PERGAMON

Atmospheric Environment 34 (2000) 3723–3733

ATMOSPHERIC ENVIRONMENT

www.elsevier.com/locate/atmosenv

A modelling system for predicting urban air pollution: model description and applications in the Helsinki metropolitan area

A. Karppinen[a,*], J. Kukkonen[a], T. Elolähde[b], M. Konttinen[a], T. Koskentalo[b], E. Rantakrans[a]

[a]*Finnish Meteorological Institute, Air Quality Research, Sahaajankatu 20 E, 00810 Helsinki, Finland*
[b]*Helsinki Metropolitan Area Council, Opastinsilta 6 A, 00520 Helsinki, Finland*

Received 24 November 1999; accepted 11 December 1999

Abstract

We have developed a modelling system for evaluating the traffic volumes, emissions from stationary and vehicular sources, and atmospheric dispersion of pollution in an urban area. The dispersion modelling is based on combined application of the urban dispersion modelling system (UDM-FMI) and the road network dispersion model (CAR-FMI). The system includes also a meteorological pre-processing model and a statistical and graphical analysis of the computed time series of concentrations. The modelling system contains a method, which allows for the chemical interaction of pollutants, originating from a large number of urban sources. This paper presents an overview of the modelling system and its application for estimating the NO_x and NO_2 concentrations in the Helsinki metropolitan area in 1993. A companion paper addresses comparison of model predictions with the results of an urban measurement network. This modelling system is an important regulatory assessment tool for the national environmental authorities.

Keywords: Model; Urban; Air pollution; Nitrogen oxides; Chemical transformation

1. Introduction

Urban scale dispersion modelling systems have been developed in many European countries; these have recently been reviewed by Moussiopoulos et al. (1996). The latest generation of local scale models is used in combination with meteorological pre-processing models, which are based on scaling theories of the atmospheric boundary layer (ABL). Examples of such models are the Danish OML model (Olesen, 1995a), the UK-ADMS system of the United Kingdom (Carruthers et al., 1995) and the models applied in this study, UDM-FMI (Karppinen et al., 1998c) and CAR-FMI (Härkönen et al., 1995, 1996). On the other hand, various local scale Gaussian models using the Pasquill (or equivalent) stability classes are still widely used in practical applications in many European countries.

The urban scale modelling systems should be able to allow for the various local scale effects, for instance, the influence of buildings and obstacles, downwash phenomena and plume rise, together with chemical transformation and deposition (e.g., Kukkonen et al., 1997). The modelling systems also need information concerning the hourly traffic volumes, travel speeds and emissions from various urban mobile pollution sources.

This paper describes an integrated urban pollution modelling system and discusses predicted concentration distributions of nitrogen oxides in the Helsinki metropolitan area in 1993. Some selected results have been published previously by Karppinen et al. (1997a, Karppinen et al. (1997a, 1998a, b). A companion paper

* Corresponding author. Fax: + 358-9-1929-5403.
E-mail address: ari.karppinen@fmi.fi (A. Karppinen).

1352-2310/00/$ - see front matter
PII: S1352-2310(00)00074-1

addresses the testing of the model against the results of an air quality monitoring network.

This atmospheric dispersion modelling is based on a combined application of the urban dispersion modelling system UDM-FMI and the road network dispersion model CAR-FMI (contaminants in the air from a Road), both of these models have been developed at the Finnish Meteorological Institute (FMI). Both dispersion models include a treatment of the chemical transformation of nitrogen oxides. The dispersion modelling system was refined to take into account the chemical interaction of pollutants from a large number of individual sources. This novel modelling method allows for the interdependence of urban background NO, NO_2 and O_3 concentrations, and NO and NO_2 emissions from various sources.

In a previous study by Valkonen et al. (1995, 1996), the spatial distribution of concentrations and the statistical parameters of CO, NO_2 and SO_2 concentrations were numerically evaluated in the city of Espoo in southern Finland in 1990, with the UDM-FMI model. The combined dispersion modelling system UDM-FMI and CAR-FMI has been applied in air quality assessments, which have been conducted nationally in numerous cities (for instance, Pietarila et al., 1997).

2. The mathematical models

2.1. Comparison of various modelling approaches

An advantage of Gaussian modelling systems is that these can treat a large ensemble of emission sources, dispersion situations, and a receptor grid network, which is sufficiently dense spatially (of the order of tens of meters). For instance, an hourly time series of one year is required, in order to evaluate statistical concentration parameters, which have been defined in air quality guidelines (for instance, various percentile values).

Computations of the present study contain approximately 5000 line sources, a couple of hundred stationary sources, an hourly time series of one year (8760 meteorological and emission situations) and about 10 000 receptor points. The dispersion equations therefore had to be solved separately of the order of $5 \times 10^3 \times 10^4 \times 10^4 = 5 \times 10^{11}$ times. The transformation chemistry computations increased the computational time even further. The computations therefore required 24 h CPU-time on a super-computer Cray C94. The corresponding computations using an Eulerian grid model (for instance, Yamartino et al., 1992; Nikmo et al., 1999) would not be numerically possible using the presently available computer resources.

Limitations of Gaussian urban modelling systems include that these are based on so-called quasi-steady-state assumptions (for instance, Seinfeld and Pandis, 1998). It is assumed that pollutant concentrations can be treated as though they resulted from a time sequence of different steady states (commonly taken as 1 h). However, this assumption can be invalid particularly during peak concentration episodes, caused by accumulation of air pollution in an urban area. In contrast, in Eulerian grid or Lagrangian models, meteorology, dispersion and chemistry can be described (at least in principle) in real time, selecting a suitable numerical time step.

Clearly, modelling of chemical transformation and deposition processes is not so straightforward in Gaussian modelling systems, compared with Eulerian grid models or Lagrangian models. However, some solutions have been presented (for instance, Benson, 1984, 1992; Härkönen et al., 1996).

2.2. The integrated modelling system

Fig. 1 shows the overall structure of the modelling system applied in this study. The system includes the following models: (i) the estimation of traffic volumes and travel speeds with the EMME/2 transportation planning system (INRO, 1994), (ii) the computation of emissions from vehicular sources, using the EMME/2 and LIISA systems (Mäkelä et al., 1996), (iii) the model for evaluating the emissions from stationary sources, (iv) the meteorological pre-processing model MPP-FMI, developed at the FMI (Karppinen et al., 1997b, 1998c), (v and vi) the dispersion models for stationary and mobile sources, UDM-FMI and CAR-FMI, and (vii) post-processing models, including a statistical and graphical analysis of the computed time series of concentrations.

The urban dispersion modelling system UDM-FMI takes into account all stationary sources and the road network dispersion model CAR-FMI all traffic sources, respectively. The programs have been executed on the Cray C94 supercomputer.

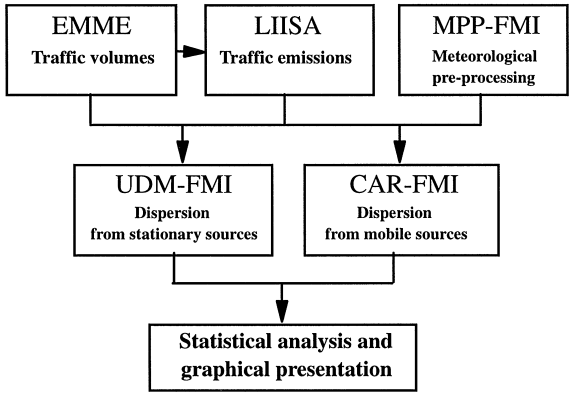

Fig. 1. An overview of the modelling system.

2.3. Traffic volumes and emissions

We conducted an extensive emission inventory in the Helsinki metropolitan area in 1993, which included the emissions from various mobile sources (road traffic, harbours and marine traffic, and aviation) and stationary sources (power plants, other point sources and residential heating). Table 1 presents a summary of the total emissions of NO_x. Approximately, one half of the total NO_x emissions originates from stationary sources.

In the spatial concentration distributions presented in this paper, we have neglected the emissions from marine traffic and from aviation, as their influence is local and in general small (about 6% of all the emissions of NO_x within the study area), compared to emissions from roads and streets. The influence of marine traffic and aviation on the NO_x and NO_2 concentrations in the Helsinki metropolitan area has been discussed in more detail by Pesonen et al. (1996).

The emissions from vehicular traffic were estimated as follows. First, the so-called traffic demand matrices (which describe the vehicle trips from every spatial zone to all the other zones within the area considered) were formed, based on a comprehensive inquiry in 1988. The matrices were first evaluated for three specific times: morning and afternoon traffic peak hours and an average non-peak hour. We then utilised the regression coefficients, which were based on traffic countings performed by the city of Helsinki and the Finnish National Road Administration. We formed 24 new matrices: 10 for weekdays, seven for Saturdays and seven for Sundays. For a more detailed description, the reader is referred to Elolähde and Koskentalo (1996).

The traffic volumes and average travel speeds of each traffic link were computed using EMME/2 transportation planning system (INRO, 1994), which assigns the trips (from zone to zone) to links of a network model. The model allows for the diurnal and daily variations both in traffic volumes and speeds, and traffic emissions.

Table 1
A summary of the NO_x emissions in the Helsinki metropolitan area in 1993

Source	Emissions (as NO_2)	
	t a^{-1}	%
Public power generation	13 305	47.7
Other point sources and residential heating systems	855	3.0
Road transport	12 090	43.3
Aviation	437	1.6
Marine traffic and harbours	1223	4.4
Total	27 910	100

The emission factors of cars in city traffic are based on traffic cycle measurements in Helsinki. The emission factors of the LIISA system (Mäkelä et al., 1996) were used for road traffic and for heavy duty vehicles.

The stationary sources considered include energy production, industry and residential heating systems. These are considered as point or area sources. The computations included 5000 line sources, 169 point sources, area sources and the regional background concentrations.

2.4. Atmospheric boundary-layer scaling

The relevant meteorological parameters for the models are evaluated using data produced by a meteorological pre-processing model (Karppinen et al., 1997b, 1998c). The model is based mainly on the energy budget method of van Ulden and Holtslag (1985). The model utilises meteorological synoptic and sounding observations, and its output consists of estimates of the hourly time series of the relevant atmospheric turbulence parameters (the Monin-Obukhov length scale, the friction velocity and the convective velocity scale) and the boundary layer height.

We have made use of the meteorological database of our institute, which contains routine weather and sounding observations. Fig. 2 shows the location of the study area, the meteorological stations and the background air quality measurement stations. We used a combination of the data from the stations at Helsinki-Vantaa airport (about 15 km north of Helsinki downtown) and Helsinki–Isosaari (an island about 20 km south of Helsinki). The mixing height of the atmospheric boundary layer was evaluated using the meteorological preprocessor, based on the sounding observations at Jokioinen (90 km northwest) and the routine meteorological observations.

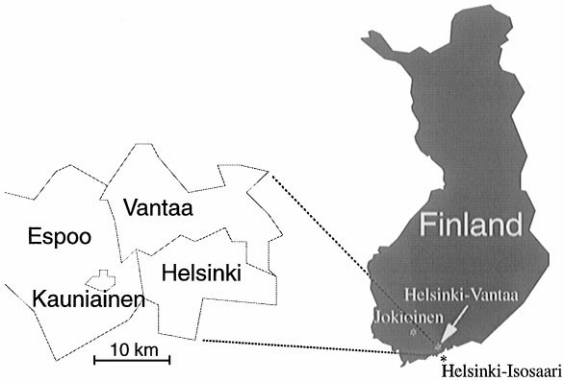

Fig. 2. The location of the cities of Helsinki, Vantaa, Espoo and Kauniainen and the meteorological stations of Helsinki–Vantaa, Helsinki–Isosaari and Jokioinen.

2.5. Road network dispersion model (CAR-FMI)

The dispersion from a road network is evaluated with the Gaussian finite-line source model CAR-FMI (Contaminants in the Air from a Road; Härkönen et al., 1995, 1996). The model includes an emission model, a dispersion model and statistical analysis of the computed time series of concentrations.

The dispersion equation is based on an analytic solution of the Gaussian diffusion equation for a finite line source (Luhar and Patil, 1989):

$$C = \frac{Q_l}{2\sqrt{2\pi}\,\sigma_z\,u\sin\theta}\left[\exp\left(-\frac{(z-H)^2}{2\sigma_z^2}\right) + \exp\left(-\frac{(z+H)^2}{2\sigma_z^2}\right)\right]\left[\operatorname{erf}\left(\frac{\sin\theta\,(p-y)-x\cos\theta}{\sqrt{2}\,\sigma_y}\right) + \operatorname{erf}\left(\frac{\sin\theta\,(p+y)+x\cos\theta}{\sqrt{2}\,\sigma_y}\right)\right], \quad (1)$$

where C is the concentration, Q_l is the source strength per unit length, u is the average wind speed, θ is the angle between the wind direction and the road, x, y and z are the coordinates, H is the effective source height, p is the half-length of the line source, erf is the error function and σ_z and σ_y are the vertical and lateral dispersion parameters, respectively. The solution (1) allows for any wind direction with respect to the road.

The dispersion parameters are modelled as function of the Monin–Obukhov length, the friction velocity and the mixing height (Gryning et al., 1987). These quantities are computed by the meteorological pre-processing model (Karppinen et al., 1997b, 1998c). Traffic-originated turbulence is modelled with a semi-empirical treatment (Petersen, 1980).

The model includes the basic reactions of nitrogen oxides, oxygen and ozone:

$$\mathrm{NO_2} + h\nu \overset{K_R}{\rightarrow} \mathrm{NO} + \mathrm{O}, \quad (2a)$$

$$\mathrm{O} + \mathrm{O_2} + \mathrm{M} \rightarrow \mathrm{O_3} + \mathrm{M}, \quad (2b)$$

$$\mathrm{NO} + \mathrm{O_3} \overset{K_F}{\rightarrow} \mathrm{NO_2} + \mathrm{O_2}, \quad (2c)$$

where M is a molecule and K_F and K_R are reaction rate constants, which are functions of ambient temperature and solar radiation intensity, as presented by Hertel and Berkowicz (1989).

The influence of hydrocarbons on the transformation of nitrogen oxides is important in the regional and long-range transport scales. In the urban scale, their influence is less significant, due to short transport times. Their influence in Northern European urban areas may be substantial in episodic conditions, during prevailing stable atmospheric stratification and low wind speed.

The system of Equations (2a)–(c) can be solved analytically (Benson, 1984, 1992). Appendix A presents a method for including the chemical transformation module corresponding to these equations into a Gaussian line source dispersion model.

The advantage of using such fairly simple chemical modules is that dispersion and chemistry computations are sufficiently efficient numerically. For the number of sources and receptor grid points applied in this study, the numerical requirements of more complex numerical chemistry schemes would be unreasonable, even for numerical program execution on a super computer.

The predictions of the CAR-FMI model have been previously compared with the results of two measurement campaigns, conducted near major roads (i) in a suburban area in the city of Espoo in 1994 (Walden et al., 1995; Härkönen et al., 1997) and in a rural area in southern Finland in 1995. For both of these measurement campaigns, the predicted NO_x and NO_2 concentrations agreed well with the experimental data in most cases considered; however, there were some specific meteorological and traffic conditions, in which there was substantial disagreement.

2.6. Urban dispersion modelling system (UDM-FMI)

The urban dispersion modelling system (Karppinen et al., 1998c, 1997b) includes a multiple source Gaussian plume model and the meteorological pre-processor. The dispersion model is an integrated urban-scale model, taking into account all source categories (point, line, area and volume sources).

For the most general case involving volume sources, it is assumed that these are evenly distributed within the volume $\Delta x\ \Delta y\ \Delta z$, i.e. $\{x, y, z \mid x \in [x_1, x_2] \wedge y \in [y_1, y_2] \wedge z \in [z_1, z_2]\}$. Assuming a total reflection of pollutants from the ground, the resulting ground-level concentration can be written as (Karppinen et al., 1998c):

$$C(x, y, 0) = \frac{Q}{\pi\,\Delta x\,\Delta y\,\Delta z\,u} \times \int_{y_1}^{y_2}\int_{x_1}^{x_2}\int_{z_1}^{z_2} \frac{\exp(-y^2/2\sigma_y^2(x))}{\sigma_y(x)} \times \frac{\exp(-H^2/2\sigma_z^2(x))}{\sigma_z^2(x)}\,\mathrm{d}z\,\mathrm{d}x\,\mathrm{d}y, \quad (3)$$

where Q is the total emission strength of the individual sources. If the sources are of equal height, the volume source reduces to an area source, and Eq. (3) can be written in terms of error functions.

As previously, the dispersion parameters are modelled as function of the Monin–Obukhov length, the friction

velocity and the mixing height (Hanna, 1985; Gryning et al., 1987). These quantities are computed by the meteorological pre-processing model.

The transformation of NO–NO_2 is described by the simple relation (Janssen et al., 1988)

$$\overline{C}(NO_2)/\overline{C}(NO_x) = B_1(1 - \exp(-\beta_1 x)), \quad (4)$$

where $\overline{C}(i)$ is the concentration of the species. B_1 and β_1 are parametrized in terms of local atmospheric conditions, ozone concentration, wind speed and season of the year (Janssen et al., 1988). Eq. (4) has been derived from measurements over a period of ten years in the Netherlands.

The model also includes a treatment of dry and wet deposition for nitrogen oxides and SO_2, plume rise, downwash phenomena, the dispersion of inert particles and the influence of a finite mixing height. The system computes an hourly time series of concentrations and statistical parameters, which can be directly compared to air quality guidelines.

The model predictions have been compared with the tracer experiments of Kincaid, Copenhagen and Lilleström, presented by Olesen (1995b). The predictions were well in agreement with the Kincaid data; somewhat larger differences were found for the Copenhagen and Lilleström data.

2.7. Chemical interaction of pollutants originating from various sources

Both dispersion models applied allow for the chemical transformation of nitrogen oxides, by considering a single plume in a background air with a uniform constitution. However, the plumes originating from various sources interact chemically also with each other. For instance, it is not uncommon that urban background O_3 concentration vanishes, caused by the depletion of O_3 in the oxidation of NO into NO_2. However, many regulatory modelling systems assume for simplicity that the urban background O_3 concentration is equal to the regional O_3 background concentration.

We have therefore developed a modelling system, which allows for the chemical interdependence of the NO_x concentrations originating from various sources and the O_3 concentrations. Fig. 3 illustrates the main principles of the modelling system.

(i) First, the modelling system evaluates the time variation of the regional background concentrations. The regional background concentrations are based on the data from the monitoring station of Luukki, situated in the North-Eastern part of the Helsinki metropolitan area. In order to filter out temporally high episodic concentrations, originating from local sources, we computed diurnal hourly average concentrations for each month. This procedure produced a matrix of 12 times 24 regional background concentration values, for each pollutant and each measurement station.

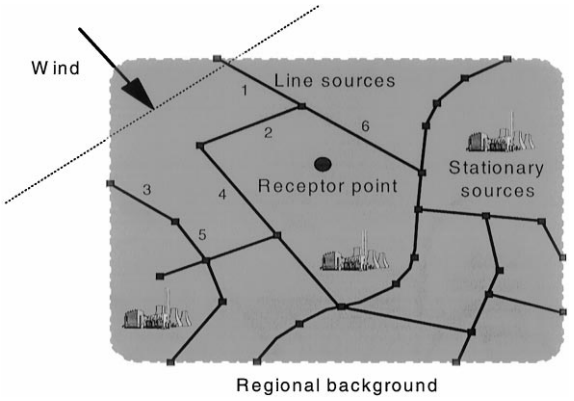

Fig. 3. A schematic presentation of the model for evaluating the chemical interaction of pollutant plumes originating from various urban sources.

(ii) Secondly, we summed the spatial pollutant distribution from all urban stationary sources to the regional background. This simplification is reasonable, as most of the emissions of stationary sources are released from higher altitudes. Their chemical transformation therefore mostly takes place before substantial interaction with pollutant plumes from other urban sources. This procedure produces a so-called first-order spatial background distribution for each pollutant (NO, NO_2 and O_3).

(iii) Thirdly, the system evaluates the contribution on the background caused by the mobile sources. The urban mobile sources and the receptor points are sorted out in terms of their location with respect to the wind direction. The plume from the most upwind mobile source (a link of road or street) is allowed to interact chemically with the first-order spatial background distribution. This produces a second-order spatial background distribution. Correspondingly, the second most upwind mobile source is allowed to interact with the second-order spatial background distribution. All the mobile sources are then treated consecutively.

The system properly takes into account, for instance, the depletion of O_3 in the urban area. The influence of the chemical interactions of the sources is largest during the rush hours and air quality episodes, in which atmospheric diffusion conditions are unfavourable. We have tested this assumption with numerical computations, using the modelling system (i) including the above-mentioned modelling method, and (ii) excluding it from the

computations. For the Helsinki Metropolitan Area in 1993, the influence of the chemical plume interactions on the hourly NO_2 concentrations during rush hours exceeds 2 and 5% at 10 and 2% of the grid points, respectively. For O_3 concentrations, the relative influence of the chemical plume interactions can be substantially larger.

3. The numerical results

Helsinki metropolitan area is situated by the Baltic Sea at the latitude of 60°N. The climate is relatively milder compared with many other areas in the same latitudes, largely because the Gulf Stream and the prevailing global atmospheric circulation have a warming effect. The Helsinki metropolitan area comprises four cities: Helsinki, Espoo, Vantaa and Kauniainen. The population of the Helsinki metropolitan area is 850 000 and the area covers 743 km^2.

The dispersion models applied are Gaussian and these do not generally take into account the influence of individual buildings on the atmospheric dispersion (although the UDM-FMI model allows for the influence of the source building itself on the dispersion). However, the terrain in the area is relatively flat and the average height of the buildings is fairly low (most buildings are lower than 15–20 m). There is only a moderate number of street canyons in the area. We have used the roughness length of 0.7 m in the numerical computations.

We have computed the concentrations of nitrogen oxides (NO_x) and nitrogen dioxide (NO_2) in the Helsinki metropolitan area for one year, 1993. The concentration time series were computed on a receptor grid, which contains approximately 10 000 receptor points. The receptor point network covers the whole area, and the largest grid intervals are 500 m. A more densely spaced grid was applied in the Helsinki downtown area, the grid interval being 100 m. In the vicinity of the major roads in the area, the smallest grid interval was 50 m. The variable receptor grid is required in order to evaluate isoconcentration curves with adequate accuracy from the computed data.

The NO_2 concentrations in Helsinki are generally comparable with those in the major Central European cities (Jol and Kielland, 1997; Kukkonen et al., 1999). In other Finnish cities the NO_2 concentration levels are usually somewhat lower than those in the capital (Kukkonen et al., 1999).

3.1. The emissions

Figs. 4a–b show the evaluated NO_x emissions of mobile and major stationary sources in the Helsinki metropolitan area in 1993. The size of the depicted area is 35 km × 25 km and the legend in the top left-hand corner

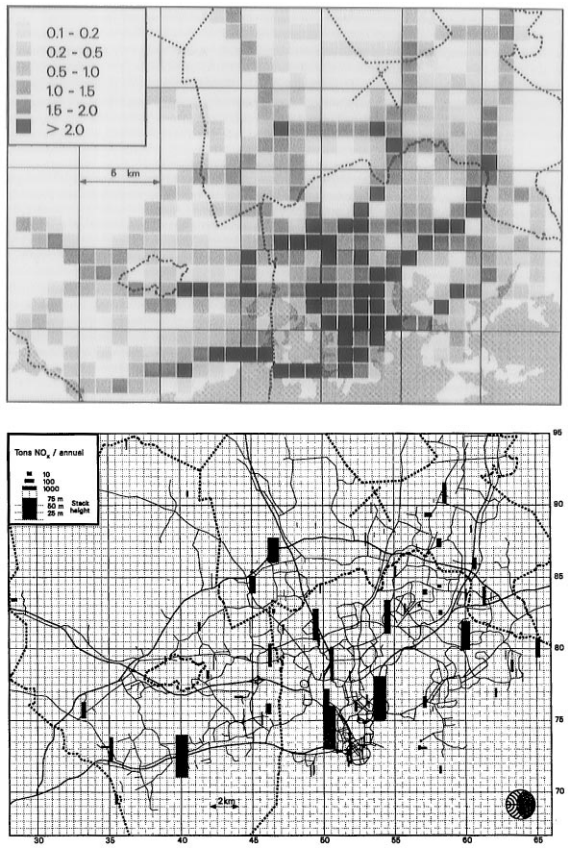

Fig. 4. NO_x emissions from mobile (upper figure) (μg m^{-2} s^{-1}) and stationary (lower figure) (t a^{-1}) sources in the Helsinki metropolitan area in 1993.

of the figures shows the absolute values of the pollutant emissions. In Fig. 4b, the height of each rectangle is proportional to the stack height, and its width is logarithmically proportional to the emission rate. The dotted lines indicate the city borders.

The emissions from mobile sources are mainly dependent on traffic densities and driving speeds. The emissions of NO_x increase continuously with increase in vehicle travel speed in the speed range from 40 to 120 km h^{-1} (Mäkelä et al., 1996), caused by the enhanced oxidation of air-originated nitrogen with the increasing combustion temperature. The largest emission densities occur in the Helsinki city centre area, and along Ring Road 1, situated at a distance of 8–10 km from the city centre. There are also substantial emissions along the major roads leading to the Helsinki city centre, and at major crossroads.

All the largest stationary sources are coal-fired power plants, also using heavy fuel oil as an additive fuel. The stack heights of the three largest power plants are about

150 m; the emissions of the largest individual unit ("Hanasaari B") are approximately 6200 tonnes NO_x (as NO_2) annually. The contribution of minor residential heating plants and industrial plants (not shown in Fig. 4b) on the total NO_x emissions is only a few per cent.

3.2. The annual average spatial concentration distributions

Figs. 5a–b show the computed annual means of NO_x and NO_2 concentrations at the ground level in the Helsinki metropolitan area in 1993. The legend in the top left-hand corner shows the absolute values of the pollutant concentration.

Clearly, the traffic emissions have a larger relative influence on the ground-level concentrations, compared to the stationary emissions, which are mostly released from higher altitudes. Although the contribution of traffic on the total emissions is slightly less than a half, approximately 80–95% of the ground level NO_x concentrations originate from traffic sources. The modest contribution of stationary sources is caused by the strong concentration of the NO_x emissions to large power plants, which have substantial stack heights, within the study area. The concentrations of NO_x and NO_2 are the highest in the vicinity of the main roads and streets, and in the downtown area of Helsinki. The figures show the distinct influence of the ring roads (situated at the distances of about 8 and 15 km from the city centre), the major roads leading to the Helsinki city centre, and the junctions of major roads and streets. The concentrations of NO_2 are strongly distributed along major roads, with higher vehicle travel speeds, due to the increase of the NO_x emissions with the travel speed, and in the downtown area, caused by the largest traffic volumes within the area.

Fig. 6 illustrates the NO_2 concentrations at the ground level in the Helsinki downtown area in 1993. Again, the influence of the main traffic routes is clearly visible.

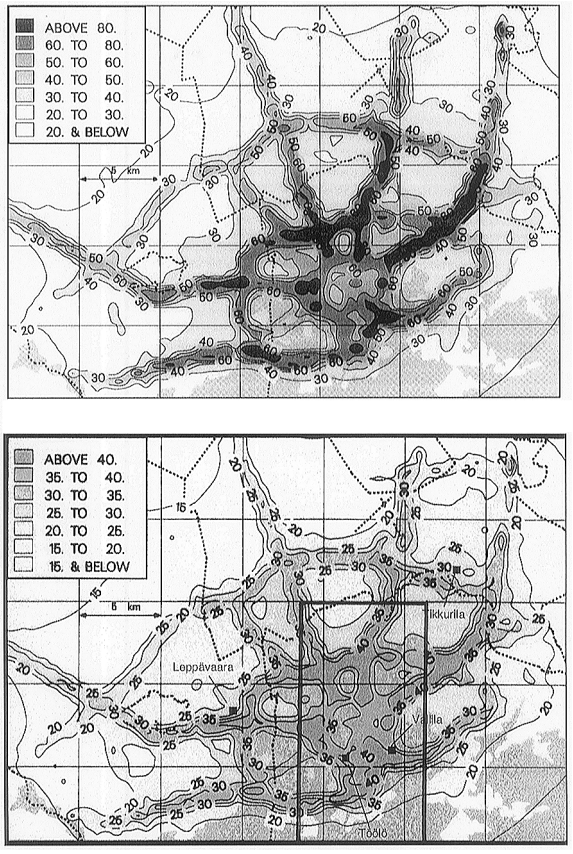

Fig. 5. Predicted spatial distribution of the yearly means of NO_x (upper figure) and NO_2 (lower figure) concentrations ($\mu g\ m^{-3}$) in the Helsinki metropolitan area in 1993. The location of the monitoring stations of YTV (Töölö, Vallila, Leppävaara and Tikkurila) has also been indicated. The size of the depicted area is 35 km × 25 km. The lower figure shows an outline of the Helsinki downtown area, presented in Fig. 6.

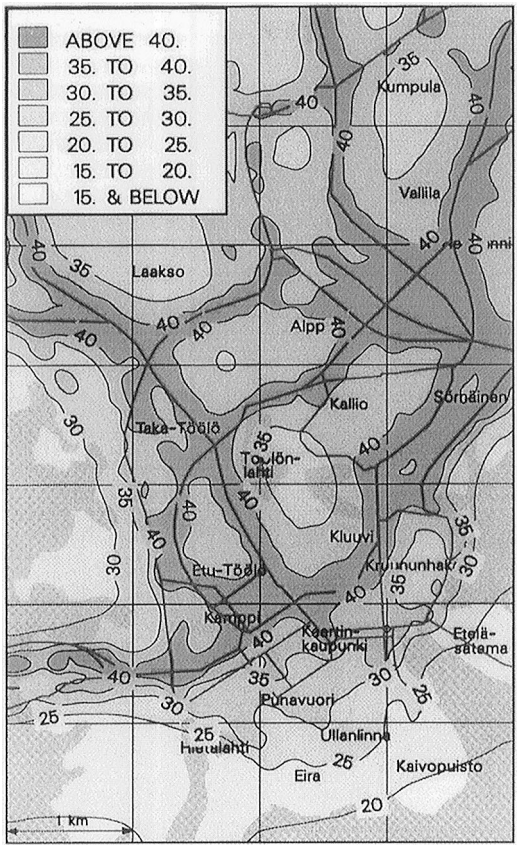

Fig. 6. Predicted spatial distribution of the yearly mean concentration of NO_2 ($\mu g\ m^{-3}$) in the Helsinki downtown area in 1993.

3.3. The chemical transformation

The ratio of NO_2–NO_x is approximately 10% in traffic emissions, and somewhat smaller in power plant emissions. Most of NO released from traffic is oxidised to NO_2 within a time scale of a few minutes (e.g., Härkönen et al., 1996), while for stationary releases this time scale is of the order of 10–20 min (Janssen et al., 1988). Far from the sources, an approximate photochemical equilibrium is reached, resulting in an NO_2/NO_x ratio of about 90% in typical ambient conditions.

The predicted concentration ratio NO_2/NO_x varies from approximately 50% in the vicinity of busy roads to approximately 90% at the outer edges of the computational regime. As expected, the NO_2/NO_x concentration ratios are substantially higher than the corresponding ratios in the emissions, even close to the traffic emission sources. This is caused by the influence of the more "aged" air masses from urban background and stationary sources, and partly also by the numerical averaging caused by the computational procedure (with grid sizes of 50–100 m).

3.4. The spatial distribution of the statistical concentration parameters

Figs. 7a–b show the predicted distribution of statistical parameters for the highest hourly 99-percentile and the second highest daily mean NO_2 concentrations in 1993. These parameters can be compared to the corresponding hourly (150 $\mu g\,m^{-3}$) and daily (70 $\mu g\,m^{-3}$) national health-based air quality guidelines. The guideline values for the daily NO_2 concentrations were exceeded fairly extensively, in the vicinity of the main roads and in the Helsinki downtown area. The corresponding guideline values for the hourly NO_2 concentrations were exceeded only at a few limited areas.

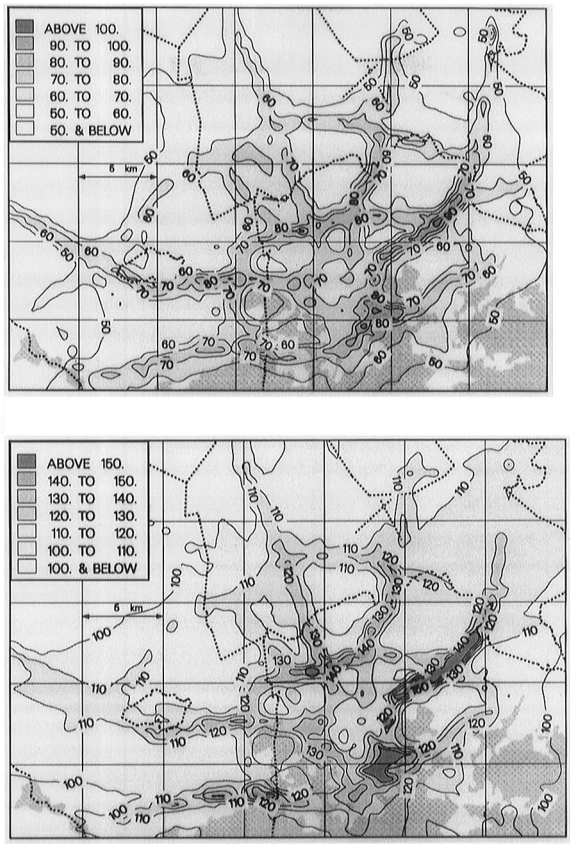

Fig. 7. Predicted spatial distribution of the second highest daily mean NO_2 concentration ($\mu g\,m^{-3}$) (upper figure) and of the highest 1-h 99-percentile NO_2 concentration ($\mu g\,m^{-3}$) (lower figure) in the Helsinki metropolitan area in 1993.

4. Conclusions

The modelling approach adopted has certain inherent limitations, both concerning the evaluation of emissions and atmospheric dispersion. Gaussian dispersion modelling does not allow for the detailed structure of buildings and obstacles. However, the terrain in the area is flat and the average height of the buildings is fairly low (most buildings are lower than 15–20 m). The computed concentrations should be interpreted as spatially averaged values (on the scale of the grid spacing, varying from 50 to 500 m), while for instance, inside a street canyon the actual concentrations can vary substantially on the scale of tens of meters.

On the other hand, the use of fairly simple dispersion models facilitates the evaluation of the hourly time series of meteorological and emission conditions for one year, which is required for the computation of statistical concentration parameters, defined in national health-based air quality guidelines. It is not at present possible to conduct such an analysis for an agglomeration of cities using, for instance, street canyon dispersion models or computational fluid dynamics models.

It was also possible to include emissions from a large number of sources (this study included 5000 line sources, 169 point sources and area sources), a substantial number of receptor points (10 000), and to use a sufficiently dense computational grid. We also allowed for the essential chemical transformation processes in the computations. The central processing (CPU) time required for the computations on the Cray C94 supercomputer was approximately 20 h.

Estimation of emissions also contains inherent limitations. Near the junctions of roads and streets there is acceleration and deceleration of the traffic flow, as well as stops and occasional congestion, which causes increased emissions. The emission modelling takes properly into

account the influence of vehicle acceleration and deceleration on the emissions. The emissions, however, are assumed to be distributed evenly along each line source in the numerical computations, although these can be strongly concentrated in the immediate vicinity of the junctions. This effect can cause an underprediction of traffic emissions near major junctions.

The modelling system contains a novel method, which allows for the chemical interaction of pollutants, originating from a large number of urban sources. The system properly takes into account, for instance, the depletion of O_3 in the urban area. This phenomenon can have a substantial influence on the computed results particularly in episodic conditions, in which the atmospheric diffusion conditions are unfavourable. However, the state-of-the-art modelling systems do not take these dependencies into account.

It can be shown numerically that although the contribution of traffic on the total emissions is slightly less than a half, approximately 80–95% of the ground level NO_x concentrations originate from traffic sources. The modest contribution of stationary sources on ground level concentrations is caused by the strong concentration of the NO_x emissions to large power plants within the study area.

The concentrations of NO_x and NO_2 are strongly distributed in the vicinity of the main roads and streets, and in their junctions. The NO_x emissions increase with the vehicle travel speed, which are substantially higher at major roads, compared with minor roads and streets. The concentrations of NO_x and NO_2 were also higher in the downtown area of Helsinki, caused by the large traffic volumes. The national air quality guidelines of the daily NO_2 concentrations were exceeded fairly extensively, in the vicinity of the main roads and in the Helsinki downtown area.

The annual average of the NO_2/NO_x concentration ratio varies from approximately 50% in the vicinity of busy roads to approximately 90% at the outer edges of the computational regime. The latter limit corresponds approximately to photochemical equilibrium conditions. The NO_2/NO_x ratio is substantially higher in the predicted concentrations, compared with the corresponding ratio in the emissions, even near the sources. This is caused by the influence of the more 'aged' air masses from urban background and stationary sources, and partly also by the numerical averaging caused by the computational procedure.

The modelling system developed has been an important assessment tool for the local environmental authorities. The system has been applied in order to evaluate the compliance of air quality with the guidelines and limit values (together with the measured concentrations), the influence of various emission categories on air quality, and the representativity of the urban monitoring stations. The modelling system has also been used in the environmental impact assessment of the different transportation system scenarios in the Helsinki Metropolitan Area (Hämekoski and Sihto, 1996).

Acknowledgements

This study is by the cooperation between the Finnish Meteorological Institute (FMI), the Helsinki Metropolitan Area Council (YTV) and the Technical Research Centre of Finland (VTT). We would like to thank our coworkers in this study, Ms. Päivi Aarnio (Helsinki metropolitan Area Council) and Mr. Juhani Laurikko (VTT Energy). We also wish to acknowledge Mr. Harri Pietarila, Mr. Risto Pesonen, Mr. Jari Härkönen and Mr. Esko Valkonen (FMI) for their cooperation. The study has been part of the national research programme "MOBILE – Energy and the environment in transportation". The funding from the Technology Development Centre (TEKES) is gratefully acknowledged.

Appendix A. Inclusion of the chemical transformation module into a Gaussian line source dispersion model.

The chemical transofrmation equations are included to the dispersion model based on a modified version of the discrete parcel method (DPM); the original version has been presented by Benson (1984, 1992). This method considers air parcels, in which the emissions and background air are assumed to be instantaneously uniformly mixed. The chemical reactions in each parcel are then assumed to proceed independently of the dispersion process.

In this model, the initial mixing zone concentration from a line source can be written as

$$C_i = C_{ia} + \frac{\chi_i Q}{u 2 H_0}. \quad \text{(A.1)}$$

where i refers to the species, C_{ia} is the ambient background concentration, χ_i is the mass fraction of the species i in the emissions, Q is the emission strength per unit length of the source (kg/(ms)), u is the wind velocity at the height H_0 and $2H_0$ is the source height. The mass fraction χ_i is needed, for instance, to take into account the fraction of NO_2 in the total NO_x emissions.

The size of the reaction volume is therefore determined by the height of the source and the wind velocity. The chemical reactions are then allowed to proceed during the travel time, i.e., the time of transport from the source to the receptor.

We suggest a revised version of the discrete parcel method, in which the reaction volume is defined separately for each receptor location, instead of the source

location. This method is named receptor-oriented DPM (R-DPM); we call the original method source-oriented DPM (S-DPM).

The relevant length scales of the reaction volume in the R-DPM are determined by the length of the finite line source, and the lateral and vertical dispersion parameters σ_y and σ_z. We define the vertical length-scale simply as $H_v(x)H_0 + k\sigma_z$, where k is a constant (selected here as $k = 3$) and x is the distance fro the source to the receptor. The horizontal length scale is defined as $H_h(x) = 2$ $(p \sin\theta + k\,\sigma_y)$, where p is the half-lengtht of the line source and θ is the angle between the wind direction and the road.

The initial concentration in the reaction volume in the R-DPM is equal to the source mass flux ($2\,p\chi_i Q$) divided by the volume flux perpendicular to the line source at the receptor location, $H_v(x)\,H_h(c)\,(u \sin\theta)$, i.e.,

$$C_i(t) = C_{ia} + \frac{\chi_i Q_p}{(H + k\sigma_2)(p \sin\theta + k\sigma_y)(u \sin\theta)}. \qquad \text{(A.2)}$$

The effluents are then assumed to be instantaneously mixed into the reaction box, and the chemical reactions are assumed to proceed during the travel time.

In bothe versions of the model, the time available for reactions is a function of the transport distance. However, for longer transport distances, more background air is available for the reactions. This is allowed for in the R-DPM, as the reaction volume is a function of the transport distance, while in the S-DPM, the reaction box volume is determined only by source properties. This is the main physical difference between the two models.

Bothe DPMs do not explicitly allow for the interaction of the chemical reactions and the physical mixing processes, or the influence of the pollutant concentration profiles within the plume on the chemical transformation. In order to allow for these effects, a substantially more complex numerical transformation model would be required (for instance, Kerminen and Wexler, 1996).

For a more detailed discussion of these models, the reader is referred to Härkönen et al. (1996).

References

Benson, P.E., 1984. CALINE4 – a dispersion model for predicting air pollutant concentrations near roadways. FHWA/CA/TL-84/15, California Department of Transportation, Sacramento, CA.

Benson, P.E., 1992. A review of the development and application of the Caline3 and 4 Models. Atmospheric Environment 26B, 379–390.

Carruthers, D.J., Edmunds, H.A., Ellis, K.L., McHugh, C.A., 1995. The atmospheric dispersion modelling system (ADMS): comparisons with data from the Kincaid experiment. International Journal of Environment and Pollution 5, 382–400.

Elolähde, T., Koskentalo, T., 1996. Calculating emissions of a transport network. Magazine of the Finnish Air Pollution Prevention Society 5–6, 26–28.

Gryning, S.E., Holtslag, A.A.M., Irwin, J.S., Sievertsen, B., 1987. Applied dispersion modelling based on meteorological scaling parameters. Atmospheric Environment 21, 79–89.

Hanna, S.R., 1985. Air quality modelling over short distances. In: Houghton, D.D. (Ed.), Handbook of Applied Meteorology. University of Wisconsin, Wisconsin (Chapter 25).

Hämekoski, K., Sihto, S., 1996. Transportation System 2020 and Air quality in the Helsinki metropolitan area, Finland. In: Baldasano Recio, J.M., Sucharov, L.J. (Eds.), Urban Transport and the Environment II. Computational Mechanics Publications, Southampton.

Härkönen, J., Valkonen, E., Kukkonen, J., Rantakrans, E., Jalkanen, L., Lahtinen, K., 1995. An operational dispersion model for predicting pollution from a road. International Journal of Environment and Pollution 5, 602–610.

Härkönen, J., Valkonen, E., Kukkonen, J., Rantakrans, E., Lahtinen, K., Karppinen, A., Jalkanen, L., 1996. A model for the dispersion of pollution from a road network. Finnish Meteorological Institute, Publications on Air Quality 23. Helsinki, 34 pp.

Härkönen, J., Walden, J., Kukkonen, J., 1997. Comparison of model predictions and measurements near a major road in an urban area. International Journal of Environment and Pollution 8 (3–6), 761–768.

Hertel, O., Berkowicz, R., 1989. Modelling NO_2 concentrations in a street canyon. DMU Luft-A131. National Environmental Research Institute, Roskilde, 31pp.

INRO, 1994. EMME/2 User's Manual. INRO Consultants Inc., Montréal, Canada.

Janssen, L., van Wakeren, J., van Duuren, H., Elshout, A., 1988. A classification of NO oxidation rates in power plant plumes based on atmospheric conditions. Atmospheric Environment 22, 43–53.

Jol, A., Kielland, G. (Eds.), 1997. Air Pollution in Europe 1997. EEA Environmental Monograph No. 4. European Environment Agency, Copenhagen, 107 pp.

Karppinen, A., Kukkonen, J., Konttinen, M., Rantakrans, E., Valkonen, E., Härkönen, J., Koskentalo, T., Elolähde, T., 1997a. Comparison of dispersion model predictions and the results from an urban air quality measurement network. In: Power, H., Tirabassi, T., Brebbia, C.A. (Eds.), Air Pollution. V. CMP, Southampton, pp. 405–411.

Karppinen, A., Joffre, S., Vaajama, P., 1997b. Boundary layer parametrization for Finnish regulatory dispersion models. International Journal of Environment and Pollution 8(3–6), 557–564.

Karppinen, A., Kukkonen, J., Konttinen, M., Härkönen, J., Valkonen, E., Rantakrans, E., Koskentalo, T., Elolähde, T., 1998a. The emissions, dispersion and chemical transformation of traffic-originated nitrogen oxides at the Helsinki metropolitan area. International Journal of Vehicle Design 20 (1–4), 131–136.

Karppinen, A., Kukkonen, J., Konttinen, M., Härkönen, J., Valkonen, E., Koskentalo, T., Elolähde, T., 1998b. Development and verification of a modelling system for predicting urban NO_2 concentrations. In: Gryning, S.-E.,

Chaumerliac, N. (Eds.), Air Pollution Modelling and its Application XXII, Vol 22. NATO, Challenges of Modern Society, New York, pp. 567–574.

Karppinen, A., Kukkonen, J., Nordlund, G., Rantakrans, E., Valkama, I., 1998c. A dispersion modelling system for urban air pollution. Finnish Meteorological Institute, Publications on Air Quality 28. Helsinki, 58 p.

Kerminen, V.-M., Wexler, A.S., 1996. The occurrence of sulfuric acid–water nucleation in plumes: urban environment. Tellus B 48, 65–82.

Kukkonen, J., Härkönen, J., Valkonen, E., Karppinen, A., Rantakrans, E., 1997. Regulatory dispersion modelling in Finland. International Journal of Environment and Pollution 8 (3–6), 782–788.

Kukkonen, J., Salmi, T., Saari, H., Konttinen, M., Kartastenpää, R., 1999. Review of urban air quality in Finland. Boreal Environment Research 4 (1), 55–65.

Luhar, A.K., Patil, R.S., 1989. A General finite line source model for vehicular pollution prediction. Atmospheric Environment 23, 555–562.

Moussiopoulos, N., Berge, E., Bohler, T., de Leeuw, F., Gronskei, K.-E., Mylona, S., Tombrou, M., 1996. Ambient air quality, pollutant dispersion and transport models. European Environment Agency, Topic Report 19, Air Quality. Copenhagen, 94 pp.

Mäkelä, K., Kanner, H., Laurikko, J., 1996. Road traffic exhaust gas emissions in Finland – LIISA 95 calculation software. (in Finnish). VTT Communities and Infrastructure, Transport Research, Research Notes 1772, Technical Research Center of Finland, Espoo, 45 pp. + App. 51 pp.

Nikmo, J., Tuovinen, J.-P., Kukkonen, J., Valkama, I., 1999. A hybrid plume model for local-scale atmospheric dispersion. Atmospheric Environment 33, 4389–4399.

Olesen, H.R., 1995a. Regulatory dispersion modelling in Denmark. International Journal of Environment and Pollution 5, 412–417.

Olesen, H.R., 1995b. Datasets and protocol for model validation. International Journal of Environment and Pollution 5, 693–701.

Pietarila H., Pesonen R., Rasila, T., Rantakrans, E., 1997. Air quality in the surroundings of the city of Turku (in Finnish). Finnish Meteorological Institute, Air Quality, Report. Helsinki, 81 pp. + A pp.

Pesonen, R., Rantakrans, E., Pietarila, H., Karppinen, A., 1996. Computations of the dispersion of nitrogen oxides in the Helsinki metropolitan area, (in Finnish). Finnish Meteorological Institute, Air Quality, Report. Helsinki, 61 pp. + A pp.

Petersen, W., 1980. User's guide for HIWAY2, a highway air pollution model. EPA-600/8-80-018, U.S. Environmental Protection Agency, Research Triangle Park, NC, 69 pp.

Seinfeld, J.H., Pandis, S.N., 1998. Atmospheric Chemistry and Physics. ISBN 0-471-17815-2. Wiley, New York, USA, pp. 945–947.

Valkonen, E., Härkönen, J., Kukkonen, J., Rantakrans, E., Jalkanen, L., Haarala, S., 1995. Application of dispersion models for evaluating the influence of urban air pollution on human health. International Journal of Environment and Pollution 5, 557–566.

Valkonen, E., Härkönen, J., Kukkonen, J., Rantakrans, E., Jalkanen, L., Haarala, S., 1996. Modelling urban air pollution in Espoo, Finland. The Science of the Total Environment 189/190, 205–211.

van Ulden, A., Holtslag, A., 1985. Estimation of atmospheric boundary layer parameters for diffusion applications. Journal of Climate and Applied Meteorology 24, 1196–1207.

Walden, J., Härkönen, J., Pohjola, V., Kukkonen, J., Kartastenpää, R., 1995. Vertical concentration profiles in urban conditions – comparison of measurements and model predictions. In: Anttila, P., et al. (Ed.), Proceedings of the 10th World Clean Air Congress, Espoo, Finland, May 28–June 2, 1995, Vol. 2. The Finnish Air Pollution Prevention Society, Helsinki, pp. 268 (4 p.).

Yamartino, R.J., Scire, J.S., Carmichael, G.R., Chang, Y.S., 1992. The CALGRID mesoscale photochemical grid model – I. Model formulation. Atmospheric Environment A (26), (8), 1493–1512.