of wind and concentration fields. The system presented a good performance for PM_{10} and CO results, revealing the effect of fire emissions on the concentration of these two pollutants. Ozone results were satisfactory although a significant under-estimation was verified.

This application allowed the analysis of the importance of urban/wildland fires on air quality at the urban level. The complexity of the coastal zone under study justified the use of AIRFIRE. Future studies can improve the results through a reduction of the uncertainty in emissions calculation and an increase of the numerical system's spatial resolution.

5. ACKNOWLEDGEMENTS

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URL3: http://weather.uwyo.edu/upperair/ (Wyoming University)