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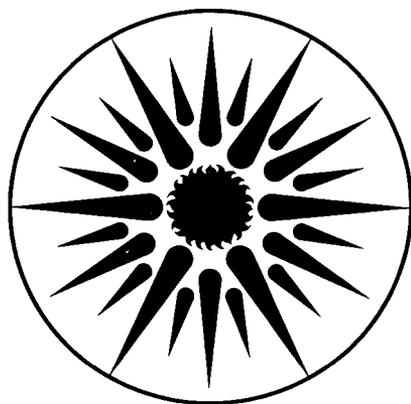
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Indoor Ozone Concentrations: Ventilation Rate Impacts and Mechanisms of Outdoor Concentration Attenuation

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AND MECHANISMS OF OUTDOOR CONCENTRATION ATTENUATION**

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ABSTRACT

The classification of outdoor (ambient) air as fresh for the purposes of ventilation is not always appropriate, particularly in urban areas. In many cities of the world, urban air frequently violates health-based air quality standards due to high ozone concentrations. The degree of protection from exposure to ozone offered by the indoor environment depends on the relationship between indoor and outdoor ozone levels. Existing concentration data indicates that indoor/outdoor ozone ratios range between 10 and 80%. This paper analyzes several of the key issues influencing indoor ozone concentrations, including: 1) the degree of penetration of outdoor ozone indoors, 2) removal within the indoor environment (removal at surfaces and within air distribution systems), and 3) the correlation in time between outdoor ozone levels and ventilation rates. A model for calculating the degree of ozone removal in typical building leaks and air distribution systems is described and applied to a range of typical cases. This model indicates that the degree of removal is minimal for most wooden building cracks, but could be significant in leaks in concrete or brick structures, and is strongly dependent on the lining material for air distribution systems. Indoor ozone exposure estimates based on hourly outdoor ozone monitoring data and hour-by-hour weather-based simulations of infiltration rates and building operation are reported for a few residential scenarios. These estimates serve as a basis for exploring the impact of energy-efficient ventilation strategies on indoor ozone exposures.

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NOMENCLATURE

Symbols

| | |
|---------------------|------------------------------------------------------------------------------------------------------------------|
| A | Cross sectional area (m^2) |
| $\sum A_j$ | Total indoor surface area (m^2) |
| C | pollutant concentration (moles m^{-3} , or parts per billion by volume) |
| d | parallel plate separation distance (m) |
| \mathcal{D} | pollutant molecular diffusion coefficient ($0.182 \times 10^{-4} \text{ m}^2 \text{ s}^{-1}$ for O_3) |
| D_h | hydraulic diameter [$4A/P$ ($2d$ for parallel plates)] (m) |
| f | friction factor (-) |
| f | reaction probability (-) |
| h | heat transfer coefficient ($\text{W m}^{-2} \text{ K}^{-1}$) |
| J | pollutant flux (moles $\text{m}^{-2} \text{ s}^{-1}$) |
| K_m | minor head loss coefficient (-) |
| k_g | thermal conductivity of fluid ($\text{W m}^{-1} \text{ K}^{-1}$) |
| L | length of parallel plates or ducts in the flow direction (m) |
| Nu | Nusselt number [hD_h/k_g] (-) |
| n | flow exponent in infiltration power law [$u_m \propto (\Delta p)^n$] (-) |
| Δp | Pressure difference (Pa) |
| P | leak or duct perimeter (m) |
| Pr | Prandtl number [ν/α] (-) |
| q | volumetric flow rate ($\text{m}^3 \text{ s}^{-1}$) |
| R | pollutant loss rate (moles s^{-1}) |
| Re | Reynolds number [$u_m D_h/\nu$] (-) |
| s | pollutant generation rate (moles s^{-1}) |
| Sc | Schmidt number [ν/\mathcal{D}] (-) |
| Sh | Sherwood number [$(J/C)D_h/\mathcal{D}$] (-) |
| u | fluid velocity in the flow direction (m s^{-1}) |
| V | volume (m^3) |
| v | fluid velocity (m s^{-1}) |
| $\langle v \rangle$ | Boltzmann velocity of pollutant (362.5 m s^{-1} for O_3 molecules at 298 K) |
| v_d | pollutant deposition velocity (m s^{-1}) |
| x | coordinate in the flow direction (m) |
| y | coordinate in the direction normal to the wall (m) |
| α | thermal diffusivity of fluid ($0.21 \times 10^{-4} \text{ m}^2 \text{ s}^{-1}$ for air at room temperature) |
| η | pollutant removal efficiency [$(C_{\text{entrance}} - C_{\text{exit}})/C_{\text{entrance}}$] (-) |
| μ | dynamic viscosity of fluid ($1.8 \times 10^{-5} \text{ kg m}^{-1} \text{ s}^{-1}$ for air at room temperature) |
| ν | kinematic viscosity of fluid ($0.15 \times 10^{-4} \text{ m}^2 \text{ s}^{-1}$ for air at room temperature) |
| ρ | density of fluid (1.2 kg m^{-3} for air at room temperature) |

Subscripts

| | |
|-----|---------------------|
| a | air handling system |
| app | apparent |
| c | combined |

| | |
|---|---------------------------|
| i | indoors |
| L | laminar |
| M | transport limited |
| m | mean value |
| o | outdoors |
| T | turbulent |
| S | surface-uptake limited |
| w | wall or fluid at the wall |
| x | local value |

INTRODUCTION

Although outdoor (ambient) air is generally considered fresh for the purposes of ventilation, this classification is not always appropriate, particularly in urban areas. Ozone (O_3) is frequently the pollutant responsible for the violation of health-based air quality standards in many cities around the world. For example, in 1988 ozone levels in more than 100 cities in the United States exceeded the federal standard (hourly averaged concentrations not to exceed 120 parts per billion by volume more than once a year, U.S. Environmental Protection Agency, 1989). The air basin in which Los Angeles is located exceeded the 120 ppb level on 196 days of 1989 (California Air Resources Board, 1991). In Mexico City, hourly-averaged ambient ozone concentrations typically exceed 110 ppb on more than 250 days of the year (Bravo *et al.*, 1991), and concentrations twice as high are common. Since ozone is a powerful oxidant, exposure to elevated levels in urban air presents a serious public health problem. Short-term health effects associated with exposure to concentrations in the range 80-200 ppb include irritation to the airways and decrease of respiratory function, while long-term exposure can result in accelerating the aging of the lung (Lippman, 1989).

The prevailing policy for dealing with severe smog episodes is to advise people to decrease their physical activity (to reduce respiration) and remain indoors, since indoor ozone concentrations are lower than those outside. However, indoor/outdoor concentration ratios vary over a wide range, and can be as high as 80% (Weschler *et al.*, 1989). Evidently, the degree of protection afforded by the indoor environment against exposure to ozone depends on the actual concentrations indoors. The goal of this paper is to examine some of the key issues surrounding the attenuation of ozone from outdoor air to indoor spaces, emphasizing the penetration of outdoor ozone into the living or working environment. Specific aspects of this problem addressed here include: 1) the degree to which ozone is removed in building leaks, 2) removal in ducts and unoccupied spaces through which air is flowing (e.g. the crawlspace in a house), and 3) the coincidence between the times of high outdoor ozone concentrations and high air entry rates into buildings.

BACKGROUND

Ozone in the indoor environment has been studied for about 20 years. Mueller *et al.* (1973) investigated the rate of decay of ozone in an office, a bedroom and several tests rooms, and showed that this pollutant decomposes rapidly after it enters living spaces, following first-order decay. Sabersky *et al.* (1973) studied ozone decay in homes and used a test chamber to evaluate the rate of ozone decomposition on several common materials. Further data were obtained by Allen *et al.* (1978) and Selway *et al.* (1980) for the decay of ozone in offices in their study of the emissions from photocopying machines. Thompson *et al.* (1973) conducted measurements of indoor ozone in several types of buildings and compared them to outdoor levels. Shair and Heitner (1974) developed a general single-chamber indoor air quality model that included the effects of ventilation, filtration, and heterogeneous surface losses, and applied it successfully to the prediction of indoor ozone levels in office rooms supplied by a mechanical ventilation system. Based on this model, Shair (1981) designed and operated an activated carbon filter to keep indoor levels below 80 ppb in an office/laboratory building in Pasadena, CA. Researchers at the California Institute of Technology (Shaver *et al.*, 1983; Druzik *et al.*, 1990, Cass *et al.*, 1991) studied indoor ozone concentrations in museums in relation to the deterioration of works of art. Nazaroff and Cass (1986) expanded the model of Shair and Heitner to an arbitrary number of chambers and included explicitly the effect of gas-phase chemical reactions, obtaining good agreement between their model results and measurements in an art gallery in Southern California. Weschler *et al.* (1989) conducted continuous measurements of indoor and outdoor ozone concentrations for a period of 5 months in an office complex in New Jersey, and compiled ozone indoor/outdoor data from the literature. Hayes (1991) used a single-chamber air quality model similar to that of Shair and Heitner to estimate peak concentration indoor/outdoor ratios in various home, office and vehicle microenvironments, and to analyze the sensitivity of these ratios to various parameters.

The mass balance of ozone in a building with a simple mechanical ventilation system can be represented by a model such as the one shown in Fig. 1 (Druzik *et al.*, 1990). The indoor ozone concentration would be governed by the differential equation:

$$V \frac{dC_i}{dt} = (1-\eta_{oa})q_{oa}C_o + q_{oi}C_o - \eta_{ia}q_{ia}C_i - q_{io}C_i + s - R \quad (1)$$

Inherent in this model are the following assumptions: 1) Indoor air can be considered well mixed, 2) ozone is unaffected by its passing through the building envelope into the building, 3) the only components in the

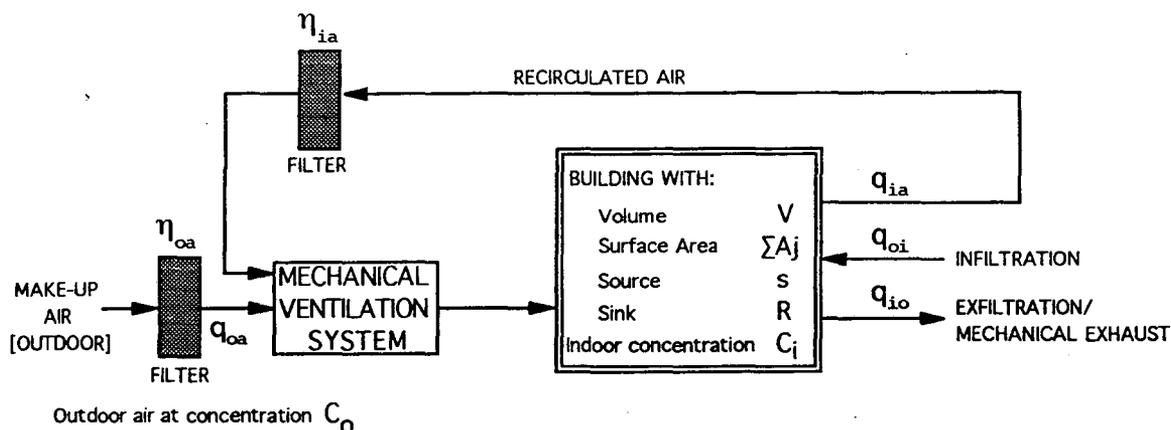


Fig. 1: Representation of a building for modeling indoor ozone concentrations

ventilation system where O_3 may be destroyed are the filters, 4) buildings can be modeled as single chambers for the purpose of predicting indoor ozone concentrations, and 5) air has constant density throughout the system (no contraction or expansion). Although not explicitly stated in the model, it is frequently assumed that the only indoor sink is the heterogeneous reaction of ozone with surfaces (modelled in terms of an average deposition velocity as $R = v_d(\sum A_j)C_i$), and that indoor ozone sources are nonexistent, except for the case when there are photocopying machines or electrostatic precipitators in operation. This paper focuses on the validity of assumptions 2, 3 and 4. Nazaroff and Cass (1986) found that the treatment of a building as a single chamber is adequate, for the case of a building having high recirculation rates through the mechanical ventilation system, but the question of whether the modeling of a residence as a single chamber is adequate has not been addressed.

ANALYSIS

The penetration of outdoor ozone indoors depends on the product of three factors: 1) outdoor concentration levels, 2) the rate of infiltration of outdoor air and/or the rate of supply of make-up air by the ventilation system, and 3) the efficiency of removal of ozone in each path through which air flows. Until now researchers have assumed that ozone penetrates without loss along all air flow paths, except through activated carbon filters, which are very efficient in removing ozone (Shair, 1981); however, the question of removal in building leaks or ventilation system ducts merits closer examination. Another important question to be addressed is the relationship among weather, outdoor ozone concentrations, air flows and ventilation rates. Consideration is also given to the decay of ozone in

unconditioned spaces that may serve as a significant source of the air infiltrating into the living areas. For example, if air infiltrating into a house passes through the basement or crawlspace first, then the ozone concentration of infiltration air would be that prevailing in the basement/crawlspace instead of the outdoor value.

Model for the removal of ozone in building leaks

We first address the issue of ozone entry through building leaks, since it is possible that scrubbing of ozone in the building envelope could at least partially account for some of the observed reduction of ozone concentration from outdoor to indoors, especially for cases in which the main mechanism for outdoor air entry is infiltration through cracks. We present results of a theoretical analysis of the potential for ozone scrubbing in typical residential building leaks based on the following simplifications: 1) a building leak can be modelled as a uniform-width gap between parallel plates, 2) flow in the leaks is laminar, but may not be fully developed, 3) the air velocity profile at the leak entrance is uniform, 4) air can be considered to have constant physical properties (ρ , μ) during passage, and 5) pollutant diffusion in the flow direction can be neglected.

The first step in solving this problem is the prediction of the flow rate in a leak based on its geometry and the pressure difference between the living space and outdoors. The solution presented here expands on the work of Baker *et al.* (1987) and Sherman (1990), by explicitly taking into account the effects of developing flow. The total pressure drop resulting from laminar flow between parallel plates is given by:

$$\Delta p = f_L \frac{L}{d} \frac{\rho u_m^2}{2} + (\sum K_m) \frac{\rho u_m^2}{2} \quad (2)$$

The first term accounts for losses due to shear stress at the walls and, in the case of developing flow, for the change in momentum required to accelerate the fluid from a flat velocity profile at the entrance to its exit profile. The second term accounts for pressure drops associated with flow contraction and expansion at the entrance and exit of the leak. The coefficients for minor losses (K_m) at sharp entrances and exits are 1.0 and 0.5, respectively. For fully developed laminar flow between parallel plates, the friction factor in Eq. (2) is the Fanning friction factor (defined as the ratio of wall shear stress to the flow kinetic energy per unit volume) and has the value $f_L = 24/Re$. In the case of developing flow, an apparent friction factor (f_{app}) may be defined so that Eq.(2) still holds. Shah (1978, p.168) gives the following formula for obtaining the value of f_{app} :

$$f_{\text{app}} \text{Re} = \frac{3.44}{(x^+)^{0.5}} + \frac{24 + 0.1685(x^+)^{-1} - 3.44(x^+)^{-0.5}}{1 + 0.000029(x^+)^{-2}} \quad (3)$$

$$\text{where } x^+ = \frac{\nu L}{4u_m d^2}$$

In order to use Shah's formula for computing the mean air flow velocity in the leak, Eq. (2) may be rewritten as:

$$\Delta p = \left(\frac{f_{\text{app}} \text{Re}}{2} \right) \frac{\mu L u_m}{d^2} + (\Sigma K_m) \frac{\rho u_m^2}{2} \quad (4)$$

Solving Eq. (4) for the mean velocity yields:

$$u_m = \left(\frac{f_{\text{app}} \text{Re}}{2} \right) \frac{\mu L}{(\Sigma K_m) \rho d^2} \left[\sqrt{1 + \frac{2\Delta p (\Sigma K_m) \rho d^4}{\left(\frac{f_{\text{app}} \text{Re}}{2} \right)^2 \mu^2 L^2}} - 1 \right] \quad (5)$$

The mean velocity can then be obtained as a function of the applied pressure and the geometry of the leak, by first assuming $(f_{\text{app}} \text{Re}) = 24$ and then iterating on Eqs. (5) and (3) until convergence is achieved. It should be noted that u_m depends on L and d in the ratio d^2/L , so that the leak geometry dependence can be captured by plotting the flow variables of interest against this ratio.

It is useful to relate the above results to observed flow exponents from leakage measurements in residences, for which leakage data are normally fit to a power-law relationship between flow and pressure. The relationship is depicted in Fig. 2, which was obtained by computing mean velocities for the range of pressure differentials observed between indoors and outdoors (0 to 10 Pa), and fitting the power-law model to the velocity-pressure data for different d^2/L ratios. The flow exponents measured in the field are typically between 0.55 and 0.75 (Sherman *et al.*, 1984), suggesting that leaks representative of the bulk of the leakage area in a residence should have d^2/L ratios in the range 10^{-4} to 10^{-3} m.

The second step in understanding ozone removal in building-envelope leaks is to examine pollutant transport from air to the leak walls. Under our model assumptions, and at steady state, the advection-diffusion equation describing ozone conservation in air assumes the form:

$$\mathbf{v} \cdot \nabla C = \mathcal{D} \frac{\partial^2 C}{\partial x^2} \quad (6)$$

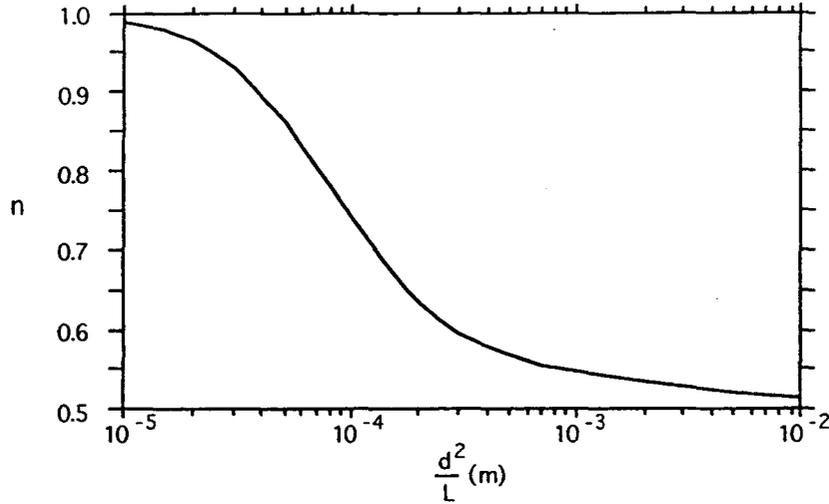


Fig. 2: Flow exponent as a function of parallel plate geometry

The solution to this equation under developing flow conditions is not trivial. However, the problem has been solved numerically for several sets of boundary conditions in the heat transfer literature (e.g. Hwang and Fan, 1964; Siegel and Sparrow, 1959). As discussed by Nazaroff and Cass (1989), the governing equations for the distribution of temperature and the distribution of concentration of dilute, reactive gaseous pollutants in a forced convection flow are of analogous form. In order to use the heat transfer results (usually given as Nusselt numbers), we need to relate the boundary conditions for ozone uptake at the wall to the analogous wall boundary conditions in heat transfer. Two possibilities are considered here for the outcome of the interaction of ozone with the leak walls: either ozone is perfectly removed at the walls, or only a fraction f of the collisions between ozone molecules and the walls result in removal. The first case is analogous to the heat transfer boundary condition of constant wall temperature (the concentration at the surface is constant and equal to 0). The second case corresponds to the boundary condition of finite resistance to heat transfer through the wall. At steady state, the flux of ozone towards the wall must be equal to the rate at which ozone is removed by a unit area of wall surface. This rate is given by the rate of collisions of ozone molecules with the wall per unit area, as predicted by the molecular theory of gases, multiplied by the reaction probability f :

$$-D \frac{\partial C_w}{\partial y} = f \frac{\langle v \rangle}{4} C_w \quad (7)$$

The boundary condition in Eq. (7) may be cast in dimensionless form to yield a wall Sherwood number analogous to the heat transfer wall Nusselt number:

$$Sh_w = f \frac{\langle v \rangle 2d}{4 \mathcal{D}} \quad (8)$$

At steady state, the equation describing pollutant concentration in a leak of arbitrary cross-section with uptake at the walls is:

$$\frac{dC_{m,x}}{dx} = -\frac{J_x P}{u_m A} \quad (9)$$

where $C_{m,x}$ is the flow-averaged pollutant concentration at a distance x from the leak entrance. Expressing Eq. (9) in terms of a local Sherwood number and the hydraulic diameter, we obtain after rearranging:

$$\frac{dC_{m,x}}{C_{m,x}} = -4 \frac{Sh_x \mathcal{D} dx}{u_m D_h^2} \quad (10)$$

Integrating Eq. (10) from the entrance ($x=0$) to the exit ($x=L$) of the leak yields:

$$\frac{C_{m,x=L}}{C_{m,x=0}} = \exp\left(-\frac{4\mathcal{D}}{u_m D_h^2} \int_{x=0}^L Sh_x dx\right) \quad (11)$$

so that the removal efficiency of the leaks can be expressed as a function of mean Sherwood number and a dimensionless leak length:

$$\eta = 1 - \exp(-4 Sh_m L^*) \quad (12)$$

$$Sh_m = \frac{1}{L} \int_{x=0}^L Sh_x dx \quad (13)$$

$$L^* = \frac{L/D_h}{ReSc} = \frac{\mathcal{D}L}{4u_m d^2} \quad (14)$$

Shah (1978, p. 190) gives an approximate formula obtained by Stephan (1959) for the values of Nu_m for the case of constant wall temperature. Substituting the Schmidt number for the Prandtl number, the formula becomes:

$$Sh_{m,M} = 7.55 + \frac{0.024(L^*)^{-1.14}}{1 + 0.0358(L^*)^{-0.64} Sc^{0.17}} \quad (15)$$

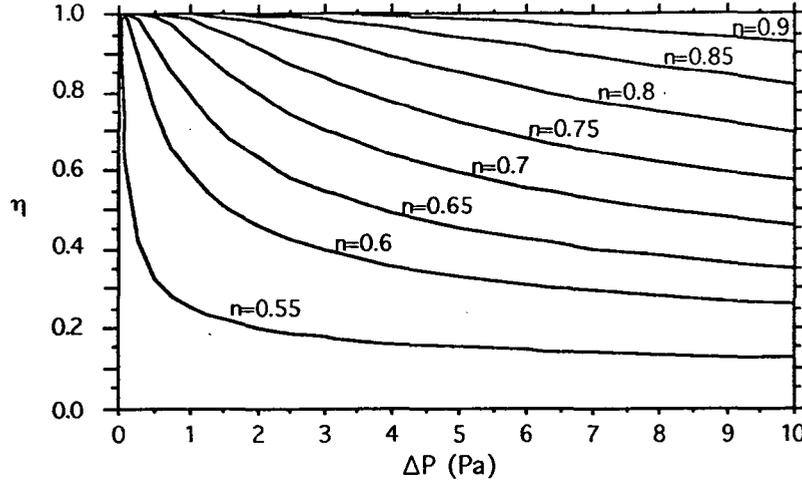


Fig. 3: Ozone removal in parallel-plate leaks for the case of perfect removal at the walls ($f = 1.0$), as a function of applied pressure and flow exponent.

Using appropriate values of L and d , one can obtain ozone removal efficiency values for parallel plate leaks as a function of the leak flow exponent and the pressure difference between the spaces connected by the leak. These values are shown in Fig. 3, and imply that if every collision of ozone molecules with the wall resulted in removal, scrubbing of ozone in typical building leaks ($n=0.66\pm 0.09$, Sherman *et al.*, 1984) would be significant. However, ozone has been shown not to be removed at the mass-transport limited rate in many settings (e.g. Cox and Penkett, 1972). For the case of finite wall resistance to ozone uptake, the solution to the penetration problem can be given in terms of a combined Sherwood number, analogous to the overall Nusselt number in heat transfer for the case of finite wall thermal resistance:

$$\eta = 1 - \exp(-4 \text{Sh}_{c,m} L^*) \quad (16)$$

$$\frac{1}{\text{Sh}_{c,m}} = \frac{1}{\text{Sh}_{m,S}} + \frac{1}{\text{Sh}_w} \quad (17)$$

Since no approximate formulas are available that allow an easy determination of $\text{Sh}_{m,S}$ (the mean fluid Sherwood number for limited surface-uptake) as a function of L^* , Sc and Sh_w , we will use the values of $\text{Sh}_{m,M}$ (the zero wall resistance Sherwood number) instead as a first approximation for the purpose of computing η . This approximation is based on the fact that the analogous Nu for finite wall heat transfer resistance approaches the constant wall temperature Nu asymptotically for small values of $1/\text{Nu}_w$; for large values of $1/\text{Sh}_w$ (f small), $\text{Sh}_{c,m}$ is approximately equal to Sh_w .

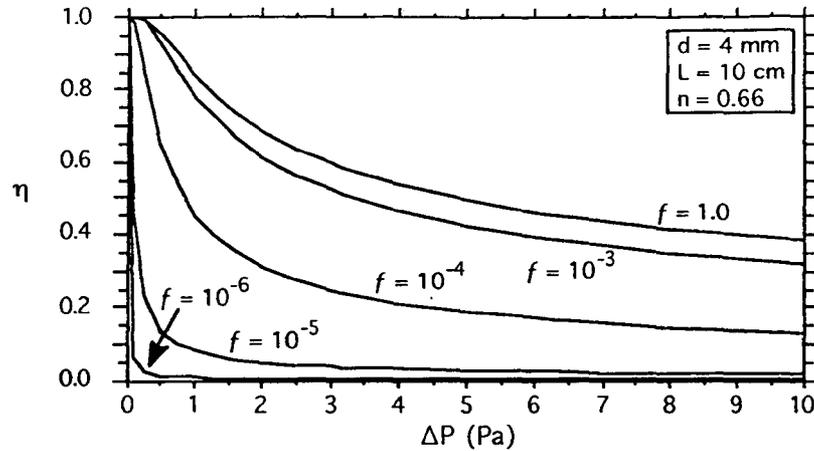


Fig. 4: Ozone removal efficiency of a typical parallel plate leak ($n = 0.66$) as a function of applied pressure and ozone-wall reaction probabilities.

Figures 4 and 5 are plots of ozone penetration values in two different parallel plate leaks as a function of pressure difference and reaction probability. Crack dimensions were chosen to yield flow exponents characteristic of a typical ($n=0.66$) and a very tight leak ($n=0.9$). These figures show that if the reaction probability of ozone with wall materials is below 10^{-6} , ozone removal would be minimal even in tight leaks. Table 1 lists f values for various materials calculated on the basis of experimental data for the decay of ozone in test chambers and the penetration of ozone in tubes (Cano-Ruiz *et al.*, 1992). Based on our model and these data, we conclude that ozone removal would be minimal in wooden cracks ($f < 10^{-6}$), and that there is a potential for significant scrubbing of outdoor ozone in leaks present in concrete or brick structures, for which f could be as large as 10^{-4} .

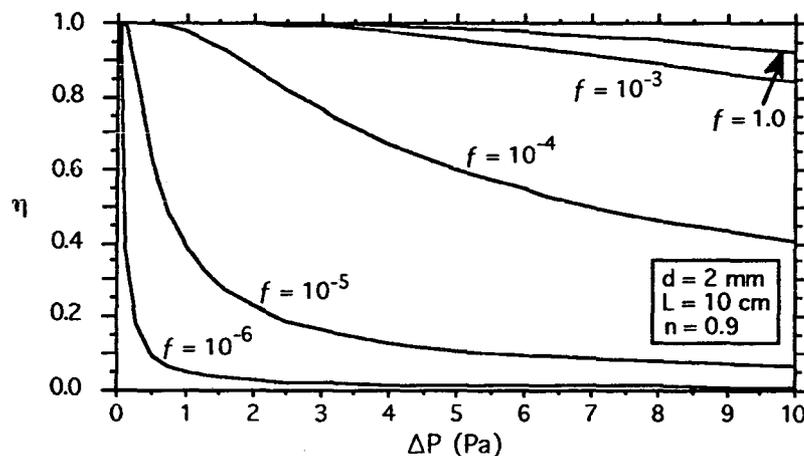


Fig. 5: Ozone penetration in a tight parallel plate leak ($n = 0.9$) as a function of applied pressure and ozone-wall reaction probabilities.

Table 1: Reaction probability of the collision of ozone molecules with various materials (calculated by Cano-Ruiz *et al.*, 1992)

| Material | f | Data used to obtain f |
|-----------------|----------------------------------------|---------------------------------|
| Teflon | $7 \cdot 10^{-9}$ | Altshuller <i>et al.</i> (1961) |
| " | $5 \cdot 10^{-7}$ | Simmons and Colbeck (1990) |
| Glass | 10^{-9} to $5 \cdot 10^{-7}$ | Altshuller <i>et al.</i> (1961) |
| " | $6 \cdot 10^{-8}$ to 10^{-7} | Sabersky <i>et al.</i> (1973) |
| " | $3 \cdot 10^{-6}$ to $5 \cdot 10^{-6}$ | Simmons and Colbeck (1990) |
| Aluminum | $6 \cdot 10^{-8}$ to $5 \cdot 10^{-7}$ | Altshuller <i>et al.</i> (1961) |
| " | $6 \cdot 10^{-8}$ to 10^{-7} | Sabersky <i>et al.</i> (1973) |
| Stainless steel | $5 \cdot 10^{-8}$ to $4 \cdot 10^{-6}$ | Altshuller <i>et al.</i> (1961) |
| Polyethylene | $2 \cdot 10^{-7}$ to 10^{-6} | Altshuller <i>et al.</i> (1961) |
| " | 10^{-6} to $3 \cdot 10^{-6}$ | Sabersky <i>et al.</i> (1973) |
| PVC | $6 \cdot 10^{-7}$ to $4 \cdot 10^{-6}$ | Altshuller <i>et al.</i> (1961) |
| Plywood | $6 \cdot 10^{-7}$ to $4 \cdot 10^{-6}$ | Sabersky <i>et al.</i> (1973) |
| Neoprene | $2 \cdot 10^{-6}$ to $9 \cdot 10^{-5}$ | Sabersky <i>et al.</i> (1973) |
| Concrete | $4 \cdot 10^{-5}$ to 10^{-4} | Simmons and Colbeck (1990) |
| Bricks | $2 \cdot 10^{-4}$ to $4 \cdot 10^{-4}$ | Simmons and Colbeck (1990) |

Extension of the model to analyze ozone removal in ducts

The model for pollutant removal in leaks may be extended to analyze the removal of ozone in ventilation system ducts. The main difference between the two cases is that flow in ventilation ducts is turbulent. A large number of semiempirical relationships for computing the Nusselt number can be found in the heat transfer literature, and have been compiled by Bhatti and Shah (1987). For the case of fully developed turbulent flow in smooth ducts they recommend using the formula:

$$\text{Nu}_m = \frac{(f_T/2)(\text{Re} - 1000)\text{Pr}}{1 + 12.7(f_T/2)^{0.5}(\text{Pr}^{0.67}-1)} \quad 2300 \leq \text{Re} \leq 5 \cdot 10^6; 0.5 \leq \text{Pr} \leq 2000 \quad (18)$$

$$\text{where } f_T = 0.00128 + 0.1143\text{Re}^{-0.311}$$

Substituting the ozone Schmidt number (0.82) for the Prandtl number in Eq. (18), the Nusselt numbers obtained can be used as Sh_m in Eq. (12) and $\text{Sh}_{m,S}$ in Eq. (17), since the Nusselt numbers for both of the analogous heat transfer boundary conditions are the same when the flow is turbulent. Ozone removal efficiency of ducts as a function of Reynolds number and duct geometry are shown in Fig. 6 for the case of perfect removal at the walls. This shows that in a typical residential ventilation system supply duct

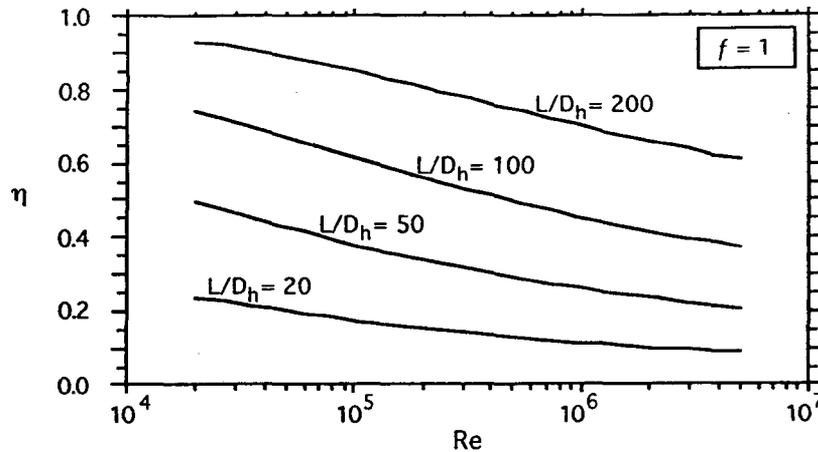


Fig. 6: Ozone removal in ducts with walls acting as perfect sinks, as a function of Reynolds number and duct geometry.

($D_h = 16$ cm, $L = 8$ m, $f_{xi} = 180$ m³/hr), about 50% of the ozone entering the duct would be removed if the walls acted as perfect sinks. However, actual ozoneremoval is dependent on the reaction probability of the lining material, as is illustrated by Fig. 7. For the duct parameters given in the example, ozone removal would decrease to about 30% for $f=6 \cdot 10^{-5}$, and would be negligible with a reaction probability of 10^{-7} . Common materials used in the manufacture and lining of ducts include sheet metal (galvanized steel), fiber glass, aluminum, maylar and neoprene, with f values from Table 1 ranging from $6 \cdot 10^{-8}$ to $9 \cdot 10^{-5}$ for some of these materials. These data and our analysis indicate that ozone scrubbing in distribution systems is strongly dependent on the lining material of the ducts, and that it could be significant, especially in the case of commercial buildings, where L/D_h ratios are larger than in residences.

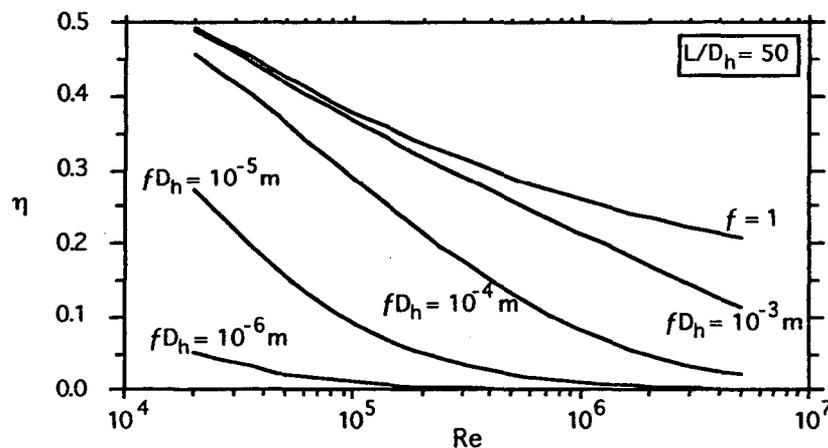


Fig. 7: Effect of ozone-wall reaction probabilities on ozone removal in ducts.

Simulation runs

In this section we give the results of a few indoor ozone simulation runs designed to explore the relationship among weather, air conditioning strategies (and the resulting ventilation rates), and indoor exposures to ozone. This effort was motivated by the interest in the impact of energy-conserving measures on indoor air quality. The pollutant transport module of the COMIS airflow network code (Feustel and Raynor-Hoosen, 1990) was modified and used to obtain hour-by-hour indoor ozone concentrations for a period of one year in each run. Inputs to this model included: 1) hourly ambient ozone data for 1989 monitored by the California Air Resources Board, and 2) hourly air flow rates based on heating/cooling loads and on weather data from the CTZ weather tapes, which contain one year of hour-by-hour data for each of the 16 climate zones in California, assembled by the California Energy Commission.

We ran simulations of indoor ozone concentrations in a one-story ranch house with an attic, a crawlspace and a garage. This prototype house has an area of 144 m², and is equipped with a central furnace/air conditioning unit (located in the garage), ten supply ducts and one return duct. Flows among the house, crawlspace, attic, garage, and outdoors were obtained by using the DOE-2 thermal simulation code (Birdsall *et al.*, 1990) in conjunction with the COMIS multizone infiltration model and a the duct-performance simulation model DUCTSIM. A flow chart of the use of these three codes and a full description of the house are presented elsewhere (Modera and Jansky, 1992). We present selected results from three simulation runs, corresponding to: 1) a house located in Riverside, CA with windows closed throughout the year, 2) the house in Riverside with windows open to provide natural ventilation cooling whenever the enthalpy of outdoor air is lower than that indoors, and 3) the same house located in San Jose, CA using natural ventilation cooling. The ozone deposition velocity in the house and the unconditioned zones (attic, garage, crawlspace) was assumed to be equal to 0.06 cm/sec in all the simulations (based on the observations of Sabersky *et al.*, 1973), and ozone scrubbing in leaks and ducts was ignored in order to isolate the effects of ventilation.

Figure 8 shows the results from these three runs expressed in terms of a monthly exposure index, defined as the time integral of ozone concentrations (units: ppb hr) throughout each month. We call this measure an exposure index because the true exposure depends on the respiration rate of the exposed individuals. For comparison, the monthly exposure indices based on 1989 outdoor concentrations are shown in Fig. 9 for both Riverside and San Jose. Times between 11 pm and 7 am were not included in the calculations, since O₃ concentrations at those times are very low and people

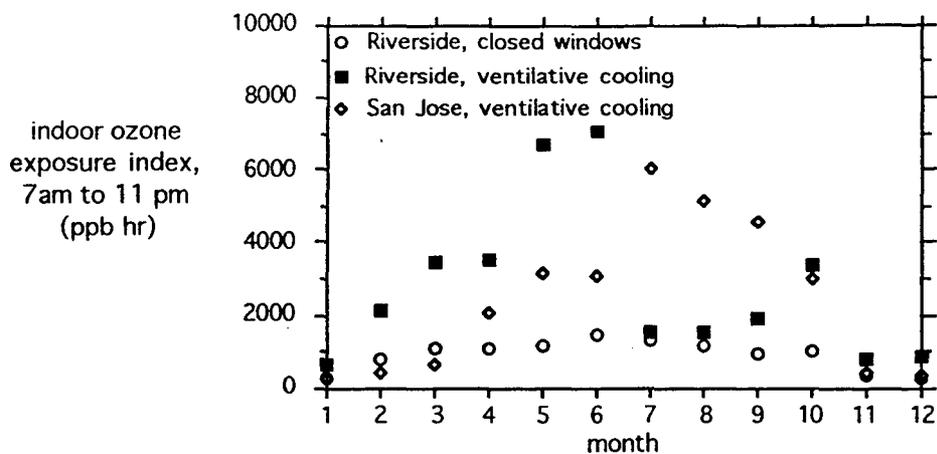


Fig. 8: Predicted monthly indoor ozone exposures for prototype residence in Riverside and San Jose.

are expected to remain indoors with the windows closed. Examination of both figures reveals that:

- Monthly indoor exposures can be almost 5 times higher if windows are opened than if air conditioning alone is used for cooling (compare both Riverside exposures for June),
- Even though the highest outdoor ozone concentrations in Riverside occur from June to September, indoor exposures in July, August and September are very low. This is because during these months outdoor air cannot be used to cool the house during the times of high outdoor ozone concentrations, and the windows are kept closed. As the weather in San Jose is cooler, window opening can be efficiently used there for cooling, and the monthly indoor ozone exposure index can be 2 to 4 times the Riverside value in July-

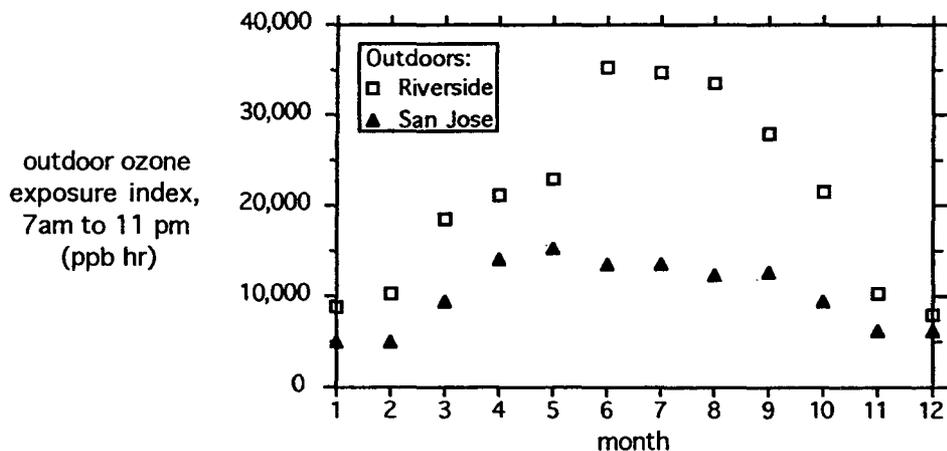


Fig. 9: Monthly outdoor exposure index from monitoring data (1989).

September, even though outdoor ozone concentrations in San Jose are typically about 40% those in Riverside during this period.

With respect to acute exposures, the federal standard of 120 ppb ozone was exceeded outdoors in 1989 on 325 hours in Riverside, while no violations were reported for San Jose. The maximum outdoor ozone concentrations in that year were 290 ppb and 120 ppb, respectively. The maximum indoor ozone concentration predicted for the closed house in Riverside was 35 ppb, well below the health-based standard, but levels higher than 120 ppb occurred in the Riverside house during 7 hours when ventilative cooling was used, with a maximum predicted indoor concentration of 180 ppb.

DISCUSSION

Ozone scrubbing in leaks in wood-frame structures appears not to be significant, but could be important in reducing ozone entry into masonry buildings during periods when windows are closed. However, these are the periods when indoor ozone concentrations are likely to be low indoors as a result of low air exchange rates and the rapid decay of ozone on indoor surfaces. The importance of reducing these low concentrations depends upon the health impacts of long-term exposure to ozone at low concentrations, which is an unresolved issue in toxicology. On the other hand, removal of ozone in duct systems could have a significant effect on exposures in buildings with continuous operation systems, or systems with longer residence times (e.g. in commercial buildings). The amount of removal will depend on the lining material: for example, neoprene-lined ducts would be expected to have a more significant effect than sheet-metal ducts. Also related to air distribution systems is their impact on room air motion, which in turn affects the rate of ozone decay. One study (e.g. Sabersky *et al.*, 1973) showed a factor of three increase in the rate of ozone removal in a house when the internal circulation system was operating.

There are very few measurements in the literature of the rate of decay of ozone in residences. Mueller *et al.* (1973) measured decay rates in a bedroom and Sabersky *et al.* (1973) obtained decay rates in one house, but to our knowledge no further data has been gathered since then. There is no information about the decay rates in buffer zones (e.g. attic, basement, crawlspace), or about the effects of furniture, rugs, drapes, etc. on ozone decomposition. We ran a few sensitivity simulations in order to examine the effect of the buffer zones on indoor ozone concentrations. Assuming that no decay took place in these zones resulted in the doubling of average indoor concentrations during periods in which windows were kept closed, as compared to the assumption used in our simulations, that ozone had a deposition velocity in these zones equal to that of the conditioned space of the

house. As in the case of removal in leaks, the impact of ozone decay in buffer zones with respect to overall indoor exposure is not likely to be high if ventilative cooling is used.

Our simulations indicate that the correlations in time among weather, cooling loads and outdoor ozone concentrations should be considered when designing alternative cooling strategies, so that energy conservation may be achieved without compromising indoor air quality.

It is important to note that indoor exposure to ozone in our simulations was dominated by the periods of high ventilation rates, that is, when the windows were open. This indicates that during smog episodes it is not enough to advise people to stay indoors, but to advise them to stay indoors *with the windows closed*. Using an indoor/outdoor concentration ratio is not enough to estimate indoor exposures to ozone in residences, since indoor/outdoor ratios and outdoor concentrations exhibit large variations in time. Knowledge about the window opening behavior of people during periods of likely high outdoor ozone concentrations is probably more useful, in terms of estimating total indoor exposure to ozone, than the determination of isolated indoor/outdoor ratios.

CONCLUSIONS

Based on our analysis we conclude that: 1) the use of ventilative cooling can have dramatic impacts on indoor ozone exposures, 2) temporal variations in venting and outdoor ozone concentrations must be taken into account when evaluating indoor exposures, 3) ozone scrubbing in building leaks and distribution systems depends principally on the reaction probability associated with the collision of ozone molecules with wall and lining materials, and 4) the removal of ozone in the unconditioned spaces of a house is significant on a relative basis during periods when ventilative cooling is not used.

The results of this theoretical analysis need to be confirmed by experiment. A critical variable in the prediction of ozone removal efficiencies in leaks and ducts is the reaction probability of ozone with the wall materials, but there are not enough data to allow reliable extraction of f values for most materials of interest. Another area where experiments can improve our understanding of indoor ozone concentrations is in the measurement of ozone decay rates in the different zones of a residence. As the venting behavior of the occupants of a residence is a critical determinant of indoor ozone exposures, the applicability of our assumptions in regard to window opening need to be examined more carefully. Finally, the challenge of modelling indoor ozone concentrations in residences and confirming hour-by-hour predictions with measurements still remains.

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