PERGAMON

Atmospheric Environment 36 (2002) 1769–1781

ATMOSPHERIC ENVIRONMENT

www.elsevier.com/locate/atmosenv

Comparison of indoor and outdoor concentrations of CO at a public school. Evaluation of an indoor air quality model

A. Chaloulakou[a], I. Mavroidis[b,*]

[a] *Chemical Engineering Department, National Technical University of Athens, 9 Heroon Polytechniou Str., 15 773 Zografou, Athens, Greece*

[b] *CERTH/CPERI, 6th Km Harilaou-Thermi Road, P.O. Box 361, 57 001, Thermi, Thessaloniki, Greece*

Received 1 June 2001; received in revised form 12 January 2002; accepted 23 January 2002

Abstract

A field study was carried out to investigate the internal and external carbon monoxide (CO) concentration levels of a public school building in Athens, Greece. Simultaneous measurements of indoor and outdoor CO concentrations were conducted using a non-dispersive infrared analyzer. Measurements of mean hourly CO concentrations inside and outside the sampling room were conducted on a 24-h basis for 13 consecutive days during May and June 1999 and for 14 consecutive days during December 1999. The aim of the study was to investigate the attenuation pattern of external pollution levels within the building. The diurnal concentration variations reported for different days during the week show that indoor CO concentrations are in general lower than the respective outdoor levels, and that the morning peaks of indoor concentrations show a delay of 1 h or less compared to the morning peaks of outdoor concentrations. The measured indoor to outdoor concentration ratios show a seasonal variation. An indoor air quality model for the prediction of indoor concentration levels developed by Hayes (J. Air Pollut. Control Assoc. 39 (11) (1989) 1453; J. Air Waste Manage. Assoc. 41 (2) (1991) 161) is coded as a computer program and evaluated using the experimental data. The model results are in good agreement with the indoor concentration measurements, although in some cases the model cannot respond adequately to sharp outdoor concentration changes. The ratio between measured and predicted daily maximum indoor concentration ranges between 0.88 and 1.23. The regression curve between predicted by the model and measured hourly indoor concentrations, for a continuous period of 96 h, has a slope of 0.64 and a coefficient of determination (R^2) of 0.69.

Keywords: Indoor air pollution; Carbon monoxide diurnal cycle; Air exchange rate; Sensitive population subgroup; Exposure

1. Introduction

Atmospheric pollution constitutes one of the major problems in urban areas. Increased concern over the adverse health effects of airborne pollutants has highlighted the need for air pollution measurement campaigns to monitor air pollution levels and also to assist the development and evaluation of predictive models. The Greater Athens Area has been facing increased air quality problems during the last three decades, due to population shift, increased industrialization, continuously increasing traffic and, finally, due to its topography. Therefore, there is a need for continuous analysis of meteorological and air quality observations from the Athens area (e.g. Kassomenos et al., 1995; Chaloulakou et al., 1998). Transport is a major source of air pollution in cities and therefore special attention is given to traffic pollutants, such as carbon monoxide (CO). In the city of Athens almost 100% of CO emissions is attributed to mobile sources (PERPA, 1989a).

High CO concentrations can cause acute CO intoxication since CO is combined with the hemoglobin (Hb) of human blood to produce carboxy-hemoglobin

*Corresponding author.
E-mail address: imavr@tee.gr (I. Mavroidis).

1352-2310/02/$ - see front matter
PII: S1352-2310(02)00151-6

(COHb), and therefore disrupts the transfer of oxygen to human tissues. Inhalation of air with a volumetric concentration of 0.3% CO can result in death within 30 min. Epidemiological studies have reported increased relative risks of daily mortality and morbidity of the population by 0.9–4.7% in prevailing urban air CO concentrations in Athens (Touloumi et al., 1994) as well as in Canadian cities (Burnett et al., 1997, 1998). To prevent COHb levels in the blood to exceed 2.5%, the World Health Organization (WHO) has set specific air quality guidelines for different averaging times (e.g. $30\,mg/m^3$ for 1-h averaging time). The effects of exposure to toxic pollutants such as carbon monoxide are more acute for sensitive groups of the population. One such group is children, since they have increased oxygen needs and this renders them more sensitive to high CO levels (Alm et al., 1994, 2000).

Most measurements of traffic-related pollutants are conducted using fixed site monitors at outdoor locations. However, there is an increasing need for indoor concentration measurements, since people are often spending up to 90% of their daily time indoors (Hoppe and Martinac, 1998), depending on the climate and the time of the year. Indoor concentration measurements are needed for determining the total personal exposure. It should be noted that ambient levels of air pollutants measured by fixed monitoring sites are a poor surrogate of personal exposure, especially for short averaging times (e.g. Alm et al., 1994, 2000). Furthermore, indoor concentration measurements are used for providing information about building penetration factors, mainly through the coupling of simultaneously measured indoor and outdoor concentrations. Indoor microenvironments contain in general lower concentrations of outdoor pollutants, such as sulfur dioxide or ozone. However, the presence of indoor sources of pollutants such as formaldehyde, carbon monoxide or nitrogen oxides can lead to increased indoor concentrations of these substances (Moschandreas, 1985). If there is a high ventilation rate, then the accumulation of indoor pollutants is reduced, but at the same time higher penetration of outdoor pollutants is expected within the indoor microenvironment (Hoppe and Martinac, 1998).

The practical difficulties of continuous measurement of indoor concentration levels and the wide range of reported values of indoor-to-outdoor (I/O) concentration ratios, have led to the development of modeling techniques which would enable the prediction of indoor concentration levels. These techniques range from simple regression models (e.g. Gold et al., 1996) to more demanding computational fluid dynamic techniques (e.g. Gilham et al., 2000). The need to develop dedicated, quick, easy to apply and quite reliable prediction tools, which would take into account all the parameters determining indoor concentration levels lead to the development of indoor air quality models (IAQM). IAQM are used for the estimation of indoor concentration levels, taking into account the type of microenvironment and its configuration. A multi-compartment IAQM solves simultaneously coupled mass-balance equations in a number of different compartments. A simpler version of the IAQM is the one compartment model (OCM) which is used for the prediction of indoor concentrations in a single compartment (Wadden and Scheff, 1983).

Hayes (1989, 1991) developed an IAQM which he evaluated using ozone measurements from 10 different microenvironments and 56 different population subgroups. The model results suggested that significantly lower population exposure to ozone may be estimated when periods indoors are taken into account (ranging from 10% to 80% of outdoor levels) and that accounting for the effect of structure configuration can be important. Shair and Heitner (1974) have developed a similar model, which was applied to several indoor ozone measurement databases. Their results suggested that for air-conditioned buildings it would be possible to reduce indoor ozone levels to 0.2 of those outdoors. Esmen (1978) proposed a general mass balance model for application to conservative, non-reactive pollutants. His model was similar to that proposed by Hayes (1989, 1991), but it did not include pollutant reactivity or air filter loss terms. Ishizu (1980) applied a similar relationship in an investigation of the effects of cigarette smoking on indoor air quality. Dockery and Spengler (1981) employed a mass balance-based model to investigate indoor air concentrations of respirable particulates and sulfates. Their model however did not include a mixing factor or a recirculation filter loss term. Moschandreas et al. (1981) developed an indoor mass-balance model which did not include a mixing factor and did not distinguish infiltration from outdoor make-up air. Allen and Wadden (1982) applied a model similar to that described in Shair and Heitner (1974) to predict indoor CO and ozone concentrations in different microenvironments. The correlation coefficient between predicted and observed hourly indoor CO concentrations ranged between 0.19 and 0.71. Leaderer et al. (1984) applied a mass-balance model to estimate the ventilation requirements in buildings in order to control CO and particulate matter concentrations from cigarette smoking. Marr et al. (1998) applied a simplified model to predict CO concentrations within a residential building room and a garage. The model results were used as input to a model predicting COHb levels in human blood. Bouhambra et al. (2000) performed measurements to determine indoor benzene levels in 26 residential houses in Kuwait, and found that time-averaged indoor concentrations varied linearly with outdoor concentrations, in accordance with a mass-balance based IAQM. A similar model has also been applied to calculate hourly indoor NO_2 concentrations

in four different microenvironments by Dimitroulopoulou et al. (2001). The authors linked the model with an exposure model to estimate the exposure of different population subgroups to nitrogen dioxide.

The present work aims to determine the contribution of outdoor concentrations of pollutants to the concentration levels observed within a public school located in the vicinity of a busy street, near the center of Athens. Measured indoor CO concentrations are compared to the predictions of an IAQM developed by Hayes (1989, 1991).

2. Experimental details

The experimental site is a public school located on Mayer Street in central Athens. A map of the site with the adjacent roads is shown in Fig. 1. The nearest major road is Acharnon Street. The average traffic flow along Acharnon Street is 57,000 vehicles/day. Indoor concentration measurements were conducted in a corner room, located at the first floor of the building. The volume of the room is 225 m^3. The room has two windows and a door looking inside the building. Both indoor and outdoor concentration measurements were conducted using a Dasibi 3008 non-dispersive infrared (NDIR) CO analyzer. The operation of the analyzer is based on the ability of CO to absorb infrared radiation of a certain wavelength. The standard range of operation of the analyzer is between 0.1 and 50 ppm, and its lag time is $\sim$2 s. The instrument was located so that the indoor concentration measurements were conducted at the same height with the outdoor concentration measurements. The measurements reported in this paper were conducted in two different periods: (a) the period between Tuesday 25 May and Sunday 6 June 1999, and (b) the period between Thursday 2 December and Wednesday 15 December 1999. Each measurement period included two weekends and measurements were conducted on a 24-h basis. Thus, concentration measurements in two different seasons were conducted, which enabled comparison of the indoor and outdoor concentration levels observed in periods during different seasons. During both measurement periods, meteorological data were simultaneously measured at the fixed monitoring site of the Greek Ministry for the Environment, Physical Planning and Public Works at the nearby Patission Street. The minimum, maximum and mean daily values of the meteorological parameters during the measurement campaign are shown in Table 1, together with the minimum, maximum and mean daily values of the same parameters during summer (15 May–15 September) and winter (November–December) periods of the same year (1999).

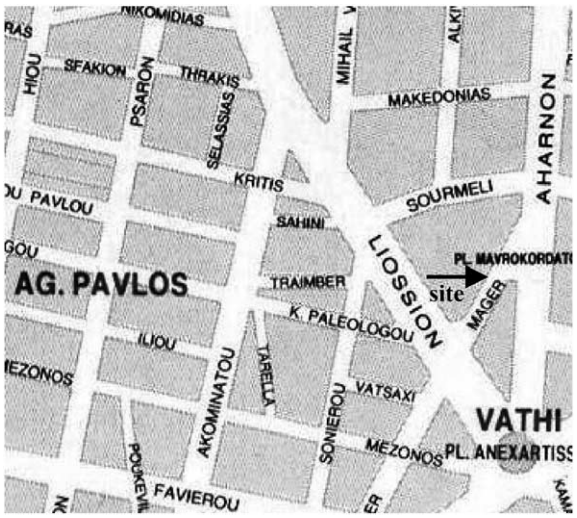

Fig. 1. Map of experimental site with adjacent roads (scale 1:7000). The location of the experimental site is indicated with the arrow.

Zero and span were checked at regular intervals using a standard CO concentration. The analyzer was equipped with two samplers and with a mechanism that enabled the sequential activation of each of the two samplers alternatively, so as to allow both indoor and outdoor concentration measurements to be conducted using the same instrument. The distance of the two samplers of the detector from the window was about 0.5 m for both indoor and outdoor concentration measurements. The instrument was recording CO concentrations every 30 s and was programmed to record every hour two 10-min periods of outdoor concentrations and two 10-min periods of indoor concentrations. Each 10-min period of outdoor concentration measurements was followed by a 10-min period of indoor concentration measurements. After each 10-min measurement period there was a 5-min interval for cleaning the samplers.

3. Model description

The model applied for the prediction of indoor concentrations is a one-compartment indoor air quality model, which solves the mass-balance equations for pollutant flow into and out of an indoor volume, including interior sources. The model was developed by Hayes (1989, 1991) in a multi-compartment and in a single-compartment version. The single-compartment version has been coded as a computer code and is used in the present paper. The single-compartment model solves the mass continuity equation for a reactive pollutant in a single compartment that is subject to infiltration from outside (both through a building

Table 1
Summary of daily values of the meteorological parameters during the experimental period and during the summer and winter periods of year 1999

	Summer			Winter			Experimental period					
	15/5–15/9/1999			1/11–31/12/1999			25/5–6/6/1999			2/12–15/12/1999		
	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean
Temperature (°C)	20.7	35.8	28.4	7.8	18.8	15.2	20.9	31.0	25.8	13.4	17.9	15.7
Wind speed (m/s)	1.1	4.8	2.3	0.7	6.5	2.5	1.7	3.8	2.4	1.0	3.8	1.9
Relative humidity (%)	26.0	77.0	48.4	57.3	87.0	67.8	34.0	63.0	46.5	62.0	82.0	70.0

ventilation system and normal building air leakage) and within which air is being recirculated. Pollutant material can be generated by indoor sources and removed from indoor air through adsorption on indoor surfaces. The single-compartment equation is (Hayes, 1991)

$$\frac{dc_i}{dt} = \underbrace{ka_M(c_o - c_i)}_{\text{Infiltration}} + \underbrace{ka_F(E_F c_o - c_i)}_{\text{Outdoor make-up air}} + \underbrace{ka_R(E_R - 1)c_i}_{\text{Recirculation air}} - \underbrace{kK(A/V)c_i}_{\text{Surface losses}} + \underbrace{S/V}_{\text{Indoor sources}}, \quad (1)$$

where

c_i is the indoor concentration (mg/m^3),
c_o the outdoor concentration (mg/m^3),
k the room mixing factor. This corresponds to the fraction of the pollutant mass which is uniformly mixed throughout the indoor microenvironment. Typical values of k range between 0.3 and 1 (Wadden and Scheff, 1983).

a_M (ACH) the infiltration flow rate. Infiltration may occur through openings and gaps (including open doors, windows, joints and cracks) allowing air either to enter or to leave the building. Typical values of a_M for a house or office are 1 ACH for closed windows, 2.4 ACH for half-open windows and 6.4 ACH for open windows (Hayes, 1989; Moschandreas et al., 1981).

a_F (ACH) the outdoor make-up air flow rate (passing through a filter). Even if a heating, ventilation and air conditioning (HVAC) system is not in operation, outdoor air enters the building through this system.

E_F the efficiency of the make-up air filter,

a_R (ACH) the flow rate of recirculated air. Typical air flow rates through HVAC systems (a_F, a_R) are 7.71 ACH for non energy efficient and 10 ACH for energy efficient buildings.

E_R the efficiency of the recirculation air filter,

S the indoor source term (mg/h),

K the pollutant reactivity factor. Concentrations of reactive pollutants decrease through heterogeneous decompositions on the surfaces of indoor microenvironments. The value of K depends on the pollutant, on the surface material and on the relative humidity (m^3/m^2 h),

V the room volume (m^3), A is the interior surface area (m^2)

(all flow rates are measured in air changes per hour (ACH)).

The mass-balance equation is integrated over an averaging period of length T, assuming that the parameters of the equation remain constant over this averaging period. Two basic approaches, analytical or recursive, may be considered for the solution of the indoor mass-balance equation. In problems like the one addressed in this paper, pollutant concentrations are time varying in ways that are not well characterized by analytical functions and also projection to cases for which measurement data are not available is necessary. Therefore, a recursive solution to the equations should be obtained, similar to the one employed by Dockery and Spengler (1981). Replacing the integrals of Eq. (1) with average values and assuming that the indoor concentration term (c_i) can be adequately represented by a linear relationship over the averaging time interval T, then the final form of the mass-balance equation is obtained (Hayes, 1991):

$$c_{i(n+1)} = D_1 c_{i(n)} + D_2 \frac{c_{o(n)} + c_{o(n+1)}}{2} + D_3 \frac{S_{(n)}/V + S_{(n+1)}/V}{2}, \quad (2)$$

where

$$D_1 = \frac{2 - b_1}{2 + b_1}, \quad (3)$$

$$D_2 = \frac{2kT(a_M + a_F E_F)}{2 + b_1}, \quad (4)$$

$$D_3 = \frac{2T}{2 + b_1}, \quad (5)$$

$$b_1 = kT(a_M + a_F + a_R(1 - E_R) + K(A/V)), \quad (6)$$

where subscript $(n+1)$ indicates the current time step, i.e. the value of the parameter at the end of the averaging time T, and subscript (n) indicates the previous time step, i.e. the value of the parameter at the start of the averaging time T.

4. Results and discussion

4.1. Comparison of indoor and outdoor concentration measurements

The use of a modified analyzer, which enabled the sequential activation of each of the two samplers alternatively, allowed the formulation of mean hourly concentration time series on a 24-h basis, for both indoor and outdoor measurement locations. To examine the weekly variation of hourly time series, using more statistically stable data, concentrations were grouped into three main categories: (a) weekdays, (b) Saturdays, and (c) Sundays, to account for the different traffic conditions and the different operation conditions of the school, which was open only during weekdays. Mean hourly concentrations for each of these three groups were averaged over several days. This analysis procedure was followed for the concentration measurements conducted both in summer and in winter.

For the summer period (25 May–6 June 1999), the diurnal cycles of indoor and outdoor concentrations for weekdays, Saturdays and Sundays are presented in Figs. 2a, b and c, respectively. The diurnal cycle of outdoor concentrations for the weekdays (Fig. 2a) has two characteristic concentration peaks, one during the morning (8:00–9:00 a.m.) and one during the night (22:00–23:00 p.m.). The morning peak corresponds to the peak traffic due to people commuting to their jobs. Furthermore, at that time of the day meteorological conditions are in general favoring the accumulation of high concentrations of pollutants (low wind, morning inversion, etc.). The daily variation of CO concentrations is attributable to a high degree on the atmospheric stability changes during the day, even more than to the traffic volume variations (Tryfonopoulos and Bergeles, 1994). The night peak, which is much lower than the morning peak, corresponds to the increased traffic due to the market closing, as well as due to people going out for entertainment. The night peak is also partly attributable to the stabilizing atmospheric conditions observed at night hours (Katsoulis, 1996). This daily variation of CO is typical of the center of Athens, as is also shown by CO concentration timeseries measured at the fixed monitoring site of the Greek Ministry for the Environment, Physical Planning and Public Works at the nearby Patission Street (e.g. PERPA, 1989b). Indoor CO concentrations follow a similar trend to that of outdoor concentrations. However, indoor concentrations are lower than outdoor concentrations, as expected, since the main source of CO is traffic. The morning peak of indoor concentrations occurs with a delay of 1 h or less as compared to outdoor concentrations. This is a typical behavior characterizing the response of indoor concentrations to changes in outdoor concentration levels (Hayes, 1991). Furthermore, indoor concentration peaks are much lower than the outdoor ones, since indoor concentrations can only mildly respond to sharp outdoor concentration changes. For example, the "instantaneous" hourly I/O ratio for the morning peak is only 0.4.

On Saturdays (Fig. 2b) outdoor concentrations also present two peaks. As for weekdays, the morning peak corresponds to people commuting to their jobs or to the market. The night peak corresponds to the increased traffic due to people going out for entertainment. The morning concentration peak is 20–50% lower than that observed during the weekdays, since on Saturdays only the market is open in the morning. Indoor concentration levels during Saturdays remain quite low, mainly because the traffic is lower and the door and the windows of the room are closed. Finally, on Sundays (Fig. 2c) there is a concentration peak at the early morning hours, corresponding to people coming back from Saturday night entertainment. Shops are closed and therefore there is no concentration peak in the morning, as was the case for weekdays and Saturdays. However, increased concentrations are observed after 20:00 p.m. in the evening, which are attributed mainly to people returning to the city at the end of the weekend and, to a smaller extent, to people going out for entertainment. The windows and the door of the room remain closed on Sundays and therefore indoor concentrations are quite low and follow only slightly the outdoor peaks.

For the winter period (2–15 December 1999), the diurnal cycles of indoor and outdoor concentrations for weekdays, Saturdays and Sundays are presented in Figs. 3a, b and c, respectively. A comparison of Figs. 2 and 3 shows that diurnal cycles follow very similar patterns in summer and winter, not only during weekdays, but also during Saturdays and Sundays.

From the above, it is clear that peak outdoor as well as indoor CO concentrations are observed during the morning hours of the weekdays, which are the hours that children are commuting to school or are within the class. The diurnal cycles of outdoor CO concentrations observed during both the summer and winter experimental periods are in agreement with the yearly averaged diurnal cycle of outdoor CO concentrations measured during 1999 at the fixed monitoring site of the Greek Ministry for the Environment, Physical Planning and Public Works at the

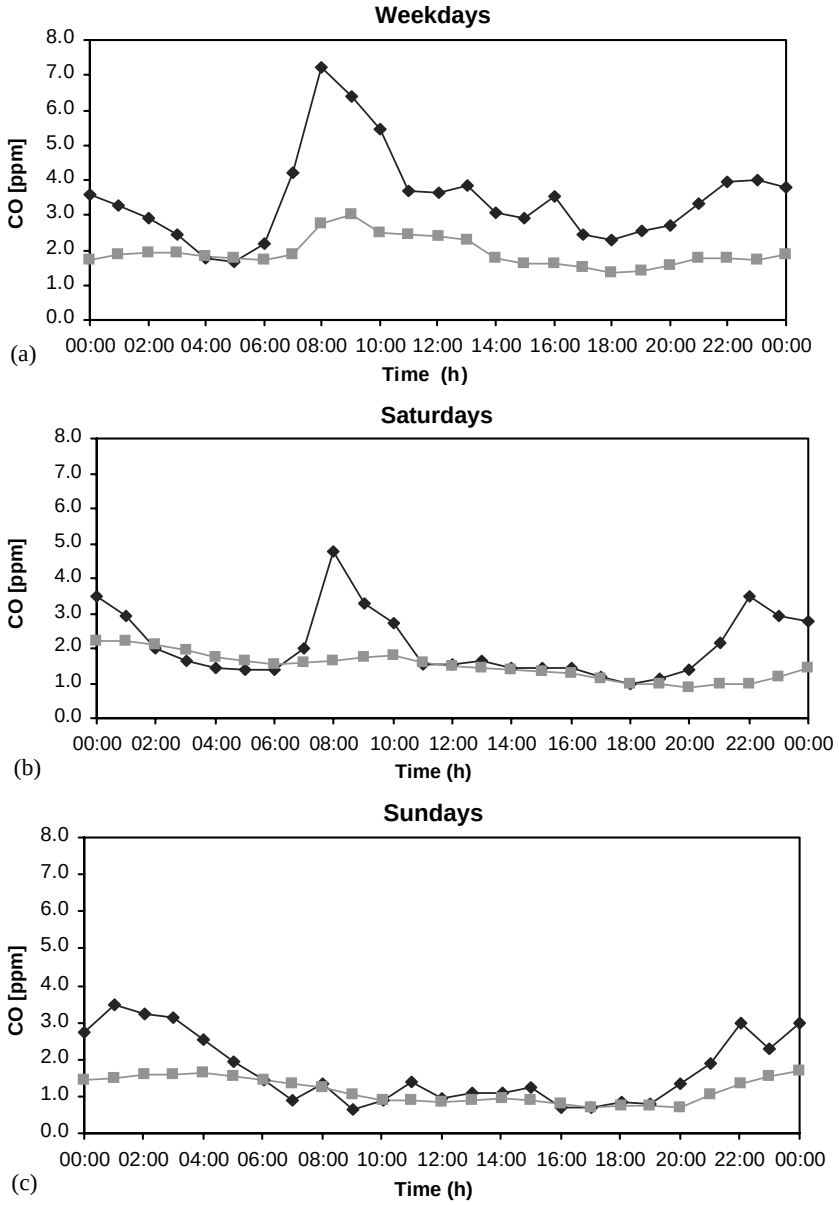

Fig. 2. Average diurnal cycle of summer outdoor (♦) and indoor (■) CO concentrations for (a) weekdays ($N = 9$), (b) Saturdays ($N = 2$) and (c) Sundays ($N = 2$).

nearby Patission Street (Direction of Air Pollution and Noise Control, 2000)

The mean indoor and outdoor concentrations observed during the winter and summer experimental periods, for weekdays, Saturdays and Sundays, together with the mean and median I/O ratios are shown in Table 2. The main difference between the summer and the winter period is that in general CO concentrations observed in the winter are higher than CO concentrations observed in the summer. The differences between seasonal concentrations were found to be statistically significant, using a paired t-test ($p < 0.05$). This is attributed (a) to the higher traffic volume observed in winter, since the weather favors the use of vehicles, (b) to the different engine operating conditions, due to the lower winter temperatures, (c) to the prevailing meteorological conditions, which favor accumulation of pollutants during the winter, and (d) to the increased depletion of CO during summer, due to the participation of CO in the photochemical atmospheric reactions.

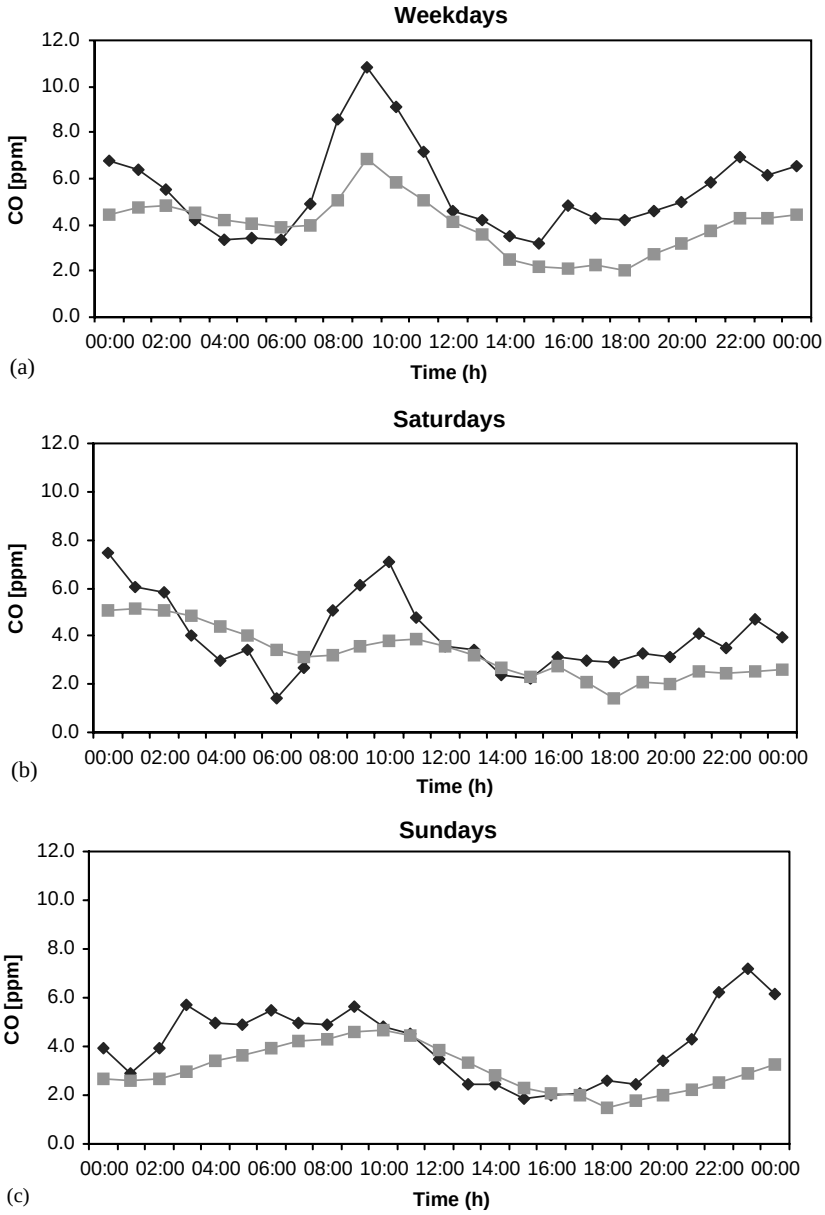

Fig. 3. Average diurnal cycle of winter outdoor (♦) and indoor (■) CO concentrations for (a) weekdays ($N = 10$), (b) Saturdays ($N = 2$) and (c) Sundays ($N = 2$).

The mean I/O concentration ratio both for summer and winter is generally lower for weekdays than it is for Saturdays and Sundays. This difference is mainly due to the higher outdoor concentration values observed during the weekdays, which are not followed by comparatively higher indoor concentrations. The observed I/O ratios are within the range reported in the literature for CO concentrations (e.g. Hoppe and Martinac, 1998). Hayes suggested that for open windows, reported I/O ozone concentration ratios range from 0.6 to 0.7, while for closed windows I/O ratios ranged from 0.65 to 0.21, which can go as low as 0.19 for energy efficient buildings. A study by Bouhambra et al. (2000) in Kuwait showed that a typical house with almost no indoor benzene releases had an I/O benzene concentration ratio of about 0.7. Differences in observed I/O ratios may be attributed to the different pollutants examined as well as to the differences in building design and construction between different countries.

Table 2
Summary of mean indoor and outdoor concentrations, standard deviation (S.D.) and I/O concentration ratios, during the winter and summer experimental periods, for weekdays, Saturdays and Sundays

	Summer			Winter		
	Weekdays ($N = 9$)	Saturdays ($N = 2$)	Sundays ($N = 2$)	Weekdays ($N = 10$)	Saturdays ($N = 2$)	Sundays ($N = 2$)
Outdoor CO (ppm)	3.47	2.09	1.71	5.50	4.00	4.13
S.D. (ppm)	0.73	0.75	0.61	2.22	1.85	1.48
Indoor CO (ppm)	1.92	1.49	1.17	3.96	3.28	3.06
S.D. (ppm)	0.30	0.41	0.22	1.91	1.48	1.15
Mean I/O ratio	0.55	0.71	0.68	0.72	0.82	0.74
Median I/O ratio	0.57	0.91	0.81	0.82	0.90	0.74

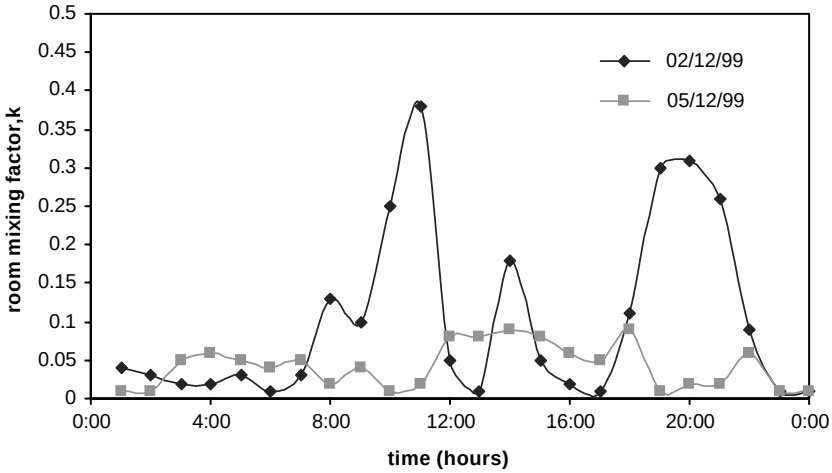

Fig. 4. Room mixing factor (k) variation for a weekday (2 December 1999) and a Sunday (5 December 1999) calculated using Eq. (7).

4.2. Comparison of measured indoor concentrations with IAQ model results

The IAQM described in Section 3 of this paper was applied for the prediction of indoor CO concentrations. Concentration measurements showed that both indoor and outdoor concentrations are higher in the winter period, and therefore the possibility that CO concentrations will exceed air quality guidelines is larger in winter. Therefore, the IAQ Model is applied for four consecutive days during the winter period, to allow comparison of the model predictions with measured indoor concentrations and to evaluate the model performance. The assumptions made for the estimation of the main model parameters in the present application are summarized below:

(i) *Infiltration flow rate* (a_M): In the present case the model is applied for the measurements conducted in the winter period, when windows were closed. Therefore, and since there are not such measurements available for buildings in Greece, the infiltration flow rate (a_M) is taken to be equal to 1 ACH. This value is also in agreement with the values reported in recent literature (Dimitroulopoulou et al., 2001).

(ii) *HVAC air flow rate* (a_F, a_R): In the present application of the model there is no air conditioning system, and therefore the HVAC air flow rate (a_F, a_R) is zero.

(iii) *Pollutant reactivity factor* (K): CO does not have a significant reactivity and can be treated as if it was conservative. Thus K is set equal to zero.

(iv) *Indoor source term* (S): The main indoor sources of CO are cigarette smoking and house appliances. The indoor source term is taken to be zero in the present application, since there were no appliances within the room and smoking was not allowed.

(v) *Room mixing factor* (k): The room mixing factor is usually not constant, since it depends in many

parameters which may vary in time. A reverse method may be used to estimate the "hourly" values of the room mixing factor using Eq. (7) (Wadden and Scheff, 1983)

$$k = \left| \frac{\log c_{i(n+1)} - \log c_{i(n)}}{T} \right|. \quad (7)$$

The above equation was applied for a typical weekday (2 December 1999) and a Sunday (5 December 1999), and the results are shown in Fig. 4. The values of the room mixing factor range from 0.01 to 0.38. The values shown in Fig. 4 indicate that k varies significantly for Thursday 2 December 1999, while it is more constant for Sunday 5 December 1999. The mean values of k are 0.10 for Thursday 2 December 1999 and 0.05 for Sunday 5 December 1999. However, it should be noted that for days with a large variation in k, such as Thursday 2 December 1999, it is not appropriate to use the mean value of k as the ideal value for this parameter, since in that case the model would not be able to respond to large outdoor concentration peaks. The subsequent underestimation of the indoor concentration peaks would be hazardous for the public health, if the model is to be used as a regulatory prediction tool. Since a single value of the room mixing factor should be used in the model, it is preferred to use as an input to the model the value 0.3, which is within the range of measured values and close to the higher "instantaneous" values of k observed. This value is also in agreement with the lower limit of the values suggested in the literature (e.g. Wadden and Scheff, 1983). Such a value of k was expected in the present case, since the volume of the room is quite large (225 m^3), the roof is high, and the windows small. Therefore, and since there is no HVAC system and the door is mainly closed, the mixing of the pollutant within the room would not be adequate.

To use the IAQM as a prediction tool for calculating indoor concentrations, the model should be applied using only the measured outdoor concentrations as input. Therefore, the model is used in a time-varying mode and the concentration calculated by the model at the previous time step is taken as $c_{i(n)}$ (Eq. (2)). For calculating the concentration at the first time step, the outdoor concentration measured at the first time step is taken as $c_{i(0)}$. The model was applied for a period of 96 h, i.e. for four consecutive days, including two weekdays with the higher concentration peaks (Thursday 2 December and Friday 3 December) followed by two weekend days (Saturday 4 December and Sunday 5 December). In Fig. 5 the daily cycles of measured indoor and outdoor concentrations, together with the calculated indoor concentrations, are presented for the four consecutive winter days.

The first 48 h correspond to two weekdays, namely Thursday 2 and Friday 3 December 1999, while the next 48 h correspond to Saturday 4 and Sunday 5 December 1999. The two concentration peaks observed during the first 24 h (2 December 1999) are similar to those observed in Figs. 2a and 3a, and correspond to the peak traffic hours and worst atmospheric conditions for pollution built up. Measured indoor concentrations respond rapidly to the changes of outdoor concentrations. The indoor concentrations predicted by the model follow the changes of the measured indoor concentrations, but the response of the model is delayed and cannot follow exactly the sharp outdoor concentration changes. For example, between 7:00 and 10:00 a.m. there is an increase of approximately 10 ppm in outdoor concentrations, followed by a subsequent decrease of outdoor concentrations to values lower than the initial ones.

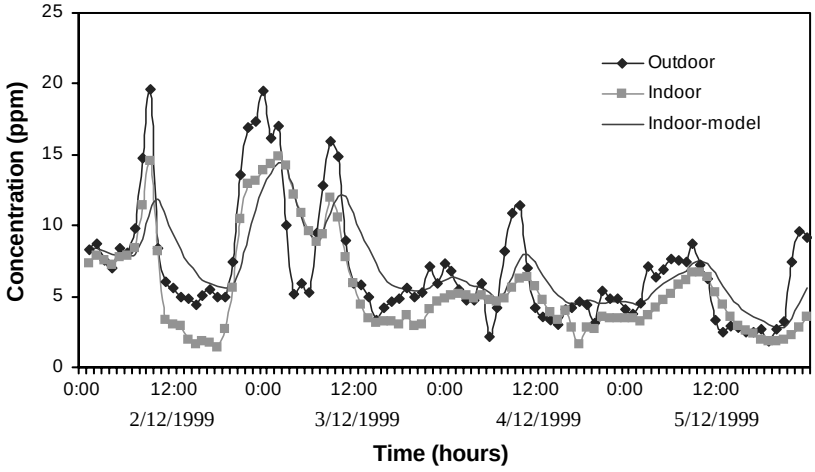

Fig. 5. Measured outdoor, measured indoor and calculated model indoor hourly CO concentrations for a period of four consecutive days (2–5 December 1999).

After the sharp concentration peak observed at 9:00 a.m., and the corresponding peak calculated by the model, the concentrations predicted by the model cannot follow the observed concentration decrease and remain higher than the measured indoor concentrations for a period of about 10 h, i.e. between 11:00 and 20:00 p.m. This indicates that the model cannot respond quickly to the decrease of concentrations, since the model predictions are affected by the measured outdoor concentrations as well as by the calculated indoor concentrations of previous time steps, and therefore have a delayed response to such changes. When the model is calculating the next indoor concentration value, the outdoor concentrations have already changed back, and the prediction is smoothed out. Inadequacies in the estimation of indoor concentration changes may be enhanced by inaccuracies in the estimation of the model parameters, e.g. by the model assuming a value of the infiltration flow rate which is lower than the actual one (e.g. Dimitroulopoulou et al., 2001). In a similar study in an urban hospital (Allen and Wadden, 1982), predicted and observed annual average CO concentrations were in good agreement, while predicted versus observed CO concentrations for small periods did not correlate as well. This was attributed to the presence of indoor CO sources, either from offices within the examined unit or from other areas on the same floor.

In the next day (3 December 1999), there are two successive concentration peaks of sustained duration during the morning hours, which are separated by a low concentration period of a short duration. In the short time period between the two peaks, measured indoor concentrations remain higher than outdoor ones. In this occasion, the model responds very well to indoor concentration changes and predicts the concentration peaks accurately. After the second concentration peak, the model cannot respond adequately to the decrease of concentrations observed. It should be noted that, in order to use such a model for regulatory purposes or as an alarm for informing the population when indoor concentrations would exceed certain limits, it is important to adjust the model parameters so as to accurately predict high concentration peaks. The fast response of the predicted concentrations to the low indoor concentration values observed after a measured concentration peak, is of secondary importance in this respect.

The concentration peak observed during the night hours of Friday 3 December is much lower, and the response of measured indoor concentrations to the outdoor concentration peak is mild. In this case the model slightly overestimates the indoor concentration peak. This is partly attributable to the fact that the model predictions could not adequately follow the drop of indoor concentrations after the previous concentration peak. A similar behavior is observed during the morning hours of the next day (4 December 1999). In this case concentration peaks are lower since this day is a

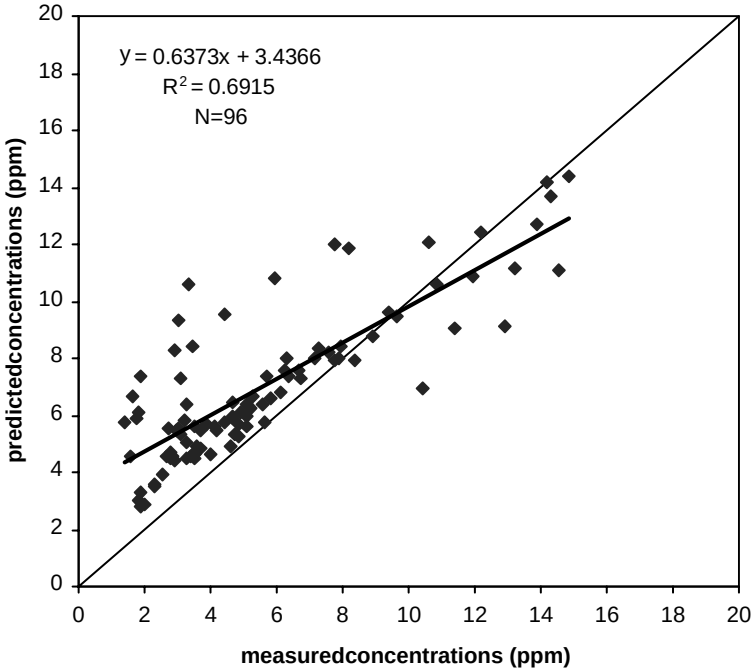

Fig. 6. Predicted versus measured hourly indoor CO concentrations, for the period of four consecutive days (2–5 December 1999).

Saturday and activity is in general reduced. The outdoor concentration peak measured during the last 24 h (5 December 1999), which correspond to a Sunday, i.e. a day with even more reduced activity, is much lower. Therefore, concentration changes are not as sharp and the response of the model is very satisfactory, although sometimes the model slightly overestimates indoor concentrations. In general, during the last 48 h, when the changes in concentrations are not as sharp, the model can respond more accurately to actual concentration changes.

Fig. 6 shows the predicted versus the measured hourly indoor CO concentrations for the entire period of 4 days presented in Fig. 5, together with the regression curve. Again it is clear that in general the model performs quite well ($R^2 = 0.69$), but it has the tendency to overpredict concentrations in the lower range, while on the other hand it underpredicts concentrations in the higher concentration range. This tendency is explained by the reasons described in the previous paragraphs, since the higher concentration range corresponds to the peak concentrations, and the lower concentration range corresponds to the low concentrations usually observed after a sudden concentration peak.

5. Summary and conclusions

Indoor and outdoor CO concentrations were measured at a school near the center of Athens on a 24-h basis, for 13 consecutive days during May and June 1999 and for 14 consecutive days during December 1999. The indoor and outdoor diurnal concentration cycles followed similar patterns, with indoor concentrations showing a mild and slightly delayed response to outdoor concentration changes. CO concentrations observed during winter are higher than the respective concentrations observed during summer. This is attributed to the higher traffic volume and the different engine operating conditions observed in winter and also to the prevailing meteorological conditions, which favor accumulation of pollutants during winter. The concentration measurement campaign also showed that the highest concentrations—both indoors and outdoors—are observed during the morning hours of weekdays, when children are commuting to school or are within the class. Further investigation of this matter is required, probably in conjunction with exposure measurement campaigns (e.g. Alm et al., 2000) or with the use of exposure models (e.g. Dimitroulopoulou et al., 2001), taking however into consideration that the relationship between personal exposure and fixed monitoring site concentrations is typically weak, especially for short averaging times.

Mean daily I/O concentration ratios ranged between 0.49 and 0.79 in the summer period and between 0.53 and 0.89 in the winter period. For weekend days the mean I/O ratios were similar for winter and summer. For weekdays the mean I/O ratios were lower in summer than in winter. I/O concentration ratios agreed with those reported in the literature in the case where indoor sources are not present (e.g. Hoppe and Martinac, 1998). The I/O concentration ratios measured were lower for weekdays than for Saturdays and Sundays and this is attributed to the fact that during the weekdays sharp, high, outdoor concentration peaks are observed and indoor concentrations cannot follow adequately these peaks.

In most cases, the I/O concentration ratios can be reduced significantly if buildings are designed and constructed in a more energy-efficient way (Hayes, 1991), since indoor peaks are dampened and lagged behind outdoor peaks. It would be advantageous therefore, to design schools in a more energy efficient way, taking into consideration, however, that the presence of indoor sources in certain microenvironments may increase dramatically indoor concentration levels, if adequate ventilation is not available. It should be also noted that when outdoor peaks are of sustained duration and separated by periods of short duration, like in the case of the morning of Friday 3 December 1999, buildings must be very tightly constructed to avoid high levels of concentrations to remain within a classroom.

An IAQM developed by Hayes (1989, 1991) is briefly reported and the parameters of the model, together with the necessary assumptions made for the application of the model in the present case, are described. The model was applied for the prediction of indoor concentrations, using measured outdoor concentrations as input, and the model predictions are compared with the indoor concentrations measured for certain days during winter, when CO concentrations are generally higher. The model predicts indoor concentrations quite accurately in general, since the regression curve between predicted by the model and measured hourly indoor concentrations, for a continuous period of 96 h, has a slope of 0.64 and a coefficient of determination (R^2) of 0.69.

It is often the case that the model cannot respond quickly to the decrease of concentrations observed after a sharp concentration peak, since the model predictions are affected by the measured as well as the calculated concentrations of previous time steps, and therefore have a delayed response to such changes. It should be noted, however, that in order to use such a model for regulatory purposes or as an alarm when indoor concentrations would exceed certain limits, it is important to adjust the model parameters so as to accurately predict high concentration peaks. This was a significant criterion for the choice of the model parameters. The performance of the model is very satisfactory in this respect, since the ratio between the measured and predicted daily maximum indoor

concentrations in the period shown in Fig. 5 ranges between and 0.88 and 1.23. The fast response of the predicted concentrations to the low indoor concentration values observed after a measured concentration peak is of secondary importance in this respect.

Certain inadequacies in the model predictions may be partly attributed to the choice of model parameters. The use of mean values for model parameters such as the infiltration flow rate (or air exchange rate) and the room mixing factor, reduces the expected variability of the model predictions. Furthermore, due to the lack of relevant data on indoor microenvironments in Greece, even the mean values of these parameters had to be estimated, or based on results from campaigns carried out elsewhere, where building design and climate may differ from that of Greece. A campaign designed to measure parameters such as the air exchange rate and the room mixing factor in typical indoor microenvironments in Greece, would be very useful in this respect.

Several studies have shown that the estimates of total population exposure taking into account indoor concentrations are very different from those which would have been made based only on outdoor concentration measurements (e.g. Hayes, 1989; Dimitroulopoulou et al., 2001). Since indoor concentration measurements are a cumbersome and time-consuming task, the use of indoor air quality models, such as the one presented in this study, constitutes a promising alternative to indoor concentration measurements. The evaluation provided in the present paper suggests that the model developed by Hayes gives a quite accurate estimation of indoor concentration peaks, taking also into consideration that it is a rather quick and easy to apply model. For a complete validation of the model, independent data measured for more extensive periods, in various climatic conditions and for various building types is needed.

Acknowledgements

The authors would like to thank the staff of the Directory of Air and Noise Pollution Control of the Hellenic Ministry for the Environment, Physical Planning and Public Works for providing the meteorological data from the fixed monitoring site of Patission Street. We would also like to acknowledge the help of Ms. Natalia Fili during the measurement campaign and the analysis of data.

References

Allen, R.J., Wadden, R.A., 1982. Analysis of indoor concentrations of carbon monoxide and ozone in an urban hospital. Environmental Research 27, 136–149.

Alm, S., Reponen, A., Mukala, K., Pasanen, P., Tuomisto, J., Jantunen, M.J., 1994. Personal exposures of preschool children to carbon monoxide: role of ambient air quality and gas stoves. Atmospheric Environment 28, 3577–3580.

Alm, S., Mukala, K., Jantunen, M.J., 2000. Personal carbon monoxide exposures of preschool children in Helsinki, Finland: levels and determinants. Atmospheric Environment 34 (2), 277–285.

Bouhambra, W.S., Elkilani, A.S., Abdul-Raheem, M.Y., 2000. Analysis of indoor concentrations of benzene using an air-quality model. Archives of Environmental Health 55 (3), 201–209.

Burnett, R.T., Dales, R.E., Brooks, J.R., Raizenne, M.E., Krewski, D., 1997. Association between ambient carbon monoxide levels and hospitalizations for congestive heart failure in the elderly in 10 Canadian cities. Epidimiology 8, 162–167.

Burnett, R.T., Cakmak, S., Raizenne, M.E., Stieb, D., Vincent, R., Krewski, D., Brooks, J.R., Philips, O., Ozkayanak, H., 1998. The association between ambient carbon monoxide levels and daily mortality in Toronto, Canada. Journal of the Air and Waste Management Association 48, 689–700.

Chaloulakou, A., Assimakopoulos, D., Lekkas, T., 1998. Forecasting daily maximum ozone concentrations in the Athens Basin. Environmental Monitoring and Assessment 56, 97–112.

Dimitroulopoulou, C., Ashmore, M.R., Byrne, M.A., Kinnersley, R.P., 2001. Modelling of indoor exposure to nitrogen dioxide in the UK. Atmospheric Environment 35, 269–279.

Direction of Air Pollution and Noise Control, 2000. Air pollution in Athens. Yearly Technical Report: Year 1999, Hellenic Ministry for the Environment, Physical Planning and Public Works, Athens.

Dockery, D.W., Spengler, J.D., 1981. Indoor–outdoor relationships of respirable sulfates and particles. Atmospheric Environment 15, 335.

Esmen, N.A., 1978. Characterization of contaminant concentrations in enclosed spaces. Environmental Science and Technology 12, 337.

Gilham, S., Deaves, D.M., Woodburn, P., 2000. Mitigation of dense gas releases within buildings: validation of CFD modelling. Journal of Hazardous Materials 71, 193–218.

Gold, D.R., Allen, G., Damokosh, A., Serrano, P., Hayes, C., Castillejos, M., 1996. Comparison of outdoor and classroom ozone exposures for school children in Mexico City. Journal of the Air and Waste Management Association 46, 335–342.

Hayes, S.R., 1989. Estimating the effect of being indoors on total personal exposure to outdoor air pollution. Journal of the Air Pollution Control Association 39 (11), 1453–1461.

Hayes, S.R., 1991. Use of an indoor air quality model (IAQM) to estimate indoor ozone levels. Journal of the Air and Waste Management Association 41 (2), 161–170.

Hoppe, P., Martinac, I., 1998. Indoor climate and air quality. International Journal of Biometeorology 42, 1–7.

Ishizu, Y., 1980. General equation for the estimation of indoor pollution. Environmental Science and Technology 14 (10), 1254.

Kassomenos, P., Kotroni, V., Kallos, G., 1995. Analysis of climatological and air quality observations from Greater

Athens Area. Atmospheric Environment 29 (24), 3671–3688.

Katsoulis, B.D., 1996. The relationship between synoptic, mesoscale and microscale meteorological parameters during poor air quality events in Athens, Greece. Science of the Total Environment 181, 13–24.

Leaderer, B.P., Cain, W.S., Isseroff, R., Berglund, L., 1984. Ventilation requirements in buildings. II. Particulate matter and carbon monoxide from cigarette smoking. Atmospheric Environment 18 (1), 99–106.

Marr, L.C., Morrison, G.C., Nazaroff, W.W., Harley, R.A., 1998. Reducing the risk of accidental death due to vehicle related carbon monoxide poisoning. Journal of the Air and Waste Management Association 48, 899.

Moschandreas, D.J., 1985. Characterization of indoor air pollution. Journal of Wind Engineering and Industrial Thermodynamics 21, 39–89949.

Moschandreas, D.J., Zabransky, J., Pelton, D.J., 1981. Comparison of indoor and outdoor air quality. Electrical Power Research Institute Report EA-1733, Research Project 1309, Palo Alto, CA, USA.

PERPA, 1989a. Air Pollution in the Athens Basin, Vol. 2: Pollution Sources. Technical Report, Hellenic Ministry for the Environment, Physical Planning and Public Works, Athens.

PERPA, 1989b. Air Pollution in the Athens Basin, Vol. 3: Primary Pollutants. Technical Report, Hellenic Ministry for the Environment, Physical Planning and Public Works, Athens.

Shair, F.H., Heitner, K.L., 1974. Theoretical model for relating indoor pollutant concentrations to those outside. Environmental Science and Technology 8, 444.

Touloumi, G., Pocock, S.J., Katsouyanni, K., Trichopoulos, D., 1994. Short term effects of air pollution on daily mortality in Athens: a time-series analysis. International Journal of Epidimiology 23, 957–967.

Tryfonopoulos, D.A., Bergeles, G.C., 1994. Temperature fields and air pollution build-up over the Athens Basin. Environmental Software 9 (4), 269–283.

Wadden, R.A., Scheff, P.A., 1983. Indoor Air Pollution: Characterization, Prediction, Control. Wiley, New York.