Particulate Matter Exposure during Domestic Work in Nepal

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Aims: To measure particulate matter (PM) exposure of people involved in domestic work (i.e. housework by a resident, not paid work) in urban and rural Nepal, with exposure to biomass smoke in the rural areas, and to examine the performance of photometric devices in collecting these data. This paper details the results of these measurements and derives calibration factors for two photometric devices compared to gravimetric measures.

Methods: Between April 2006 and February 2007, respirable dust and PM\(_{2.5}\) levels were measured over a 24-h period in 490 households in a range of urban and rural settings in the Kathmandu valley of Nepal. Sampling was carried out by photometric and gravimetric methods with the co-located gravimetric data used to derive a calibration factor for the photometric devices.

Results: The time-weighted average (TWA) (24 h) respirable dust levels measured by gravimetric sampler ranged from 13 to 2600 \(\mu g m^{-3}\) in the rural settings and 3 to 110 \(\mu g m^{-3}\) in the urban settings. The co-located photometric and gravimetric devices indicate that the SidePak Personal Aerosol Monitor device required a calibration factor of 0.48 and 0.51 for rural and urban data, respectively, whereas the DustTrak device required a factor of 0.31 and 0.35 for rural and urban settings to correct for the particle size and density of the biomass smoke. The photometric devices provide time history data on PM concentration levels and generally indicate two distinct peaks around a morning and early evening cooking time.

Conclusions: Those involved in domestic work in rural Nepal are exposed to average respirable dust concentrations of \(\sim 1400 \mu g m^{-3}\). Converted to an 8-h TWA, this equates to more than the current UK limit for respirable dust (4000 \(\mu g m^{-3}\)). Homemakers, primarily women, spend a large proportion of their lives indoors in these high respirable dust concentrations and these exposures are likely to produce respiratory illness. Exposure can be controlled by the use of different fuel types and/or the use of flued stoves.

Keywords: biomass fuels; domestic work; indoor air pollution; particulate matter (PM)

INTRODUCTION

Indoor smoke exposure is a recognized major cause of ill health in less economically developed countries (LEDCs) (Pandey et al., 1989; Smith et al., 2000; Smith, 2003). Homemakers in the rural areas of developing countries tend to be women whose prime occupation involves activities such as cooking, cleaning and childcare. The fuels for heating and cooking are primarily biomass fuels such as wood, dried cow dung and charcoal. This work involves long periods of exposure to smoke and fine particulate matter (PM) (Bruce et al., 2000; Smith et al., 2000; Ezzati, 2005). These exposures also put the children within the household at risk and the lack of resources to improve ventilation or to switch to better fuel further prolongs exposure and risk. Indoor air pollution (IAP) and its health effects is becoming a key target area for the United Nations Development Programme, World Health Organization (WHO) and other national and international agencies. The WHO listing IAP from burning solid fuels as one of the top 10 global health risks, being responsible for 1.5 million deaths per year and 2.7% of the global burden of disease, including chronic obstructive pulmonary...
around half of the world’s population, >3 billion people, use unprocessed biomass fuels such as wood, dung, crop residues and coal to meet their basic household energy demand (WRI, 1998–1999; Smith et al., 2004; Rehfues, 2006). It is estimated that world’s total energy consumption from biomass has decreased from 50% in 1900 to a current approximate value of 14% (WRI, 1998–1999; Parikka, 2004; Smith et al., 2004), but in >30 countries, primarily those that are less economically developed, wood still provides >70% of household energy requirements. Recent increases in the price of alternatives such as kerosene and bottled gas has further increased the use of biomass fuels, slowing the transition to cleaner fuels (Bruce and Schirnding, 2000).

Given the magnitude of health burden that is attributable to IAP in LEDCs, there is a need to quantify the risk of IAP more precisely to determine the degree of reduction in exposure required to significantly improve health and to establish the effectiveness of interventions (Naeher et al., 2000).

Previous work has shown high indoor concentrations of PM generated from biomass or solid fuels in different settings. A study in Nepal close to the Kathmandu valley reported PM$_{10}$ concentrations ranging from 5100 to 8100 µg m$^{-3}$ with a mean concentration of 2400 µg m$^{-3}$ in houses using biomass as their main fuel and without proper ventilation (Shrestha and Shrestha, 2005). Other studies have reported peak concentration of PM$_{10}$ in houses using biomass fuels, ranging from 3000 up to 18 900 µg m$^{-3}$ (Park and Lee, 2003; Regalado et al., 2006) and suspended particle $>50$ 000 µg m$^{-3}$ (Ezzati and Kammen, 2001) during cooking. Brauer et al. (1996) reported PM$_{2.5}$ concentration of 300–1500 µg m$^{-3}$ with an average of 560 µg m$^{-3}$ in Mexico (Brauer et al., 1996). Similarly, Naeher et al. (2000) reported average PM$_{2.5}$ concentrations of 640 µg m$^{-3}$ from open fires in highland Guatemala before going for an intervention study (Naeher et al., 2000) with similarly high levels in Zimbabwe (Collings et al., 1990) and India (Semple et al., 2008).

It is important to be sure that the methods used for assessing levels of exposure are accurate. Gravimetric techniques had been widely used to measure the mass concentration of filters in both indoor and outdoor environments. They provide a measure of the integrated sample and time-averaged concentrations over a given period and are generally regarded as the primary method in the measurement of air PM (Ayers et al., 1999).

However, photometric methods are widely used to measure the micrometre and submicrometre range of airborne particles both in indoor and outdoor environments, providing real-time measurements in a continuous and repeated manner. They have been widely used to measure aerosol concentrations in occupational and environmental exposures. This approach is satisfactory provided that there is a consistent relationship with results obtained from gravimetric dust samplers (Armbruster, 1987) as the light scattering devices rely on indirect sensing techniques that require more frequent calibration (Pui and Chen, 2000). They are simpler to use and are less labour intensive than gravimetric methods where filters need to be weighed on a high-precision scale before and after sampling in well-defined environmental conditions. In addition, volatile substances collecting in the filter sample might evaporate before weighing resulting in underestimates of the true mass (Sillanpää et al., 2002). Furthermore, gravimetric methodology cannot provide real-time measurements and require long sampling periods so that sufficient mass can be collected on the filter in order to weigh the sample. Photometric devices are commonly calibrated to ‘Arizona road dust’ and so often over-read levels compared to gravimetric measures particularly when measuring combustion-derived PM. Photometric devices thus need to be calibrated regularly depending on the source in use and the environmental conditions under study.

As part of a wider study examining the relationship between levels of PM exposure and respiratory and cardiac disease, levels of PM in indoor and outdoor air were measured in a selection of rural and urban households in the Kathmandu valley in Nepal. In all, 490 households with a total of 1648 participants were studied and the clinical results will be published at a later date. Measuring levels of pollutant exposure will help estimate the dose–response relationship of these clinical indicators but this requires accurate assessment of particle exposure. Although photometric devices have been used in measuring IAP in LEDCs (Semple et al., 2007), there is little information on the appropriate calibration factors for these instruments when used to measure PM from biomass fuel smoke in such environments. One of the aims of this work was to examine the relationship between data generated from photometric devices and co-located gravimetric sampling devices.

METHODS

Ethical approval was obtained from the Nepal Health Research Council and permission to work in the rural areas was obtained from the secretaries of two village development committees for security reasons due to political unrest within Nepal during the period of this work.

Recruitment of households and general description of sampling sites

The sampling was carried out in urban and rural areas of Kathmandu valley, Nepal from April 2006 to February 2007. The sampling sites were selected...
after careful consideration of fuel use, availability of people willing to participate and the local security situation. Both rural and urban sites were located at an altitude of between 1300 and 1600 m above sea level with the rural sampling sites between 12 and 17 km from the centre of Kathmandu metropolitan area. Locally derived wood was the predominant type of fuel used for cooking and heating in these rural locations. The urban sampling sites were within the Kathmandu metropolitan area and here liquefied petroleum gas (LPG) was used as the primary domestic fuel. The majority of the houses in the rural areas were constructed from a mud-based material with a thatched or tiled roof. In the urban areas, houses were of brick and cement construction.

Households were selected using cluster random sampling in both urban and rural areas. A total of 245 households in the rural area and an equal number of households from the urban area were sampled. Prior to sampling each household was visited personally to obtain written informed consent for the measurements to be done. PM sampling was done in households that used firewood or LPG as their main fuel in the rural and urban areas, respectively, and where at least two adults (≥16 years of age) were living. The sampling was carried out by six trained local researchers supervised by the principal researcher (O.P.K.).

**Indoor respirable particle measurements**

*Photometric methods.* The TSI SidePak AM510 Personal Aerosol Monitors and TSI DustTrak Aerosol Monitors (TSI Incorporated, Shoreview, MN, USA) are laser photometers (particle size range 0.1–10 μm and minimum resolution 1 μg m⁻³) which measure aerosol concentrations in the range of 1–20 000 and 1–100 000 μg m⁻³, respectively. Both were zero calibrated at least twice each week during the study with the flow rate set to 1.7 l min⁻¹ using a TSI flow meter. The devices were set to measure respirable dust using a Dorr-Oliver cyclone.

*Gravimetric methods.* Respirable dust was measured using standard gravimetric methods in accordance with MDHS 14/3 (HSE, 2000). Briefly, an Apex pump (Casella CEL Ltd, Bedford, UK) was attached to an aluminium Higgins Dewell cyclone containing a pre-weighed low extractable, binder-free borosilicate glass fibre filter with a 0.7-μm (nominal) pore size designed to meet the requirements of the US Environmental Protection Agency (EPA) SW-846. The flow was set to 2.2 l min⁻¹ using a calibrated rotameter and was retested on completion of the 24-h sampling. The rotameter was calibrated in Aberdeen at STP and was used to set pump flow rates on sampling location. No further correction for temperature and pressure was made. The filters were weighed twice in laboratory conditions at the Department of Environmental and Occupational Medicine in Aberdeen, with at least 24 h between the two weighing.

Filters used for the study were packed in steel containers and sealed with tapes being taken from the sealed pack just before use at the sampling sites. After sampling, each filter was repacked immediately in the field into the steel container and sent back to the UK for re-weighing, weights being recorded twice at least 24 h apart. All the filters were weighed before and after the experiments with a five-point scale balance (Scaltec Instruments Inc., Brentwood, NY, USA) (detection limit 0.00001 g = 10 μg). The weighing was carried out at the same temperature and relative humidity both before and after sampling. Seven field blanks and five laboratory blanks were used. The transport losses were not evaluated directly but we used both laboratory and field blanks to correct the data.

*Side-by-side co-location of samplers.* In a number of settings, gravimetric samples were co-located with either a DustTrak (DT) or a SidePak (SP). Studies need to have identical spatial positioning for the monitor and monitoring duration for a comparison to be made. The spatial positioning of the instruments has been mentioned below for a comparison to be made in similar type of studies in future. Both samplers were placed on a wooden stand at a height of 1 m from ground level and between 0.5 and 1.0 m from any cooking stove over a period of ~24 h. The distance 0.5–1.0 m is the typical distance between the stove and the individual performing domestic cooking work.

*Development of calibration factors.* Thirty-five rural households were sampled with co-located gravimetric sampler (GMS) and photometric devices. Out of 35 rural households, 18 households were sampled with co-located GMS and DT and remaining 17 households were sampled with co-located GMS and SP. Similarly, 25 urban households were sampled with co-located GMS and photometric devices. Out of 25 urban households, 7 households were sampled with co-located GMS and DT and remaining 18 households were sampled with co-located GMS and SP.

Twenty-two (rural = 16 and urban = 6) out of 25 co-located GMS and DT and 31 (rural = 14 and urban = 17) out of 35 co-located GMS and SP samples were used to derive the calibration factors for the photometric devices. Two calibration factors were obtained for each photometric device, one each for the rural and urban areas. A total of seven pairs (n = 4 for GMS + SP and n = 3 for GMS + DT) were excluded because of very large differences (>3 SD of the mean of the difference) between the gravimetric and photometric devices. In all six cases, activities within the household that could have resulted in contamination of the gravimetric sample or tampering with the photometric device were identified and also could be possibly due to haze.

In a further 430 (rural = 210 and urban = 220) households, PM₂.₅ levels were measured using either a SP (rural = 189 and urban = 140) or DT
Sampling was performed for around a 24-h period in order to capture data on all daily cooking activities. The exact length of time of sampling for each house was recorded. The start and end time for sampling in rural areas was between 09:30 and 10:00 h. In urban areas, start/end times were usually between 15:30 and 16:00 h. Different start and end time were chosen for rural and urban areas to accommodate participants’ work times. Sampling for <24 h may have introduced some degree of error. However, in our study, all the cooking time data were captured and the average measurement time was >22.5 h. While this would likely lead to a slight overestimate of the concentrations within the household over a full 24-h period, we think that the size of this error is likely to be much smaller than the day-to-day variability in exposure levels within any given household.

Statistical analysis

Data were downloaded using the TRAKPRO v3.61 software and analysis to compare with gravimetric results was carried out using SPSS v15.

Descriptive statistics like arithmetic means, minimum and maximum values, range and standard deviation were generated for rural and urban respirable PM data. Gravimetric results were plotted against those from each photometric device and the calibration factors were obtained by applying linear regression methods. Pearson correlation coefficient and its two-tailed P-values were calculated for the continuous bivariates to check the degree of association and its significance, respectively.

The TWA for PM values for each household were calculated from the total time of measurement and the mean of the PM data for every minute measurement by using equation (1)

$$C_{\text{TWA}} = \frac{\sum C_1 + C_2 + \ldots + C_n}{N}, \quad (1)$$

where, $C_{\text{TWA}} = $ concentration of PM measurement in a household for the measured time period in $\mu g m^{-3}$; $C_1 = $ mean PM concentration for time $T_1$ ($T_1 = 1\text{st min}$) in $\mu g m^{-3}$; $C_2 = $ mean PM concentration for time $T_2$ ($T_2 = 2\text{nd min}$) in $\mu g m^{-3}$; $C_n = $ mean PM concentration for time $T_n$ ($T_n = nth\text{ min}$) in $\mu g m^{-3}$; $N =$ total number of measurement.

Similarly,

$$C_{24-\text{TWA}} = \frac{C_{\text{TWA}} \times T_S}{24}, \quad (2)$$

where, $C_{24-\text{TWA}} = 24\text{-h TWA in } \mu g m^{-3}; T_S = \text{total sampling duration.}$

RESULTS

Photometric calibration

Table 1 shows 30 and 23 measurements from rural and urban dwellings, respectively, with GMSs co-located with either SP or DT. Respirable dust concentrations in the rural homes were much higher than those found in urban house (Table 1) concentrations in the kitchen being several fold higher during periods of cooking. In most of the co-located samples, the ratio of SP to GMS measures lay close to either 2 or 3 but on three occasions they exceeded 35. Similarly, the DT overestimates respirable particle mass levels compared to the GMS by a factor that lies close to either 3 or 4 on most occasions.

There was a reasonable linear relationship as seen in the regression lines between the co-located SP and GMS data for rural areas [Fig. 1; Pearson correlation coefficient $r = 0.538\ (n = 14), P\ (\text{two tailed}) = 0.047$] providing a calibration factor for SP measurements in rural areas of 0.48. Similarly, co-located SP and GMS data for urban areas are weakly correlated [Pearson correlation coefficient $r = 0.428, P\ (\text{two tailed}) = 0.086$] but provided a calibration factor for SP measurements in urban areas of 0.51.

There was a strong linear relationship between the co-located DT and GMS data for rural areas [Fig. 2; Pearson correlation coefficient $r = 0.686\ (n = 16), P\ (\text{two tailed}) = 0.03$] providing a calibration factor for DT measurements in rural areas of 0.31. Similarly, there was a very strong linear relationship between the co-located DT and GMS data [Pearson correlation coefficient $r = 0.947\ (n = 6), P\ (\text{two tailed}) = 0.004$] for urban areas providing a calibration factor for DT measurements in urban areas of 0.35.

These calibration factors were then applied to the remaining 430 SP or DT household measurement data.

Table 1. TWA respirable particles concentrations ($\mu g m^{-3}$) measured by SP, DT and GMS in rural and urban settings

<table>
<thead>
<tr>
<th>Location</th>
<th>Combination</th>
<th>n</th>
<th>Mean SP or DT</th>
<th>Mean GMS</th>
<th>Standard deviation SP or DT</th>
<th>Standard deviation GMS</th>
<th>Minimum SP or DT</th>
<th>Minimum GMS</th>
<th>Maximum SP or DT</th>
<th>Maximum GMS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rural</td>
<td>SP–GMS</td>
<td>14</td>
<td>893</td>
<td>339</td>
<td>508</td>
<td>256</td>
<td>283</td>
<td>13</td>
<td>1850</td>
<td>912</td>
</tr>
<tr>
<td>Rural</td>
<td>DT–GMS</td>
<td>16</td>
<td>3580</td>
<td>1050</td>
<td>2830</td>
<td>685</td>
<td>747</td>
<td>171</td>
<td>8410</td>
<td>2560</td>
</tr>
<tr>
<td>Urban</td>
<td>SP–GMS</td>
<td>17</td>
<td>121</td>
<td>51</td>
<td>45</td>
<td>29</td>
<td>56</td>
<td>3</td>
<td>198</td>
<td>111</td>
</tr>
<tr>
<td>Urban</td>
<td>DT–GMS</td>
<td>6</td>
<td>231</td>
<td>79</td>
<td>49</td>
<td>20</td>
<td>187</td>
<td>59</td>
<td>319</td>
<td>109</td>
</tr>
</tbody>
</table>
Personal exposure to PM

The 24-h mean of respirable dust presented in Table 2 is the mean data from urban and rural sampled households. There is no air quality guidelines for indoor PM$_{10}$ and respirable particles but when compared to the US EPA standard for outdoor PM$_{10}$, the respirable data lie in the range that are from unhealthy to possessing hazardous health effects. Nepal does not have workplace exposure limits for respirable particles. In the UK, a TWA of 4000 µg m$^{-3}$ brings exposure into the scope of the Control of Substances Hazardous to Health Regulations. Our measurement indicated that the respirable dust levels in the households of rural areas lie within 410–7820 and 80–300 µg m$^{-3}$ in households of urban areas. These figures were calculated from the 24-h mean and by then comparing to the 8-h occupational UK limit (i.e. multiplying by 3 to produce an 8-h TWA equivalent exposure), nearly 40% of the homemakers in rural areas have exposure to respirable dust that exceeds the UK limit of 4000 µg m$^{-3}$.

Figure 3 shows a typical real-time diurnal pattern of indoor PM$_{2.5}$ measured by photometers (SP and DT) in a rural and an urban home after correcting with the calibration factor derived from the co-located sampling data. The high peaks show the level of PM$_{2.5}$ during the morning and evening cooking periods in both cases. These data demonstrate the temporal variability of concentrations over the 24-h period and show that urban areas have a higher background (probably due to outdoor air pollution) but that peaks during cooking times are much higher in rural settings.

DISCUSSION

To the best of our knowledge, this is the first study that has measured levels of PM exposure of homemakers in both rural and urban Nepal. This cross-sectional study has provided important quantitative exposure data for both respirable particles and PM$_{2.5}$ from biomass burning in rural areas and from LPG use in urban homes within Nepal for the first time. The difference in exposure to respirable particles and PM$_{2.5}$ in rural and urban Nepal is large, with 24-h TWA respirable particle levels in 30 rural
Nepalese households of 792 μg m⁻³ (range 136–2610 μg m⁻³) >10 times higher than that measured in urban households at 67 μg m⁻³ (range 29–112 μg m⁻³). These rural exposures are primarily due to biomass (wood) burning for cooking and heating and can be compared to respirable particles measured elsewhere such as rural areas of China (2600 μg m⁻³), Kenya (1300 μg m⁻³), Gambia (2100 μg m⁻³) and India (1442 μg m⁻³) (Smith and Liu, 1994; Brauer et al., 1996; Balakrishnan et al., 2002; Khalequzzaman et al., 2007). PM concentrations are high in these settings due to the source strength and poor room ventilation.

This study also provides calibration factors for two photometric devices (SP and DT) by calibrating them with gravimetric methods in two different settings based on the types of fuel use. Both the photometric equipments were calibrated in the factory using ISO 12103-1, A1 (formerly known as ultra-fine Arizona test dust or Society of Automotive Engineers ultra-fine). The dust size distribution as well as particle refractive index and particle density of the dust from combustion sources found in Nepal homes are likely to be very different from the above test dust and without applying proper calibration the results of photometric measurements would be inaccurate (Dasgupta et al., 2006).

Results from this study in Nepal shows that light scattering devices overestimate the respirable concentration of indoor aerosols in comparison to the gravimetric method, the SP and DT devices giving

Table 2. UK occupational limit, US EPA outdoor standard and arithmetic mean of respirable indoor particles in urban and rural Nepal

<table>
<thead>
<tr>
<th>Location</th>
<th>Sample (n)</th>
<th>UK limit</th>
<th>US EPA standarda</th>
<th>Respirable dust 24-h mean</th>
<th>Standard deviation</th>
<th>Range</th>
<th>8-h TWA mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Urban</td>
<td>23</td>
<td>4000</td>
<td>150</td>
<td>67</td>
<td>23</td>
<td>29–112</td>
<td>201</td>
</tr>
<tr>
<td>Rural</td>
<td>30</td>
<td>4000</td>
<td>150</td>
<td>792</td>
<td>738</td>
<td>136–2610</td>
<td>2380</td>
</tr>
</tbody>
</table>

All the unit are in μg m⁻³.

a0–50 are good and >50–150 have moderate effect to health.
readings approximately two and three times higher, respectively. At higher concentrations, the scatter becomes wider with less consistency, although at low (<1 mg m$^{-3}$) concentrations the relationships are much tighter. This might be also due to the variability of source type. In rural settings, the types of fuel and their physical characteristics differ from house to house and also in same house at different cooking times. For example, in rural areas the type of wood may differ between households (e.g. damp wood and different tree types) but in urban homes the particle size distribution/density will be much more uniform as it is generated from LPG, although contributions from outdoor air will be significant. LPG produces much lower particle concentrations. However, the high indoor urban concentrations seen in our data reflect the influence indoor ingress of outdoor particles, partly from vehicle emissions but in particular from disturbed road dust because of poor road conditions. The calibration factors obtained in this study are very close to some of the calibration factors derived in other studies using photometric devices but most of them cannot be compared as they were calculated in different environments by different standard methods. Calibration factors of 0.32 (Repace, 2004; Edwards et al., 2006; Wilson et al., 2007) calibrated against another laser photometer for PM data all from cigarette smoking, 0.295 (Leavell et al., 2006) for PM data also from cigarette smoking and 0.38 (Green et al., 2007) for PM data from mineral dust exposure calibrated against gravimetric methods have been reported for SP data. Similarly, calibration factors of 0.49 (Mugica et al., 2002) for TSP and PM$_{10}$ from metal contents in Mexico city and 0.51 (Englert, 1999) for IAP in Germany for the DT against the mini-volume air sampler have been reported.

The 24-h respirable dust concentrations for urban and rural areas measured in this study are in the range of 30–100 and 140–2600 lgm$^{-3}$, respectively, and are very high. Almost all the values in the rural areas are higher than the US EPA outdoor air quality index for ‘unhealthy’ fine particles. US EPA rates PM$_{2.5}$ concentrations >65–150 µg m$^{-3}$ as unhealthy, >150–250 µg m$^{-3}$ ‘very unhealthy’ and >250–500 µg m$^{-3}$ as ‘hazardous’ and most of the values are higher than 150 µg m$^{-3}$ which is rated as very unhealthy. All the values are also greater than the WHO interim target 1 24-h PM$_{2.5}$ guideline of 75 µg m$^{-3}$. Most of the 24-h TWA respirable dust concentrations measured in rural Nepalese homes exceeds the air quality standards on respirable PM of 100 µg m$^{-3}$, a standard set by the Central Pollution Control Board of India for outdoor air in rural residential areas in India (Central Pollution Control Board, 2008).

The indoor PM generated from domestic cooking in Nepal can also be compared to other studies of occupational exposure during cooking tasks. The mass concentration of PM$_{2.5}$ generated in commercial kitchens in Singapore (premises of National University of Singapore) by Malay (245 µg m$^{-3}$), Chinese (202 µg m$^{-3}$) and Indian (187 µg m$^{-3}$) cooking differs from one another due to the different methods in

![Fig. 3. Example of diurnal pattern of indoor PM$_{2.5}$ in both rural and urban sites.](image-url)
cooking, the amount of food cooked and the cooking time (Wei See et al., 2006). Similarly, PM$_{2.5}$ measured in four different restaurants in Hong Kong with different cooking styles gave concentrations of 1167, 81, 29 and 22 $\mu$g m$^{-3}$, respectively (Lee et al., 2001).

The rural study was completed by the end of the first week of September 2006. Most of the gravimetric measurements in the rural areas were carried out during the local rainy season but it is not believed that this would have had a major influence on either cooking/heating behaviour or the measurements obtained because people in Nepal cook indoors and hardly use biomass fuel for heating in any room other than the kitchen. The fuels are usually dry as they are stored indoors for the rainy season. People use moist fuel only if there is shortage of fuel but again usually these fuels are dried during sunny periods in the rainy season. Similarly, the urban sampling was started in the first week of September 2006 and completed in the second week of February. Most of the sampling was carried out during the winter seasons but it is believed that this is unlikely to have had a major effect on urban data as people rarely burn wood in rooms other than the kitchen.

**Strengths and weakness of our study**

The measurements were made during a period of conflict and unrest in Nepal which forced to sample in the rural areas quite close to Kathmandu city, although the area sampled was well away from urban vehicle pollution. While this might have led to a degree of sampling bias in representing the average Nepalese rural population and the rural environment but it is believed that any such effect was extremely small. Attempts were made to start and stop the measurements at the same time in every household and were successful in the great majority of occasions, although in some cases the start and stop times were slightly later in the morning due to the remote locations of the sampled households.

The SP and the DT are relatively portable and easy to use in remote areas, although sometimes electricity supply problems in the rural areas were experienced and had to operate the equipment on batteries that did not last for 24 h. This required temporary cessation of sampling to replace the batteries resulting in small periods of incomplete data.

SP failed to register the data measured in rural sites if the concentration of PM was $>20$ $\mu$g m$^{-3}$. It is recommended using equipment with higher range of measurement while measuring concentration in very highly polluted environment.

This study focuses only on the calibration of DT and SP against the gravimetric technique in the rural and urban settings in Nepal. It is highly recommended that a local calibration factor be calculated for future or similar studies.

**CONCLUSIONS**

People involved in domestic work in rural Nepal are exposed to levels of respirable dust several times greater than those involved in similar domestic tasks in urban areas in Nepal. The concentrations of respirable dust in rural Nepalese homes are high, even when averaged over the whole 24-h daily cycle, and results in ‘occupational’ exposures that exceed the UK limit for respirable dust. We would recommend that future studies using photometer devices calibrate results against locally derived gravimetric data especially when measuring PM generated from burning of different types of fuel, different burning conditions as this will have different dust size distribution with different particle density and refractive index.

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**REFERENCES**


