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Integrated assessment of brick kiln emission impacts on air quality

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Abstract This paper presents monitoring results of daily brick kiln stack emission and the derived emission factors. Emission of individual air pollutant varied significantly during a firing batch (7 days) and between kilns. Average emission factors per 1,000 bricks were 6.35–12.3 kg of CO, 0.52–5.9 kg of SO_2 and 0.64–1.4 kg of particulate matter (PM). PM emission size distribution in the stack plume was determined using a modified cascade impactor. Obtained emission factors and PM size distribution data were used in simulation study using the Industrial Source Complex Short-Term (ISCST3) dispersion model. The model performance was successfully evaluated for the local conditions using the simultaneous ambient monitoring data in 2006 and 2007. SO_2 was the most critical pollutant, exceeding the hourly National Ambient Air Quality Standards over 63 km^2 out of the 100-km^2 modelled domain in the base case. Impacts of different emission scenarios on the ambient air quality (SO_2, PM, CO, PM dry deposition flux) were assessed.

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Introduction

Traditionally, brick manufacturing is a small-scale and unorganised industry which is mainly concentrated in the rural and peri-urban areas of developing countries. The brick industry is rapidly developing to meet the increasing brick demand for construction. The industry brings in overall development to the areas where it is located (Singh and Asgher 2005). However, the adverse environmental effects from brick manufacturing are also significant and include loss of land, change of the land cover, removal of nutrients and humus from soil, increased erosion and environmental effects (Brick Industry Association 2006; FAO 1993; RERIC 2003).

Recently, the air pollution emission from brick kilns has gained international attention (CAI-Asia 2008; Ferdausi et al. 2008). Simple kiln technologies used for brick firing and the lack of emission control devices often result in a large amount of released air pollutants. These pollutants include a wide range of incomplete and complete combustion products emitted during the brick firing process. They originate from both the

fuel used for brick firing and the raw brick materials (RERIC 2003; Zhang 1997). Emission from multiple brick kilns that are densely distributed in manufacturing villages would cause severe adverse effects to human health (Blackman 2000; Zhang 1997) and agricultural crops (Singh and Asgher 2005). Atmospheric dispersion processes can bring the emission to the surrounding urban areas which can further worsen the air quality in adjacent cities. However, there is quite limited information on the emission factors and environmental impacts of brick making industry. Often, when emission inventory is conducted, the emission factors have to be drawn from various literature sources which results in ambiguity and the potential for error because of differences in manufacturing technologies.

The brick industry in Vietnam is rapidly developing with the major share from the non-state sector. This sector includes the medium, small and family-based enterprises with the share increasing from 62% in 2001 to over 80% in 2005 (total product of almost 17,000 million brick pieces in 2005; General Statistics Office 2005). A wide range of the kiln technologies exists in Vietnam with the most common type being traditional-improved kilns which are small in size and low cost, hence, more affordable to low income farmers (Dung and Son 2004). The traditional-improved kilns are arranged in pairs. Each pair shares a common chimney located between the kilns which operate on rotation (Co et al. 2009). Coal briquettes, made from coal particles and different types of binding materials, are commonly used for brick firing.

Bac Ninh province is located in the Red River Delta in the north of Vietnam. The province has a population of 976,700 and an area of 804 km^2 of which 64.7% is for agricultural use. There are over 1,500 brick kilns in the province which are distributed in six districts along the banks of the Duong and Cau Rivers. Normally, about 40–60 kilns form a brick making commune. High air pollution emissions from the brick kilns have caused concern to surrounding communities. In order to protect rice plantations, a regulation was promulgated by People's Committee of Thuan Thanh district that bans brick manufacturing in the whole district during the period from March to September each year, which is the growing period of the two major rice crops (AIRPET-VN 2007). Song Ho village, the study area, is one of 17 villages of the Thuan Thanh district hence is also affected by the ban.

This study was designed to develop integrated management strategies for air pollution emission reduction from brick kilns. The research was conducted within the "Asian Regional Air Pollution Research Network (AIRPET)" which is coordinated by the Asian Institute of Technology (http://www.serd.ait.ac.th/airpet). The present paper focuses on emission monitoring and dispersion modelling for assessment of the environmental effects caused by the brick manufacturing in a selected village of Song Ho. Integrated management strategies and a more detail description of the selected village are provided in Co et al. (2009) and Le (2007), respectively.

Research methodology

Study area

Bac Ninh province has a typical climate of the Northern Vietnam which is under the influence of the NE monsoon during winter (November–January) and SW monsoon during summer. The annual average temperature recorded for Bac Ninh province is 24.3°C (30–36°C in summer and 15–20°C in winter). Average relative humidity is 79% and annual rainfall is 1,800 mm (General Statistics Office 2005). In the Song Ho village, a high density of kilns is observed on the right bank of the Duong River (Fig. 1). The number of operating kilns (excluding the obsolete) in this village has increased quickly, i.e. from 35 kilns in April 2006 (AIRPET-VN 2007) to 45 in January 2007 (Le 2007). These kilns were arranged in clusters and each cluster may consist of one to six kilns located nearby and often belong to one owner. No air pollution emission control device was applied in any kiln in the village. The production per batch varies from 200,000 to 1,000,000 bricks with an average of 450,000. The average coal consumption is 8.53 $\pm$ 3.0 tonnes per 1,000 bricks. Most of the kilns in the village were equipped with rectangular stacks of 10–21 m high and the stack tip cross-sectional area of around 1 m^2 (equivalent diameter 1–1.2 m). Some kilns were not equipped with

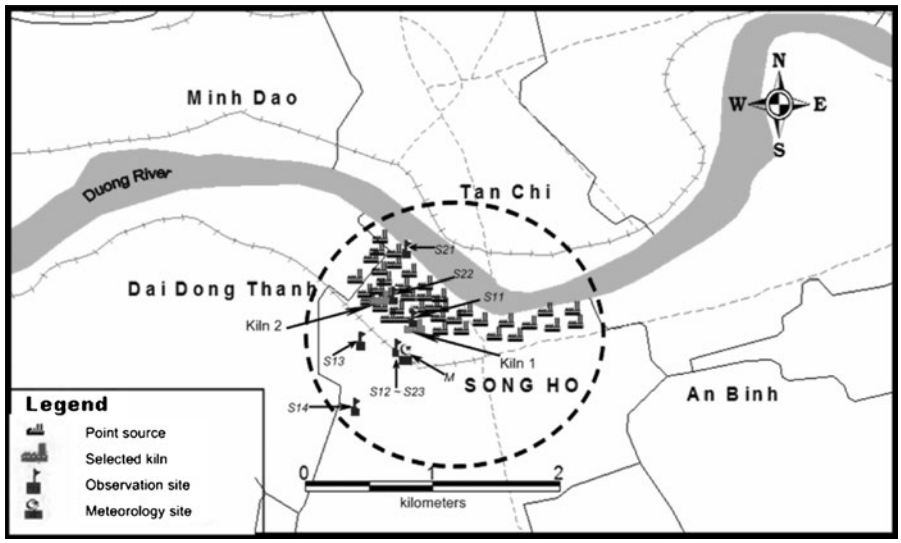

Fig. 1 The study area (the ISC model domain) with the Song Ho brick making village and ambient air monitoring sites. *M* was meteorological measurement site in both campaigns. Ambient sites S11, S12, S13 and S14 were used in April 2006 whilst S21, S22 and S23 were used in January 2007

stacks; hence, the emission was released directly from the rooftop of the kilns, at a height of around 10 m (Le 2007).

Research design

Integrated monitoring and modelling tools were employed. Two monitoring campaigns were conducted with simultaneous measurements of source emission (kiln stacks) and ambient air pollution. The overall research design is presented in Fig. 2.

Sampling and analytical techniques

The first monitoring campaign was done during 11–16 April 2006 when only one kiln was operated (kiln 1) in the entire village. Stack emission results

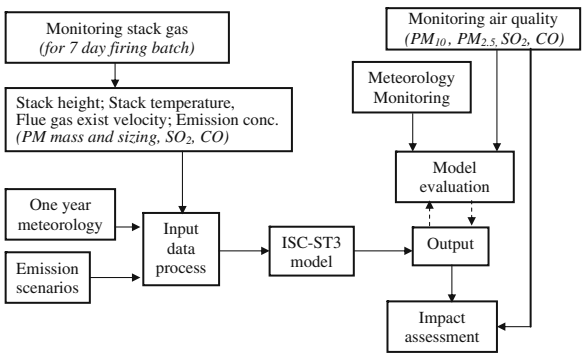

Fig. 2 Framework of the research methodology

of this campaign are briefly summarised in Co et al. (2009). The monitoring was conducted during the period banned operation, and the kiln was operated under a special permission. The main purpose was to evaluate the ambient air quality when only one kiln was operated with and without a test run of a control device. In this period, the simultaneous ambient air and stack measurements were made for 2 to 3 h/day during the last 4 days of the 7-day firing batch (13–16 April 2006). Four ambient monitoring sites located within a radius of 500–700 m around kiln 1 were used that included S11 (NE to the kiln, generally upwind) and others downwind sites (S12, S13 and S14).

The second campaign monitoring was conducted for 7 days (24–31 January 2007) for 1 h (9:00 a.m. to 10:00 a.m.) daily. There were 21 kilns operated in the village which represents the normal manufacturing condition. The stack gas monitoring was conducted for kiln 2 (Fig. 1), which was about 300 m from kiln 1. Simultaneous ambient monitoring was conducted at three sites S21 (upwind), S22 (inside village) and S23 (downwind) shown in Fig. 1. In fact, monitoring sites S12 and S23 and the meteorology measurement site (M) were the same in both campaigns. Monitoring detail of both campaigns is given in Table 1. Note that in January 2007, more parameters were measured, i.e. flue gas particulate matter (PM) sizing and ambient PM_{10} and $PM_{2.5}$. The monitoring equipment was calibrated before use.

Table 1 Sampling and analytical methods in the monitoring campaigns

Monitoring	Parameter	Sampling method	Analysis	Monitoring period
Ambient air	$PM_{2.5}$	MiniVol 1100, Ecotech, filters: QFF	Gravimetric (microbalance)	January 2007
	PM_{10}	SL-20, Sibata (20 L/min), filters: QFF	Gravimetric (microbalance)	January 2007
	TSP	Low-volume sampler, Sibata, filters: GFF	Gravimetric (5-digit balance)	April 2006
	CO	NDIR, ML—9830, Monitor Labs	Automatic	January 2007
	SO_2	UV-fluorescence, ML—9850, Monitor Labs	Automatic	Both
Stack gas	CO	Quintox, KM9006	Automatic	Both
	SO_2	Quintox, KM 9006	Automatic	Both
	PM	STL-Combi dust sampler, manual isokinetic, filters: GFF	Gravimetric (microbalance)	Both
	PM sizing	Anderson cascade impactor: TISCH, 8-stage, filters: GFF	Gravimetric (microbalance)	January 2007
Meteorology	Monitoring	Portable equipment, GroWeather	Hourly, on-line	Both

QFF quartz fiber filters, *GFF* glass fiber filters

Two types of filter—glass fiber filters and quartz fiber filters—were used for PM sample collection. Clean filters as well as sampled filters were desiccated in the laboratory for 24 h before weighing. A microbalance was used for weighing the $PM_{2.5}$ and PM_{10} mass.

For stack gas sampling, a port was made into the stack following the requirement of isokinetic sampling (US EPA 1986). Thus, the port was located at a distance of more than eight equivalent diameters from the upstream disturbance (flue gas entrance point) and above two equivalent diameters from a downstream disturbance (stack tip). The stack sampling velocity was adjusted manually to achieve the isokinetic condition. The filter box was heated at a temperature of around 100°C. For PM sizing, an arch (almost L-shape) stainless pipe probe was connected to an eight-stage Anderson cascade impactor. The probe was edged shape and was aligned at the right angle to the cross-sectional area of the stack. The average stack gas velocity measured during the first campaign was around 8–10 m s^{-1}. Accordingly, the diameter of the sampling probe was selected to be of 0.8 cm to get the isokinetic sampling velocity for the fixed pumping rate of 28 L min^{-1} used in the cascade impactor.

In old kilns, flue gas can be channelled through various openings on kiln roof, and the leaks may significantly affect stack flow rate measurement results. New kilns are observed to have less flue gas leaking. In this study, to minimise leaks, openings on the roofs of the selected kilns were sealed with clay. The fire appeared to be normal during the experiments; hence, the sealing effects on kiln operation should be minimal as the stack opening was large enough to handle the exhaust gas. Accordingly, its effects on combustion conditions and, consequently, on air pollution emission are considered to be insignificant. Both kilns selected for stack monitoring were of the traditional-improved type that is most common in the village, i.e. around a 90% share (Le 2007).

Dispersion modelling

The Industrial Source Complex Short-Term (ISCST3) model (US EPA 1995) was applied to estimate ambient air concentration in the surrounding area of the studied brick making village for different emission scenarios. The domain was of 100 km^2 (Cartesian 40 × 40 grids of 250 m size) of flat terrain covering Song Ho village (Fig. 1).

For the scenario analysis, the year 2003 reanalysed meteorological data obtained from the National Oceanic and Atmospheric Administration (NOAA) for the Noi Bai Airport station (both surface and upper air data), which is the

nearest meteorology station to the study site (about 40 km away), was used. PCRAMMET meteorological pre-processor (US EPA 1999) was applied to produce the meteorological input file for ISCST3. The emission input data were prepared based on the results of our survey (number of kilns operating per day, their locations by global positioning system, stack height, etc.) and the emission rates from the stack monitoring. The particle size mass distribution in the flue gas, obtained from the stack monitoring by the cascade impactor, was used to model the dry deposition of PM.

The model output includes PM concentration, PM deposition flux, CO and SO_2 concentrations over the domain. For the model evaluation purpose, simultaneous measurement results of meteorology, emission and ambient air pollution during two sampling campaigns were used. The first campaign was for simulation of a single stack emission, and the second campaign was used for base case emission.

Results and discussion

Ambient air quality

In the second campaign (January 2007), the daily monitoring was conducted for one morning hour (9:00 a.m. to 10:00 a.m.) that yielded hourly ambient concentrations of CO, SO_2, $PM_{2.5}$ and PM_{10} during the firing batch (7 days) at selected sites. As mentioned above, this was a period of normal brick making operation in the village (representing the base case emission) when around 21 kilns were operating. Note that the CO concentration did not exceed the hourly National Ambient Air Quality Standards of Vietnam (NAAQS) at all three sites (Table 2). However, SO_2 levels were observed (87–514 μg m^{-3}) with a high frequency of exceeding NAAQS, i.e. ten out of 21 measurements or 48%. There is no hourly NAAQS for PM_{10} and $PM_{2.5}$, but three measurements of hourly PM_{10} at S23 (downwind during January 2007) even exceeded the 1-h NAAQS for total suspended particulates (TSP) of 300 μg m^{-3}.

During the January 2007 monitoring period, the prevalent wind directions were generally

Table 2 Hourly average pollutant concentrations in the ambient air (micrograms per cubic metre; 25°C; 1 atm), measured at 9:00 a.m. to 10:00 a.m. daily, 24–30 January 2007

Dates (2007)	CO, mg m^{-3} (1-h NAAQS = 30)			SO_2, μg m^{-3} (1-h NAAQS = 350)			PM_{10}, μg m^{-3} (1-h NAAQS for TSP = 300)			$PM_{2.5}$, μg m^{-3} (24-h US NAAQS = 35)			Wind	
	S21	S22	S23	S21	S22	S23	S21	S22	S23	S21	S22	S23	Speed (m s^{-1})	Direction
24 January	1.03	2.00	2.93	197	150	231	54	144	229	11	24	37	2.2	N–NW
25 January	2.01	2.31	2.67	87	102	224	50	136	234	40	34	67	0.4	NW
26 January	1.14	2.68	2.46	374	293	351	49	153	274	48	43	45	1.6	N
27 January	2.30	2.75	3.08	117	369	418	52	176	388	51	45	66	2.1	N–NE
28 January	2.07	1.04	2.13	351	356	362	35	151	344	34	59	63	1.3	N–NE
29 January	1.37	2.38	3.35	162	168	303	43	184	342	37	46	41	1.2	NW
30 January	2.23	3.15	3.20	431	514	502	54	234	235	50	67	68	1.1	NW
Average (STD)	1.74 (0.54)	2.33 (0.68)	2.83 (0.43)	293 (134)	298 (139)	274 (135)	47 (7)	168 (34)	292 (65)	40 (14)	45 (14)	55 (14)	1.4 (0.6)	

S21, S22 and S23 locations are shown in Fig. 1

STD standard deviation

northerly (Table 2). Site S21 was the upwind site and was located close to the bank of the river, about 300 m, from the NE edge of the village (Fig. 1). S22 and S23 were located on the downwind side of the village: S22 was in the SW part of the village whilst S23 was about 500 m away to the south of the village. The latter (S23) was on the dike which served as a local road used for bricks transport (light duty diesel vehicles) and general commuting purposes (mainly motorcycles). The downwind locations of S22 and S23 explained the generally higher pollutants concentrations compared to S21. The most significant differences between the three sites were observed for PM_{10} which may be caused largely by the local sources inside the village such as road dust re-suspended by traffic, vehicle exhaust and emission from the processing of brick raw materials and coal briquettes. The coarse particles of PM_{10} would tend to deposit nearby the emission sources. For fine particles ($PM_{2.5}$), the difference between the three sites was not so significant. These fine particles may be transported far hence can be originated from both local and regional/remote sources.

Higher $PM_{2.5}/PM_{10}$ ratios were obtained at S21, averaged at 0.82, which are somewhat higher than those reported in urban areas, 0.5–0.7 (Kim Oanh et al. 2006). When located upwind to the brick making village, this site can be considered as a rural remote site (far from urban sources and industry) which explains this high $PM_{2.5}$ fraction in PM_{10}. The $PM_{2.5}/PM_{10}$ ratios for site S22 and S23 were low with the average of 0.27 and 0.2, respectively, which was due to the high contribution of local PM_{10} sources as mentioned above.

The range of hourly SO_2 and TSP measured at four sites in April 2006 was 45–112 and 150–280 $\mu g\ m^{-3}$, respectively, which did not exceed the corresponding NAAQS (Co et al. 2009). The SO_2 levels during April 2006 when only one kiln (kiln 1) operated (Fig. 3) were significantly lower than those in January 2007. Wind in April 2006 was variable and site S11 was generally upwind whilst site S14 was generally downwind to the monitored kiln; hence, SO_2 levels in site S14 were higher than other sites.

Stack gas emission

Emission rates

Considerable variations of all measured parameters during a firing batch were observed. In January 2007, the stack gas monitoring (for kiln 2) was generally conducted at 9:00 a.m. to 10:00 a.m. each day except for the last 2 days when measurements were done for 4–6 h/day. Several (two to six) gaseous pollutant measurements were obtained daily at 30-min interval, and the average and standard deviation of the results are presented in Table 3. For PM, however, only one filter sample was collected per day. The flue gas temperature increased from the ambient temperature on the first day (i.e. around 20°C in January) to above 235°C on the seventh day of the batch reaching the highest temperature just before the fire stopped. Stack gas velocity during the batch was 6–10.5 m s^{-1} with the corresponding volumetric flow rate of 6–10.5 $m^3\ s^{-1}$ (Table 4). The high values at the beginning (days 1 to 2) were most

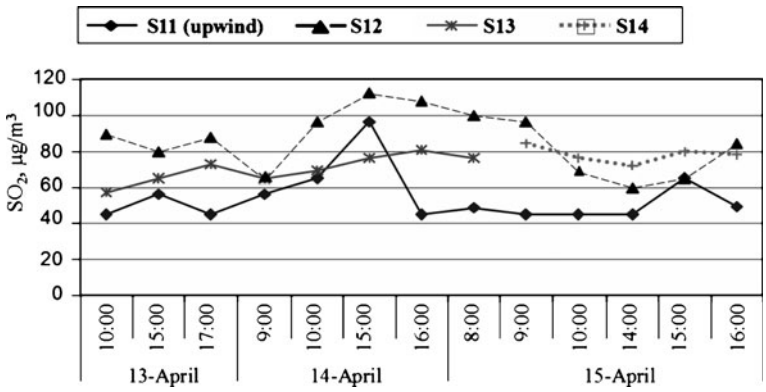

Fig. 3 Hourly SO_2 (25°C and 1 atm) at four sites around the monitored kiln (only one kiln operated) in April 2006: S11 (generally upwind site), S12, S13 and S14 (see Fig. 1)

Table 3 Daily stack gas emission (average and standard deviation) during 24–30 January 2007 (concentrations reported at 0°C, 760 mmHg)

Day	Stack gas temp (°C)	CO		SO_2		PM	
		mg m^{-3}	kg day^{-1}	mg m^{-3}	kg day^{-1}	mg m^{-3}	kg day^{-1}
1st	20	4 ± 1.4	3 ± 1	2.9 ± 0	2 ± 0	13	9
2nd	33	1,867 ± 18	1,266 ± 12	23 ± 4.2	16 ± 3	36	24
3rd	50	1,770 ± 36	997 ± 20	11 ± 0	6 ± 0	108	61
4th	46	1,277 ± 30	624 ± 15	6 ± 0	3 ± 0	105	51
5th	51	1,712 ± 355	1,033 ± 214	150 ± 3	91 ± 2	157	95
6th	180	1,212 ± 139	850 ± 97	199 ± 17	140 ± 12	367	257
7th	235	339 ± 32	306 ± 29	179 ± 6.1	162 ± 6	17	15
Emission standard[a]		900	–	1,350	–	360	–

[a]Industrial ES adapted for this type of sources (Kp = 0.9 for the range of flue gas flow rate from brick kilns, Kv = 1.0 for peri-urban areas) at 0°C, 760 mmHg. Coefficient (Kp) is related to the air emission flow rate and coefficient Kv is related to the regions, where facilities are located (MONRE 2005)

probably due to the intensive use of fans to start up the fire. The higher values towards the end of the batch (days 6 to 7) were mainly due to intensive combustion. CO and PM concentrations were low during the startup days and higher in the middle of the firing cycle then reduced when burning stopped at the end of the batch. However, the variation in SO_2 was different which may be due to the low combustion temperature at the beginning, and there may be a possibility that SO_2 released was initially absorbed by the upper unfired bricks layers in the kiln (and would be emitted later when upper layers were being fired). Thus, only during the last 3 days when the top layers were being fired a high SO_2 emission was observed. No monitoring was conducted after day 7 when the kiln was in the cooling period as the emission was assumed to be low. Compared to the Vietnam industrial emission standards (ES) for inorganic substances and dusts (MONRE 2005) applied for this type of sources, i.e. located in the suburban areas (Kv = 1.0) and for the flow rate of around 30,000 m^3 h^{-1} (Kp = 0.9; see Table 3), CO appeared to exceed ES most frequently, i.e. for 5 out of 7 days of the firing batch, whilst PM exceeded ES on 1 day and SO_2 remained below ES during the firing batch. Note that the opposite was observed for the ambient air quality where CO was not observed exceeding the NAAQS whilst SO_2 exceeded the NAAQS most frequently. A tighter SO_2 ES for the local sources in this case may be required, and considerations should be

Table 4 Stack air pollution emission from brick kilns

Parameter		Kiln 1[a] (April 2006)	Kiln 2[b] (January 2007)	Data for comparison[c]
Size (W × L × H), m^3		9 × 13 × 10	9 × 17 × 10	
Stack height, m		27	22	
Stack equivalent diameter, m		~1	~1	
Production, bricks/batch		500,000	800,000	
Volumetric flow, m^3 s^{-1} (0°C, 1 atm)		10–11	6–10.5	
Coal consumption, tonnes		45	60	
SO_2 emission	kg/1,000 bricks (kg/batch)	5.9 (2,940)	0.52 ± 0.02 (420 ± 14)	0.8–1.2
CO emission	kg/1,000 bricks (kg/batch)	12.3 (6,150)	6.35 ± 0.3 (5,080 ± 240)	11–12
PM emission	kg/1,000 bricks (kg/batch)	1.4 (695)	0.64 (510)	3.5–7.5

Average weight of finished product (fired and cool bricks)—2.2 ± 0.02 kg per piece

[a]Co et al. (2009)

[b]Average and standard deviation based on repeated measurements for each day

[c]RERIC (2003)

given to the regional sources within and outside the province when developing the management strategies to attain NAAQS for SO_2.

Results of the stack gas monitoring for kiln 2 are presented in Table 4 in comparison with kiln 1 (Co et al. 2009) and other published data. Note that in April 2006, CO monitoring was done for 4 days (third–sixth day) of the batch which also showed CO levels above the ES (1,300–1,600 mg m^{-3}). PM concentration in the flue gas during 2 days (third and fourth) was 120 and 220 mg m^{-3}, respectively, i.e. below the ES. However, much higher levels of SO_2 were obtained during this period, i.e. 50, 100, 900 and 2,200 mg m^{-3} on the third, fourth, fifth and sixth days, respectively, as compared to kiln 2. The highest SO_2 concentration in the emission of kiln 1 of 2,200 mg m^{-3} did exceed the ES. Note that the pattern of SO_2 variation of kiln 1 is similar to kiln 2 (January 2007), i.e. higher at the end of the batch.

The total emission per production batch in January 2007 was lower than in April 2006 (Table 4). Kiln 2 was larger in size and having larger production than kiln 1 hence consuming more coal. Total emission per batch estimated for kiln 1 would be 6.2 tonnes CO, 2.9 tonnes SO_2 and 0.7 tonnes of PM as compared to kiln 2 of 5 tonnes of CO, 0.42 tonnes of SO_2 and 0.51 tonnes of PM. Due to the difference in the production rate, the emission per 1,000 bricks between the two kilns differs significantly. Kiln 1 has higher emission rates than kiln 2 for all three pollutants, i.e. about 2-fold for CO and PM and above 11-fold for SO_2. Our emission results for CO appear to be comparable to the reported average emission factors of brick kilns in India (RERIC 2003) but lower for PM. In the vertical kiln type used in our study, a part of the PM emission may also be retained in the unburned brick layers located above the firing zone; thus, it may reduce the PM emission. SO_2 emission from kiln 2 was comparable to RERIC (2003) whilst SO_2 emission from kiln 1 was significantly higher (Table 4). Note that the average weight of finished product (fired and cool bricks) is 2.2 ± 0.02 kg per piece that may be used to obtain mass-based emission factors if desirable.

The obtained (uncontrolled) emission factors differ significantly between two kilns which may be due to the different operating conditions and also the quality of fuel. In particular, the SO_2 variation indicates the coal sulphur (S) content. Coal used for brick firing in this village was purchased from different mining places with the coal quality varying greatly. Before use, coal particles are mixed with some binding materials, commonly mud, and made into briquettes similar in shape to but thinner than raw bricks. These briquettes are placed in between raw bricks for firing. According to a survey conducted by AIRPET-VN (2007), the coal has an average heating value of around 5,000 Kcal kg^{-1} and varying S content starting from as low as 0.4–0.6% by weight. Assuming that 100% of S in coal would be converted to SO_2 during firing, kiln 2 used 60 tonnes coal of 0.4–0.6% S would result in 0.48–0.72 tonnes SO_2 per batch, which is close to the measurement result of 0.4 tonnes per batch. However, kiln 1 had a much higher SO_2 emission for less coal consumed that would correspond to an S content in coal of 3% if all SO_2 emission being originated from coal only. This shows that S content in coal used in the village may be highly variable.

Size distribution of suspended particulate matter

In January 2007, mass size distribution of PM was determined based on the samples collected using an eight-stage cascade impactor. The results (Table 5) show that most (65–77%) of the particles emitted were of the coarse fraction (>2.1 μm). High mass fractions were consistently observed for the size range of 2.1–5.8 μm. The contribution of finer particles (PM2.1, i.e. with size below 2.1 μm) generally increased towards the end of the firing batch, and on the seventh day, PM2.1 consists of 34.5% of the total PM mass concentration. The PM emission from brick kilns was mostly of the aerodynamic diameter less than 9 μm hence can be regarded as PM_{10}.

Modelling results

Base case emission estimate

The detailed survey made in January 2007 shows that out of 45 available kilns in the village, 21 kilns were operating during the period. These

Table 5 Mass size distribution of PM in stack gas, January 2007 (see Table 3 for PM concentration)

Size range (μm)	Mass fractions, %			
	1st day	2nd day	6th day	7th day
≥9.0	2.7	13.0	0.14	0.9
5.8–9.0	26.8	13.1	10.76	0.9
4.7–5.8	14.1	13.8	11.43	22.1
3.3–4.7	17.0	16.1	23.77	20.3
2.1–3.3	14.9	20.8	21.15	21.2
1.1–2.1	5.1	20.3	19.19	0.9
0.7–1.1	4.1	1.4	9.68	10.9
0.4–0.7	13.5	1.7	3.47	9.4
Backup (<0.4)	2.7	0.03	0.42	13.9
Coarse (>2.1 μm), %	75.4	76.8	67.2	65.5
Fine (≤2.1 μm), %	24.6	23.2	32.8	34.5

kilns were operated on various days of a firing batch, hence, for the base case, these 21 kilns were divided evenly into seven groups, each group operating on a particular day of the 7-day firing batch. Data on the exact location of kilns (by global positioning system), brick quantity per batch, amount of coal used, stack characteristic (height, diameter), kiln operation cycles etc. were collected. To account for the variation in emission rates between the kilns, the average values of the monitored two kilns (kiln 1 and kiln 2) were used with the emission rate on different days of a batch shown in Fig. 4. This was considered necessary especially for SO_2 due to the variations in S in coal used in the kilns, as discussed above. Note that only the stack kilns emission was simulated. In the model performance evaluation, the measured upwind concentration was used as the background. In the scenario study, no background concentration was considered; hence, only contribution from the kiln stacks was evaluated.

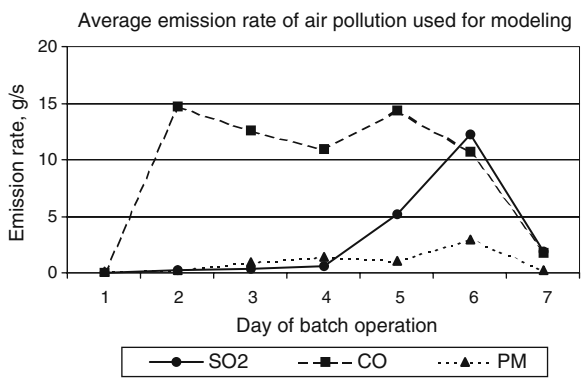

Fig. 4 Average emission rates from a brick kiln during the firing period used in modelling for the base case

Model evaluation

The input meteorology was prepared from the simultaneous measurement data into the ISCST3 format. Model evaluation was done using measurements in April 2006 (one kiln operated) and in January 2007 (base case—21 kilns operated). Only kiln emission was simulated; hence, contribution from other sources to the ambient pollution should be principally excluded. Therefore, upwind ambient measured concentrations were considered as the background levels and were subtracted from simultaneously measured downwind concentrations for a given sampling time to obtain the net contribution from the kilns. The latter were compared with the model output for performance evaluation purpose.

Scatter plots between the measured and modelled concentrations of SO_2 (Fig. 5a, b) show agreement. High correlation coefficients between the measured and modelled were observed, i.e. 0.85 and 0.98 for the April 2006 and January 2007 campaigns, respectively.

For CO and PM_{10} of January 2007 (Fig. 6a, b), the model appeared to under-predict the high values. Other local emission sources (than the kiln stacks) that were not included in the input data may be the reason for this discrepancy. In particular, many vehicles transporting the raw materials and fuels to and bricks from the manufacturing

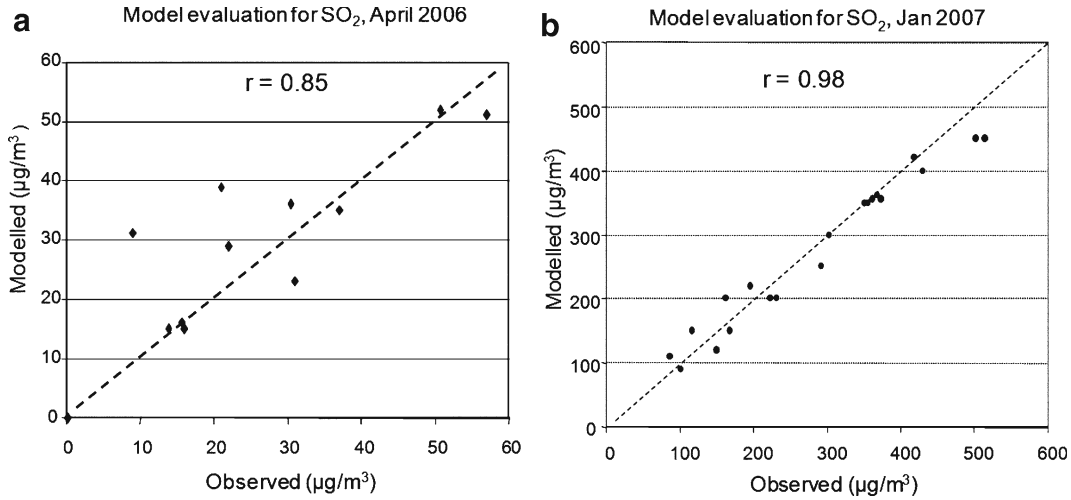

Fig. 5 Modelled vs. observed SO_2 at downwind sites (S12 and S23, respectively) when one brick kiln was operated in April 2006 (**a**) and for base case emission in January 2007 (**b**). Note the different scales in these two *figures*

village during the normal manufacturing operation in January 2007 may contribute substantially to CO and PM in the area. In addition, a substantial PM amount was re-suspended by passing vehicles. PM was also emitted from various raw material preparation activities in the village. These activities would occur simultaneously and would tend to increase with the kiln operation intensity; hence, the modelled and measured values still remained in a good correlation ($r > 0.9$). Overall, the agreement between the measured and modelled values was better for SO_2. That may be due to the fact that kiln emissions were the major SO_2 source in the study area.

Spatial distribution of pollution in base case

The dispersion pattern of the pollutants (shown in Fig. 7 for SO_2) for the base case in January 2007 agrees with the prevalent wind directions shown in Fig. 8. The prevalent wind directions in a year are SSE–SE causing the plume to extend to the NW direction. Some expansion towards the SW direction is consistent with the NE monsoon

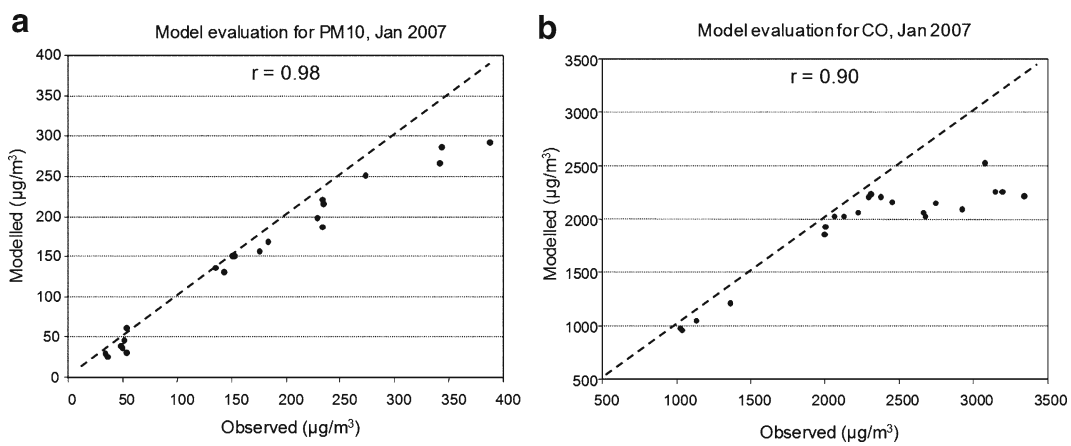

Fig. 6 Scatter plots between observed and modelled PM_{10} (**a**) and CO (**b**) for the base case emission, January 2007

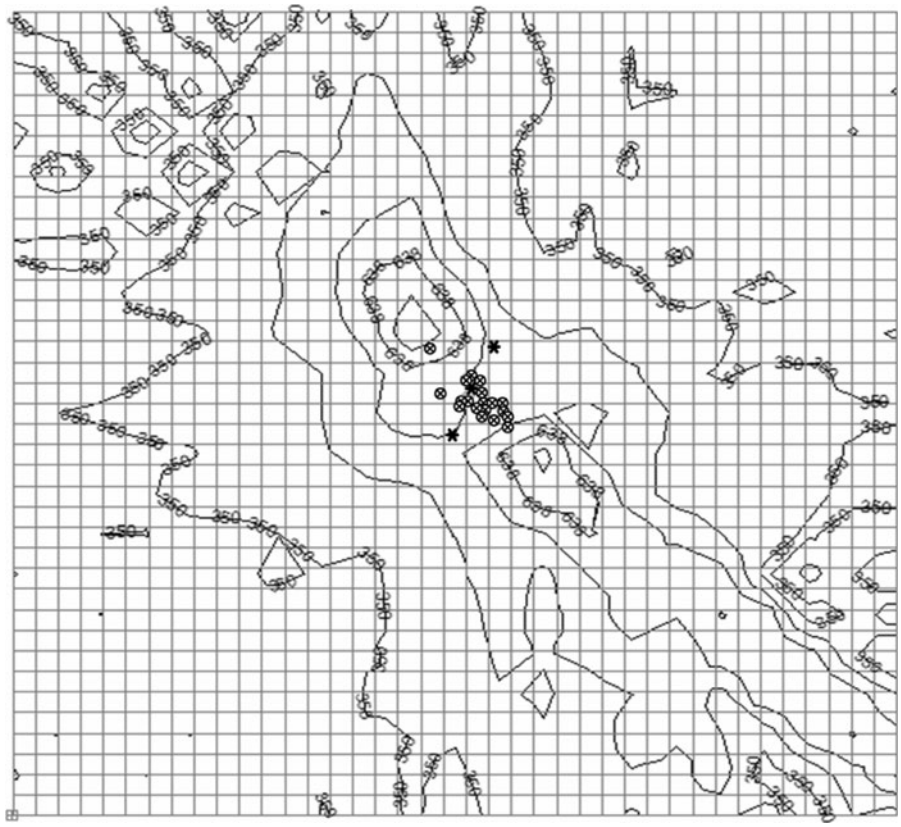

Fig. 7 The first highest SO_2 concentrations in the 10×10-km^2 model domain in the base case. The outermost contour is 1-h NAAQS of 350 μg m^{-3}. The Song Ho village is in the center of the domain. Locations of brick kilns (⊗) and ambient monitoring sites (∗) are marked

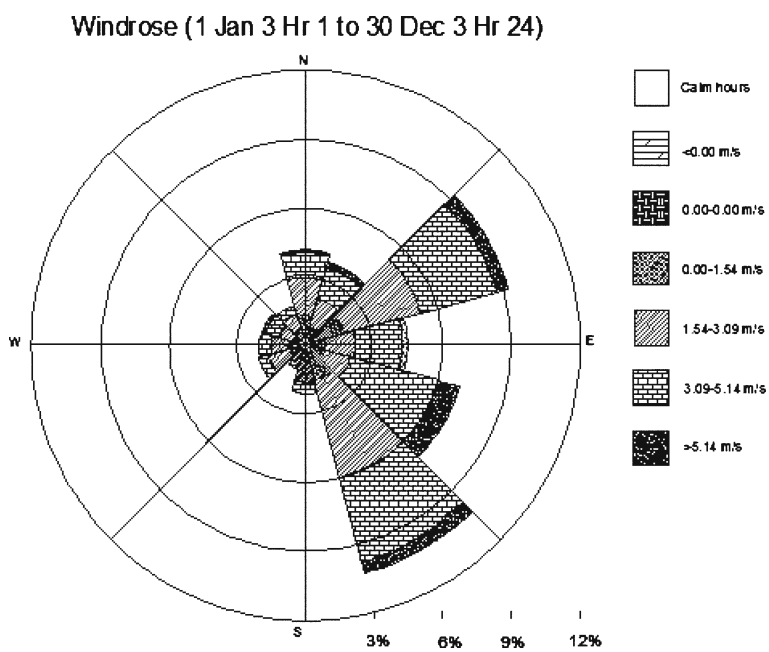

Fig. 8 Windrose for 2003 for Noi Bai Airport (NOAA data)

during winter. The outermost numbered contour is the hourly SO_2 NAAQS of 350 μg m^{-3}. Thus, a large area in the domain would have the pollutant exceeding the standard in the base case which also agrees with the ambient monitoring results presented above (Table 2).

Emission reduction scenarios

To determine the maximum number of kilns that may be operating simultaneously in the Song Ho village without violating the NAAQS, the number of kilns was reduced proportionally into 14 kilns and seven kilns. In addition, a conservative scenario was tested when all 21 kilns were assumed to operate only on the last 4 days (four each on the fourth, fifth and seventh days and nine kilns on the sixth day). Note that the highest number of kilns in the conservative scenario was assumed to operate on the sixth day when the PM and SO_2 emission rates were the highest. As mentioned above, other emissions (than the kiln stack) as well as the upwind concentration were not included in this modelling exercise; hence, the model would underestimate the ambient air pollution in this scenario study. The model was run with year 2003 meteorology data from the NOAA, as mentioned earlier.

The first highest 1 h, the first highest 24 h and the annual average ambient concentrations of CO, SO_2 and PM_{10} and the PM deposition fluxes are presented in Table 6. The highest levels, as expected, were obtained in the conservative case, followed by the base case. The lowest ambient levels were obtained when only seven kilns operated (in different days of a firing batch) for which all pollutants in the domain would not exceed the NAAQS for the entire year. However, the modelled 1-h SO_2 level was already quite close to the standard, and adding one more kiln would make the level exceed this limit.

The most serious pollutant in the study area was SO_2 which in the base case had the maximum 1-h concentration of 1,006 μg m^{-3} and 24-h concentration of 407 μg m^{-3}, i.e. well above the NAAQS of 350 and 125 μg m^{-3}, respectively. The area with the highest 1 h and 24 h SO_2 exceeding the standard was 63 and 12 km^2, respectively. However, the maximum annual SO_2 average in the domain is 36 μg m^{-3}, which was lower than the NAAQS of 50 μg m^{-3}.

For PM_{10}, the output ambient levels in the base case for all averaging times would not exceed the

Table 6 Maximum modelled levels of pollutants (micrograms per cubic metre) and the area exceeding corresponding NAAQS (square kilometres) in the 100-km^2 study domain for different emission scenarios

Parameters	NAAQS (μg m^{-3})	ISC results	Base case (21 kilns)	14 kilns	7 kilns	Conservative case
1 h PM_{10}	300 (for TSP)	Concentration	184	137	59	337
		area	0	0	0	0.19
24 h PM_{10}	150	Concentration	67	51	24	127
		area	0	0	0	0
Annual PM_{10}	50	Concentration	10	7	3	17
		area	0	0	0	0
PM annual deposition flux	No standard	Max value (g/m^2/year)	0.28	0.20	0.09	0.53
1 h SO_2	350	Concentration	1,006	524	325	1,684
		area	63	4	0	95
24 h SO_2	125	Concentration	407	202	97	723
		area	12	1	0	48
Annual SO_2	50	Concentration	36	27	15	99
		area	0	0	0	1.5
1 h CO (mg m^{-3})	30	Concentration	2.16	1.71	0.81	2.65
		area	0	0	0	0
8 h CO (mg m^{-3})	10	Concentration	1.18	0.67	0.31	1.13
		area	0	0	0	0

respective NAAQS. Thus, the measured high 1-h PM_{10} levels at the monitoring sites were possibly caused also by other sources (than kilns emission) inside and outside the village, as mentioned earlier. However, in the conservative scenario, the maximum highest 1-h PM_{10} in the domain would even exceed 1-h NAAQS for TSP of 300 $\mu g\ m^{-3}$ (no 1-h PM_{10} standard is available for comparison). For CO, similarly to the monitoring results, the modelled concentrations were all below the NAAQS in all scenarios presented.

High concentration of SO_2 in the domain would be of concern as SO_2 is known to be toxic to human health and plants thus may also eventually affect the agricultural crop yield. A significant deposition flux of PM was obtained with the annual highest flux values in the domain of around 0.28 $g\ m^{-2}\ year^{-1}$ in the base case. The effects of the deposition would be substantial since the brick manufacturing village is surrounded by rice paddies. PM deposited on the leaf surfaces would reduce the gas exchange, increase surface temperature and reduce the photosynthesis (Wark et al. 1998) that eventually would affect the crop growth and yield. In addition, acidic compounds (e.g. sulphate and nitrate) and phytotoxic pollutants, such as fluoride compounds in the deposit particles, may affect the garden and rice crops. Quantification of acidic and fluoride compounds in deposit PM should be carried out to better understand the potential effects. Deposition of phytotoxic SO_2 and other gases would be required to have a better overall picture of the effects of the brick manufacturing on the agricultural crops and the ecosystem.

The modelling results imply that, currently, no more than seven kilns should be operating simultaneously in order to meet the NAAQS of PM_{10}, CO and SO_2. This is valid when no emission reduction measures are applied. Also, kiln operation should be scheduled so that the seven are distributed evenly on different days of a firing batch. Further, if control devices are applied and low sulfur fuel is used, then more kilns can be operated. Even during the rice-growing season, the operation ban can be relieved for the kilns that apply emission reduction measures. This would create incentives for kiln owners to go for the emission reduction (Co et al. 2009).

Conclusions

High air pollution around brick manufacturing village was observed with SO_2, and PM_{10} often exceeded the NAAQS. Ambient levels of SO_2 in the study village are mainly caused by the brick firing operation whilst PM_{10} and CO may also be originated from other related activities. Pollutant concentrations in the kiln stack emission vary significantly during 7 days of a firing batch with higher emission of PM and SO_2 observed towards the end. In particular, the elevated SO_2 emission during the last 3 days of a firing batch may be due to the increase in combustion temperature as well as the release of the SO_2 that was initially absorbed in unfired brick layers above. The emission varies from one kiln to another especially for SO_2 which is most probably due to the variation in the sulphur content of coal. Most of PM emitted from kiln stacks are of PM_{10} size with the fine fraction contributing 23–35% by mass which is a health hazard and potential for the regional transport. ISCST3 model performs reasonably for the study area with a satisfactory agreement between modelling and monitoring results, especially for SO_2 that was mainly contributed by the kiln stack emission simulated by the model. The pollutant spatial distribution appears to be consistent with the prevalent wind directions in a year. If no emission reduction measures are taken, the maximum number of kilns to operate in the village should be regulated (not above 7) to avoid violation of the Vietnam NAAQS. Further assessment of the potential effects from ambient air pollution and deposition fluxes on human health and agricultural crops is required to provide information for the decision-making process to improve air quality in the area. Large fluctuations in obtained emission factors between the kilns suggest that more emission measurements are still required to produce representative ranges of the values. The emission factors obtained in this study can be useful for other places where similar brick kiln technologies exist.

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