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Monitoring and control of particulate matter in indoor air: A review

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Abstract: The review discusses the sources and concentrations of particulate matter in different microenvironments, their effect on human health, monitoring and their possible control measures particularly in office and residential buildings. Effect of various ventilation strategies on particulate concentration in air has been discussed. The calculations suggest that if the outdoor particle concentration is not more than 5-10 times the indoor particle generation rate, ventilation could be a very effective tool to control the indoor particle concentration. For coarse particles the choice of ventilation strategy is not very crucial, but for fine particulate slow rate of ventilation is more effective.

Keywords: Particulate matter, Indoor air, Air speed, Ventilation, Fine particle

INTRODUCTION

It's true that all of us face a variety of risks to our health due to air pollution as there is a variety of pollutants present in lower atmospheric zone, which may cause mild to severe human health hazards. Major sources of air pollutants are mainly coal combustion, diesel engines, vehicular emissions, construction, industrial smokes etc. Most of the people spend nearly 80% of their time indoors and often the air within homes, offices, and other buildings may be more seriously polluted than the outdoor air. Hence polluted indoor air presents higher risk to human health and deserves attention. Volatile organic compounds and Radon (Bhandari and Gupta, 2009, 2010) are the potential indoor air pollutants arising from sources within the building; whereas other pollutants like combustion products (CO, SO_2, NO_2, CO_2 environment tobacco smoke, fine particles), ozone etc. arise from both indoor and outdoor sources. Small particles are among the pollutants that represent one of the biggest health hazards (CAI, 2004). Particulate matter (PM) is an air pollutant consisting of a mixture of particles that can be solid, liquid or both, are suspended in the air and represent a complex mixture of organic and inorganic substances. These particles vary in size, composition and origin. The particle size for PM can vary from 0.005 μm to 100 μm in diameter. All ambient PM are referred to as total suspended particulate matter (TSP). PM less than 10μm in diameter is referred to as PM_{10}, that less than 2.5 μm in diameter as $PM_{2.5}$ and so on. Particles less than 0.1 μm are called ultrafine particles (UFP). PM_{10} are inhalable and are referred to as respirable particulate matter (RPM). This work presents a discussion of various sources of

PM in indoor environment and its effects on human health along with a review of various monitoring studies. The effects of ventilation on deposition rates and airborne concentrations of various particle sizes are critically discussed.

PARTICULATE SOURCES

Outdoor sources: The outdoor sources of particles are coal combustion, diesel engines, vehicular emissions, windblown dust, agriculture, and construction and industrial activities such as mechanical grinding etc. PM in urban areas is mainly made up of metals, organic compounds, biological material, secondary PM (formed due to reactions of various atmospheric gases) and pure or elemental carbon. TSP may also contain fungal spores and pollen which cause allergic diseases. A major fraction of diesel particles are PM_1 on which many organic compounds like carcinogenic polycyclic and nitro-polycyclic hydrocarbons are adsorbed (Kittelson, 1998). Toxic elements like Arsenic contribute appreciably to the airborne particulate matter (Deb *et al.,* 2002). Particulates, after being emitted into the atmosphere from various sources, remain suspended in the urban atmosphere for an appreciable time. Subsequently they are deposited on various near ground surfaces and on street sediments. These PM enter the indoors through various passages and pollute the indoor air too.

Indoor sources: Indoor particle sources such as tobacco smoke, cleaning, cooking etc. have an appreciable effect on indoor particle concentrations and personal exposure. Cooking and heating with solid fuels such as dung, wood, agricultural residues or coal is probably the largest source of indoor air pollution globally and nearly half the world

continues to cook with solid fuels (WHO, 2002). Different types of office equipment (including fax machines, laser printers, ink-jet printers, scanners, and photocopying machines) have been found to emit RPM along with volatile organic compounds and ozone (Lee *et al.*, 2001). It is currently a topic of high concern to monitor the release of UFP from hardcopy devices such as laser printers into the indoor environment. The general emission behavior of a printer can be examined by conducting emission test chamber measurements with particle-counting devices. Chamber experiments with modified laser printers operated without toner or paper also revealed UFP emissions. On the basis of these results Wensing *et al.* (2008) assumed the high-temperature fuser unit, instead of the toner, to be a source of UFP emission. An increase in the concentration of ozone and UFP numbers in the indoor air of a room during printing processes of a laser printer/ink-jet printer have also been observed by Kagi *et al.* (2007).

HEALTH EFFECTS

Exposure to PM is hazardous to health (Dockery *et al.*, 1993). According to the World Health Organization, indoor air pollution from solid fuels ranks amongst the ten highest risks to human health in Africa and Asia (WHO, 2002). The size and the shape of the particles as well as their physico-chemical properties determine the depth of inhalation, the extent of inhalation, and the deposition rate in the airways (Yeh *et al.*, 1976). The smaller the particle, the greater is the fraction of particles deposited in airways and lungs, and the greater is the surface area available for interaction with biological systems. Particulates larger than 10 μm are generally filtered in the nose and throat and do not cause problems, but particulate matter smaller than about 10 μm can settle in the bronchi and lungs and cause health problems. Time-series epidemiological studies have shown statistical relations between health outcomes and outdoor PM concentration with mass median diameter of <10 μm in urban areas (USEPA, 1996; Vedal, 1997 and Samet *et al.*, 2000). A statistical relation was found to exist between PM_{10} and hospital admissions for chronic lung diseases even in those areas which have relatively low ambient PM concentrations by Schwartz (1994) and Moolgavkar *et al.* (1997). More recent studies, however, indicate that smaller particles such as $PM_{2.5}$ may be more closely linked with health effects (Schwartz *et al.*, 1996; Borja-Aburto *et al.*, 1998). After adjustment for $PM_{2.5}$, Peng *et al.* (2008) found no statistically significant associations between coarse particulates ($PM_{10-2.5}$) and hospital admissions for cardiovascular and respiratory diseases. On the other hand $PM_{2.5}$ leads to high plaque deposits in arteries, causing vascular inflammation and atherosclerosis-a hardening of the arteries that reduces elasticity, which

can lead to heart attacks and other cardiovascular problems (Pope *et al.*, 2002). UFP ($PM_{0.1}$) are still more harmful as these may pass through the lungs to affect other organs. Also fine particles are carriers of toxic air pollutants including heavy metals and organic compounds which can lead to mutations and cause cancer (Mage *et al.*, 1996). Indoor air pollution (IAP) from biomass fuels contains high concentrations of health damaging pollutants and is associated with an increased risk of childhood pneumonia.

COMPOSITION OF INDOOR FINE PARTICULATES

The main components of $PM_{2.5}$ are organic matter (30-60%), nitrates and sulphates (25-30%), elemental carbon (5%), and metals (1%) (USEPA, 1995). Geller *et al.* (2002) studied the chemical composition of fine and coarse particles in indoor and outdoor air. They observed that organic carbon was the most significant component of $PM_{2.5}$ mass concentrations both outdoors (41±14%) and indoors (61±17%). The average indoor-to outdoor $PM_{2.5}$ organic carbon concentration ratio was higher than one (1.77±0.36) which is due to contribution of indoor sources like cooking, waxes, cleaners, polishes, plasticizers and pesticides. As for elemental carbon indoor-to outdoor concentrations for fine PM was 0.84 (±0.32). The main source of elemental carbon is vehicular emissions outdoors which contributes substantially to indoor concentrations (Jones *et al.*, 2000). Elemental carbon exists mostly in the ultrafine range, between 0.05-0.2 μm and this size range has the highest outdoor-to-indoor penetration, varying generally from 75 to 100% (Seinfeld and Pandis, 1998).

Abt *et al.* (2000) studied the effect of indoor particle sources on indoor particle size distributions and concentrations. They observed that cooking, cleaning, and indoor work (characterized by movement of people etc.) significantly increased $PM_{0.7-10}$ concentrations by 0.27, 0.27, and 0.25 $μm^3cm^{-3}min^{-1}$, respectively. Cooking was the only variable significantly associated with generation of particles less than 0.5 μm in diameter. Outdoor particles of size<10 μm were found to contribute significantly to indoor particle levels. Effective penetration efficiencies ranged from 0.38 to 0.94 for $PM_{0.02-0.5}$ and from 0.12 to 0.53 for $PM_{0.7-10}$. Concentrations of various particle sizes for $PM_{0.7-10}$ particles decreased with increasing particle size, reflecting the influence of deposition losses from gravitational settling.

MONITORING

Monitoring and control of indoor PM concentrations is important due to the fact that personal exposure monitoring studies have found only weak statistical associations between outdoor PM concentrations and

Table 1. Respirable particulate matter concentration in various countries (World Bank, 2007).

Country	Particulate matter ($\mu g/m^3$) 2004	Country	Particulate matter ($\mu g/m^3$) 2004
Argentina	58	Ireland	19
Australia	12 - 20	Italy	29 - 44
Austria	41	Japan	31 - 40
Belgium	28	Kenya	43
Brazil	35 - 40	Korea, Rep	41 - 50
Bulgaria	61	Malaysia	29
Canada	13 - 22	Mexico	51
Chile	61	Netherlands	34
China	50 - 125	New Zealand	14
Colombia	31	Norway	14
Croatia	33	Philippines	39
Cuba	21	Poland	39 - 43
Czech Republic	23	Portugal	23
Denmark	21	Romania	18
Ecuador	30	Russian Federation	21 - 22
Ecuador	23	Singapore	44
Egypt, Arab Rep.	169	Slovak Republic	15
Finland	21	South Africa	16 - 33
France	11	Spain	30 - 35
Germany	19 - 22	Sweden	11
Ghana	33	Switzerland	23
Greece	43	Thailand	79
Hungary	19	Turkey	46 - 55
Iceland	18	Ukraine	35
India	37 - 150	United Kingdom	15 - 25
Indonesia	104	United States	21 - 34
Iran, Islamic Rep.	58	Venezuela, RB	10

personal exposure to PM.

Geller *et al.* (2002) measured fine (0-2.5 μm) and coarse (2.5-10 μm) RPM concentrations concurrently indoors and outdoors during the seasons when maximum outdoor PM penetration in indoor environments was expected (due to minimized use of heating/cooling systems). They found that coarse indoor concentrations were markedly lower than those outdoors with the average indoor-to-outdoor mass concentration ratio equal to 0.66 (±0.27). They also found that indoor $PM_{2.5}$ levels were very similar to those measured outdoors, the average indoor-to-outdoor $PM_{2.5}$ mass concentration ratio being equal to 1.03 (±0.29). In Geller's measurements outdoor concentrations could explain only about 37% of the variation of the indoor concentrations, thereby suggesting that there may be significant contributions by indoor sources to the overall indoor $PM_{2.5}$ concentrations. Fine RPM concentrations contributed, on average, 74.3(±11.0) % of the total PM_{10} concentrations indoors and 61.3 (±13.1) % of the total PM_{10} concentrations outdoors. Their studies revealed that indoor-to-outdoor concentrations of trace elements and metals in fine PM were well correlated, whereas this correlation was poor in the case of coarse PM.

Lai *et al.* (2006) performed a large population based study of indoor air concentrations of $PM_{2.5}$, black smoke and NO_2 in various European cities. Their study revealed that the characteristics of different populations, such as cultural practices and preferences and living styles, can affect the magnitude of various indoor determinants, eventually influencing the indoor exposure levels. On the other hand smoking, gas-stove usage, outdoor temperature and wind speed were found to be the common determinants of indoor fine particle concentrations.

Adgate *et al.* (2003) attempted to document outdoor (O), indoor (I) and personal (P) $PM_{2.5}$ levels in a population of healthy non-smoking adults over multiple days and seasons for three different communities. They found I concentrations in all the localities to be higher than O concentrations. The P concentrations were even higher than I concentrations particularly in subjects with active lifestyles, which may be attributed to occupational and environmental tobacco smoke exposures outside the residences. Communities with higher household ventilation/day (through open doors and/or windows) had higher I and P concentrations of $PM_{2.5}$ in all seasons. Season variability shows the lowest P and I concentrations during summer, which seems to be due to more hours of air conditioning/day and hence lower household ventilation/day during summer as compared

Table 2. Ten most polluted cities by RPM (World Bank, 2007).

S. No.	Particulate ($\mu g/m^3$) 2004	City	City population (thousands) 2005
1	169	Cairo, Egypt	11,128
2	150	Delhi, India	15,048
3	128	Kolkata, India	14,277
4	125	Tianjin, China	7,040
5	123	Chongqing, China	6,363
6	109	Kanpur, India	3,018
7	109	Lucknow, India	2,566
8	104	Jakarta, Indonesia	13,215
9	101	Shenyang, China	4,720
10	97	Zhengzhou, China	2,590

to spring and fall seasons.

Eight homes in the Bangkok Metropolitan Region (BMR) were investigated to study the relationship between indoor and outdoor concentration of airborne TSP and associated organochlorine pesticides (OCP). For TSP, on average, the higher levels were observed outdoors than indoors for urban area (69-95 $\mu g/m^3$ outdoors vs. 51-69 $\mu g/m^3$ indoors). For rural homes, the outdoor levels were however lower than the indoor levels suggesting the predominant influence of indoor sources activities. Overall, TSP levels in this study were lower than those observed in other studies in Thailand (Pentamwa and Oanh, 2008).

Measurements of carbon monoxide (CO) and $PM_{2.5}$ in 13 households using bio-mass fuel in Gambia were done. Average 48-h $PM_{2.5}$ concentration in the cooking area was 361±312 $\mu g/m^3$ and its exposure was an estimated 219 $\mu g/m^3$ for children and 275$\mu g/m^3$ for their mothers. The continuous $PM_{2.5}$ concentration curve had peaks in all households representing the morning, midday, and evening cooking periods, with the largest peak corresponding to midday (Dionisio *et al.,* 2008).

The effect of charcoal smoke exposure on risks of acute upper and lower respiratory infection (AURI and ALRI) among children under age 18 months in Santo Domingo, Dominican Republic (1991–1992) was investigated. It was observed that exposure to charcoal smoke increases the

risk of ALRI in young children, an effect that is probably mediated by RPM (Bautista *et al.,* 2009). After adjustment for other risk factors, exposed children had no significant increase in risk of AURI but were 1.56 times more likely to develop ALRI. RPM concentrations were higher in charcoal-using households (27.9 $\mu g/m^3$) than in non-charcoal ones (17.6 $\mu g/m^3$), and ALRI risk increased with increased RPM exposure. Studies by Jiang *et al.* (2008) indicate that rural kitchen PM_{10} levels are three times higher than those in urban kitchens during cooking. PM_{10} was 6.1 times higher during cooking periods than during non cooking periods for rural kitchens. Personal $PM_{2.5}$ levels for rural cooks were 2.8-3.6 times higher than for all other participant categories. $PM_{2.5}$ exposures were the highest during cooking periods for both urban and rural cooks. However, rural cooks had 5.4 times higher $PM_{2.5}$ levels during cooking than did urban cooks. Also rural cooks spent 2.5 times more hours per day cooking than did their urban counter parts. The combustion of incense, wood, cigarette, and candles are major sources of residential indoor particulate matter, especially in the 2.5 µm size range and below (Fang *et al.,* 2002, 2003; Brauer *et al.,* 2000). Mannix *et al.* (1996) reported that burning incense could generate PM greater than 45 mg/ g burned, as compared to 10 mg/g burned for the cigarettes. Liao *et al.* (2006) found that incense burning had size integrated source emission rates of 0.038±0.026 particles/second. For indoor particles ranging from 0.5 to 5 µm, 62–92% is from indoor sources, including cooking, incense burning, and other residential activities. Table 1 presents the compiled data of RPM concentration in various countries in the year 2004 (World Bank, 2007).

PM STUDIES IN INDIA

Table 2 gives a list of ten most polluted cities in the world by RPM according to 2007 World Development Indicators of World Bank (2007). Four Indian cities – Delhi, Kolkata, Kanpur, and Lucknow figure in this list with RPM concentrations of 150, 128, 109, and 109 $\mu g/m^3$, respectively. These values are much higher than both the national and international standards. The concurrent World Health Organization (WHO) air quality guidelines

Table 3. Experimental deposition loss rates, $_g$ (/hr) as a function of particle size and ventilation strategy (Bouilly *et al.,* 2005).

Ventilation strategy	Bottom- top		Top- bottom		Top- top	
Ventilation rate, $_v$ (/hr)	0.5 ach	1.0 ach	0.5 ach	1.0 ach	0.5 ach	1.0 ach
$PM_{0.3-3}$	0.53-0.86 (0.64)	0.19-0.78 (0.42)	0.49-1.05 (0.73)	0.41-1.17 (0.68)	0.20-0.65 (0.40)	0.21-0.72 (0.40)
PM_{3-10}	1.10-3.72 (1.96)	1.22-2.91 (1.85)	1.19-3.33 (1.98)	1.46-3.61 (2.20)	0.87-2.90 (1.59)	0.84-3.47 (1.80)
PM_{10-15}	6.46	5.58	5.98	8.12	6.34	6.66

Note: Values in parentheses are the averages for the respective particle size ranges.

Table 4. Variation of I/O ratio for PM3 for top-bottom ventilation strategy for different ventilation rates (C_i at zero hours is taken equal to $24g_i$).

t Hours	I/O Cocentration ratio for PM$_3$							
	$_v$ = 1.0ach, $_g$=0.68, and C_o =				$_v$ = 0.5ach, $_g$=0.73, and C_o =			
	g_i	2.5 g_i	5 g_i	25 g_i	g_i	2.5 g_i	5 g_i	25 g_i
0	24.00	9.60	4.80	0.96	24.00	9.60	4.80	0.96
1	13.61	5.88	3.30	1.24	12.63	5.27	2.81	0.85
2	8.35	3.99	2.54	1.38	7.15	3.18	1.85	0.79
3	5.68	3.04	2.16	1.46	4.51	2.17	1.39	0.76
4	4.33	2.56	1.96	1.49	3.24	1.68	1.17	0.75
5	3.64	2.31	1.87	1.51	2.63	1.45	1.06	0.75
6	3.30	2.19	1.82	1.52	2.33	1.34	1.01	0.74
7	3.12	2.12	1.79	1.52	2.19	1.28	0.98	0.74
8	3.03	2.09	1.78	1.53	2.12	1.26	0.97	0.74
9	2.99	2.08	1.77	1.53	2.09	1.24	0.96	0.74
10	2.96	2.07	1.77	1.53	2.07	1.24	0.96	0.74
11	2.95	2.06	1.77	1.53	2.06	1.24	0.96	0.74
12	2.95	2.06	1.77	1.53	2.06	1.23	0.96	0.74

for PM_{10} are annual mean concentrations of 20 µg/m³ (World Bank, 2007). For fine particulate matter ($PM_{2.5}$) mean annual levels the guidelines of the European Commission, the European Parliament, US EPA, and WHO are respectively 25 µg/m³, 20 µg/m³, 15 µg/m³ and 10 µg/m³ (Ballester *et al.*, 2008). CPCB notification of November, 2009 on the national Ambient Air Quality Standards set a guideline of 60 µg/m³ and 100 µg/m³ repectively for annual and 24-h average of PM_{10}, and 40 µg/m³ and 60 µg/m³ respectively for annual and 24-h average of $PM_{2.5}$ (CPCB, 2009). Yet there are only few air quality studies particularly PM_{10} and $PM_{2.5}$ characterization in India (TERI, 2001; Balachandran *et al.,* 2000; Venkataraman and Kulkarni 2000; Kumar *et al.,* 2001; Sharma *et al.,* 2003; Sharma and Maloo, 2005; Bhanarkar *et al.,* 2005, Ratan and Kumar 2005; Gupta and Kumar, 2006). Sharma and Maloo (2005) studied particle size distribution in terms of PM_{10} and $PM_{2.5}$ and their chemical composition in the city of Kanpur, India. They concluded that the overall quality of air in Kanpur was much inferior to other cities in India and abroad. The concentrations of PM_{10}, $PM_{2.5}$ and heavy metals were 5-10 times higher than the corresponding values in European cities.

Gupta and Kumar (2006) studied the trends of particulate matter in four metropolitan cities in India and found that PM in all the four cities was higher than the prescribed standards of CPCB, India as well as WHO guidelines. They found that overall PM_{10} level in the four mega cities of India was showing a decreasing trend whereas the TSP showed steady levels. They attributed the improved PM_{10} levels to the lighter emission standards for automobiles and closing down of many industries in these cities. A relative comparison of ambient air concentration of pollutants emitted from transport sector

during the years 1995-2000 (without CNG) and the year 2001 (with CNG) has suggested that the air was much cleaner after the introduction of CNG. The annual average concentration of TSP came down from 405 to 347µg/m³ but it was still beyond the permissible limits (Goyal and Sidharatha, 2003). The non decreasing trend of TSP as against decreasing PM_{10} trends was attributed to the large scale construction activities which contribute to particles larger than 10 µm. Ratan and Kumar (2005) monitored the air quality in 7 different locations in Delhi and found that the individual pollutant index for RPM is greater than TSP for industrial and commercial areas in Delhi. Delhi is the fourth most polluted city by TSP (Gadhok, Risks in Delhi) and second most polluted city by RPM in the world (Table 2). Constantly rising population levels, their haphazard distribution and growth, consequent rise in the levels of infrastructure and transportation activities needed to support them are the main reasons behind this pollution. Diesel electricity generating sets and vehicles, particularly auto rickshaws, are major sources which not only generate significant amount of air pollution, but also lead to highly uncomfortable levels of noise pollution. This polluted outdoor air penetrates inside buildings adding to the indoor air pollution.

Saksena and Uma (2008) monitored Indoor levels of TSP and RSP in an office building and just outside the building in New Delhi for over three and a half years. The mean values of indoor TSP and RSP concentrations were less than the outdoor concentrations but higher than the Indian ambient standards, and were much higher than those observed in similar micro-environments in developed countries. The variation in outdoor concentration of RSP could explain only 25% of the variation in indoor concentration. They concluded that

Table 5. Variation of I/O ratio for PM_{3-10} for top-bottom ventilation strategy for different ventilation rates (C_i at zero hours is taken equal to $24g_i$).

t Hours	I/O Cocentration Ratio for PM_{3-10}								
	$_v$ = 1.0ach, $_g$=2.20, and C_o =				$_v$ = 0.5ach, $_g$=1.98, and C_o =				
	g_i	2.5 g_i	5g_i	25 g_i	0.8g_i	g_i	2.5g_i	5g_i	25 g_i
0	24.00	9.60	4.80	0.96	30.00	24.00	9.60	4.80	0.96
1	3.47	1.63	1.02	0.53	4.90	3.97	1.72	0.97	0.37
2	1.19	0.75	0.60	0.48	1.44	1.20	0.63	0.44	0.29
3	0.94	0.65	0.55	0.47	0.96	0.82	0.48	0.37	0.27
4	0.91	0.64	0.55	0.47	0.89	0.77	0.46	0.36	0.27
5	0.91	0.64	0.55	0.47	0.89	0.76	0.46	0.35	0.27
6	0.91	0.64	0.55	0.47	0.88	0.76	0.45	0.35	0.27
7	0.91	0.64	0.55	0.47	0.88	0.76	0.45	0.35	0.27
8	0.91	0.64	0.55	0.47	0.88	0.76	0.45	0.35	0.27
9	0.91	0.64	0.55	0.47	0.88	0.76	0.45	0.35	0.27
10	0.91	0.64	0.55	0.47	0.88	0.76	0.45	0.35	0.27
11	0.91	0.64	0.55	0.47	0.88	0.76	0.45	0.35	0.27
12	0.91	0.64	0.55	0.47	0.88	0.76	0.45	0.35	0.27

indoor- outdoor relationships cannot be generalized, but are strongly dependent on factors such as type of building, ventilation, climate etc.

Concentration of fine particulate matter in other Indian cities is also well above recommended WHO levels. While industrial and vehicular emissions are the major contributors to PM concentrations in outdoor air, the domestic fuel combustion does so to the indoor air particularly in the residential areas. And it is the latter which contributes more to human exposure to PM as the time spent indoors is much higher than outdoors. The risk of PM exposure is even higher for women as their role in the family is of primary cook. P.R. Andresen *et al.* (2005) studied personal and indoor $PM_{2.5}$ exposures for women using different kinds of cooking fuel in Mysore. They found that both personal and indoor $PM_{2.5}$ exposures were much lower for LPG users [71±15 µg/m³ and 71 ± 9 µg/m³, respectively] than for kerosene users [111±13 g/m³ and 98 ± 9 µg/m³, respectively in summers, and 177 ± 21 µg/m³ and 155 ±13 µg/m³, respectively in winters]. The values for LPG users did not have much impact of seasons - these values are much lower than the kerosene users but are still higher than those found in the US (Quackenboss *et al.,* 1989; Ozkaynak *et al.,* 1990; Kim and Stock, 1986; Clayton *et al.,* 1993; Rojas-Bracho *et al.,* 2000; Jantunen *et al.,* 2002; Adgate *et al.,* 2003). This may be attributed to different cooking styles or more open kind of residences in India which provide an easy access to outdoor particulates to enter the houses.

In Mumbai a pilot survey of the indoor and the monitoring site (outdoor) air quality was carried out indicated good correlation between RSP at the monitoring site & indoors during winters and summers but only a weak correlation during Monsoons (Deshpande *et al.,* 2002). The emission of various pollutants from industrial sources in Greater Mumbai was studied by Bhanarkar *et al.* (2005). Their estimates suggested that an erroneous 9.794 Gg/year of PM was emitted into the Greater Mumbai atmosphere, out of which 19% was contributed by thermal power plants. This suggests that the industries should be more isolated from the residential and commercial sites than required by the present policy. Also the industrial workforce should be required to wear nose masks in order to protect them from RSP. The above discussion suggests a need to address the issue of fine PM, particularly $PM_{2.5}$, monitoring and its effective control in the indoor air.

Studies done in Andhra Pradesh by Balakrishnan *et al.* (2002) revealed that the concentrations of RPM ranged from 500-2000 µg/m³ during cooking in biomass-using households, and average 24-hr exposures ranged from 90±21 µg/m³ for non-cooks to 231±109 µg/m³ for cooks. In households using clean fuels the 24-hr exposures were around 82±39 µg/m³ (with similar exposures across household subgroups). Daily average concentrations of respirable particulates in 412 rural homes from three districts of Andhra Pradesh were recorded and the mean 24-h average concentrations ranged from 73-732 µg/m³ in gas versus solid fuel-using households, respectively. Concentrations were significantly correlated with fuel type, kitchen type, and fuel quantity. The mean 24-h average exposures ranged from 80-573 µg/m³. Among solid fuel users, the mean 24-h average exposures were the highest for women and were significantly different for men and children. Among women, exposures were the highest in the age group of 15-40 years (most likely to be involved in cooking or helping in cooking), while among men, exposures were highest in the age group of 65-80 years (most likely to be indoors).
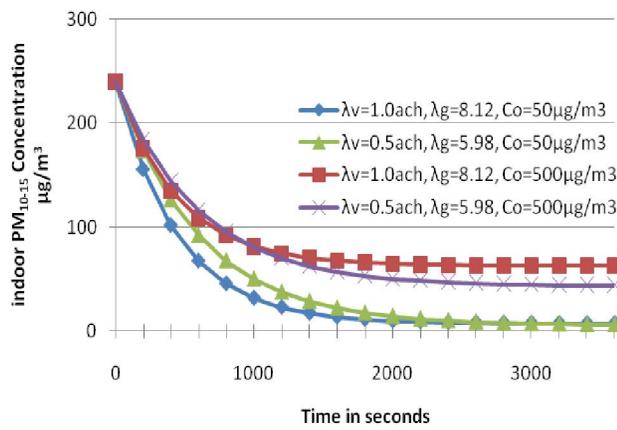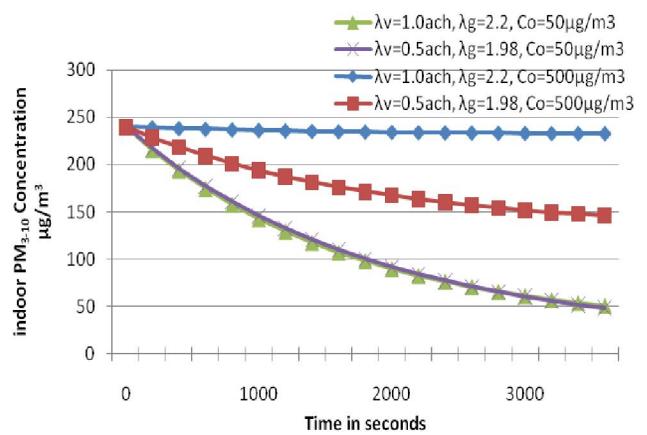
Fig. 1. *Variation of PM$_{10-15}$ concentration with time with top-bottom vantilation parameters : g$_i$10 µg/m³h, c$_i$ = 240 µg/m³.*

In the study by Balakrishnan *et al.* (2003, 2004) two parameters – type of fuel used and ventilation – emerged as the key determinants of exposure. Nearly 90% of the households sampled by Balakrishnan *et al.* (2002) in south India used solid fuel. In these households average 24-hour exposures to RPM were the highest amongst women cooks 442±37 µg/m³ compared to all the other household members. Amongst non-cooks, older women (61-80 years) experienced the highest exposures (337±57 µg/m³), followed by children under five (262±55 µg/m³). This is presumably because older women remain indoors for larger periods of time. Exposures of female and male children were similar. Men of 16-60 years experienced the least exposures owing to greater likelihood of working outdoors (148±5 µg/m³). In households using solid fuels, kitchen configuration played an important role in affecting exposures of all household members, including children under five. Average living area RPM concentrations were the highest in households having indoor kitchens without partitions (280±17 µg/m³) followed by households with indoor kitchens with partitions (264±17 µg/m³). Enclosed outdoor kitchens or simply outdoor cooking resulted in lower levels of indoor exposure (178±11 µg/m³ and 175±10 µg/m³, respectively) but still exceeded national health guidelines for outdoor air pollution (24-hour Indian standard for PM$_{10}$ is 100 µg/m³; CPCB, 2009). Thus, dispersion considerably affected indoor levels even during outdoor cooking. In households using LPG, children were exposed to three to four times lesser RPM levels (76±6 µg/m³) compared to solid fuel using households and these levels were similar to all the other population subgroups.

INTERVENTION AND CONTROL

In general, the smaller and lighter a particle is, the longer it will stay in the air. Larger particles (> 10 µm in diameter) tend to settle to the ground by gravity in a matter of hours whereas the smallest particles (< 1 µm) can stay in the atmosphere for weeks and are mostly removed by
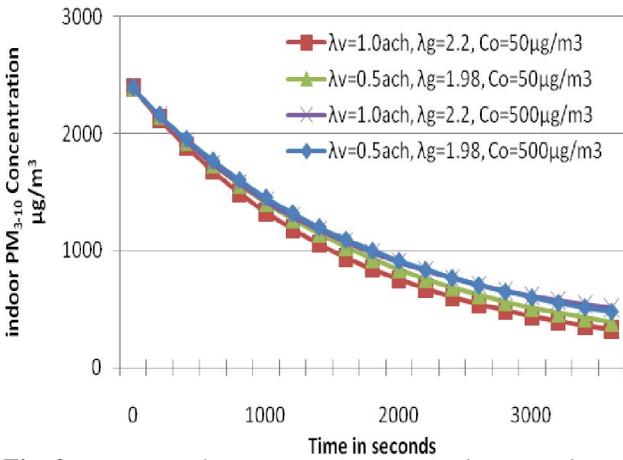
Fig. 2. *Variation of PM$_{3-10}$ concentration with time with top-bottom vantilation parameters : g$_i$10 µg/m³h, c$_i$ = 240 µg/m³*

precipitation. Diesel particulate matter is highest near the source of emission. Potential interventions for reducing human exposure to indoor air pollution are controlling the source (emissions), the local environment (concentrations) and the user (exposure).

Based on the evaluation it appears that for residential areas the most effective interventions and most beneficial to the user and society as a whole would be a shift from wood or charcoal to kerosene, LPG, biogas or grid electricity. Other more progressive alternatives such as ethanol (gel) fuel, or possibly biomass gasification, could not be effectively evaluated, but should be considered in greater detail in the future. Another intervention that is economic and appears to offer promising benefits is the use of a cooking window which is a form of hood built into a window (an alternate to chimney). However, this needs further investigation and evaluation for its applicability to a range of different housing types (Tremeer *et al.*, 2000).

Indoor particle sources such as tobacco smoke and cooking vapours can have a great effect on personal exposure (Abt *et al.*, 2000). Particle deposition on surfaces and adapted ventilation strategy can substantially reduce indoor particle concentrations (Bouilly *et al.*, 2005; Wallace, 1996; Lai, 2002). The indoor particle deposition rate is influenced by the properties of the deposition environment such as size and shape of the room, roughness of surfaces, airflow rates, inlet/outlet locations, surface-to-air temperature difference, furniture in the room, etc. (Nazaroff *et al.*, 1993; Abadie *et al.*, 2001; Jamriska *et al.*, 2000; Mundt, 2001; Nomura *et al.*, 1997; Fogh *et al.*, 1997; Zhao *et al.*, 2004; Thatcher *et al.*, 2002). But most of all it depends on the particle mass which directly depends on particle size. To evaluate the particle concentration evolution within a room, under isothermal conditions, the mass balance of the pollutant can be written as a function of the incoming polluted air, the particle deposition on the walls, the mass of pollutant

Fig. 3. *Variation of PM_{3-10} concentration with time with top-bottom vantilation parameters : g_i 100 µg/m³h, $c_i = 2400$ µg/m³*

leaving the zone, and particle generation and coagulation in the room. Bouilly *et al.* (2005) presented a model for ventilation effects on indoor particulate concentration ignoring the last of these factors, i.e., particle generation and coagulation in the room. They found that the ventilation acts differently on different particle size: the smaller the particles are, the more important the ventilation strategy is on particle deposition velocity (Bouilly *et al.*, 2005). They found that the influence of inlet/outlet locations is stronger for fine particles (<5 µm in diameter) than for coarse particles and that an increase of ventilation rate does not necessarily lead to higher deposition. Thus RPM could effectively be controlled by a careful choice of ventilation strategy.

VENTILATION MODELING

Taking particle generation rate due to indoor sources (g_i µg/m³h) also into consideration in the model of Bouilly *et al.* (2005), the time-dependent particle concentration can be written as

$$\frac{dC_i(t)}{dt} = \lambda_v C_o(t) - \lambda_v C_i(t) - \lambda_{de} C_i(t) + g_i \quad \text{... (1)}$$

Where t is the time in hours (h), $C_i(t)$ and $C_o(t)$ are, respectively, the indoor and outdoor particle concentrations (m⁻³) at time t, λ_v is the air change rate (h⁻¹), and $_{de}$ is the particle deposition loss rate coefficient (h⁻¹).

Thus $$\frac{dC_i(t)}{dt} = \left[\frac{dC_i(t)}{dt}\right]_+ - \left[\frac{dC_i(t)}{dt}\right]_- \quad \text{... (2)}$$
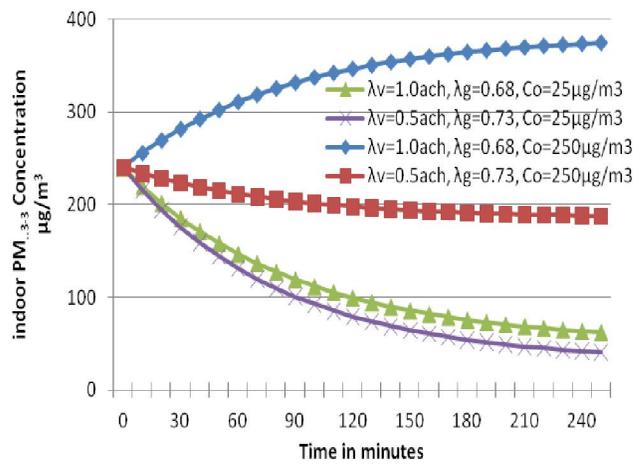
where $$\left[\frac{dC_i(t)}{dt}\right]_+ = \lambda_v C_o(t) + g_i \quad \text{... (3)}$$

is the gain in indoor particulate concentration per unit time,

and $$\left[\frac{dC_i(t)}{dt}\right]_- = \lambda_v C_i(t) + \lambda_{de} C_i(t) \quad \text{... (4)}$$

is the loss in indoor particulate concentration per unit time.

If we consider $C_o(t)$ to be constant with respect to time,

Fig. 4. *Variation of PM_3 concentration with time with top-bottom vantilation parameters : g_i 100 µg/m³h, $c_i = 240$ µg/m³*

then the gain rate/volume may be taken a constant, K_g (say), i.e.,

$$\left[\frac{dC_i(t)}{dt}\right]_+ = \lambda_v C_o(t) + g_i = K_g \quad \text{... (5)}$$

Also we can write $$\left[\frac{dC_i(t)}{dt}\right]_- = \lambda_g C_i(t) \quad \text{... (6)}$$

where $_g = \lambda_v + _{de}$ represents the overall loss rate (h⁻¹). In a non-ventilated room, $\lambda_v = 0$;

therefore, $$\frac{dC_i(t)}{dt} = g_i - \lambda_{de} C_i(t) \quad \text{... (7)}$$

Whereas in a ventilated room,

$$\frac{dC_i(t)}{dt} = K_g - \lambda_g C_i(t)$$

On integrating we get, where $C_i(0)$ is the indoor concentration at the time of switching on the ventilation. This gives

$$C_i(t) = \frac{K_g}{\lambda_g}\left[1 - e^{-\lambda_g t}\right] + C_i(0)e^{-\lambda_g t} \quad \text{... (8)}$$

K_g being given by eq. (5).

or $$C_i(t) = C_i(t)_+ + C_i(t)_- \quad \text{... (9)}$$

such that $$C_i(t)_+ = \frac{K_g}{\lambda_g}\left[1 - e^{-\lambda_g t}\right] \quad \text{... (10)}$$

and $$C_i(t)_- = C_i(0)e^{-\lambda_g t} \quad \text{... (11)}$$

Bouilly *et al.* (2005) measured the deposition loss rates ($_g$) for various particle sizes and ventilation strategies. Their results are summarized in Table 3. These data suggest that the highest loss rates for RPM are obtained with top-bottom ventilation. For larger particles top-bottom ventilation gives slightly lower deposition loss rate than other ventilation strategies.

Using eq. (8) and the mean values of $_g$ obtained by Bouilly *et al.* (2005) for top-bottom ventilation, the variation of indoor particle concentration with time has been plotted for various values of ventilation rate ($_v$), outdoor concentration (C_o), and indoor particle generation rate (g_i). Fig.1 shows the variation of indoor

coarse particles (PM_{10-15}) with ventilation for indoor generation rate of $10 \mu g/m^3$ and an indoor concentration of $240 \mu g/m^3$ when the ventilation is started. The plot suggests that although the concentration decreases faster with higher ventilation rate (1.0 air change per hour) in the beginning but after an hour of ventilation result is slightly better for lower ventilation rate (0.5 ach). The indoor coarse particle concentration decreases by ~97% if the I/O concentration ratio is ~5 and by ~75% for I/O ratio of ~0.5.

Figs. 2 and 3 show similar plots for PM_{3-10} for initial indoor concentrations of $240 \mu g/m^3$ and $2400 \mu g/m^3$, respectively. If the outdoor concentration is lower than the indoor concentration, the ventilation rate is immaterial and the indoor concentration decreases to ~20% for 0.5ach as well as 1.0ach. However if the outdoor concentration is higher than the indoor concentration, higher ventilation rate is ineffective, whereas with slow ventilation the indoor concentration decreases by a moderate factor.

Ventilation affects the fine particles (PM_3) quite differently (Fig.4). If outdoor concentration is lower than the indoor concentration, the latter decreases slowly for both fast and slow ventilation rates. For an initial I/O ratio of 10:1, it takes ~60 & 90 min. for the indoor concentration to decrease to half the initial level for 0.5ach and 1.0 ach ventilation rates, respectively. On the other hand if the outdoor concentration is of the order of the indoor concentration, the latter decreases marginally for 0.5ach ventilation rate, whereas it increases (to ~150% in 2½ hours) for 1ach ventilation rate.

Tables 3 and 4 give the variation of I/O ratio with time for PM_3 and PM_{3-10} for top-bottom ventilation strategy for two different ventilation rates for various ratios of outdoor concentration and indoor generation rate. Indoor concentration (C_i) at zero hours is taken equal to $24g_i$ assuming one full day of closed doors (no ventilation) and zero deposition owing to various activities like cleaning etc. before switching on the ventilation. The calculations suggest that both slow and fast ventilation reduce the PM I/O ratio with time for all particle sizes; except for fine particles (PM_3) in the event of the outdoor concentration being twenty five times or more as high as indoor generation rate. However, in all the cases the slow ventilation is more effective.

Let us compare these calculations with some observed results. The measurements of Geller *et al.* (2002) gave an average indoor-to-outdoor $PM_{2.5}$ mass concentration ratio of 1.03 (±0.29). This is in agreement with the I/O PM_3 mass concentration ratios obtained by our model for 0.5 ach ventilation rate after 12 hours of ventilation when the outdoor concentration is more than the indoor PM_3 generation rate ($C_o = 2.5 g_i$ to $25 g_i$; see table 4). This suggests that the 0.5 ach top-bottom mechanical ventilation may be considered equivalent to the natural ventilation of the houses monitored by Geller *et al.* (2002). The I/O PM_{3-10} mass concentration ratios obtained by our model with 0.5 ach top-bottom mechanical ventilation after 12 hours of ventilation are 0.88, 0.76, 0.45, 0.35, and 0.27 for $C_o = 0.8g_i$, g_i, $2.5g_i$, $5g_i$, $25g_i$, respectively. In Geller's (2002) measurements $PM_{2.5-10}$ indoor concentrations were found markedly lower than those outdoors with the average indoor-to-outdoor mass concentration ratio equal to 0.66 (±0.27). Thus our calculated values for PM_{3-10} for 0.5 ach ventilation after 12 hours of ventilation also agree with the Geller's observations when the outdoor concentration is nearly equal to the indoor PM_{3-10} generation rate ($C_o = 0.8g_i$ to 2.5 g_i; see table 5).

PARTICULATE CONTROL IN INDUSTRY

Various techniques are used to control air pollutants in industrial establishments depending on the size and properties of the PM (USEPA/BCES: module 6, 2010). These include

Gravity settling chamber: This can be used only for very large particles (75 μm).

Mechanical collectors: These are used to remove particles larger than 5 μm. These equipments provide control efficiency of 50 to 90%.

Particulate wet scrubbers: Their collection efficiency varies with particle size. It is good for PM_{5-10} (90-100%) but is very limited for PM< 0.3 μm for some types.

Electrostatic precipitators (**ESPs**): They can have very high efficiencies for PM_{1-10}, but it decreases for UFPs and is minimum (20-60%) for 0.1 to 0.5 μm size range. For $PM_{0.1}$ the collection efficiency of various ESPs is low-to-moderate (40-75%).

Fabric filters: This technique is highly efficient for the entire particle size range of interest in air pollution control. The control efficiencies usually range from 99% to greater than 99.5% depending on the characteristics of the particulate matter and the fabric filter design.

Conclusion

The above studies and discussion provide us an understanding on how a deadly combination of solid fuels and poor ventilation triggers off an alarming increase of health damaging RPM in homes. Though dynamic and complex relationship exists between sources, air change rate and other variables but it is important to control indoor PM concentration. Awareness programs should therefore be undertaken to take proper measures to reduce RPM in indoor air so as to develop an environment which is suitable for its residents. The most effective control measure in residences and offices is a proper ventilation strategy. Ventilation acts differently on different particle sizes: the smaller the particles are, the more important the ventilation strategy is on airborne particle concentration. Ventilation plays a major role in controlling the indoor

coarse particulate ($PM_{>10}$) irrespective of the outdoor concentration and ventilation rate. But the choice of inlet/outlet locations and ventilation rate become important for finer particles. In case of fine particulate lower rate of ventilation is preferable.

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