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Spatial variability and population exposure to PM2.5 pollution from woodsmoke in a New South Wales country town

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Abstract

A portable radiance research nephelometer was used to measure the variation in woodsmoke pollution in Armidale (a small town of 22,000 people), New South Wales, Australia, on 14 winter nights in 1996. Winter nights are characterised by inversions that trap the air within the valley and reduce winds to very low speeds (averaging $0.15\,m\,s^{-1}$). Pollution varied considerably with location. Mean scattering coefficients (bsp/10 km) for 14 measurement nights ranged from less than 1.0 on the undeveloped fringes of the city to 8.7, the latter representing a 14-night average of $200\,\mu g\,m^{-3}$ of PM2.5. Pollution was generally highest in the residential areas on either side of the valley, where the smoke was generated, rather than the low-lying central creeklands. In places, average pollution levels increased 4-fold within 41 m. The correlation between nephelometer and gravimetric pollution measurements ranged 0.95–0.99. The presence of large, sudden but repeatable changes in air pollution, and high correlations between nephelometer and gravimetric measurements, indicate that mobile pollution monitoring devices provide a useful and accurate estimate of spatial variability. Estimated exposure for the town as a whole was 1.02 for the 6 months from April to September, 0.25 in October as heater use declines, and 0.12 in normal summer months. For comparison, published 25th, 50th and 75th percentiles of the distribution of nephelometer coefficients in Sydney were 0.15, 0.24 and 0.37, respectively. Thus annual exposure to PM2.5 pollution in Armidale from woodsmoke is more than double that from all sources in Sydney, a city of 4 million. Overseas estimates of 6% increased mortality for each additional $10\,\mu g\,m^{-3}$ of PM2.5 suggest that wood heaters in Armidale may increase mortality in Armidale by about 7%, with estimated cost of about \$4270 per woodheater per year. Alternative cheap and environmentally friendly methods of keeping houses warm in winter, such as solar heating, should therefore be developed.

Keywords: Wood smoke; PM2.5; PM10; Spatial variability; Health costs; Mapping

1. Introduction

Armidale, New South Wales, is a city of some 22,000 people. The town is situated in a bowl-shaped valley on the Northern Tablelands at an elevation of approximately 1000 m. The winter climate is characterised by mild days (average daily maxima 12–14 °C) and cool, often frosty nights (average daily minima 0–2 °C). Although many countries introduced insulation standards for new buildings and major renovations in the 1970s, insulation was not required in Armidale until 1997. Consequently, a large proportion of the

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7000 houses have little or no thermal insulation. Armidale City Council reported that 47% of new houses built in 1992 had no ceiling insulation other than reflective foil and 67% had no wall insulation other than foil. Houses usually also have large eaves (600 mm) that not only shade north facing windows in December and January, but prevent 80% of solar energy at midday from entering a 1.8-m window in September (average daily minimum temperature 4 °C) and increase the need for heating throughout winter.

Woodheaters became popular in the 1970s as the cheapest method (apart from retrofitting insulation) of overcoming the deficiencies in housing. Coal and diesel are prohibitively expensive, but firewood from land clearing and tree dieback was abundant and cheap. Woodheaters can produce a considerable amount of heat—15–25 kW on high burn (Todd, 2003). Gilmour and Walker (1995) noted that the average output of a woodheater on its lowest setting was 4.2 kW, significantly higher than the average home heating requirement (3.6 kW) in both Oregon and New York State, USA, where the colder climate is offset by better insulation.

Although some houses use gas or electric heating, surveys in 1995 and 1996 by the Armidale City Council showed that 55% of households had some form of solid fuel heating and 47% of these (i.e. 26% of households) used the heater 24-h a day. Population density in rural towns such as Armidale is lower than in large cities, most houses having about 0.25 acres (about 1000 m^2) of land. Buildings are usually single story, with chimneys discharging smoke about 4 m above ground level. Except for the small amount of traffic pollution typical of a small country town and a coal-fired boiler (now replaced by gas) for hot water and central heating at the university (on a hill some distance from residential areas, the stack discharging above the inversion layer) no other sources of particulate pollution have been identified and none are listed in the National Pollutant Inventory.

Emissions from woodstoves are a serious health hazard (see e.g. Robinson and Campbell, 1998 for a literature review). The particles are almost entirely less than 2.5 μm in diameter (Larson and Koenig, 1994), so, like all particles of this size, are known as PM2.5. Long-term cohort studies (e.g. Pope et al., 2002) suggest that these tiny particles are the most hazardous air pollutant.

PM2.5 penetrate deep in the lungs where they cause inflammation and increase the risk of heart and respiratory diseases (Godleski et al., 1996; Pope et al., 2004). Three independent US long-term cohort studies found significant relationships between PM2.5 and cardiopulmonary mortality (Dockery et al., 1993; McDonnell et al., 2000; Pope et al., 2002). The largest involved 500,000 subjects and 120,000 deaths. A 10 $\mu g\,m^{-3}$ increase in annual PM2.5 exposure increased cardiopulmonary mortality by 6–9% and lung cancer mortality by 8–14% (Pope et al., 2002).

PM2.5 or the broader class of PM10 (particles $<10\,\mu m$) can also cause substantial increases in the incidence of minor ailments. Lewis et al. (1998) reported that, for each additional 10 $\mu g\,m^{-3}$ of annual PM10 pollution, children suffered a 43% increase in chest colds and a 34% increase in night-time coughs. In Armidale, a significant correlation was observed between woodsmoke measurements and visits to GPs for respiratory complaints (Khan et al., 2001).

In addition to measuring average pollution levels in each community, it is important to take account of spatial variation. Another cohort study (Hoek et al., 2002) found that living within 100 m of a freeway or 50 m of a major road was associated with a relative risk of 1.95 (95% CI 1.09–3.51) in cardiopulmonary mortality. Estimates of the relative risk for a 10 μgm^{-3} increase in community average pollution (measured by black smoke) were considerably higher (1.71 vs. 1.34) if both local sources and background concentrations were included in the model, demonstrating the importance of local effects.

Jerrett et al. (2005) interpolated measurements from 23 PM2.5 and 42 ozone monitors in Los Angeles to obtain annual pollution exposure of 22,905 subjects (5856 deaths) in the American Cancer Society study. Estimated effects derived from the more accurate interpolated PM2.5 measurements were nearly 3 times greater than published estimates from models using community-wide PM2.5. The authors concluded that the chronic health effects of PM2.5 pollution could be much larger than previously reported.

Because of its bowl-like topography and the occurrence in winter of frequent inversions that can trap PM emissions close to ground level, spatial variability of particulate pollution was of considerable interest in Armidale. A pollution mapping exercise was therefore undertaken to measure average pollution levels on a number of transects across the valley and determine the variation with height and land use, from the sparsely developed fringes to the central business district (CBD) and the residential areas. Spatial variation was then used to predict population exposure to PM2.5 pollution and the effect on health.

2. Materials and methods

2.1. Air pollution mapping

A portable Radiance Research M903 integrating nephelometer was installed in an insulated box together with a heater to warm incoming air to approximately 25 °C and evaporate any water droplets (Photo 1). The set-up was designed to fit in the back of a small vehicle. A 3-m inlet pipe to the nephelometer was positioned through the top of the driver-side front window, facing forwards so the vehicle's motion assisted the fan to draw air through the instrument (Photo 2).

Four transects, from the elevated areas on south hill across the valley to the elevated areas north of the city, plus a shorter north–south transect and a circular route in East Armidale, were chosen to study the variation in pollution with height and land use, from the sparsely developed edges of the city to the CBD and residential areas. Measurements were taken between approximately 11 pm and 2 am on 14 nights between 17 July 1996 and 10 September 1996.

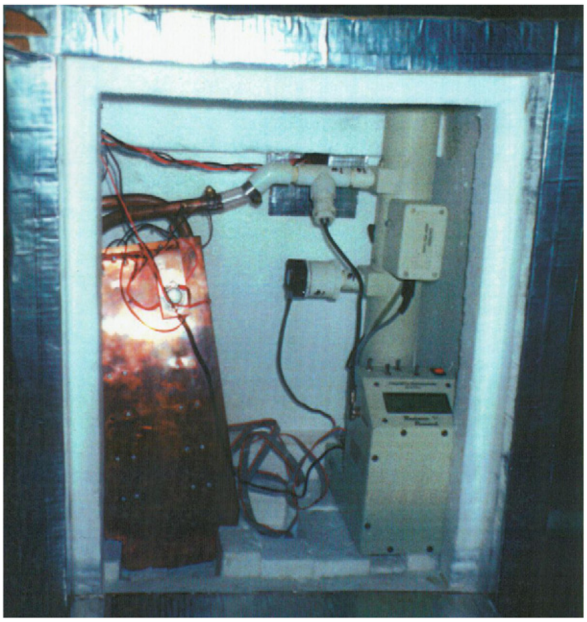

Photo 1. Insulated box with front cover removed, showing air intake through heater (left) to the RR neph (right).

Photo 2. Inlet pipe to the nephelometer positioned through the top of the front driverside window, facing forwards, so the vehicle's motion aided the fan in drawing air through the instrument.

To synchronise timings and ensure nephelometer readings had time to stabilise, each transect had up to 15 specified stopping points (SP), at which the vehicle paused for 88 s. Start and finish times of each transect, and all intermediate stopping times were recorded on a laptop computer, which also displayed the name of the SP and beeped when it was time to continue the drive. Between SPs, a slow driving speed of approximately 20 km h^{-1} was maintained. Nephelometer and laptop time were synchronised before each run. Average pollution values were calculated for all periods of 4 s during the run.

The order of measuring transects was varied, starting at either the east-most or west-most transect on different nights and driving either from north to south or vice versa, to remove any possible biases due to measurement order. After all data had been collected, the vehicle's emissions were tested by placing the inlet tube to the nephelometer close to the vehicle's exhaust. At the time, pollution levels were negligible and no increase could be observed in the air sampled near the vehicle exhaust. Pollution from the vehicle was therefore not considered to have any effect on the results reported here.

2.2. Statistical analysis of mapping data

After all runs had been completed, the average number of 4-s readings between SPs was calculated. These were on average 623, 562, 485, 467, 303 and 145 4-s records (41.5, 37.5, 32.3, 31.1, 20.2 and 9.7 min for transects 1–6, respectively). A linear interpolation algorithm was used to convert values from each individual run to the above numbers of readings per transect and translate these into distances from the start of each SP. Complete data were available for 78 transect passes; equipment failures on four nights resulted in a small amount of data (a total of 6 transect passes) not being collected. Two models were fitted to determine the best way of estimating the missing data:

$$\log(\text{neph\_coeff} + 0.1) = \text{night} + \text{location} + \text{error}, \quad (1)$$

$$\text{neph\_coeff} = \text{night} + \text{location} + \text{error}, \quad (2)$$

where neph_coeff (bsp/10 km) and location represent each individual interpolated 4-s value for the location being driven through. A log transformation was used because night effects were expected to be multiplicative, with low pollution nights being low throughout, and because, the higher the overall pollution, the greater the difference between the relatively unpolluted outer areas and locations where pollution can build up. The log-transformed model explained a greater proportion of total variation, but backtransforming fitted values to their original scale did not always produce lower residual variation, especially on low-pollution nights. For nights with missing data, residuals from the two methods (fitted using all available data on all nights) were calculated as the difference between observed data and fitted values (backtransformed for model 1 to the original scale and constrained to be non-negative). Whichever resulted in smaller residuals for the night was used to estimate that night's missing data; means were also computed for the 10 nights with no missing data. Spot heights of SPs were estimated from a contour map, enabling mean air pressure to be translated into elevation using the least-squares fit of the relationship between pressure and height.

2.3. Daily average pollution measurements

When not required for pollution mapping, the Radiance Research nephelometer (RR neph) was used to measure outdoor air pollution in East Armidale at a residential property that did not use wood heating. The nephelometer was calibrated at the start of the winter and checked again at the end of the winter when it was found to be reading 4% high, which was considered acceptable. To allow for this small deviation in calibration, measurements were reduced by an average of 2%.

Air pollution was also measured in Armidale by the NSW EPA using a Belfort nephelometer, located in the outdoor swimming pool complex, on the creeklands floodplain, close to the valley floor and immediately north of Armidale's CBD. To ensure measurements from the two nephelometers were comparable, the RR neph was co-located with the Belfort neph for 26 h from 14:00 on 9/8/96. During this period, mean pollution-weighted temperature at the RR neph inlet was 292.1 K and pressure 91.53 hPa. The correlation between hourly average measurements from the two nephs was 0.9997, implying the two sets of measurements were equivalent.

Australian air quality data are reported at conditions of standard temperature and pressure (STP, Ayers et al., 1999). After adjusting RR measurements to STP, the regression equation was

$$\text{bsp(Belfort)} = 1.025\text{bsp(RR, STP)},$$

where bsp is the nephelometer scattering coefficient per 10 km. Thus, after correcting measurements from the RR neph to STP, the two nephelometers were essentially measuring the same thing.

In 1995, a high-volume sampler was co-located with the Belfort nephelometer to ascertain the relationship of nephelometer and gravimetric measurements for seven periods of time (days or overnight periods). After excluding an obvious outlier (coinciding with a fire that burned down a warehouse not far from the nephelometer), the correlation between HiVol and neph measurements was 0.999 and the regression relationship was

$$\text{PM10(HiVol, STP)} = 3.90 + 23.2\text{bsp(Belfort)}.$$

A Series 1400 TEOM (Rupprect and Patashnick) with a PM2.5 cyclone was also co-located with the nephelometer in East Armidale to compare gravimetric and optical methods of measuring woodsmoke pollution. Initially, the TEOM proved unreliable due to frequent blockages of the filter and highly negative readings during the day, coinciding with periods when the volatile compounds in woodsmoke evaporate off the filter. On advice from the manufacturer (based on their experiments measuring woodsmoke, see Meyer et al., 1992), the flow rate was reduced to $1\,\text{L}\,\text{min}^{-1}$ and chamber temperature to 25 °C. Humidity in Armidale is low in winter, so this set-up ensured that water droplets were vapourised, while lessening the incidence of blockages and negative readings. The lowest was $-59\,\mu\text{g}\,\text{m}^{-3}$, averaged over the 6 h from 11 am to 5 pm on 25 August, following a peak the previous night of 183 $\mu g\,m^{-3}$ between midnight and 2 am.

The RR neph and TEOM were compared for a period of 308 hourly measurements. The correlation between TEOM and neph was slightly higher ($r = 0.955$) if measurements for generally low-pollution period from 11 am to 5 pm were averaged and any that were still negative set to zero, compared to using the untransformed data ($r = 0.950$). After omitting the intercept (which was non-significant), the regression relationship was

$$\mathrm{PM(TEOM,\ STP)} = 21.42\mathrm{bsp(RR\ neph,\ STP)}. \tag{3}$$

This relationship was broadly similar to that obtained by the NSW EPA HiVol sampler and Belfort neph, though marginally lower, as would be expected given the frequent negative pollution measurements recorded by the TEOM. The cyclone particle size cutoff for 1 $L\,min^{-1}$ was about PM8, rather than PM2.5. However, this should have made little difference because woodsmoke essentially has no particles greater than 1 μm (Larson and Koenig, 1994).

In woodsmoke-affected areas, TEOMs at 50 °C have considerable bias. DEH (2005) reported an empirical relationship between a HiVol and co-located TEOM (with flow rate of 2 $L\,min^{-1}$ because of the high particle loading) of HiVol $= 2.2 \times$ TEOM$-0.08 \times T$, when the daily average temperature, T, is between 0 and 15 °C. For the average of 6.4 °C in July in Armidale (Table 1), this equates to an average adjustment factor of 1.69, varying daily according to temperature. In this study, with lower TEOM operating temperatures, observed relationships between TEOM, nephelometer and HiVol measurements were more consistent, as has been found elsewhere (Meyer et al., 1992).

Table 1
Mean daily temperatures in Armidale by month and percent of solar energy blocked from a north-facing living room

Month	Ave. Temp.	Month	Ave. Temp	% of window area in shade[a]	
				150 mm eave	600 mm eave
		December	19.4	100	100
January	20.1	November	16.9	100	100
February	19.9	October	14.1	66	100
March	17.9	September	10.5	38	80
April	14.5	August	7.8	25	54
May	10.0	July	6.4	18	38
June	7.8			16	34

[a]Percent of north-facing window (1.8 m, directly under 150 mm or 600 mm eave) shaded at midday in a brick-veneer dwelling. Calculated from the height of the vertical shadow, $h = w\tan(\alpha)$, where α is the average angle of the sun at midday on 21st of the month and w = width of eave + gutter (150 mm) + indentation of window in brickwork (100 mm).

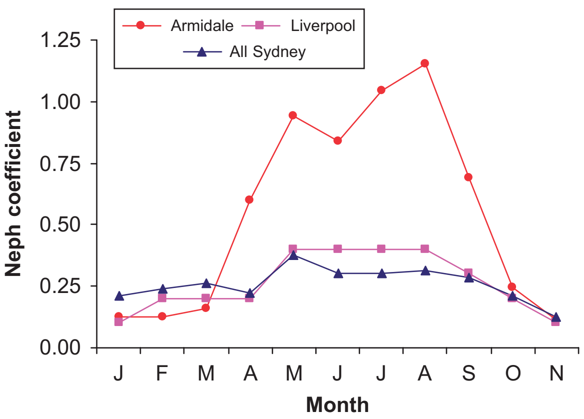

Fig. 1. Average pollution by month (nephelometer coefficient, bsp $(10\,km)^{-1}$, 1996) at the Armidale creeklands (some distance from residential areas) compared to Liverpool, Sydney (a suburb where some households use woodheaters) and the average of all monitoring stations in Sydney in 1999.

3. Results

3.1. Measurements at fixed locations

Fig. 1 compares monthly average pollution in Armidale, recorded by the EPA nephelometer located on the creeklands floodplain away from residential areas, with the average of all monitoring stations in Sydney (Australia's largest city, population 4 million) and Liverpool (a Sydney suburb where some households use woodheaters). Summer days in Armidale (November–March) are usually pollution free (about 0.05 bsp/10 km), though occasional bushfires and burnoffs have a small impact on monthly averages. Smoke levels rise significantly in April, remaining elevated until September. By October, wood heaters are used only on cold days, so pollution is only slightly higher than in summer.

Sydney has marginally higher summertime pollution than Armidale. In winter, about 13% of households in Sydney use woodheaters (NSW EPA, 1996) so pollution tends to increase (Fig. 1). The source of Sydney's wintertime pollution was determined by carbon dating PM10 samples taken

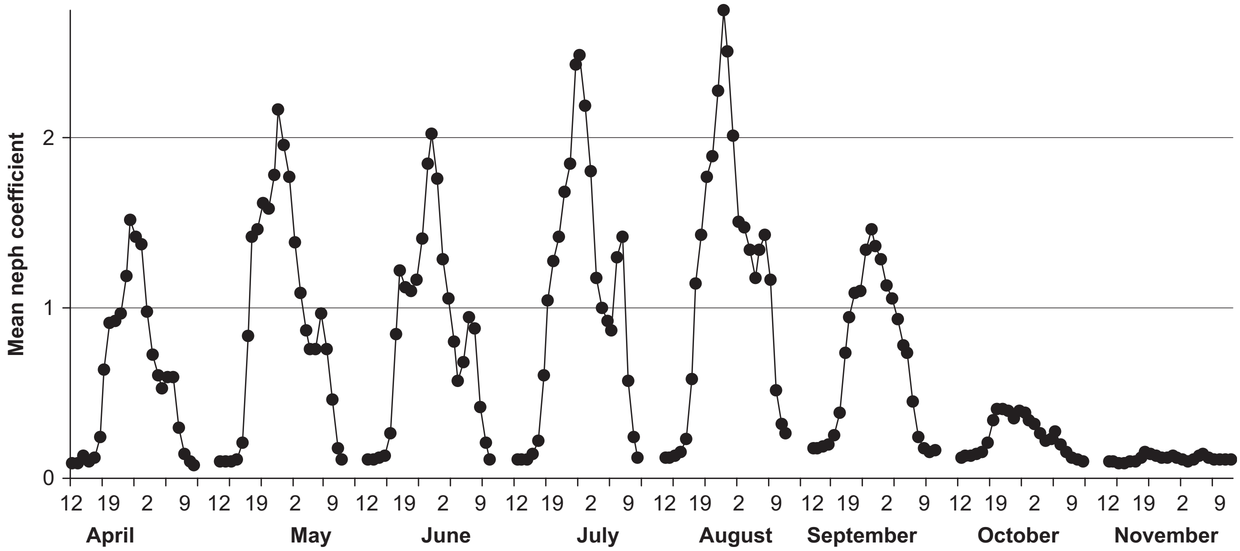

Fig. 2. Average pollution levels (nephelometer coefficient, bsp $(10\,\text{km})^{-1}$) for the months April–November, by hour of the day (starting 12 noon), Armidale creeklands.

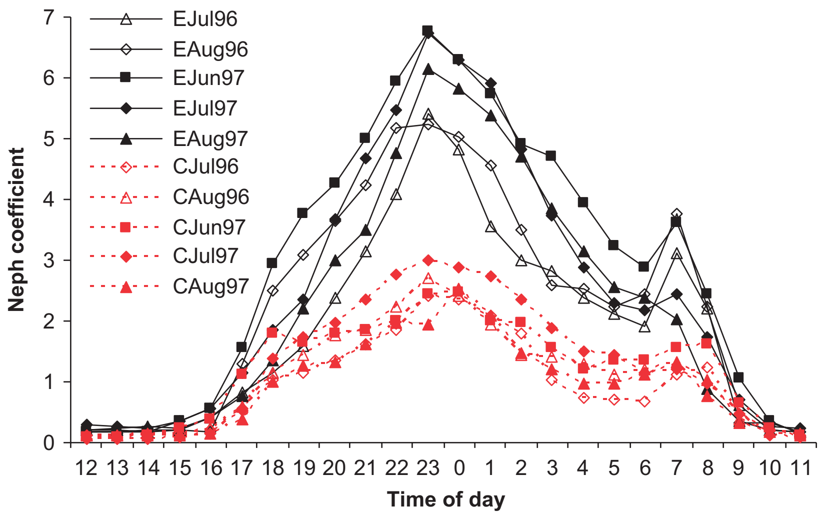

Fig. 3. Mean pollution (nephelometer, bsp $(10\,\text{km})^{-1}$) by month and time of day for the winter months of June–August in the residential area of East Armidale (EJul96, EAug96, EJun97, EJul97, EAug97) and the central creeklands location (CJul96, CAug96, CJun97, CJul97, CAug97).

from 4 pm to 8 am the following day. At Rozelle, near the CBD, 67% of all carbon was modern, i.e. from wood, not coal, diesel or petrol. In the Blue Mountains, the proportion was 81% (NSW EPA, 1996).

An interesting pattern is seen by averaging Armidale's measurements by month and time of day (Fig. 2). Pollution is very low around midday, when there are no inversions to trap the air in the valley. Woodheater use may also be lower at this time of day, the surveys showing that only 47% of woodheaters were used 24-h a day. There is a steep rise from early evening onwards, peaking around midnight. A small peak is also seen in the morning at about 7 am, when fires are refuelled or relit, followed by a decline as people travel to work.

Fig. 3 compares monthly averages by time of day in the winter months of June–August for the residential area of East Armidale and the central creeklands location. Although the two sites are only about 1.4 km apart, daily average pollution in E Armidale was more than double that at the creeklands (Table 2). However, the temporal distribution was almost identical for both sites, pollution peaking late in the evening, then declining overnight until the small peak at about 7 am.

3.2. Air pollution mapping

Weather on the mapping nights was generally comparable to conditions for the winter as a whole (Table 3). The three nights in July had generally lower winds and higher pollution levels than was typical for the month. Those in August had marginally higher pollution than the month as a whole (24 h average of 1.3 vs. 1.11 for the creeklands nephelometer), but the three nights in September were less polluted than the average for September (Table 3).

Fig. 4 shows a contour map of Armidale with the location of built-up areas and average pollution readings for all transects over the 14 nights. The correlation between means for the 10 nights with no missing data and 14-night means was 0.99, so only the latter are shown. Lowest readings were for the undeveloped edges of the city, e.g. the area northwards up the brow of the hill on transect 3 (a lookout and parkland). The northern end of transect 2, and the northern and southern ends of transect 1 were also surrounded by open fields and

Table 2
Average 24-h pollution levels at the creeklands (C) and E Armidale nephelometers (bsp/10 km) and average daily ratios by month

Month	Mean (bsp/10 km)		Ratio means		Mean ratio[c]	Month	Mean (bsp/10 km)		Ratio means		Mean ratio[c]
	C	E Arm	All[a]	Same D[b]			C	E Arm	All[a]	Same D[b]	
Jul 96	1.05	1.92	1.83	2.01	2.59	Aug 96	1.11	2.38	2.14	2.16	2.86
Jun 97	1.24	2.97	2.40	2.97	2.31	Jul 97	1.29	2.55	1.98	2.22	1.94
Aug 97	0.96	2.26	2.35	2.65	2.18	Sep 97	0.47	0.82	1.74	1.97	1.60

[a]Mean for E Armidale divided by the creeklands mean for the same month.
[b]Ratio of means for the 24, 28, 22, 31, 30 and 18 days in Jul, Aug 96, June, July, August, and September 97 with data for both locations.
[c]Mean value of (E/C) each month, where E and C are the 24-h means for East Armidale and the creeklands, respectively.

Table 3
Comparison of pollution levels and weather data on the 14 mapping nights with monthly averages

Month	Ave. daily pollution levels (bsp/10 km)					Weather data—airport			Weather data—CBD	
	No. of nights	Creeklands		East Armidale		Wind (knots) 9 pm–3 am	Temp. (°C)		Temp. (°C)	
		Mean	1-h max	Mean	1-h max		Min.	Max.	Min.	Max.
Means for all nights by month										
July	31	1.05	3.61	1.92	8.55	8.0	2.3	10.4	0.0	11.5
August	31	1.11	3.46	2.38	8.72	7.1	3.5	13.2	−0.1	14.4
Sept	30	0.66	2.02	0.94	3.52	9.9	6.2	15.5	3.3	17.1
Mean	91	0.94	3.04	1.99	7.86	8.3	3.9	13.0	1.1	14.3
Means over 14 mapping nights by month										
July	3	1.64	6.77	3.31	9.18	6.1	2.3	10.0	−2.0	13.2
August	8	1.30	3.81	3.21	10.54	6.3	2.2	12.2	−1.9	14.0
Sept	3	0.42	1.58	0.82	3.19	12.4	6.3	14.7	4.4	16.3
Mean	14	1.19	3.97	2.72	8.67	7.6	2.6	12.3	−1.3	13.4

The airport, 1084 m above sea level, is situated on a plateau about 5 km away from the creeklands monitor (situated on the valley floor, 967 m above sea level). The lower minimum temperatures at the CBD (compared to the airport) indicate the presence of nighttime temperature inversions; the CBD also has lower overnight wind speeds.

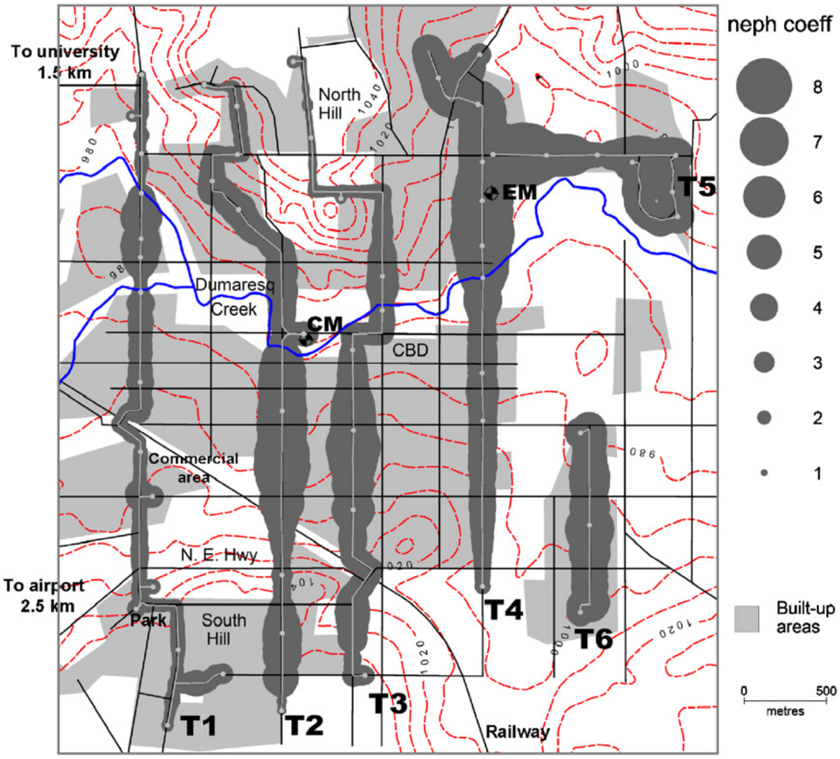

Fig. 4. Contour map of Armidale, together with average pollution readings over the 14 nights (shown by width of grey shading) and stopping points (white dots) for transects 1–6 (T1–T6). CM and EM indicate locations of fixed monitors on the creeklands and in East Armidale; the railway and roads are shown as black lines.

vacant blocks, so relatively unpolluted. The commercial/industrial area south of the railway line on the west-most transect (T1) also has few residential houses and relatively low pollution levels.

Fig. 5 illustrates the suddenness of transitions for the northern section of transect 1 between the two creeks shown on the map (Fig. 4). There are only a few houses north of first creek, so the area is generally unpolluted. At the creek, average bsp was 1.5 (about 35 $\mu g\,m^{-3}$ PM2.5), but over the next 41 m the average for the 14 nights rose to 4 (90 $\mu g\,m^{-3}$ PM2.5). Average pollution remained between 4 and 6 for the entire residential section, falling to just under 2 as houses give way to the floodplain of the southmost creek. The three peaks in the graph probably correspond to individual chimneys or groups of chimneys that have a noticeable effect on pollution of the surrounding area.

Transect 2 (Figs. 6 and 4) also shows that smoke pollution does not vary uniformly with height above the valley but is influenced by land use. The lowest pollution reading (stopping point SP1) corresponds to vacant land on the side of the hill. The area between SP2 and SP3 had below-average pollution; some of the land is used for non-residential purposes so there are fewer emissions and smoke may also drain downhill. Average pollution levels increase through the steeply sloping area to SP5. The area above and east of SP4 and SP5 is a lookout and park (no houses), so easterly winds will reduce the amount of smoke in this area. Pollution remains relatively constant for the next 700 m of residential area, falling away somewhat as houses give way to the creeklands floodplain (SP6), south of which is the CBD. Smoke pollution is substantially higher immediately south of the CBD in the gently sloping residential area (surrounding SP7). This corresponds to an older style residential area of weatherboard (timber-clad) houses. Pollution then falls away, as the land starts to rise more steeply up to the railway and New England Highway, reaching a minimum somewhat below the level at the creeklands nephelometer. Though the slope of the land is virtually unchanged, pollution starts to rise immediately after the highway, peaking at a slight

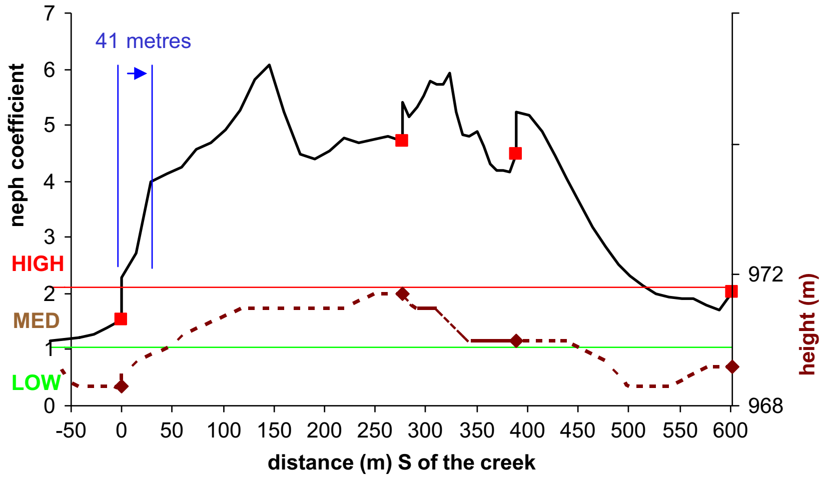

Fig. 5. Air pollution readings, transect 1 (solid line, squares indicate stopping points) by distance south of the creek together with height above sea level (dashed line/diamonds). Horizontal lines represent the NSW EPA air pollution index (NSW EPA, 2003). If particles are the worst pollutant, readings less than 1.05 are LOW, 1.05–2.1 are MEDIUM, and readings >2.1 are HIGH. Values are not strictly comparable, because the EPA categories are calibrated to STP and refer to the worst 1-h of the day, whereas values in Fig. 5 are 14-night averages for heated air at ambient pressure (mean 907 hPa) at the time pollution was measured. On some nights, pollution for the worst hour would have been substantially higher than the 14-night averages.

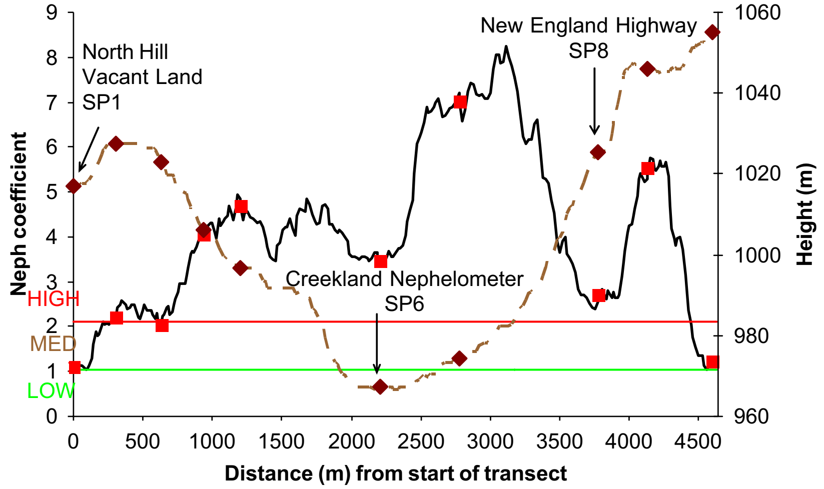

Fig. 6. Mean nephelometer coefficients (solid line, squares indicate stopping points) and height above sea level (dashed line, diamonds) for all locations along transect 2. Horizontal lines depict the NSW EPA air pollution categories (LOW, MEDIUM, HIGH), based on the maximum 1-h average (see Fig. 5 for details).

hollow where the land starts to flatten off on the south side of the valley.

The low pollution reading at New England Highway is interesting. The railway crossing is approximately 250 m north of the highway. Although there are some houses in this area, population density is less than the main residential area immediately south of the CBD. Thus, although there may be some tendency for smoke to drain away from this area, the main reason for the relatively low pollution is probably lack of emissions in the area. In contrast, the area immediately south of the highway is a popular residential area with high emissions and consequently high pollution levels.

Observation of smoke plumes suggest the inversions may become "layered", with smoke from each residential area generally trapped in the layer closest to the ground, only a few metres from chimney height. This may explain the relative lack of pollution in the creek bottoms, which might trap a layer of even colder air that mixes only slowly with the more polluted residential air. Wind measurements at a location next to the creeklands nephelometer were available for a number of days from June to August 1997. In contrast to the relatively high winds at the airport, night-time winds averaged only $0.15\,m\,s^{-1}$ at the rooftop inlet of the nephelometer. Thus, even if the wind direction remains constant, smoke at rooftop level would travel only 540 m in an hour, probably lesser at the height (approximately 1.5 m above ground) that pollution was measured by the mobile nephelometer. In the absence of wind, pollution builds up where it is emitted, so areas with many chimneys generally have pollution levels 3 or 4 times the general background woodsmoke pollution.

The magnitude of the buildup of pollution may be more readily understood from the data for individual nights. For the vacant land at the start of transect 2, the average over all 14 nights was 1.1 (Fig. 6). However, pollution on one night was 4.4, higher than the CBD reading of 4.1, presumably because of a local source and conditions that encouraged smoke to drain or drift into the area on that particular night. One the same night, further along the transect, nephelometer coefficients in excess of 20 were recorded for approximately 0.25 km between SPs 7 and 8. A single wood heater can substantially increase local pollution, as demonstrated by short-term measurements of $4000\,\mu g\,m^{-3}$ on the balcony of a house exposed to smoke from a neighbour's heater (Todd, 2005).

3.3. Estimated population exposure

The mapping data, together with daily average pollution measurements at the two fixed locations (Fig. 3), shows that many places, especially older residential areas, have consistently higher pollution than the creeklands nephelometer.

Averages for the 14 nights at the SPs next to the nephelometer/TEOM in East Armidale and at the creeklands nephelometer were 8.58 and 3.35, respectively. Mean pollution on the mapping nights at East Armidale was therefore 2.56 times that at the creeklands. This value is similar to the mean pollution ratio of 2.59 for July 1996 and 2.86 in August 1996 (Table 2). Thus average pollution on the 14 nights is indicative of the relative amounts of pollution at each location.

Mean pollution at all locations on all transects, except the industrial area on transect 1 and outlying areas with no houses, was 4.08, 22% higher than that in the creeklands, compared to the 156% noted above for East Armidale. It was assumed that the same proportions would apply to monthly averages. Thus, although pollution at two locations varies with season, and the need to use heating, the city-wide average (CWA) will equal the creeklands average plus $22/156 = 0.141$ of the difference between East Armidale and the creeklands, i.e. $\mathrm{CWA} = C + 0.141(E - C) = C + 0.141(R - 1)C$ where C and E represent mean monthly pollution at the creeklands and East Armidale, respectively, and $R = E/C$.

Results are given in Table 4, for the observed ratio of monthly means, and after smoothing R by using averages for the three coldest months of June–August. Both methods lead to an estimated population exposure of 1.01–1.02 neph coefficients for April–September with about 0.25 in October, compared to an expected background of about 0.05 in the absence of bushfires and burnoffs, i.e. an estimated population exposure of $11.5\,\mu g\,m^{-3}$ PM2.5 pollution from woodheating.

Table 4
Average pollution at the creeklands (means for 1995–1997), ratio (R) of means for E Armidale and the creeklands, and predicted averages for the town as a whole (CWA, CWA1)

Month	Creeklands ave. C	Ratio of means R	City-wide average CWA	Smoothed ratio SR	City-wide average CWA1
Apr	0.48	(1.6)	0.52	1.6	0.52
May	0.79	(2.0)	0.90	2.0	0.90
June	1.02	2.97	1.30	2.5	1.24
July	1.24	2.14	1.44	2.5	1.50
Aug	1.05	2.40	1.26	2.5	1.27
Sep	0.59	1.80	0.66	1.8	0.66
Mean	0.86		1.01		1.02

R = mean(East Armidale)/mean(creeklands), where the means include all days in 1996–1997 with measurements at both locations (see Table 3). Values in brackets are estimates. $\mathrm{CWA} = C + 0.141(R-1)C$ where C is the creeklands average and R is the ratio of means for East Armidale to the creeklands (see text). SR = R with values for June–August replaced by the mean for all three months. CWA1 calculated as CWA, but with SR instead of R.

3.4. Estimated health consequences

Kunzli et al. (2000) stressed the importance of estimating the costs of air pollution. Without economic signals, important resources such as clean air are wasted. Several teams of researchers have now used the methodology pioneered by Kunzli et al. (2000) to estimate the health effects of air pollution (e.g. DEC NSW, 2005; Fisher et al., 2005; WHO, 2006; Wang and Mauzerall, 2006). The latest studies (WHO, 2006; Wang and Mauzerall, 2006) used exposure–response relationships (ERR) for PM2.5 from a cohort of half a million adults with about 120,000 deaths. Pollution measurements included PM10, TSP, NO_2, CO, O_3, SO_2, sulphates (a subset of PM.25), and PM2.5; only the last three were significantly associated with mortality. Total mortality increased by 6% (cardiopulmonary mortality 9%, lung cancer mortality 14%) for each increase of $10\,\mu g\,m^{-3}$ in PM2.5 pollution (calculated as averages of 1979–1983 and 1999–2000 measurements, see Pope et al., 2002).

Based on the above ERR, Armidale's average population exposure of $11.5\,\mu g\,m^{-3}$ PM2.5 pollution equates to increased overall mortality of 7% (10% in cardiopulmonary mortality, 16% in lung cancer mortality). As with all estimates of pollution-related health effects, these values should be treated with caution. The true effects could be lower. On the other hand, the results of Jerrett et al. (2005) using more accurate estimates of PM2.5 exposure including within-city variation, suggest that the effects of PM2.5 could be nearly 3 times greater than estimates (including those reported above for Armidale) derived using ERR from comparisons between communities.

4. Discussion

4.1. Health effects

Although there is still a degree of uncertainty about the health effects of air pollution, it is useful to have some estimate of the costs, in order to compare and evaluate potential control measures. This is analogous to policies on tobacco control, which were guided by estimates of the number of deaths due to tobacco smoking (in Armidale, 19.5% of all deaths in the 1980s, Cancer Council, 1989). In Europe, estimates of the cost of traffic pollution (Kunzli et al., 2000) were followed by initiatives such as low-emission zones, lower taxes for low-emission vehicles, and government subsidies to install oxidation catalysts or particle traps in diesel vehicles (Energy Saving Trust, 2003).

For Armidale's death rate of about 177 per year (Khan, 2002) the estimated 7% increase in mortality due to woodheating equates to 11.5 premature deaths per year. In Australia, the cost of a premature death from air pollution was estimated at AU$1.3 million (US$0.97 million, BTRE, 2005). So 11.5 premature deaths represent an annual cost of AU$14.95 million, or $4270 per heater per year. Although this estimate is simplistic and ignores the substantial cost of morbidity (Hurley et al., 2005), estimates using the Clean Air for Europe methodology to value life-years lost (LYL, with 1 premature death equal to about 10 LYL, see Rabl, 2003) are similar. Median and mean values for a year of life are respectively 52,000 and 120,000 euros (Hurley et al., 2005), i.e. 0.52–1.2 million euros (AU$0.87–2.0 million) for a loss of 10 years.

The first study to identify long-term relationships between annual PM2.5 pollution and mortality (the Six Cities Study, Dockery et al., 1993) also provided evidence of cause and effect. A follow-up study was conducted when PM2.5 had dropped substantially in one city, moderately in another, remaining stable elsewhere. Death rates fell in the first two cities relative to the other four (Schwartz, 2002).

When PM2.5 pollution was reduced in Dublin by banning non-smokeless coal in September 1990, there were 15.5% fewer respiratory and 10.3% fewer cardiovascular deaths in the 6 years after the ban, compared with the previous 6 years (i.e. 116 fewer respiratory and 243 fewer cardiovascular deaths per year, Clancy et al., 2002). This reduction, sustained over 6 years, represents a substantial benefit far greater than the cost of switching to less polluting heating.

A review of nine studies where woodsmoke was a major source of particles reported that relative risks for ambient particulate matter and adverse health effects were stronger than general estimates from the literature. The authors concluded: "*there seems to be no reason to assume that the effects of particulate matter in areas polluted by wood smoke are weaker than elsewhere*" (Boman et al., 2003). Researchers in Spokane, Washingon, apportioned PM2.5 into its different sources. Emergency department visits for respiratory complaints were significantly associated with PM2.5 from woodsmoke/vegetative burning, but not motor vehicles, airborne soil or all PM2.5 (Schreuder et al., 2006).

Woodsmoke is the main source of air pollution in some cities, e.g. Christchurch (population 333,000), NZ. A 157-page report on the health effects of air pollution in Christchurch (Fisher et al., 2005) estimated that emissions from domestic solid-fuel heaters cause 124 premature deaths per year and cost the community NZ$127 million (US$89 million per year). Other sources of Christchurch's pollution were industrial (18 premature deaths, total cost NZ$22 million), diesel vehicles (15.6 deaths, $18.5 million) and petrol vehicles (0.4 deaths, $0.5 million, see Fisher et al., 2005).

The adverse health effects of PM2.5 pollution have no known threshold, below which adverse health effects are not observed. Consequently, in 2000, the WHO chose not to set guidelines for fine particle pollution (WHO, 2000). In Australia, the NEPC set advisory standards for PM2.5 but acknowledged that even when the standards are met, PM2.5 will cause 1000 premature deaths per year (NEPC, 2003).

Rather than achieving specific standards, it would therefore be advantageous for air quality strategies to focus on costs relative to benefits. Christchurch's domestic pollution is caused by 8570 open fires and 38,184 wood or multi-fuel burners; average fuel consumption is about 15 kg per day for both enclosed heaters and open fires (Scott, 2005a). A small study of in-home measurements found no relationship between in-home emissions and ratings by the Australian/New Zealand standards test; AS4013 heaters rated $<1.5\,\mathrm{g\,kg^{-1}}$ fuel had real-life emissions averaging $15.5\,\mathrm{g\,kg^{-1}}$ (median $13.0\,\mathrm{g\,kg^{-1}}$, Scott, 2005b), compared to an estimate of $9\,\mathrm{g\,kg^{-1}}$ for open fires (Scott, 2005a). This suggests that open fires have similar or lower emissions to new and old heaters. Estimated health cost per solid fuel heater or open fire equate to about NZ$2700 heater^{-1} year^{-1}. Christchurch aims to replace at least 29,600 heaters or open fires with non-polluting heating by 2013.

Wood consumption in Armidale averages about 4 tonnes per heater per year. Emissions of $13.0\,\mathrm{g\,kg^{-1}}$ fuel correspond to 52 kg of PM2.5 per year. For comparison, a typical petrol-fuelled passenger car emits about 0.15 kg per year (15,000 km at $0.01\,\mathrm{g\,kg^{-1}}$). Air toxics, e.g. polycyclic aromatic hydrocarbons (PAH), are another concern. Even in Sydney, where about 13% of households use woodheating, PAH concentrations averaged $4.47\,\mathrm{ng\,m^{-3}}$ in winter, seven times the summer average of $0.62\,\mathrm{ng\,m^{-3}}$ (NSW EPA, 2002). Despite these measurements, a common perception is that cars are worse for the environment than woodheaters.

The benefits of wood heating depend mainly on intangibles such as ambience, though this is offset by the effort required to chop wood, dispose of ash and light the heater. In Armidale, supplies of local firewood are becoming scarce and environmentalists are concerned that harvesting fallen or mature trees in old-growth forests deprives wildlife, including threatened species, of hollow logs for homes (Environment Australia, 2004). Nowadays, non-polluting alternatives such as reverse cycle heating have lower purchase and running costs. Innovations such as inexpensive rooftop solar heaters (essentially an airspace sandwiched between a black absorber and a clear polycarbonate cover plus a small fan to transport heated air to living areas) can considerably reduce the need for heating (Solar Project, 2001) and hence greenhouse gas emissions.

Thus the benefits (apart from intangibles) of woodheating range from zero, to the cost of alternative heating for households that currently collect their own firewood—a few hundred dollars per year. Estimated health costs are considerably higher. Yet, there is currently little public awareness of the problem. A survey in 1999 found that only 53% of all residents and 48% of woodheater users in Armidale considered woodsmoke to be harmful (Khan, 2002). Earlier surveys in 1996 and 1995 found that 60% and 55%, respectively, of Armidale residents thought that woodsmoke was harmful. Although $46,000 was spent on education, mainly in Armidale after the 1995 survey, programmes tended to focus on how to operate heaters correctly, not why this is important. The failure to emphasise health effects (unlike campaigns against tobacco) may have contributed to the perception that woodsmoke is not harmful. Perhaps the best way to counter this misunderstanding would be to explain that woodsmoke contains many of the same and similar chemicals to tobacco smoke and that it is linked to the same diseases—heart and respiratory diseases and lung cancer. In babies, particulate pollution, like tobacco smoke, is also linked to increased risk of cot deaths.

In Australia, pollution-reduction strategies, like all government spending, have limited funding. A user-pays policy is therefore needed, e.g. a small annual levy on woodheater use (say, $100 per year, and a nominal $10 for low-income families).

This could provide subsidies to replace heaters with non-polluting alternatives, incentives to develop a marketable rooftop solar heater, fund education programmes on the health effects of woodsmoke and also act as an incentive for owners to operate heaters correctly. Careless operation can increase emissions more than 10-fold (Todd, 2002). A licence to pollute, which could be withdrawn if heaters continue to be used incorrectly, would serve as a reminder that woodsmoke is hazardous and that owners have a duty to minimise emissions.

4.2. Spatial variability

The large variation in smoke concentrations demonstrates that ambient air quality standards cannot protect people living in the worst areas. The most important determinant of annual exposure is the quantity of *local* emissions. Similar results were found in Christchurch; average annual PM10 concentrations varied from less than $1\,\mu g\,m^{-3}$ to more than $20\,\mu g\,m^{-3}$ (Fisher et al., 2005). The equipment used in the Armidale study provides a simple, convenient way to map emissions over a wide area.

Such information could be very useful for people who are looking to buy or rent housing, especially if they suffer asthma or other respiratory diseases. The survey in 1999 (Khan, 2002) showed that 41% of people in Armidale suffered a respiratory illness during the previous winter. Of these, 90% thought that woodsmoke was harmful. One lady responding to a radio discussion said her family lived in central Armidale but were forced to move out of Armidale every winter for health reasons (ABC, 2005). Patients suffering respiratory diseases have been told by their doctors to move out of Launceston (another woodsmoke-affected city) in winter for similar reasons (Markos, 2004). Information on spatial variability of pollution may help people choose the best place to live, and so minimise these problems.

Average measurements of indoor fine particle concentrations in Launceston were more than $30\,\mu g\,m^{-3}$ higher in winter than summer (NHT, 2004). The spatial variability observed in this study, and elsewhere, suggests that exposure in the worst areas is likely to exceed $60\,\mu g\,m^{-3}$. This is substantially higher than PM2.5 exposure from living with a smoker (about $30\,\mu g\,m^{-3}$, Neas et al., 1994). Increasing public awareness of the considerable spatial variation in woodsmoke pollution levels, and the similar nature of wood and tobacco smoke, may encourage people to reduce their emissions and so reduce health impacts, as well as enable people with respiratory problems to choose to live in less-polluted areas.

5. Conclusions

Considerable variability was observed in winter woodsmoke pollution levels, from relatively low levels on the outskirts of the city to very high levels in older residential areas. In general, smoke tends to build up in the area where it is emitted, with some drainage according to local terrain and wind directions. The 4-fold increase in smoke levels within 41 m suggests that even a small number of badly operated heaters can have a large influence on local air quality.

Australia's National Environment Protection Council was set up to ensure that "all Australians enjoy the benefits of equivalent protection from air, water, soil and noise pollution" (NEPC, 2002). The substantial variability observed in this study implies that this objective cannot be achieved simply by measuring ambient air. It is also necessary to identify and target hot spots, e.g. using a portable nephelometer.

Estimated population exposure from this study suggests that woodheaters increases mortality by about 7%, or about 11.5 premature deaths per year. If the cost of a premature death is assumed to be AU$1.3 million, this equates to about $4270 per heater per year.

More effort is required to reduce woodsmoke pollution. Funding is required to help people understand that woodsmoke is harmful, to ensure all homes have adequate insulation, and that environmentally friendly alternatives such as solar heating are developed. An annual "polluter-pays" levy of, say, $100 per year, with a rebate for low-income families, could provide much-needed funds for an effective woodsmoke-reduction programme.

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