

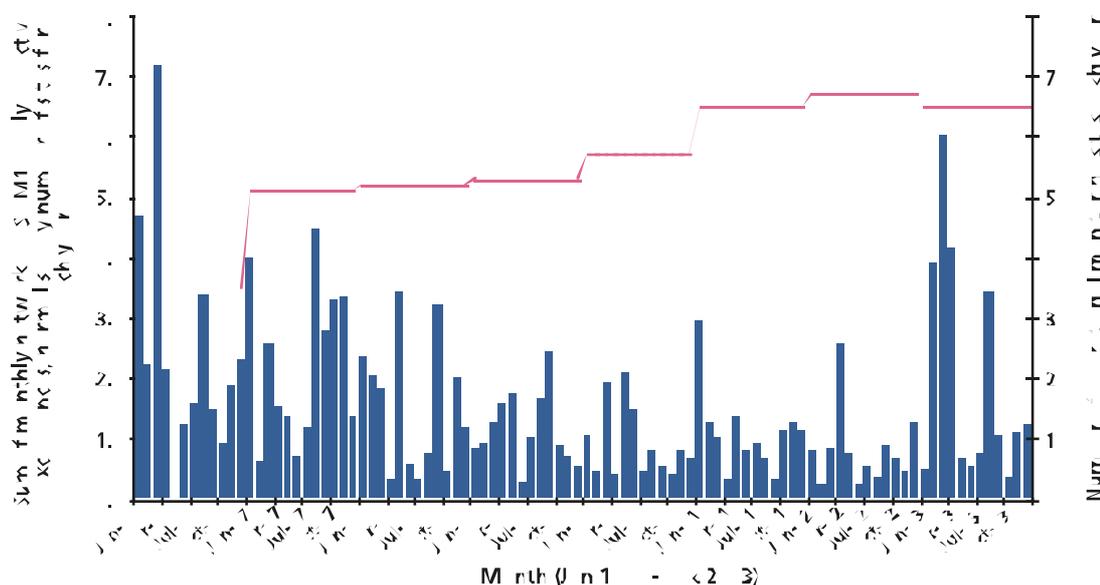
585. Chloride concentrations have also been measured at these sites. Chloride concentrations have more small-scale spatial variation than the secondary sulphate and nitrate concentrations. In particular the 12-site network does not provide enough information on the gradients in chloride concentrations close to the coast. The spatial pattern of chloride concentrations is also more variable from year to year. Chloride concentrations have, therefore, not been interpolated in Figure 6.60. In 2002 there are indications of a southwest to northeast gradient.

6.3 Episodicity of particle concentrations

6.3.1 Monthly exceedences of air quality objective concentrations

586. Figure 6.61 shows the number of exceedences of $50 \mu\text{g m}^{-3}$ (TEOM * 1.3) measured each month between January 1996 and December 2003 at national network sites. There was an increase in the number of monitoring sites over this period, so the number of sites operational during each year is also shown. The total number of measured exceedences tended to increase as the number of sites in the network increased, since an exceedence can occur at several sites on the same day. Figure 6.61 shows the number of monthly exceedences divided by the number of operational sites in each year. This should provide an estimate of the number of exceedences that is reasonably independent of the number of sites.

Figure 6.61 Number of daily PM₁₀ AQS exceedences summed across AURN by month (1996–2003).



587. Figure 6.61 shows that there is a considerable month-to-month variation in the number of exceedences. The number of exceedences has generally declined since 1996 but the number of exceedences during 2003 was unusually high. The episodes seen in Figure 6.61 had a range of causes. There were notable winter secondary PM episodes in January and March 1996 and in early 2003. The photochemical episode that happened in August 2003 is also clearly shown. Episodes were also caused by poor dispersion of primary pollutants, such as during the autumn of 1997. Long range transport dust events such as those that occurred in March 2000 and early 2003 are also shown.

Table 6.13 Annual number of AQS exceedences normalised by the number of sites in each year.

Year	Number of exceedences
1996	29
1997	27
1998	19
1999	14
2000	11
2001	13
2002	10
2003	24

588. Table 6.13 lists the annual number of exceedences per site summed from the monthly counts in Figure 6.61. There were more exceedences in 2003 than in any year since 1997.

6.3.2 Episodicity of specific components

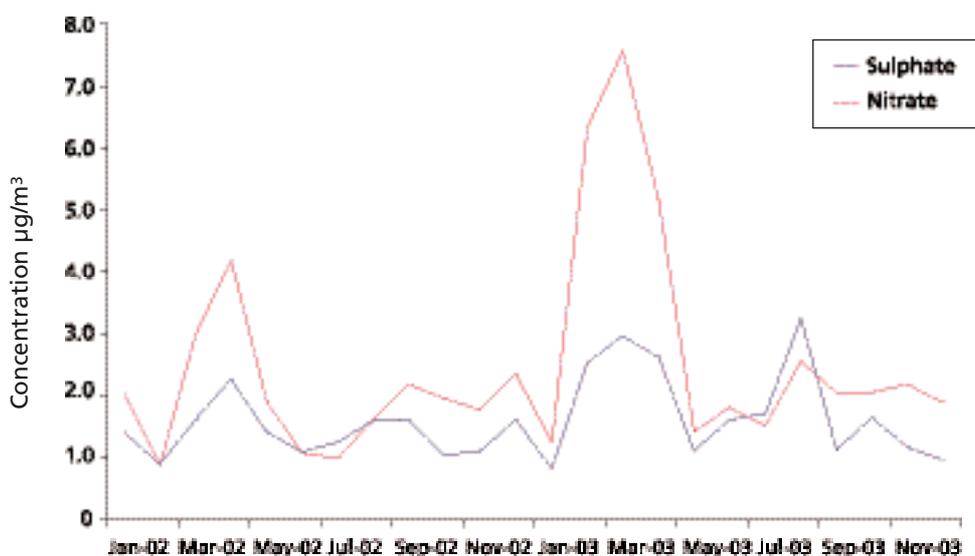
- 589.** Short-term excursions of primary emissions (such as fireworks and bonfires) influence the UK's ability to meet the current EU limit value for 24 h PM_{10} , but the episodicity of individual chemical components may have an important influence also. Thus, the frequency distributions of concentrations of PM_{10} and three of its major components are examined at Belfast and Harwell (see Table 6.14).
- 590.** Episodicity is expressed in terms of the ratio of the higher percentile values (90% and 99% in this example) to the mean concentration. Table 6.14 shows that at the 90%ile level, chloride, nitrate and sulphate all have greater episodicity than PM_{10} and, therefore, make greater contributions than some other components to high PM_{10} concentrations. The same is true at the 99%ile level (except for chloride in Belfast), but nitrate shows far greater episodicity than either chloride or sulphate: this component may make a substantial contribution to very high PM_{10} concentrations.
- 591.** Figure 6.62 indicates monthly levels of sulphate and nitrate from CEH, averaged over the network. There were notable rises in sulphate and nitrate levels during the spring seasons of both 2002 and 2003, with nitrate exhibiting the more marked increases. Levels peaked in April of 2002 and in March of 2003. This rise was particularly marked for the 2003 data. The increase in sulphate and nitrate associated with the August 2003 heatwave can also be observed in Figure 6.62. Sulphate at this time rose to exceed the monthly concentrations of that spring, whereas nitrate levels remained significantly lower than the dramatic rises seen earlier in the year.

Table 6.14 Statistical properties of 24-h air quality data for Belfast and Harwell.

Site	Period	Component	Mean	90%ile	99%ile	90%ile ÷ mean	99%ile ÷ mean
Belfast	23/10/01–30/06/03	PM ₁₀ ^a	17.5	29.0	56.7	1.66	3.24
		Chloride	1.92	3.90	6.04	2.03	3.14
		Nitrate	2.22	5.19	21.56	2.34	9.71
		Sulphate	2.33	5.02	8.54	2.15	3.66
Harwell	17/11/01–30/6/03	PM ₁₀ ^a	12.91	21.00	37.22	1.63	2.88
		Chloride	1.05	2.59	4.58	2.47	4.36
		Nitrate	3.07	7.95	21.24	2.59	6.92
		Sulphate	2.49	5.10	9.66	2.05	3.88

^aMeasured by TEOM.

Figure 6.62 Monthly concentrations of sulphate and nitrate across the network during 2002–2003.

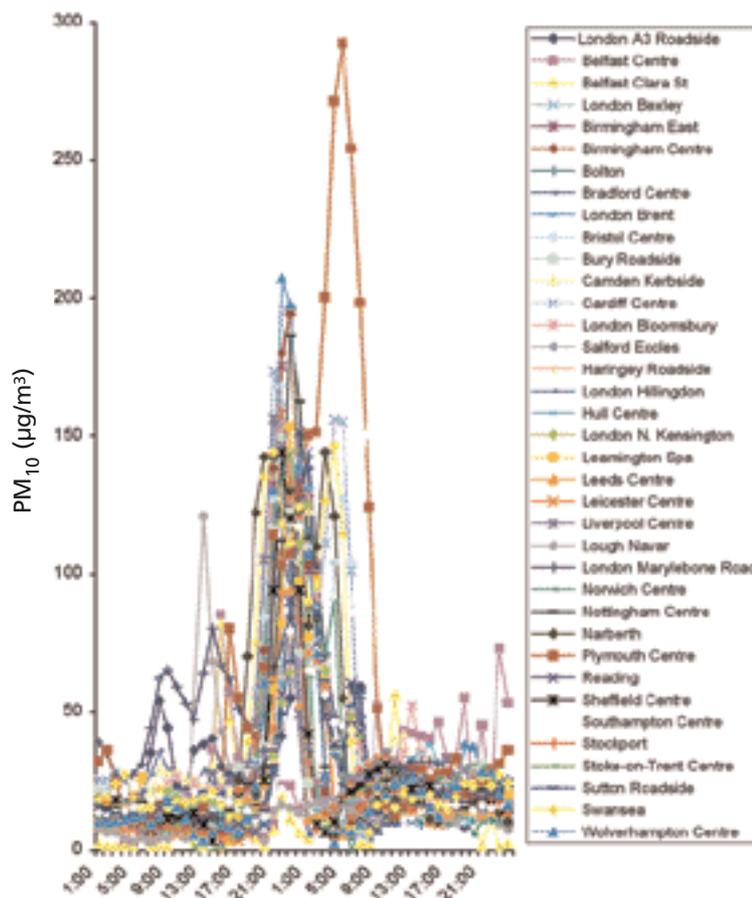


6.3.3 Saharan dust episodes

592. On a global scale, wind-blown dust is the largest natural source of particles after sea spray, with a source strength of about 2150 Tg yr⁻¹ (IPCC, 2001). Dust source regions are mainly deserts, dry lake beds and semi-arid desert fringes. The atmospheric lifetime of dusts depends on particle size: large particles are removed quickly from the atmosphere by gravitational settling, whereas sub-micron particles can have atmospheric lifetimes of weeks. Long-range transport of Saharan dusts occurs across the Mediterranean Sea into southern and central Europe (Rodriguez *et al.*, 2001) and across the tropical Atlantic Ocean into the Caribbean (Rajkumar and Siung Chang, 2000) and into both North and South America (Prospero *et al.*, 1981). Long-range transport of Saharan dust into the British Isles has been identified (Stevenson, 1969) and red dust deposits are regularly reported following rainfall in air masses originating over the Sahara.

- 593.** Figure 6.63 shows the observed values of PM_{10} at all the AURN monitoring sites in Northern Ireland, Wales and England (excluding the northeast of England) on the 2nd and 3rd March 2000. Significant peaks were observed at all sites except those in the northeast of England and Scotland, whether rural, suburban, roadside or kerbside sites. No corresponding peaks were observed in NO_x . $PM_{2.5}$ levels followed the PM_{10} levels at the London Bloomsbury and Rochester sites where peak concentrations were in the range $40\text{--}60\ \mu\text{g m}^{-3}$, about one-half of the PM_{10} concentrations observed at these sites. The highest hourly mean peak PM_{10} concentration observed was $292\ \mu\text{g m}^{-3}$ at Plymouth, the highest concentration observed at that site since measurements began there. The peak concentrations generally reduced across the British Isles from west to east, with maximum concentrations observed at eastern sites of $\sim 100\ \mu\text{g m}^{-3}$. The rise in PM_{10} concentrations from their baselines occurred some 2–3 h earlier in the west, indicating rapid west-east transport. The end of the episode occurred later in the south, resulting in a longer episode in the south and west and a shorter episode in the north and east of the British Isles.
- 594.** The Meteorological Office dispersion model NAME has been used to study the origins of the high particle concentrations observed on the 2nd and 3rd March 2000 (Ryall *et al.*, 2002). Assuming a source in Africa, the model is able to explain the timing of the PM_{10} episodes at the different sites across the British Isles with close correspondence. An eruption was reported from Mount Hekla in Iceland on 26th February 2000 and this was also followed using the NAME model. From the model results, it was clear that the observed PM_{10} peaks on 2nd and 3rd March could be clearly associated with Saharan dust rather than volcanic ash from Mount Hekla.

Figure 6.63 PM_{10} concentrations observed during 2nd and 3rd March 2000.

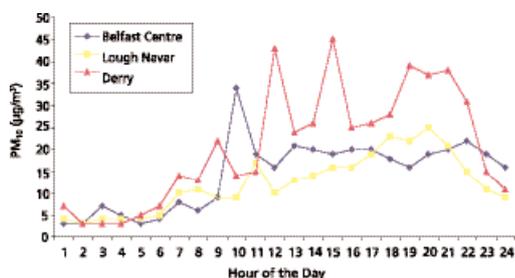


595. Ryall *et al.* (2002) considered the frequency of such dust episodes in the British Isles and predicted the occurrence of just one or two episodes per year with PM₁₀ concentrations exceeding 50 $\mu\text{g m}^{-3}$. In contrast, Rodriguez *et al.* (2001) identified up to 23 events resulting in daily mean concentrations exceeding 50 $\mu\text{g m}^{-3}$ in eastern and southern Spain.

6.3.4 Sea salt aerosol

596. On a global scale, sea spray is the largest natural source of particles with a source strength of 3,300 Tg yr⁻¹ (IPCC, 2001). Sea salt particles cover a wide diameter size range of 0.05–10 μm ; the majority fall into the 1–16 μm size range. They are generated by various physical processes, especially the bursting of entrained air bubbles during whitecap formation, resulting in a strong dependence on wind speed (IPCC, 2001).
597. Because of the long exposed Atlantic Ocean coastline, sea spray is an important source of coarse particles in the British Isles. Figure 6.64 shows the PM₁₀ concentrations observed at three locations in Northern Ireland during 4th May 1998. Concentrations reached 40 $\mu\text{g m}^{-3}$ at the Derry site, which was the site most exposed to the strong winds. Lough Navar is an inland site and the peak concentrations were somewhat lower. The levels observed at the Belfast centre site were about one-half of those observed at the coastal site.

Figure 6.64 Hourly PM₁₀ measurements at three sites in Northern Ireland during 4th May 1998.



598. In most years, gales are more generally observed over the British Isles during autumn and winter, and the high winds that caused the sea salt episode shown in Figure 6.64 are a relatively uncommon during May. Figure 6.65 shows the PM₁₀ concentrations observed during the period 25th to 26th November 2000. Early winter gales are a more common occurrence and this example recorded a gust of 78 knots at the Mumbles near Swansea. The peak PM₁₀ concentrations reached 50–70 $\mu\text{g m}^{-3}$ at exposed sites in Wales and Southwest England. Elevated levels were also observed at the Port Talbot site, and the sea salt episode was clearly detectable despite the strong local particle emissions from industrial activities in the local area.
599. Severe and strong gales are reported with a frequency of between one- and about five-times during each year and they are likely to bring sea salt episodes to exposed coastal locations with a similar frequency. It is important to determine how far inland these episodes penetrate and whether they have a wider impact on air quality than in the coastal fringe. Figure 6.66 shows the PM₁₀ concentrations recorded at a number of sites in the southern part of the British Isles during 10th and 11th December 2000. A depression travelled

in a northeasterly direction across Ireland and Scotland, with severe gales on its southern flank. The wind gusted at up to 80 knots at the Mumbles near Swansea and at Cranwell in Lincolnshire. Peak PM_{10} concentrations reached $50\text{--}60\ \mu\text{g m}^{-3}$ at the more exposed sites, such as Blackpool and Swansea. However, there were a significant number of urban sites in the path of the gales that reached $30\text{--}40\ \mu\text{g m}^{-3}$ during this episode that could not be classified as especially exposed. These included sites in Birmingham, Reading and London. Indeed, there is evidence that the PM_{10} concentrations observed at the London Marylebone Road, Camden and Haringey roadside sites were also influenced by the sea salt episode.

Figure 6.65 Hourly mean PM_{10} concentrations observed at five sites during 25th and 26th November 2000.

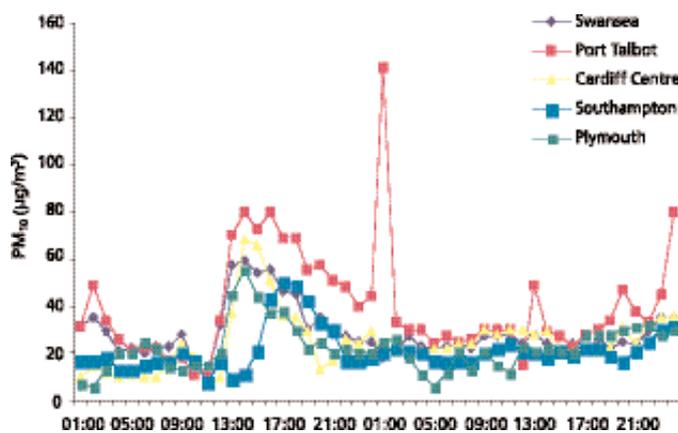
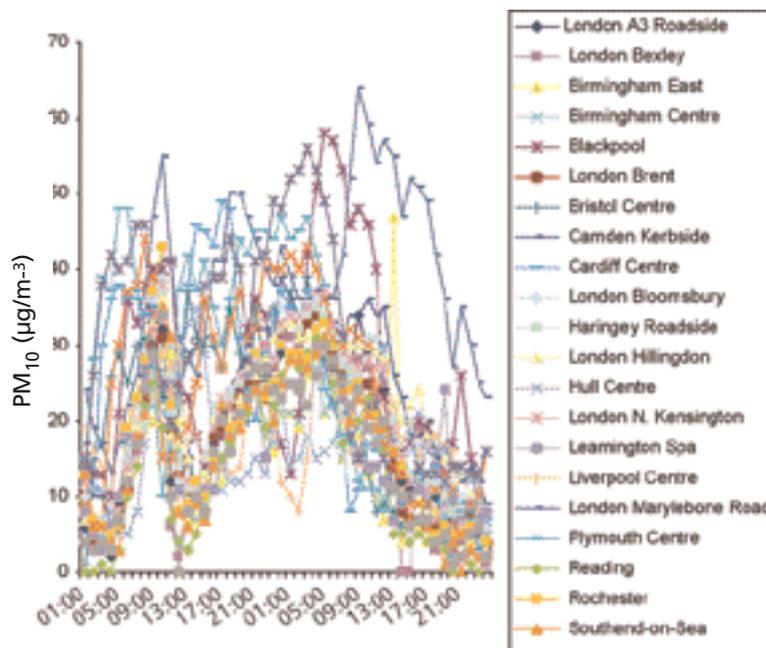


Figure 6.66 Hourly mean PM_{10} concentrations observed during 10th and 11th December 2000.

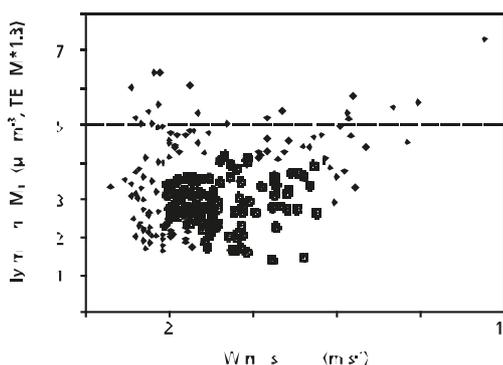


600. It is concluded that there are natural episodes of coarse sea salt particles in the UK and they occur with a frequency of between one- and five-times each year. They generally occur during the autumn and winter months and

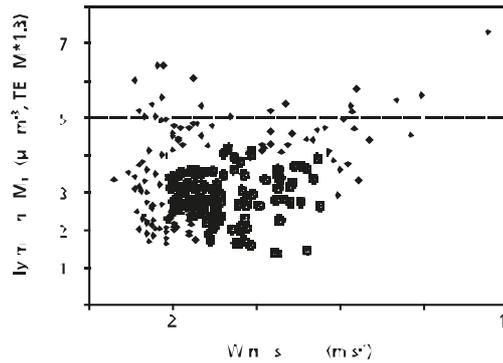
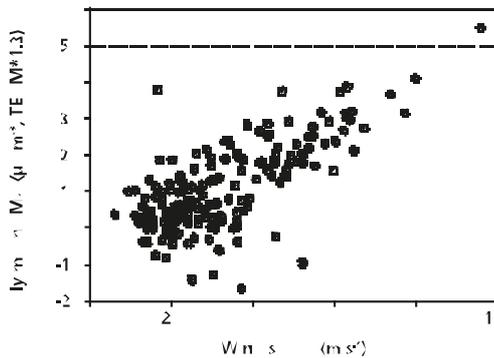
strongly influence the most exposed coastal sites, bringing peak hourly PM_{10} concentrations to in excess of $40 \mu g m^{-3}$. Urban background sites in the paths of severe gales, far removed from the coastline, will experience hourly peak PM_{10} concentrations that are about one-half of the coastal values.

- 601.** An analysis of PM_{10} data from a monitor sited on the coastal fringe of Sussex has also highlighted distinct sea salt episodes. Elevated concentrations of PM_{10} have been measured at the Lewes monitoring site during high wind speeds that result in daily mean concentrations above $50 \mu g m^{-3}$ (all data reported as TEOM * 1.3). During the eight months from September 2002 to May 2003, five such daily means were measured. The Lewes site is located in a roadside location on top of cliffs at Peacehaven in East Sussex, around 250 m from the sea. Figure 6.67 shows the relationship between daily mean PM_{10} and wind speed measured at the coast. Two groups of daily means above $50 \mu g m^{-3}$ can be seen. One group of means occur at low wind speeds and is due to primary and secondary PM_{10} . A second group of daily means above $50 \mu g m^{-3}$ occurs at higher wind speeds (and, therefore, rougher sea conditions) with an apparent wind speed dependency. The PM_{10} wind speed dependency at the Peacehaven site contrasts with the measurements from inland sites, such as the Horsham roadside site, shown in Figure 6.68, which is located ~30 km from the sea. At Horsham, daily mean concentrations above $50 \mu g m^{-3}$ are confined to low winds speeds, with PM_{10} concentration being inversely proportional to wind speed.
- 602.** Source apportionment of daily mean PM_{10} at Peacehaven has been undertaken using measurements of NO_x at nearby Brighton and Hove roadside sites to provide an estimate of local primary NO_x at Peacehaven to eliminate this source from the measurements shown in Figure 6.67. Local primary PM_{10} has an inverse relationship to wind speed and makes a negligible contribution to PM_{10} concentrations during high wind speeds.

Figure 6.67 Daily mean PM_{10} at Lewes (Peacehaven) as a function of wind speed.



- 603.** The PM_{10} coastal increment (Lewes – Horsham) is shown in Figure 6.69 and exhibits strong wind speed dependence. The coastal increment reached a maximum of $>50 \mu g m^{-3}$ at the highest wind speeds. Several negative artefacts in Figure 6.69 suggest that the analysis may not be accurate for these low wind speed conditions. This is probably due to the distance between the monitoring sites and the greater prevalence of inversion conditions at the Horsham site, which is located in the Sussex Weald. Further measurements made during winter 2003–2004 will enable a better characterisation of this coastal effect, including comparison of TEOM and 'gravimetric' measurements.

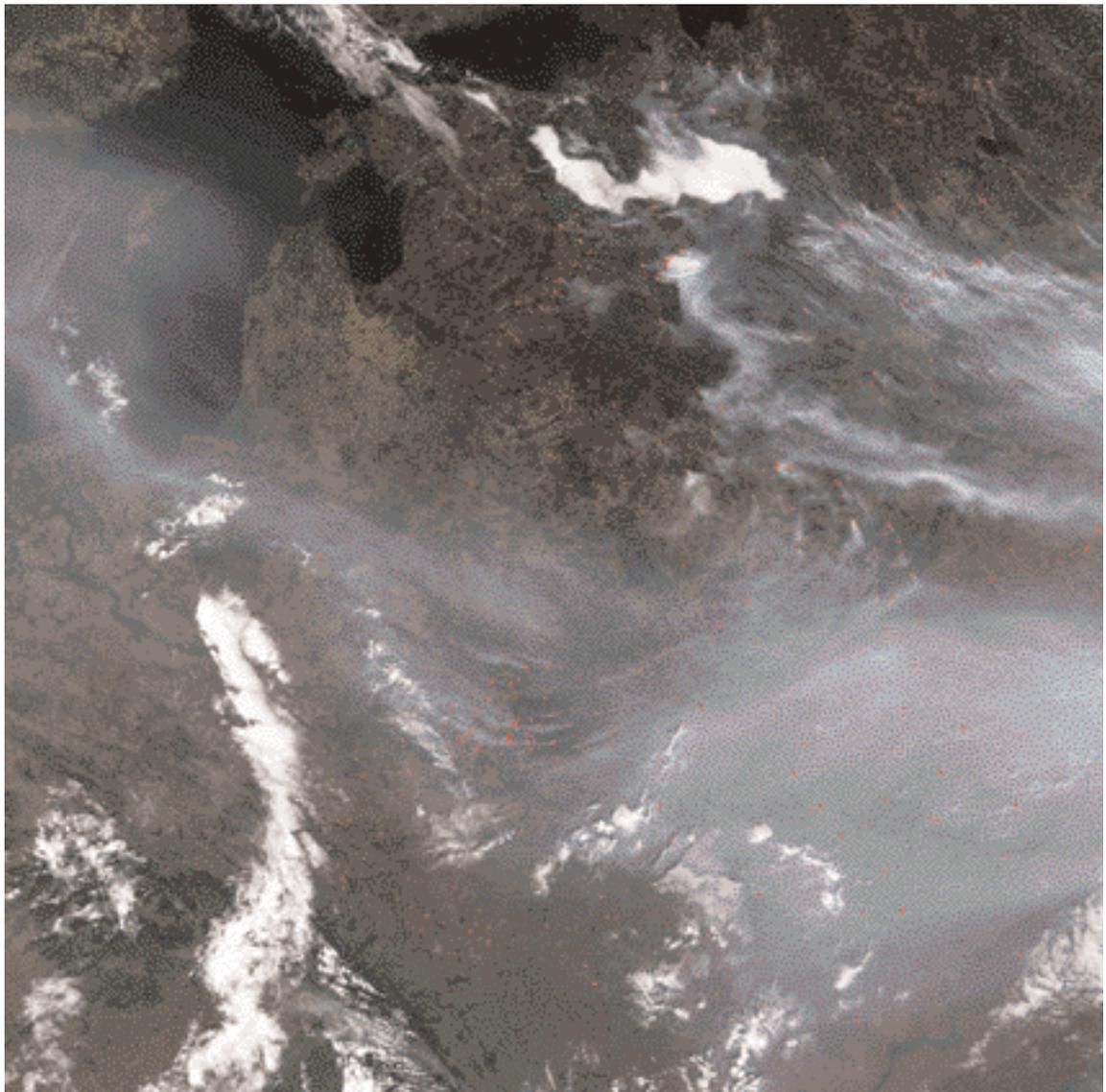
Figure 6.68 Daily mean PM_{10} at Horsham as a function of wind speed.**Figure 6.69** PM_{10} daily mean coastal increment as a function of wind speed.

6.3.5 Biomass-burning fires as a source of particles

- 604.** Towards the end of August 2002, reports were received of extensive forest fires in Western Russia and consequential poor air quality in Moscow due to smoke. Around the end of August/early September elevated PM_{10} levels were reported in Helsinki where levels reached close to $100 \mu g m^{-3}$. Figure 6.70 presents a satellite image from the Moderate Resolution Imaging Spectroradiometer (MODIS) rapid response team at NASA-GSFC: <http://earthobservatory.nasa.gov/NaturalHazards>, which clearly shows the wide extent of the forest fires on 4th September 2002 within the region 28° to 40° E, 52° to 60° N.
- 605.** The smoke-laden air masses were transported in a westerly direction and crossed the North Sea during the 11th and 12th of September. PM_{10} peaks were observed at all the automatic monitoring stations in the British Isles at some time during these two days. Figure 6.71 presents the PM_{10} data for 17 monitoring locations in northeastern and northwestern England. Peak hourly concentrations here were reported on the 12th of September and were in the range 70 – $125 \mu g m^{-3}$. Peaks were also seen on the 11th September, but these were generally smaller in magnitude ~ 40 – $80 \mu g m^{-3}$ and their timings showed less coherence between the sites than they did on the 12th.
- 606.** Two sites were operational in Scotland at Edinburgh centre and Glasgow centre. These sites showed peaks during the morning and late evening of the 11th and substantial peaks of $>100 \mu g m^{-3}$ during the afternoon of the 12th. Four sites were operating in Northern Ireland and they showed no evidence of the pollution event on the 11th. Peaks of $>100 \mu g m^{-3}$ were observed during the afternoon of the 12th in Belfast and of about $80 \mu g m^{-3}$ at Lough Navar. Two peaks were observed at the

Welsh sites Cardiff Centre, Cwmbran and Port Talbot and reached over $50 \mu\text{g m}^{-3}$. The rural site Narberth in the west of Wales showed peaks that arrived significantly later than the other Welsh sites and with much reduced magnitudes. The smoky air masses reached London late on the evening of the 11th and again during the afternoon of the 12th. These peaks were similar in magnitude, both being $\sim 60\text{--}70 \mu\text{g m}^{-3}$ at the seven AURN sites. Sites in the Midlands showed a morning peak on the 11th and elevated concentrations throughout the morning into the early afternoon of the 12th, which reached $40\text{--}80 \mu\text{g m}^{-3}$.

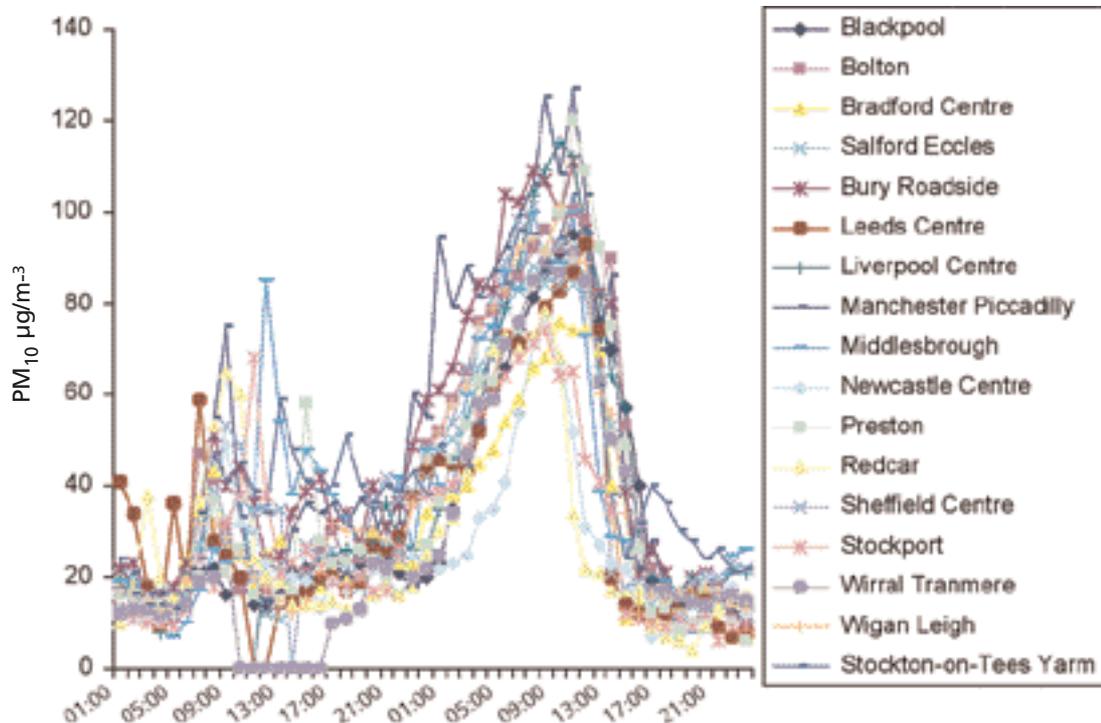
Figure 6.70 Satellite image of western Russia from MODIS for the 4th September 2002, covering the region 28° to 40° E, 52° to 60° N and showing the extent of the forest fires.



- 607.** The Meteorological Office Lagrangian dispersion NAME model was run for 10 days for each 3-h period from midnight on 11th September 2002 to midnight on 14th September 2002 for three locations: London, Lough Navar and Edinburgh. Global meteorology was used, which has a horizontal resolution of ~ 60 km and 32 vertical levels. A total of 45,000 air parcels were released over each 3-h period of interest between the ground and 80 m. Air parcel position information was collected on a ~ 60 km horizontal by 80 m vertical grid extending from 40°W to 60°E and 20°N to 80°N . The dispersed and transport material was assumed

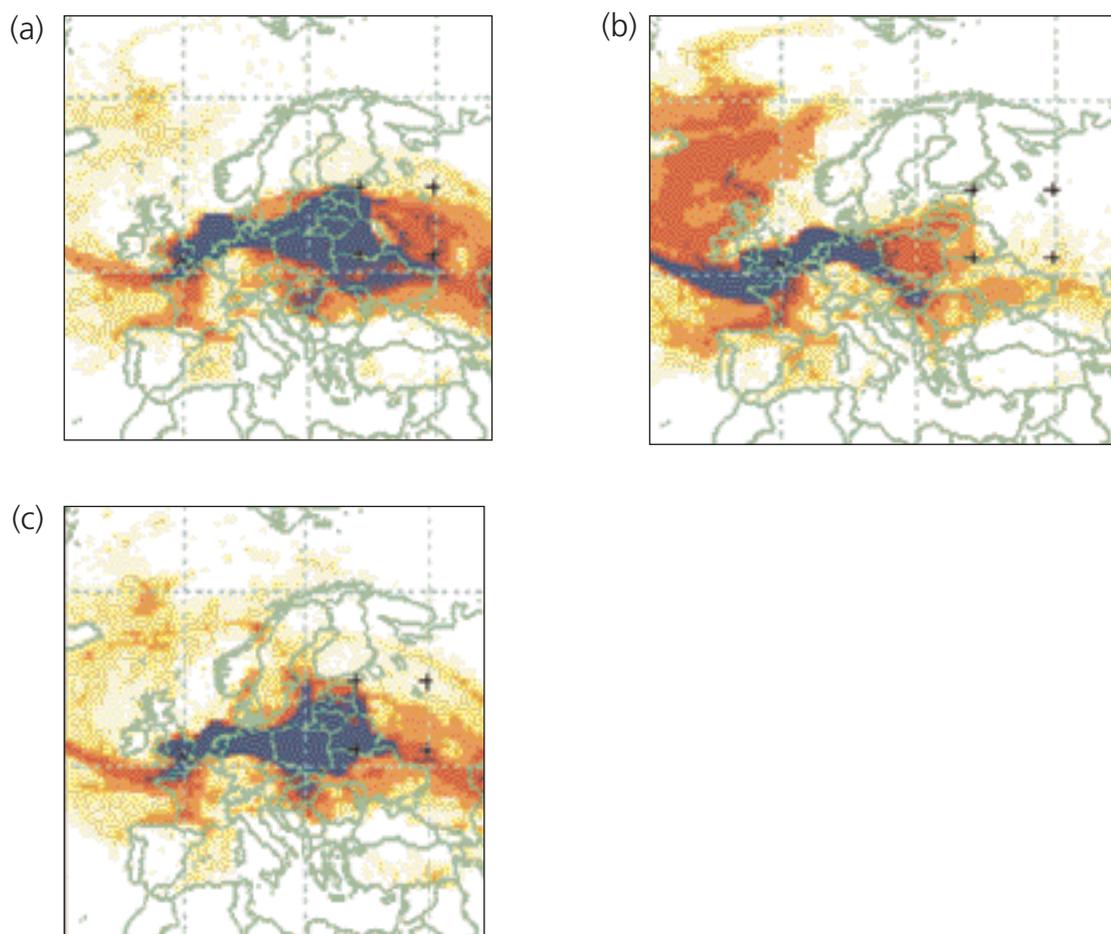
to be inert and, therefore, no account was taken of dry or wet deposition, sedimentation or chemical changes. The amount of material released was arbitrarily set to 1 g/s.

Figure 6.71 Hourly mean PM_{10} concentrations monitored at 17 locations in northeast and northwest England during September 11th and 12th 2002.



- 608.** Figure 6.72 shows the modelled air origin maps for London for the 3-h periods ending at (a) 00Z 12/09/02, (b) 06Z 12/09/02 and (c) 12Z 12/09/02. The four corners of the domain encompassing the fires identified in the satellite images have been marked (+) in each plot and the location of the receptor has also been highlighted (x). The plot is contoured with darker shades representing greater contributions.
- 609.** Figure 6.73 shows the time series of observations of PM_{10} and the modelled contribution from the domain encompassing the fires (see Figure 6.72) at London, Edinburgh and Lough Navar. The observations shown for London are the average PM_{10} values measured at the Bloomsbury and North Kensington sites.
- 610.** The double peak structure (Figure 6.73a) in the London observations is reflected in the modelled concentrations. The air origin maps for the times of the two peaks and the low point in-between are shown in Figure 6.72. The map of the 3-h period marking the low point (Figure 6.72b) shows that the origin of the air was more diffuse, with significant contributions from the Atlantic. The observations were uniformly elevated by background levels of 15–30 $\mu\text{g}/\text{m}^3$, reflecting the normal diurnal pattern of PM_{10} concentrations at London during an easterly flow. The second observed peak for the period 09:00 to 12:00 h will have a significant extra burden from local traffic sources compared to the first peak at 21:00 to 24:00 h. The timings of the peaks in the observations are identical to those modelled, namely at the 3-h periods centred on 22:30 h 11/9/02 and 10:30 h 12/9/02.

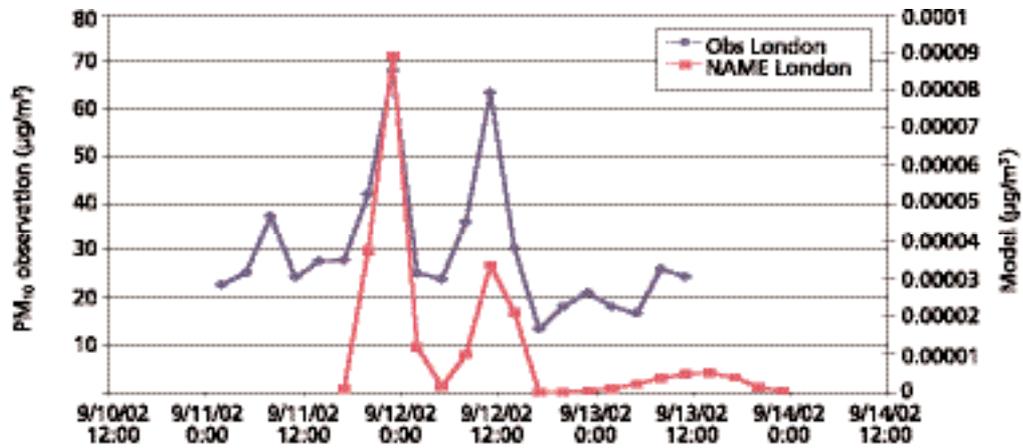
Figure 6.72 Ten-day back-attribution maps for London, 3-h periods ending (a) 00z 12/09/02, (b) 06z 12/09/02 and (c) 12z 12/09/02.



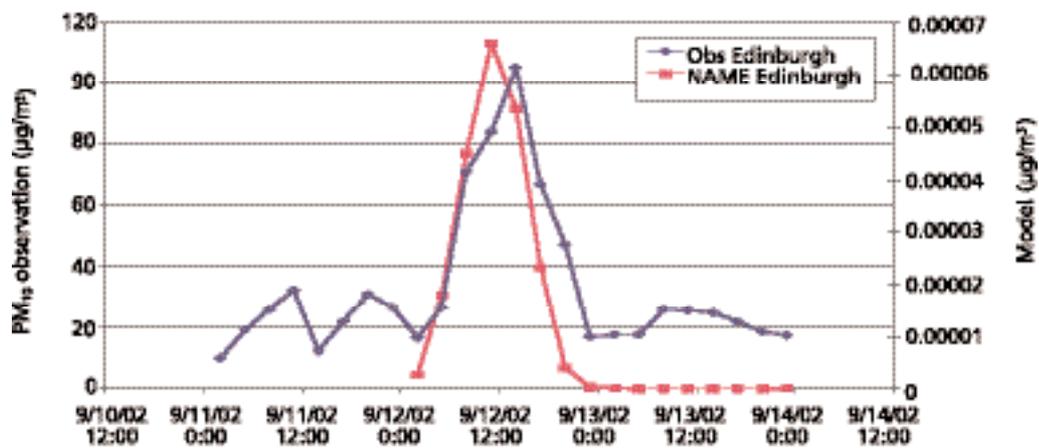
- 611.** The start times of the episodes at the three locations are significantly different. At London it began at 18:00 to 21:00 h on 11th September; at Edinburgh it was 03:00 to 06:00 h on 12th September and at Lough Navar the episode did not start until -09:00 to 12:00 h on 12th September. The observed episode start times were all mirrored by the modelled arrival times of air from the identified fire domain, as shown in Figure 6.73.
- 612.** The conclusions drawn from the above analysis of the PM₁₀ pollution episode on the 11th and 12th of September 2002, is that particles can clearly travel distance of over 1000 km across Europe from Western Russia into the British Isles. Furthermore, natural sources of particles from biomass burning can produce elevated hourly PM₁₀ concentrations of >100 µg m⁻³ at sites within the UK.

Figure 6.73 Time series of observed background PM₁₀ and modelled air concentration from the estimated area on fire in Western Russia at (a) London (b) Edinburgh and (c) Lough Navar. (The model results correspond to a notional emission rate of 1 g/s, as the true source strength is unknown.)

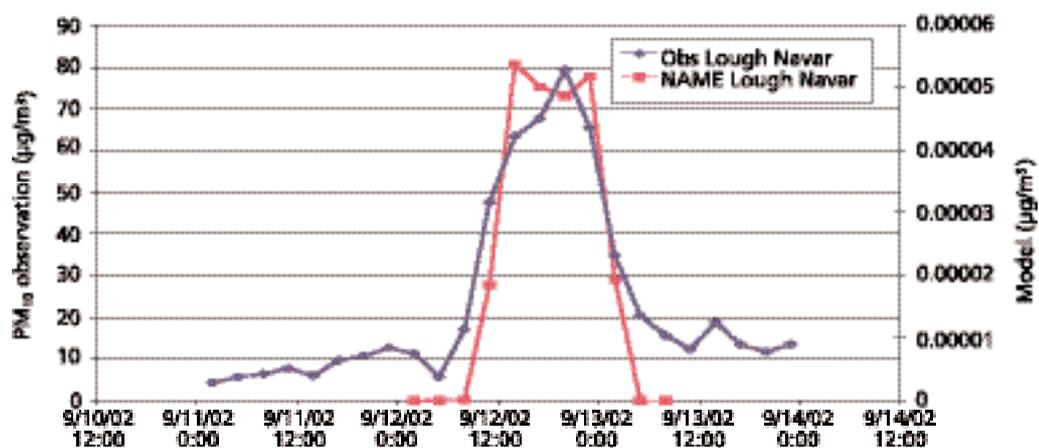
(a)



(b)



(c)

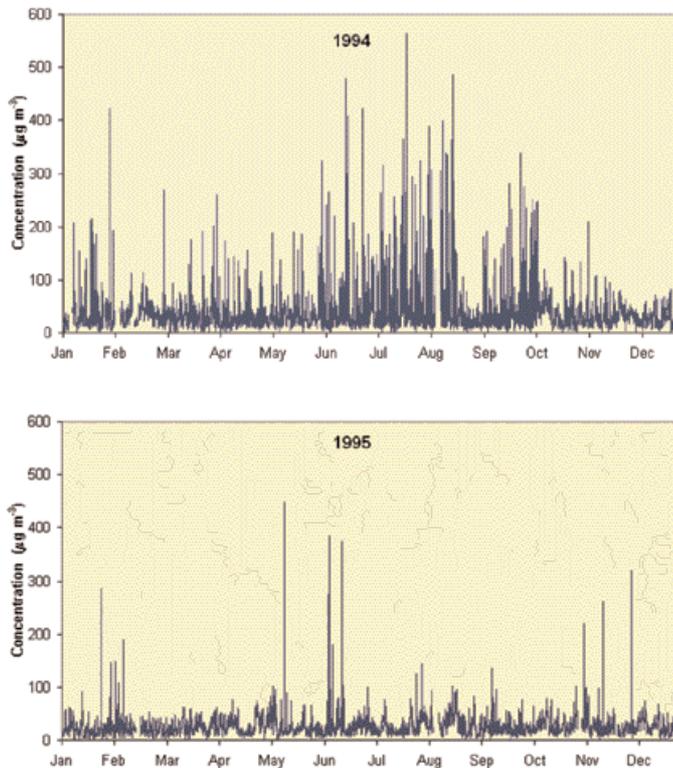


6.3.6 PM from construction activities

- 613.** Dust is generated from a number of activities on a construction site, the principal sources being: vehicle movement over dry unpaved ground; the handling of dusty materials; the cutting of concrete/stone; and the demolition of buildings. Dust is also generated by material tracked out of the site onto local roads. Typically 15–45% of construction dust is emitted as PM_{10} . Nationally, construction is not considered to be an important source of ambient PM (see Chapter 5), but locally this can be different. Unfortunately there have been few reported studies of PM concentrations near to construction activities, so it is difficult to be certain about impacts on local concentrations. These impacts would, in any case, be expected to vary from location to location, depending on the size and nature of the works, the weather conditions, and the stringency with which mitigation measures are applied. They will also be of a temporary nature, although the impacts could increase public exposure over a period of several years for major construction sites.
- 614.** One opportunity to investigate the impact very close to a construction site arose at the Cardiff centre AURN monitoring site in 1994. This city centre site is located in a pedestrianised area away from traffic. In 1994 a construction programme was started on a retail unit located alongside the monitor, 5 m away at the nearest point. The works lasted a full year and involved demolition, ground works, laying of concrete foundations, erection of a steel frame and concrete floor slabs, wall and roof construction, then finishing. A significant impact on PM_{10} (TEOM) concentrations was observed, as illustrated in Figure 6.74. (It is appropriate to use raw TEOM values, as the particles are unlikely to be volatile and the 1.3 scaling factor is unlikely to be appropriate). During 1994 there were 54 days when PM_{10} (TEOM) 24-h concentrations exceeded $50 \mu\text{g m}^{-3}$, with a maximum 24-h value of $96 \mu\text{g m}^{-3}$, compared to 12 days in 1995 and a maximum of $82 \mu\text{g m}^{-3}$. The greatest impact was on 1-h PM_{10} (TEOM) concentrations, with 89 h $>200 \mu\text{g m}^{-3}$ in 1994, compared to 11 h in 1995. The impact on annual mean values was, however, much smaller, with PM_{10} (TEOM) values of $31 \mu\text{g m}^{-3}$, $34 \mu\text{g m}^{-3}$ and $25 \mu\text{g m}^{-3}$ in 1993, 1994 and 1995, respectively. An analysis of the events showed that the 1-h exceedences of $200 \mu\text{g m}^{-3}$ nearly all occurred between 08:00 and 17:00 h, that is, during working hours (Kukadia *et al.*, 1998). The greatest number of high 1-h concentrations was during the summer months; it is not clear how much this was due to drier weather or the nature of the work activities at that time.
- 615.** The national monitoring activities have provided a further opportunity to examine the effects of construction works on PM concentrations. This occurred when demolition activities took place close to the Marylebone Road site in London, between July and November 1999 (Charron & Harrison, 2004). During this period and more especially in September, very high concentrations of PM_{10} (TEOM) were recorded, with hourly peaks of up to $800 \mu\text{g m}^{-3}$. The main contribution to these PM_{10} peaks was made by PM_{coarse} , although coincident peaks of $PM_{2.5}$ (TEOM) were also recorded (Figure 6.75). This resulted in many hourly $PM_{2.5}:PM_{10}$ ratios being lower than 0.5. All the high PM_{coarse} concentrations occurred during the working period (08:00 to 16:00 h) on weekdays, and the very high concentrations occurred for short periods of few hours or less. High concentration events did not appear to depend on wind speed or rainfall conditions and occurred on days with low or high wind speeds as well as on rainy or dry days. All high concentrations

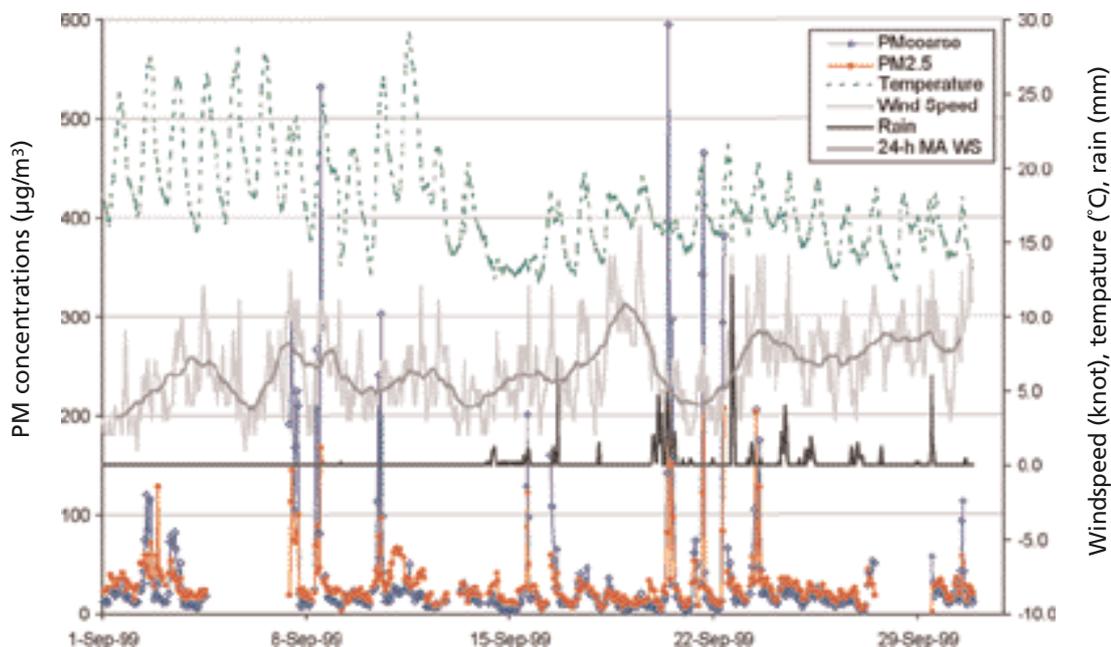
occurred when the synoptic wind was between 70° and 270° , which corresponds to wind direction associated with the southerly vortex in this canyon-like location (see Section 6.2.5) and parallel to the street. (Most of them occurred during southerly vortex situations.)

Figure 6.74 PM_{10} (TEOM) 1-h concentrations at the Cardiff centre AURN site: construction occurred in 1994, no construction occurred in 1995.



- 616.** During the period of July to November 1999, the very high hourly concentrations had a significant impact on daily PM_{10} concentrations. There were 27 days when PM_{10} (TEOM) 24-h concentrations exceeded $50 \mu\text{g m}^{-3}$, with a maximum 24-h concentration of $139 \mu\text{g m}^{-3}$; compared to 6 days and a maximum of $57 \mu\text{g m}^{-3}$ and 16 days and a maximum of $87 \mu\text{g m}^{-3}$, respectively, from July to November 1998 and 2000. From July to October 1999, the agreement between daily PM_{10} (TEOM) data and filter-based mass measurements (KFG) was good (median ratio of KFG:TEOM was 1.04), showing that particles were mainly made of non-volatile material. Although the impact of demolition activities on $PM_{2.5}$ concentrations was less spectacular than the impact on PM_{coarse} concentrations (PM_{coarse} : 44 h $>100 \mu\text{g m}^{-3}$; $PM_{2.5}$: 19 h $>100 \mu\text{g m}^{-3}$), it is nevertheless still significant. From July to November 1999, there were 8 days when $PM_{2.5}$ (TEOM) 24-h concentrations exceeded $40 \mu\text{g m}^{-3}$ with a maximum of $54.8 \mu\text{g m}^{-3}$, which compares with 1 day and a maximum of $41.9 \mu\text{g m}^{-3}$ and 3 days and a maximum of $50.2 \mu\text{g m}^{-3}$, respectively, from July to November 1998 and 2000.
- 617.** Local fugitive PM_{10} at Marylebone Road during 1999 has been modelled using a source apportionment technique (Fuller & Green, 2004). This shows that the local fugitive PM_{10} alone was sufficient to cause the daily mean PM_{10} at the site to exceed $50 \mu\text{g m}^{-3}$, reaching a maximum daily mean concentration of $133 \mu\text{g m}^{-3}$.

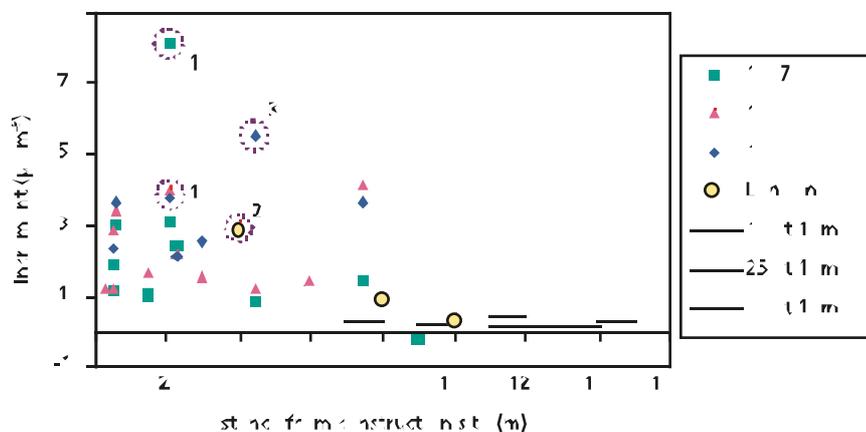
Figure 6.75 Hourly $PM_{2.5}$ and PM_{coarse} concentrations (TEOM), wind speed (including 24-h moving average), temperature and rain from 1st to 30th September 1999 at Marylebone Road.



- 618.** Monitoring data from sites across London and Southeast England have been examined for the period 1999–2001 to identify PM_{10} incidents that are considered to be unrelated to road traffic (Fuller & Green, 2004). Incidents have been attributed to local fugitive PM sources on the basis of brief high measurements of PM_{10} , using the criterion of 15-min mean concentrations in $>250 \mu\text{g m}^{-3}$. PM_{10} measurements from 81 TEOM sites during the 3 years were examined, using this criterion, to estimate the likely frequency of local fugitive incidents (many of which were likely to be related to some form of construction activity, but not exclusively so). This dataset of 156 site years included 134 daily means $>50 \mu\text{g m}^{-3}$ that might have been linked to fugitive sources. This represents an average of 0.9 days per site per year. Furthermore, the analysis suggests that fugitive sources might have been associated with 5% of the 2,470 daily means $>50 \mu\text{g m}^{-3}$ and that they occurred at almost half (37/81) of the monitoring sites, including all types of site from kerbside to rural. Two sites that measured more than ten daily means associated with fugitive sources during the 3-year period were Kensington and Chelsea 2 (19 days) and Marylebone Road (45 days). Marylebone Road measured more than 7 such days in each of the 3 years.
- 619.** Turning to more specific studies of construction sites, results are available for monitoring around a major construction programme in the centre of downtown Boston, Massachusetts (Dolan *et al.*, 2000). PM_{10} (gravimetric) was monitored at 16 locations within 90 m of an eight-lane underground roadway being constructed through the city. Monitoring using Minivol gravimetric samplers was carried out as part of a programme to control dust impacts. The programme involved monitoring 2 days a week for the 5 months of June to October over the 3 years of the project. The results are summarised in Figure 6.76, in which the increment above background is plotted against distance from the site (results for two monitoring sites that were alongside roads with well over 100,000 vehicles per day, as well as being close to the construction site, are not included).

Concentrations clearly increase closer to the construction site. Distances are from the boundary of the site, not the dust source. In some instances the dust source is material tracked out onto local roads, thus the source of the dust can be closer to the monitor than shown (see the caption for Figure 6.76). The results show clear evidence that mean PM_{10} concentrations can be significantly elevated by construction dust, potentially by some $10\text{--}50\ \mu\text{g m}^{-3}$ over 5 summer months within 90 m of a major construction site. They also show that the movement of dump trucks along off-site roads spreads the PM_{10} impact away from the site. Separate studies were carried out following a number of incidents when 24-h PM_{10} (gravimetric) concentrations exceeded $150\ \mu\text{g m}^{-3}$ – the USEPA standard at the time. These showed that additional control measures to reduce airborne dust, including improved cleaning of roads, were effective in reducing PM_{10} concentrations, confirming the role of construction dust in producing the higher levels close to the construction site.

Figure 6.76 PM_{10} increment in relation to background versus distance from boundary of a major construction site. (1997–1999 refers to the summer mean PM_{10} (gravimetric) for the site in Boston, Massachusetts, 1997–1999; London refers to the annual mean PM_{10} (TEOM) for the site in London. See text for discussion of the lines. (1) This site was close to a road being used by over 600 dump trucks a day in 1997. Lorries were idling alongside the monitor. (2) This site was five m from the road used by site lorries and 40 m from site entrance. (3) This site was affected by dirt being tracked out by dump trucks. (4) There is no explanation for the high levels recorded at this site; the lines are fitted assuming $10\ \mu\text{g m}^{-3}$ at 10 m, $25\ \mu\text{g m}^{-3}$ at 10 m and $40\ \mu\text{g m}^{-3}$ at 10 m.)



620. Monitoring of PM_{10} using TEOM instruments has recently been carried out at three locations near to a major construction site in London. Results for a 1 year period have been made available with permission of the developer and the local authority (Symonds, 2004). During this period major earthworks were under way on the site of around $0.75\ \text{km}^2$. Annual mean and daily exceedences of $50\ \mu\text{g m}^{-3}$ are set out in Table 6.15. TEOM units have been retained, as the excess above the background is unlikely to contain volatile particles, thus a factor close to 1.0 would be expected to apply to this increment. The results show that there can be a significant impact close to the site, but the impact 100 m away seems to be minimal (the site at 100 m is not downwind of the predominant wind direction). The highest concentrations are found near to the site entrance, where vehicle activity is generating significant amounts of dust.

- 621.** The results for the London construction site are also shown as an increment above the background in Figure 6.76. The lines in this figure are derived using the relationship shown for the modelled decline in concentrations with distance from a fugitive source (Defra, 2003a). This model was normalised to a concentration of one at 10 m from the source. A good fit would not be expected, as the monitoring sites are not located on a transect away from a single source, also the distance is from the boundary of the site, not the source of the dust, nevertheless the pattern of the monitoring seems to be consistent with the modelled relationship.

Table 6.15 Measured PM₁₀ (TEOM) concentrations around a major construction site in London.

Monitoring location	Distance from site boundary	Annual mean $\mu\text{g m}^{-3}$ (TEOM)	Daily exceedences of $50 \mu\text{g m}^{-3}$ (TEOM)
Site 1	40 m from site entrance to east of site and five m from road affected by dirt being tracked out of the site.	46	122
Site 2	80 m from northeast boundary of site	27	26
Site 3	100 m from site boundary to south in residential area	21	7
Site 4	Local authority background site ~7 km to north	18	4

- 622.** At a smaller scale, a study by Upton & Kukadia (2002) involved PM₁₀ monitoring at three locations around a 0.65 ha development site within a town centre. The site housed an old industrial building, which was demolished before removing 1 m of contaminated soil, after which the site was in-filled and levelled. The work took place between September and March, thus not over the season when dust would be at its worst (dust is more likely during drier summer months, May to August). One monitoring site was located a few metres from the boundary of the site, to the north, the other two were 185 m to the southwest and 130 m to the east of the site. No impacts were detected at the two distant monitors, but at the site boundary concentrations were elevated by $\sim 3 \mu\text{g m}^{-3}$ during internal stripping of the buildings (over 4 weeks), $11 \mu\text{g m}^{-3}$ during demolition (over 8 weeks) and $5 \mu\text{g m}^{-3}$ during earth-working operations (over 8 weeks). This period of works is equivalent to an increase in annual mean of $2.7 \mu\text{g m}^{-3}$ at the site boundary. There was one exceedence of the $50 \mu\text{g m}^{-3}$ objective at the site boundary, but none at the other two locations. It is not surprising with increases of such small magnitudes at the site boundary that no effects are seen at the more distant monitoring sites. This was the case even at times when the winds were blowing from the site towards the more distant monitors. Dust was controlled in a rigorous manner at this site, due to its town centre location, so it may not be typical of other sites.

623. In conclusion, it is clear that construction activities can give rise to elevated PM_{10} concentrations, both in terms of annual mean and 24-h exceedences of $50 \mu\text{g m}^{-3}$. This is certainly the case within 10 m of relatively small construction sites and may extend out to 50–100 m from major construction sites. It would seem that beyond about 100 m, construction sites are likely to make only a small contribution to local PM_{10} concentrations. The evidence is that vehicle movements are a key source of the PM_{10} , including byway or resuspended material tracked out of the site onto public roads. This track out of dust and dirt could extend the area of significant impact further than 100 m from the site boundary. The situation occurring at Manor Road, Bexley in London, where dust and dirt are being tracked out from light industrial sites onto the local roads by lorries, is similar (see Section 6.4.7). The study of monitoring sites across the southeast of England suggests that there might be fairly widespread contributions to PM_{10} made by fugitive, in many cases construction-related, sources and that these incidents can contribute to the number of days with concentrations $>50 \mu\text{g m}^{-3}$. No specific studies have been identified that show the contributions construction activities can make to $PM_{2.5}$ concentrations, although the results for the Marylebone Road site suggest that $PM_{2.5}$ concentrations are affected by construction activities, although to a lesser extent than PM_{coarse} concentrations.

6.4 Use of correlations, concentrations or chemistry to elucidate particle sources

6.4.1 Comparison of PM concentrations with measurements of other pollutants

624. Measured PM_{10} and $PM_{2.5}$ concentrations have been compared with the measured concentrations of other air pollutants at Marylebone Road and Harwell. Figures 6.77 to 6.79 show comparisons of hourly PM concentrations during 2002 with NO_x , benzene and CO. PM concentrations are more closely correlated with NO_x than with benzene or CO. This is consistent with the expected emission factors for road vehicles, which are higher for PM and NO_x for diesel vehicles and higher for benzene and CO for petrol vehicles. It confirms that traffic emissions are an important PM source at this site. The emissions of PM_{10} are greater than the emissions of $PM_{2.5}$ (the gradient of the line is greater), although this analysis does not directly inform as to whether the additional emissions of PM_{coarse} are exhaust or other emissions.

625. The gradients for PM_{10} and $PM_{2.5}$ are much more similar at the rural site at Harwell (Figures 6.80 and 6.81) and the correlation coefficients are lower because this site is less strongly influenced by traffic sources. Figures 6.84 and 6.85 show the relationships between PM concentrations and ozone. The daily maximum running 8-h ozone concentration has been compared with daily mean PM concentrations (hourly PM and ozone concentrations tend to be anti-correlated because primary PM emissions are correlated to NO emissions and thus anti-correlated with measured ozone concentrations). Figure 6.82 shows that there is some tendency for high PM concentrations to be associated with high ozone concentrations in the summer at Harwell, which reflects the photochemical generation of secondary particles. This is not the case in winter when the weak anti-correlation suggests that the higher PM days tend to be associated with lower ozone concentrations, when poor dispersion leads to a build up of primary