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Introduction

Robert JOUMARD

This conference follows the Transport and Air Pollution conferences held since 1986 in Avignon, France, Graz, Austria and Boulder, USA, and the Environment & Transport conference in Avignon in 2003. It will be aimed at contributing to a systemic approach to environmental and transportation issues. The main topics are:

- The evolution of the transportation system originating the environmental impacts: vehicle fleets, mobility, infrastructures, "emissions", covering the various transportation modes (road, rail, air, sea) with a specific attention to their long-term dynamics.
- The perception of the environment by the population, experts and decision-makers, analysing the determining, the taking into account of the environment in transportation-related public policies.
- Transport-related nuisance impact on populations and ecosystems: air pollution, health effects, greenhouse effect, noise, water pollution, space and landscape, waste, fauna and flora, etc …
- The environment in the concept of sustainable development: the richness or the vacuity of the concept, its contribution to societal debates.
- The evaluation methods for environmental sustainability: impact models, environmental indicators, decision-making tools, including external costs.
- The control and reduction technologies: new fuels, vehicle technologies, urban traffic management…
- Considering the environmental issue in scenarios of transportation policies: environmental legislation, mobility policy for people and goods, sustainable scenarios.

Cette conférence est dans la lignée des colloques Transports et pollution de l'air organisés depuis 1986 à Avignon en France, à Graz en Autriche et à Boulder aux Etats-Unis, et du colloque Environnement & Transports organisé en 2003 à Avignon. Elle a pour objectif d'établir un bilan des connaissances scientifiques sur l'approche système de l'environnement et des transports. Les principaux thèmes traités sont :

- *L'évolution du système de transport à l'origine des impacts sur l'environnement : parcs, trafics et mobilité, infrastructures, "émissions", pour les différents modes de transport (route, fer, air, mer) avec une attention particulière à leur dynamique à long terme.*
- *La perception de l'environnement par la population, les experts et les décideurs, l'analyse de ses déterminants, la prise en compte de l'environnement par les politiques publiques liées aux transports.*

- *L'impact des transports sur les populations et les écosystèmes : pollution atmosphérique, effets sur la santé, effet de serre, bruit, pollution des eaux, espace et paysage, déchets, faune, flore …*
- *La place de l'environnement dans le concept de développement durable : la richesse ou la vacuité du concept, son apport aux débats de société.*
- *Les méthodes d'évaluation de la durabilité environnementale : modélisation des impacts, indicateurs environnementaux, outils d'aide à la décision, y compris coûts externes.*
- *Les moyens de contrôle technologiques : nouveaux carburants, technologies véhicule, gestion des transports urbains...*
- *La prise en compte de l'environnement dans les scénarios de politique de transports : législation environnementale, politique de mobilité des personnes et des marchandises, recherche de scénarios durables.*

Road traffic noise annoyance around intersections in Yavatmal City: Perceptions and Attitudes

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Abstract

The present study reports the evaluation study of the traffic noise impact on the quality of life among resident's around 2 five Leg signalized road intersection at Intermediate City; Yavatmal, district place in Maharashtra state (India). A total of 276 individuals in the vicinity of major two five-leg Intersections were questioned in writing for their perceptions and attitudes towards urban traffic noise. The socioeconomic characteristics of the sample population were identified and the perceived impact of noise on their welfare and health was evaluated. While significant number of individuals were aware of the interference of traffic noise with daily activities and awareness of the health impact.

Keys-words*: Road Traffic noise, annoyance, perception*

Introduction

The primary source of noise is the individual vehicle; the nuisance is caused by the accumulation of sound of individual vehicles of the traffic stream into traffic noise. Mechanized transport is one of the major pollutants of the natural environment. Noise disturbance due to traffic has detrimental effect on the tranquility of the area and is particularly annoying in the vicinity of noise sensitive areas. It can cause hearing loss, tension, anxiety, anger, Sleeplessness and host of serious problems. Poor vehicle maintenance, poor riding surface, high speed and bad driving add to noise levels.

Yavatmal's Profile

Yavatmal is a major developing urban center; district place, growing Intermediate City in the Maharashtra State (India). The city comprises of 31 municipal wards, located at 20°23' N latitude, 78°07' E Longitude. The welfare, health effect of vehicular traffic noise pollution around major two intersections were analysed with the help of questionnaire survey. In Yavatmal many medium and small-scale industries are located in this town. It is also an important commercial center. Many educational institution i.e. school and colleges are also present in the town. Hence the town supports a large number of floating populations along with its own. The number of automobiles also increases every year due to growing population. All these human activities affect the environment and cause various types of pollution. Table 1. Shows growth of the vehicle population registered in ARTO, Yavatmal.

Table 1: Vehicle population registered in ARTO Yavatmal

Sr. No	Category of Vehicles/ Year	1999	2000	2001	2002	2003	2004
1	Motorcycle	14960	18383	24564	30352	36316	43773
2	Scooter	11459	12703	13259	13516	14288	14703
3	Mopeds	20844	22491	24893	27407	28848	30058
4	Motor cars	1254	1495	2002	2394	2833	3251
5	Jeeps	2025	2119	2387	2563	2816	3052
6	Station Wagons	84	48	48	47	47	51
7	Taxi cabs	191	243	357	537	760	843
8	Auto rickshaws	2624	2881	3623	4259	5134	6043
9	Stage carriages	400	400	253	322	399	396
10	Contract carriages	292	315	326	324	242	220
11	School Buses	11	11	18	15	15	18
12	Private service vehicles	2	1	7	7	7	8
13	Ambulance	13	16	18	18	20	39
14	Goods vehicle Trucks and lorries	1832	1869	2002	2087	2249	2388
15	Tankers	103	103	103	105	105	108
16	Delivery vans (4 Wheelers)	369	427	488	515	567	697
17	Delivery vans (3 Wheelers)	192	283	396	501	637	881
18	Tractors	1789	2026	2356	2497	2611	2816
19	Trailers	1817	2061	2429	2574	2676	2849
20	Others	42	42	43	43	46	48
	Total	60303	67917	79319	89761	100616	112242

Literature Review

Increasing urbanization, high-density traffic and rapid industrialization in the last three decades has risen to a number of environmental problems including noise. Effects of noise exposure on Human beings are generally manifested in the form of cardiovascular, psychological and physiological symptoms or disorders. Some of the studies in India,on exposure to traffic noise annoyance and its effects, noise in industrial areas . A. A. Mosed, et.al[1], Dr. A. K. Gupta et.al[2], C. Ravichandran et.al[3], D. Charkrabarty et.al[4], M.J.R. Chowdary, K. Thanasekaran et.al[5], Nalini Mohan Singh and S. Narayanrao, [6], P. Ramalingeswara Rao et.al[7], R. Edison

Raja et.al[8], R.S. Nirjar et.al[9], T.N. Tiwari et.al[10] have studied noise pollution status of some intermediate cities and metropolitan cities of India. Studies investigated the problem of increasing noise pollution levels due to roads traffic flow.

Survey of Social attitudes

A newly prepared Questionnaire was distributed to heads of households/shops residing in the vicinity of two major intersections i.e. State Bank Intersection and Bus Station Intersection. A comprehensive questionnaire sought information about traffic noise traits and its effects on exposed individuals. The questionnaire addressed two main categories. In the first category, the socioeconomic characteristics of the individual were sought. These include age, marital status, occupation, education and income. The second category included individual attitudes towards traffic noise and the interference of noise with important daily activities, such as sleeping, relaxation, speaking, telephoning, eating, studying and watching Television. The individual annoyance with noise and the effect of noise on health were also included in the survey.

Socioeconomic characteristics

Thirty four percent of the sample ranged in age between 25 and 35 years. Ninety-seven and half percent were within the working age range 15-55 years. Of the interviewed individuals, 57.5% were married. The greatest majority however had a graduation and HSC education, followed by post graduation. The majority of the sample (74.5%) was comprised with an income of Rs.5000 – 15000 and 25.5 % had a monthly income more than Rs. 15000.

Problem of traffic noise

Sample individuals were requested to rank the most important transport related urban problem. The list included noise, air, vision pollution. Noise (72%)and air (23%)pollution were recognized as the most important transport related urban problem. The reasons for noise pollution were evaluated as Horn (40%) followed by Traffic jam (26%), silencer (24%) and Engine (10%). The distribution of annoyance due to vehicle categories are as 45% due to Trucks, followed by 24% due to Bus, 20% due to motorcycle and 11 % due to car/minibus. Response to the question" Does traffic noise annoy you?" showed that 89% of sample respondent were annoyed; 07% were not annoyed; and the remaining 4 % stated, " I don't know". . The period between 6.00 pm to midnight was identified by 33 % the interviewed individuals as the period when traffic noise bothered the most. The period extending from 6 am to 12 noon was the second most disturbed period of the day (32%), followed by 12noon to 6 pm period (30%), with 12 midnight to 6 am being the least disturbed period (5%).

Perceived welfare and health impact

The response distribution of the sample population, regarding interference of routine activities by traffic noise is given in Table 2. Based on the percentage of responses in the two categories of severe interference (extremely and very much), talking on the telephone and speaking were the activities most interfered with by traffic noise (59% & 58.5%) resp. Studying was third (58%), followed by other time (57 %), watching T.V. (54%) and relaxing (52.5%). Interference with the other two categories of daily activities – sleeping and eating was reported to a lesser extent. The data indicated that at least one person in four - reported severe interference with important daily activities.

Table 2: Distribution of response about interferences of daily activities by traffic noise around both the Intersections.

Activity	Distribution of Reported Interference (%)					
	None	Little	To some extent	Very much	Extremely	Total
Sleeping	9.5	21	24	21	24.5	100
Relaxing	4.5	22	21	29	23.5	100
Speaking	3.5	16	22	28	30.5	100
Telephone	3	21	17	27	32	100
Eating	6	24	23	22	25	100
Studying	5	19	18	29	29	100
Watching T.V.	4	20	22	26	28	100
Other time	8	9	26	24	33	100

The potential health impacts of traffic noise on exposure individuals are also investigated. Results are presented in Table 3. Again, based on the severely interfered response categories of extremely and very much. 53% of the sample population reported frequent headaches as a result of being exposed of traffic noise. Nervousness was reported by 58 %, as extent of exposure to traffic noise, and 50 % believed that traffic noise causes hearing damage

Table 3: Distribution of sample responses with regards to health impacts of traffic noise around both the Intersections.

Activity	Distribution of Reported Interference (%)					
	None	Little	To some extent	Very much	Extremely	Total
Headache	5	20	22	29	24	100
Nervousness	4	18	20	26	32	100
Hearing Damage	10	17	23	24	26	100

Conclusions

More than half of the total sample population around two major Five Leg intersections in Yavatmal city expressed annoyance with traffic noise during daily activities. Of these, 27.5% were ”extremely” and 26% “ very much” annoyed, followed by 22% to “some extent”, 19 % “ little” and 5.5% “none” annoyed. The reported annoyance level reached its maximum during the evening hours for nearly 33% of the sample population. Reported interferences of traffic noise with routine activities were an order of significance, telephone, speaking, studying, other times, watching T.V., relaxing, eating and sleeping. While more than one in two sample reported that traffic noise caused headache, nervousness and hearing as a result of exposure to noise. Individuals in higher income group reported a much higher level of annoyance with traffic noise than those in lower income groups. The same was observed for the level of education.

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Gestion et valorisation des sédiments issus de l'assainissement pluvial en domaine routier

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Résumé

Pour limiter l'impact de la pollution liée au transport, des bassins de rétention et d'infiltration des eaux circulant sur les chaussées sont mis en place le long des routes et autoroutes. Pour restituer la fonction de ces ouvrages, il est nécessaire de curer les sédiments qui s'y sont accumulés. Face à des volumes importants et une pollution organique (hydrocarbures et HAP) et minérale (métaux lourds) de ces sédiments, l'enjeu économique et environnemental est important. L'objectif est de proposer aux gestionnaires des techniques de traitement permettant une valorisation des sous produits. Pour cela, une accumulation de la pollution dans les particules fines des échantillons permettrait de valoriser les particules grenues. Nos recherches s'orientent vers la conception d'un pilote basé sur la technique d'attrition et la séparation granulométrique. En parallèle, la dégradation des hydrocarbures par Landfarming est étudiée. Les sédiments valorisables doivent répondre à un cahier des charges précis et rigoureux.

***Mots clés** : sédiments, pollution, métaux lourds, hydrocarbures, traitement, valorisation*

Abstract

Valorization and management of sediments from road ponds

The development of cities and road networks over the past several decades has induced an increase in waterproofing efforts, making road runoff water management one of the priorities in the field of urban planning. For several decades, retention ponds have been built along highways and motorways, as well as in urban environments. The sediments that accumulate in the basins must then be dredged in order to maintain or restore the functions of such structures. These sediments are frequently contaminated by metallic and organic pollutants and can potentially be harmful to the environment and human health when their concentration is high. There is a lack of knowledge in the field of sediment management. The objective of this study is to characterize the sediments from stormwater sewerage in order to provide project managers with technical solutions for the treatment and reuse of these sediments.

The sediments from 2 stormwater ponds were studied. A detailed description of the ponds is given in Durand (2003). The Wissous urban retention pond is located in the Villemilan Industrial Park, east of the A6 motorway near Paris. The Cheviré pond is a motorway infiltration pond located SW of Nantes, France.

Particle size analysis of the sediment indicate that the sediment is very fine in the case of Wissous (D50: 27 µm) and coarser for Cheviré (D50: 120 µm). Metal concentrations are high and always exceed the Dutch target value for polluted soils. Zinc concentration are higher than the intervention value, this is also the case for Cu and Ni concentrations in the Wissous sediment.

After attrition, a strong decrease in organic matter (OM) and metal pollution is seen, e.g. OM drops from 6.2 to 0.8 % , Cd from 36 to 6 % and Zn from 27 to 3 % (table 6). Therefore, attrition allows to concentrate the pollutants in the finest fraction, with concentrations in the > 1250 and 250-1250 µm below the Dutch standard values, which is undoubtedly an advantage in view of a future treatment of the sediment. Further tests are needed in order to find the best fraction in which a maximum of pollutants will be concentrated.

Landfarming can help reduce the OM content, and particulary organic compounds, of a sediment. This technique was used for the Cheviré and Wissous sediments. After 2 months 83 % of the hydrocarbons were eliminated in the case of Wissous, 77 % in the case of Cheviré. However, the concentrations remain higher than the Dutch standards. The study goes on and the next months will allow to see if the decay goes on.

Relying on these results, a mobile pilot plant based on attrition and particle fractionation is under study in order to separate recoverable coarse particles and polluted fine particles. The recoverable sediments must be consistent with rigorous specifications.

Key words : *sediments, pollution, trace metals, hydrocarbons, treatment, valorization.*

Introduction

Le développement croissant des réseaux routiers et autoroutiers au cours des dernières décennies et l'imperméabilisation qui en résulte ont fait de la maîtrise du ruissellement l'une des priorités environnementales. Pour gérer ces effluents, des fossés et des bassins de retenue des eaux pluviales sont fréquemment mis en place, et ce depuis plus d'une vingtaine d'année (Nightingale, 1987; Youssef et coll., 1990). Ils jouent un rôle à la fois sur les débits (écrêtement des pics) et sur la qualité des effluents (sédimentation des matières en suspension). Les sédiments qui s'y accumulent doivent être curés pour maintenir ou restituer les fonctions de ces ouvrages. Cependant, les matériaux collectés lors de ces opérations d'entretien peuvent être contaminés (métaux lourds, hydrocarbures, pesticides…) et présentent un risque pour l'environnement et la santé humaine. Plusieurs travaux (Petterson, 1999; Bäckström, 2001; Färm, 2001; Durand, 2003) montrent un intérêt particulier pour ces sédiments mais les gestionnaires de bassins sont confrontés à un manque global de connaissance pour permettre la gestion et la valorisation de leurs sous produits. Une estimation des quantités de sédiments présents dans les fossés et bassins demeure très difficile à obtenir. Plusieurs explications peuvent être données ; parmi celles-ci la diversité des gestionnaires (directions départementales de l'équipement, collectivités, sociétés d'autoroutes…) n'aide pas à une évaluation globale ; par ailleurs, l'estimation volumique est parfois difficile (bassin en eau). Une extrapolation grossière, issue d'estimations (tab. 1) ou de diverses enquêtes (tab. 2), à l'ensemble du territoire français, permet d'obtenir des tonnages en sédiments provenant des réseaux routiers et autoroutiers. Bien que les différentes estimations donnent des résultats très hétérogènes entre les quantités déposées ou curées, elles restent très importantes (jusqu'à 5 millions de tonnes environ). A titre de comparaison, la production annuelle française de boues des stations d'épuration des eaux résiduaires urbaines (matière sèche) est estimée à 1,3 millions de tonnes en 2005 (IFEN, 2001). Face à ces tonnages élevés, l'évacuation et le devenir des résidus constituent un enjeu important pour les collectivités. L'objectif de cette étude est de caractériser ces sous produits afin de proposer des techniques de traitement appropriées respectant des critères environnementaux et économiques. Les produits traités devront répondre à des cahiers des charges bien précis adaptés à la filière de valorisation choisie.

Tableau 1 : Estimation des dépôts sur la voirie routière nationale (Fournier, 2000).

Table 1 : Estimation of an average annual sedimentation rate extrapolated on the French road network (Fournier, 2000).

	Linéaire (km)	Largeur (m)	Estimation du dépôt ($g/m^2/j$)	Quantité de sédiments (million de tonnes/an)
Autoroutes	7 500	25 à 30	2,5 à 4	0,17 à 0,33
Routes nationales	26 000	8 à 9	2,5 à 4	0,18 à 0,34
Routes départementales	300 000	6 à 7	-	-

Tableau 2: Estimation des quantités de sédiments dans les bassins et fossés en France.

Table 2: Estimation of the mass of sediment in road ponds and ditches in France.

Références	Ouvrages de curage	Tonnes de matière sèche par an	Bilan (million de tonnes/an)
SETRA (1995)	Bassins Fossés	52 000 4 145 454	4,2
ONR (2001)	Bassins autoroutiers Bassins routes nationales Fossés	3 135 000 2 280 000 304 320	5,7

Méthode

1. Sites expérimentaux

Deux sites expérimentaux ont été étudiés :

Figure 1 : Bassin de décantation de Cheviré en eau.

***Figure 1 : Cheviré infitration pond - wet weather*.**

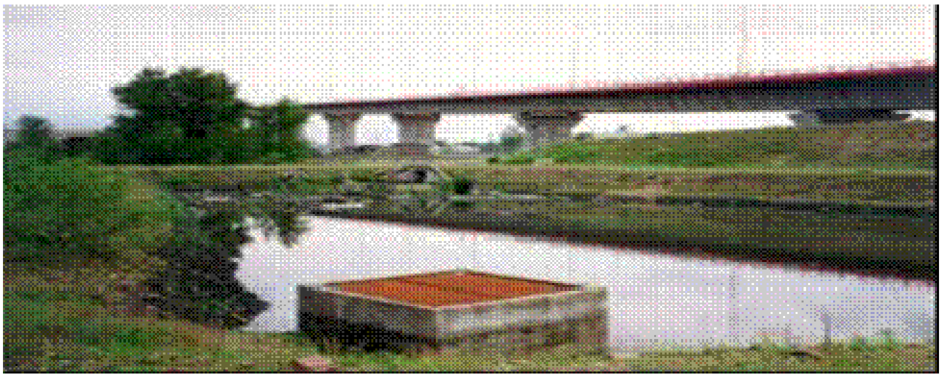


Figure 2 : Bassin de décantation de Cheviré à sec.

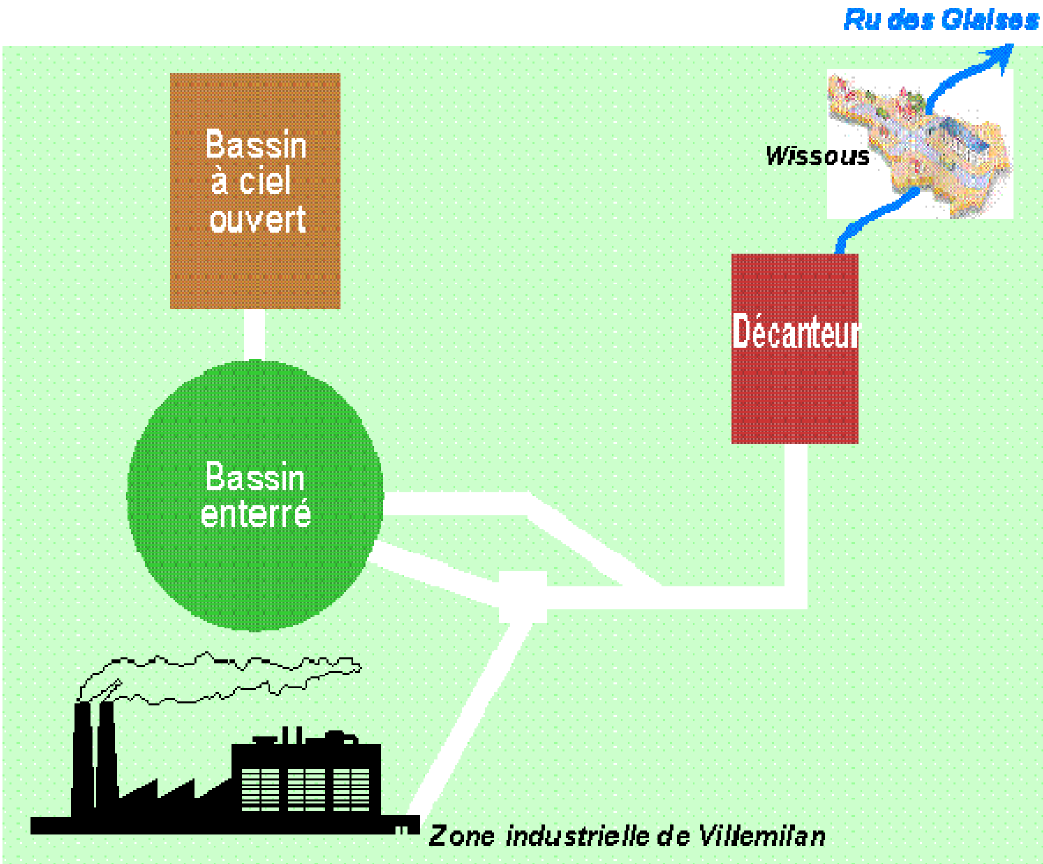
Figure 2 : Cheviré infiltration pond – dry weather.

Le bassin de décantation des eaux pluviales de Cheviré (fig. 1 et 2) est localisé au sud ouest de Nantes. De type routier, il draine le pont de Cheviré qui permet le franchissement de la Loire par la rocade ouest. Sa longueur est de 1562 m avec deux rampes à 6 % qui élèvent le tracé de 50 m au dessus du fleuve. Le pont est constitué de 2 chaussées à trois voies avec un séparateur central en béton. Le profil transversal a une largeur totale de 24,60 m. La surface de drainage des eaux est de 38 425 m2. Le trafic supporté par le pont est d'environ 70 000 véhicules par jour. Pour la partie sud de l'ouvrage, les eaux de ruissellement sont dirigées dans un bassin de décantation dont la surface est de 780 m2 et la profondeur d'environ 1,50 m. Le bassin n'étant pas imperméabilisé, une partie importante des eaux peut s'évacuer par infiltration dans le sol. Le niveau de l'eau connaît d'importantes variations au cours de l'année et le bassin peut s'assécher entre les périodes pluvieuses. Une couche de sédiments plus ou moins épaisse recouvre le fond du bassin.

Le bassin de décantation des eaux pluviales de Wissous (Fig. 3), a été crée en 1999 pour pallier aux inondations. Il est de type urbain et se situe dans la zone industrielle de Villemilan à l'est de l'autoroute A6, en région parisienne. Le dispositif mis en place est composé d'une vanne, d'un bassin enterré, d'un bassin à ciel ouvert (utilisé pour des précipitations exceptionnelles) et d'un ouvrage de dépollution (décanteur lamellaire). L'eau décantée est rejetée dans le milieu naturel. Les sédiments issus de la décantation sont acheminés vers des silos.

Figure 3 : Schéma du bassin de Wissous.

Figure 3 : Scheme of the Wissous pond



2. Protocoles Analytiques

Les analyses ont été effectuées sur la fraction inférieure à 2 mm du sédiment, selon les normes AFNOR (1999), les protocoles détaillés sont décrits dans Durand (2003). Les procédures d'assurance qualité en vigueur à la section Caractérisation et transfert de polluants du LCPC ont été appliquées ; elles concernent la préparation des échantillons pour les essais, l'exécution des essais, la conservation des produits pour les essais et l'archivage. De plus, la qualité est contrôlée par l'intermédiaire d'essais à blanc (pour toutes les méthodes appliquées au laboratoire), d'un contrôle qualité interne (solutions et matériaux de référence), et d'un contrôle qualité externe (essais inter laboratoires sur les eaux et les sédiments).

3. Attrition

Les essais de laboratoire sont réalisés en batch dans une cellule octogonale avec une capacité de 1 litre. La cuve comporte des arêtes vives et des palles d'hélices en opposition. La vitesse de rotation des palles est 2125 tr/min. La pulpe attritée contient 61,7 % de matière sèche.

4. Landfarming

Les essais sont réalisés dans des bacs en acier poudré avec un revêtement époxy ; les dimensions sont les suivantes : L=1,0 m, l=0,50 m, H=0,50 m. Un robinet est placé au fond du bac pour récolter les eaux de percolation qui seront analysées. 0,05 m de sable sont placés au fond du bac pour permettre un bon drainage puis le sédiment est déposé sur une hauteur de 0,40 m. Un retournement hebdomadaire est réalisé pour permettre une meilleure aération de la structure. Des prélèvements sont réalisés tous les mois sur toute la hauteur en 4 ou 5 points du bac pour l'analyse des sédiments. Un ajout d'engrais apporte les teneurs en C/N/P nécessaires au milieu biologique. Enfin, un mélange de produit dopant est ajouté dans certains bacs avec pour composition 50 % de gazole, 40 % d'huile de vidange et 10 % d'essence. Les compositions des différents bacs avant traitement sont résumées dans le tableau 3.

Tableau 3 : Composition des bacs et évolution de la teneur en hydrocarbures dans les sédiments lors des essais de landfarming.

Table 3 : Composition of the tanks and evolution of the hydrocarbon concentrations in the sediments during Landfarming.

	1	2	3	4	5
Sédiments	Cheviré	Cheviré	Cheviré	Wissous	Wissous
Dopant	Non	Oui	Oui	Non	Non
Paille	Oui	Non	Oui	Non	Oui
% Matière Sèche (M.S)	75	75	75	36	36
% Matière Organique (M.O)	8	12	12	24	24
Hydrocarbures totaux (HcT) en mg/kg à t = 0	7674	38570	38570	16423	16423
HcT en mg/Kg à t =1 mois	1499	11384	9670	7293	3397
HcT en mg/Kg à t =2 mois	1288	3217	3291	3789	4046

Résultats

1. Caractérisation des sédiments

Le pH est proche de la neutralité avec une valeur de 7,2 pour le sédiment de Wissous et de 6,9 pour le sédiment de Cheviré. Le pourcentage de matière sèche varie de 36 % pour Wissous à 75 % pour Cheviré et le pourcentage de matière organique de 24 % pour Wissous à 12 % pour Cheviré. Les sédiments sont fins avec un D50 de 27 µm pour Wissous et 120 µm pour Cheviré ; ils sont notamment plus fins que les sédiments de voiries (Durand et coll., 2003).

Les concentrations en métaux traces des sédiments des différents bassins sont présentées dans le tableau 4 et comparées aux valeurs cibles et d'intervention de la norme hollandaise pour les sols pollués. Si ces seuils n'ont pas de valeur légale en France, ils sont néanmoins fréquemment utilisés comme valeur de référence pour interpréter la présence de certains composés chimiques dans les sols. Les sédiments du bassin de Wissous sont très fortement contaminés par les éléments traces et plus précisément par le cuivre (324 $mg.kg^{-1}$), le nickel (499 $mg.kg^{-1}$) et le zinc (1575 $mg.kg^{-1}$) dont les teneurs sont 1,5 à 2,5 fois plus importantes que les valeurs d'intervention de la norme hollandaise. Ils présentent également des teneurs élevées en cadmium (5 $mg.kg^{-1}$), en chrome (348 $mg.kg^{-1}$) et en plomb (323 $mg.kg^{-1}$), proches des valeurs d'intervention. Le bassin de Cheviré est également pollué par les éléments traces, notamment par le zinc (1847 $mg.kg^{-1}$, soit 2,6 fois la valeur d'intervention de la norme hollandaise). Les teneurs en cuivre (271 $mg.kg^{-1}$) et en plomb (419 $mg.kg^{-1}$) sont légèrement supérieures à la valeur d'intervention pour le premier et légèrement inférieures pour le second. Le cadmium (1,8 $mg.kg^{-1}$), le chrome (88 $mg.kg^{-1}$) et le nickel (38 $mg.kg^{-1}$) ont des concentrations proches ou inférieures aux valeurs cibles.

Tableau 4 : Concentrations en éléments traces dans les sédiments ($mg.kg^{1}$ de matière sèche) et comparaison avec les valeurs de références (Durand et coll., 2004).

Table 4 : Trace metal concentrations in the sediment in $mg.kg^{-1}$ of dry matter and comparison with reference values (Durand et coll., 2004).

		Cd	Cr	Cu	Ni	Pb	Zn
Cheviré		1,8	88	271	38	419	1847
Wissous		4,5	348	324	499	323	1575
Norme hollandaise*	Valeur cible	0,8	100	36	35	85	140
	Valeur d'intervention	12	380	190	210	530	720

* Il existe une valeur cible et une valeur d'intervention de la norme hollandaise pour les sols pollués.

** There is a target value and an intervention value of the Dutch standard for polluted soils.*

2. Traitement des sédiments

Le tableau 5 présente les teneurs en éléments traces du sédiment de Cheviré dans 3 fractions granulométriques après tamisage à 250 et 1250 µm. Bien que les seuils de coupures soient différents, les résultats sont contraires à ceux de Durand (2003) qui observait peu de différence entre les teneurs de la fraction grossière (> 315 µm) et celles de la fraction fine (< 40 µm) (Durand, 2003). Les concentrations en éléments traces de la fraction fine (< 250 µm) sont très supérieures à celles des deux autres fractions. Ces deux dernières, bien que moins polluées, présentent des teneurs élevées. Des essais d'attrition ont été réalisés pour essayer de dépolluer la fraction intermédiaire comprise entre 250 et 1250 µm. Les résultats sont présentés dans le tableau 6. On note tout d'abord une production de particules fines issues de cette étape d'attrition avec un pourcentage massique de la fraction inférieure à 250 µm qui augmente de 34,1 % à 46,0 %. La fraction attritée présente une diminution importante de polluants avec des pourcentages massiques en éléments traces très faibles ; 6 % contre 36 % pour le cadmium, 8 % contre 28 % pour le chrome, 3 % contre 27 % pour le cuivre, 9 % contre 30 % pour le nickel, 8 % contre 32 % pour le plomb et 3 % contre 28 % pour le zinc. Les concentrations en éléments traces de la fraction attritée sont toutes inférieures aux valeurs cibles de la norme hollandaise. De plus, la teneur en matière organique totale de cette fraction a diminué de 6,2 à

0,8 % (tab. 6). L'attrition a permis une réduction très significative de la pollution avec des teneurs en éléments traces toutes inférieures aux valeurs cibles de la norme hollandaise et une concentration en matière organique inférieure au pourcent. Ces essais confirment la relation entre pollution organique et minérale.

Tableau 5 : Pourcentage massique (matière sèche), de matière organique (M.O) et pourcentage des éléments traces dans les différentes fractions granulométriques avant attrition.

Table 5 : Mass percent (dry matter), organic matter percent (OM) and trace element percents in the different particle size fractions before attrition tests.

Taille (µm)	%	% M.O	Cd	Cr	Cu	Ni	Pb	Zn	Fe	Al	Mn
> 1250	21,7	6,4	18,2	12,2	13,7	12,9	14,4	12,8	19,5	26,7	15,5
250-1250	44,3	6,2	36,5	27,8	26,7	29,4	32,0	27,3	41,2	47,7	31,5
< 250	34,1	17,1	45,3	60,0	59,6	57,6	53,6	59,9	39,3	25,5	53,0

Tableau 6 : Pourcentage massique (matière sèche), de matière organique (M.O) et pourcentage des éléments traces dans les différentes fractions granulométriques après attrition.

Table 6 : Mass percent (dry mass), organic matter percent (OM) and trace elements percent in the different particle size fractions after attrition tests.

Taille (µm)	%	% M.O	Cd	Cr	Cu	Ni	Pb	Zn	Fe	Al	Mn
> 1250	21,7	6,4	20	11	14	12	14	13	19	34	16
250-1250	32,3	0,8	6	8	3	9	8	3	14	12	13
< 250	46,0	20,0	74	81	84	79	78	84	67	55	71

Les teneurs en hydrocarbures totaux des cinq bacs sont dosées au début du traitement puis une fois par mois pour suivre la biodégradation des hydrocarbures dans les sédiments (tab. 3). Pour la case 1 composée du sédiment de Cheviré et de 5 % de paille, la teneur initiale de 7674 $mg.kg^{-1}$ d'hydrocarbures totaux diminue de 80 % le premier mois pour atteindre la valeur de 1499 $mg.kg^{-1}$ et de 15 % le second avec 1288 $mg.kg^{-1}$ d'hydrocarbures restant. Les cases 2 et 3 sont composées du sédiment de cheviré, d'un mélange d'hydrocarbures et de paille pour la case 3 avec une teneur initiale en hydrocarbures de 38570 $mg.kg^{-1}$. Au cours du premier mois, 70 à 75 % des hydrocarbures sont éliminés puis 65 à 70 % lors du second mois avec une teneur finale de 3217 $mg.kg^{-1}$ pour la case 2 et 3291 $mg.kg^{-1}$ pour la case 3. Pour les cases 4 et 5 composées du sédiment de Wissous et de paille pour la dernière, la teneur initiale est de 16423 $mg.kg^{-1}$ d'hydrocarbures totaux. Pour la case 4, 55 % des hydrocarbures sont éliminés contre 80 % pour la case 5 après 1 mois de traitement (tab. 3). Les teneurs en hydrocarbures totaux après deux mois de traitement sont similaires entre les deux bacs avec des valeurs comprises entre 3800 et 4000 $mg.kg^{-1}$. Les résultats pour les 5 cases montrent une élimination importante des hydrocarbures totaux (55 à 80 %) lors du premier mois puis une diminution plus faible le deuxième mois (15 à 70 %). Deux hypothèses peuvent expliquer ce phénomène ; la première est liée aux conditions météorologiques avec des températures et un ensoleillement plus important le premier mois ce qui favorise les voies d'élimination des hydrocarbures et la seconde correspond à une volatilisation de certains hydrocarbures en début de traitement. La dégradation des hydrocarbures est importante mais les teneurs sont encore très élevées (1288 à 4046 $mg.kg^{-1}$) si on les compare aux valeurs cibles et d'intervention de la norme hollandaise (50 à 600 $mg.kg^{-1}$). L'ajout de paille, agent structurant, dans certains bacs n'a pas montré d'effet important sur la dégradation excepté lors du premier mois pour le sédiment de Wissous (très humide).

Discussion

Les sédiments présentent une granulométrie très fine et des teneurs élevées en matière organique et polluants métalliques. L'attrition a permis de localiser la pollution au niveau des particules fines présentes dans l'échantillon. La concentration de la pollution dans une fraction granulométrique est un avantage indéniable pour un traitement ultérieur des sédiments de l'assainissement pluvial. Des essais complémentaires sont nécessaires pour déterminer le seuil de coupure optimal afin de réduire le pourcentage de particules fines et de concentrer la pollution dans un volume de produit minimum. De plus, seul le sédiment de Cheviré a été traité par attrition ; d'autres tests sur différents sédiments seront effectués pour vérifier si cette technique présente des résultats similaires sur une gamme hétérogène de produits. Les teneurs en polluants minéraux et organiques dans les différentes fractions ainsi que les résultats obtenus par les tests d'attrition permettent d'orienter nos recherches vers la conception d'une unité pilote basée sur une séparation granulométrique pour isoler les fractions et sur l'attrition pour désagréger les agglomérats de particules fines. Trois principales fractions seront extraites de l'unité en fonction de la granulométrie ; la première, une fraction grossière de plus de 5 mm composée des encombrants sera dirigée vers la filière des produits ménagers, la seconde avec une granulométrie comprise entre 80 µm et 5 mm constituera la fraction valorisable et une troisième composée des particules fines (< 80 µm) sera éliminée par incinération ou mise en centre d'enfouissement technique.

En complément du traitement physique, la fraction organique des sédiments peut être traitée par Landfarming, technique simple, à moindre coût mais d'une durée importante (1 à 3 ans). Il existe deux voies principales d'élimination des hydrocarbures, la volatilisation et la biodégradation. La première est problématique car les hydrocarbures vont se redéposer ailleurs ; pour cela, des essais seront mis en place pour étudier le pourcentage de volatilisation lors de l'élimination des hydrocarbures. Les deux premiers mois de traitement ont permis une élimination importante avec 83 % pour Wissous et 77 % pour Cheviré des hydrocarbures présents dans les sédiments. Les concentrations après deux mois sont encore très supérieures aux seuils de la norme hollandaise. La dégradation dépend des paramètres climatiques (température, humidité…) mais aussi de la nature des hydrocarbures (linéaires, ramifiés, cycliques…). La dégradation des premiers hydrocarbures (linéaires et ramifiés) est souvent relativement simple alors qu'elle est beaucoup plus difficile pour les cycliques ou aromatiques. Les prochains mois permettront de déterminer si la dégradation des hydrocarbures se poursuit.

Conclusion

L'étude menée sur les bassins de Wissous et Cheviré a permis de mettre en évidence la pollution organique et métallique des sédiments issus de l'assainissement pluvial. Des solutions doivent être apportées pour gérer les sédiments accumulés dans les fossés et bassins tout en respectant des critères économiques et environnementaux très stricts. Le coût du transport dans le traitement des déchets est relativement important d'où l'élaboration d'une unité pilote qui permettra le traitement in situ des sédiments. Avant valorisation comme mise en remblai technique ou en remblai dépôt, le produit traité doit répondre à un cahier des charges bien précis en ce qui concerne les caractéristiques physiques, chimiques et géotechniques.

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Caractérisation de la qualité de l'air inhalé par les agents dans les enceintes ferroviaires de la RATP - résultats de la phase 2

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Résumé

La RATP emploie 45 000 personnes, dont plus d'un quart travaille en milieu souterrain. Des concentrations 3 ou 4 fois supérieures aux valeurs urbaines de PM_{10} y ont été relevées. Ce travail a concerné 54 agents travaillant en milieu souterrain et 5 lignes représentatives du matériel ferroviaire RATP (pneus, fer, RER). Il visait à évaluer par dosimétrie individuelle l'exposition aux PM_{10}, et aux $PM_{2,5}$ de 4 catégories d'agents. Une analyse de composition sur ces particules a été faite. Des différences de niveaux de polluants entre métiers et selon les caractéristiques du matériel roulant ont été recherchées. Les résultats sont le plus souvent en deçà des normes professionnelles, mais certains affleurent ou dépassent les normes environnementales. L'interprétation de ces données reste délicate, les normes disponibles n'étant pas directement transposables dans leur intégralité à la problématique souterraine.

***Mots-clefs :** enceinte ferroviaire souterraine, particule fine, PM_{10}, $PM_{2,5}$, risque sanitaire.*

Abstract

Health surveillance and occupational underground metro dust exposure in Paris public transportation company – phase 2 results

1. Situation - paris public transportation employes 45,000 bus/metro drivers, technical employees, commercial agents, and administrative clerks. More than 10,000 persons work underground. Paris underground subway air quality is closely *monitored since more than a century. Annual occupational medicine visits have not shown any specific diseases. Current monitoring demonstrates important dust levels in some underground subway stations during rush hours as compared to outdoor levels. High concentrations of dust (fine particle) have been found in others underground metros, including helsinki, london, new york, stockholm and montreal.*

2. Methods - The present work refers to the second part of a study: characterization of workers ambient air quality and determination of factors that may alter it. Main objectives of the present work are to search for differences in air quality indicator levels within four occupations, three train types and some workplace characteristics. Time weight average exposure levels have been monitored for a sample of 54 underground workers. Work activities have been detailed in a journal.

3. Results - Lower PM_{10} and $PM_{2,5}$ ambient levels were measured within booth station agents as compared to train dispatchers, train conductors and controllers. These particle levels were smaller among tire-wheel subway line workers. No difference was observed among iron-wheel and RER workers. Copper dust levels were higher tire-wheel subway lines than other types of lines. Dust lead levels were higher among iron-wheel subway line workers while iron, manganese, chromium and nickel levels were higher among RER workers. No significant relationship was observed between tobacco smoke exposure and air quality indicators levels in this preliminary analysis.

4. Discussion/conclusion - This preliminary analysis will need to be confirmed during the third step on 280 additional personal air sampling randomly assigned within the whole subway network. A health study will be performed in the near future to evaluate potential health risks associated with Paris metro dust exposure. Exposure, health status and health risks will be characterized according to different activities (commercial, logistic, and control agents and drivers), job sites (stations/lines), metro types and time spent underground. Health risks of surface workers commuting by metro will be extrapolated to estimate risks of commuters. This surveillance system requires a crafted communication plan to facilitate understanding and clarify expectations, for employees, unions and directors.

Key Words: *underground metro, fine particle, PM_{10}, $PM_{2.5}$, health surveillance.*

Introduction

Les liens entre pollution atmosphérique ambiante et santé ont été observés par de nombreuses études menées au niveau national et international (Samet et al. 2000, Pope et al. 2000, Campagna et al. 2003, Schwartz 1995).

Le réseau souterrain du métro parisien a pour sa part fait l'objet d'une surveillance de la qualité de l'air depuis sa création. Les travaux menés par le Département sécurité environnement de la RATP ont souligné des niveaux d'empoussièrement élevés dans l'enceinte du métro et du RER. L'empoussièrement d'enceintes ferroviaires souterraines a également été observé à Helsinki (Aarnio et al. 2005), Londres (Priest et al. 1999, Seaton et al. 2005), New York (Chillrud 2005), Stockholm (Johansson et Johansson 2003) et à Montréal (Halley 2000).

La Direction Générale de la Santé a chargé le Conseil Supérieur d'Hygiène Publique de France d'apprécier la situation pour les voyageurs et a demandé l'établissement de valeurs guides de qualité de l'air des enceintes ferroviaires souterraines. Les différents avis émis entre 2000 et 2005 ont incité la RATP à conduire des actions visant à mieux connaître la nature des particules présentes dans les enceintes souterraines mais aussi à réduire leurs concentrations. Des valeurs de référence de 477 µg/m^3 dès 2001 et 347 µg/m^3 pour les PM_{10} à terme ont été proposées sur la base d'une présence quotidienne maximale dans le métro de deux heures.

Ces éléments ont interpellé les médecins du travail du Service de santé au travail de la RATP puisque du fait de leurs activités professionnelles, les agents travaillant dans ces enceintes y restent 3 à 4 fois plus longtemps quotidiennement. Ils sont de plus amenés à fréquenter des lieux souterrains différents de ceux empruntés par les voyageurs. Bien que la pollution ferroviaire diffère en terme de granulométrie et de composition de la pollution urbaine, il semblait légitime de s'intéresser à un éventuel retentissement sur la santé d'autant que début 2001, peu d'études avaient été publiées sur ce thème. La composition des particules retrouvées dans les enceintes ferroviaires étant différente de celles de l'air extérieur, leur nocivité n'est à l'heure actuelle pas connue.

Les études du Service de santé au travail ont ainsi pour objectif de caractériser la qualité de l'air inhalé par les agents RATP travaillant dans les enceintes ferroviaires souterraines et de mettre en évidence d'éventuels effets sur la santé. Les enseignements obtenus apporteront également des éléments d'information sur la problématique des voyageurs.

Deux étapes ont été ainsi individualisées : un volet métrologique destiné à apprécier la qualité de l'air inhalé par les agents et un volet sanitaire visant à apprécier d'éventuels effets sur la santé.

Méthodes

La caractérisation de la qualité de l'air ambiant des agents RATP travaillant dans les enceintes ferroviaires souterraines a été principalement effectuée à l'aide de dispositifs individuels portables. Ces dispositifs ont l'avantage, par rapport à des équipements fixes, de refléter en partie l'ensemble des micros environnements traversés par l'agent. Pour des raisons de sécurité ferroviaire, d'encombrement et compte tenu de la présence de voyageurs, ces dispositifs ont été portés au cours d'une journée de travail par deux techniciens spécialement formés et habilités à la sécurité ferroviaire. Les équipements de mesure ont été répartis dans deux sacs à dos portés par chacun des techniciens. Les cabines de conduite étant trop petites pour accueillir le conducteur et les deux techniciens à la fois, les prélèvements de la batterie ont été répartis sur deux agents ayant des horaires similaires et conduisant deux trains qui se suivent. Ainsi, les facteurs pouvant influencer les niveaux mesurés sont contrôlés au mieux.

Les techniciens accompagnateurs ont complété un journal d'activité mentionnant les différents événements caractérisant la journée de travail : heure de début et de fin des activités réalisées, de fréquentations de différents lieux et de certaines notifications spécifiques : garage, contrôle, traversée de chantier, odeur spécifique, zone enfumée, pause... Le technicien assurait également le calibrage et les contrôles de débit des dispositifs d'échantillonnage, avant et à la fin des prélèvements.

Le choix des paramètres (indicateurs de qualité de l'air) retenus a reposé en partie sur les recommandations des différents avis du Conseil d'Hygiène Publique de France concernant l'exposition des voyageurs. Parmi les indicateurs de polluants chimiques, les concentrations des particules de diamètre aérodynamique moyen inférieur à 2,5 et 10 micromètres ($PM_{2,5}$ et PM_{10}) semblent préoccupantes. Ces particules constituent un ensemble hétérogène dont les caractéristiques physico-chimiques sont influencées par les sources d'émission (matériels ferroviaires pour la RATP par exemple) et par des processus de transformation dans l'air. Leur composition et notamment les teneurs des PM_{10} en métaux (fer, nickel, chrome, manganèse, plomb, cadmium), en hydrocarbures aromatiques polycycliques (HAP), en carbone (organique et élémentaire) peuvent différer selon les sites et les lignes. Dans l'air, du fait de résultats préalables parfois discordants obtenus par des dispositifs de mesure fixe, les teneurs en Hydrocarbures Aromatiques Monocycliques (HAM) sont à clarifier.

Quatre métiers exercés dans les enceintes souterraines ont été retenus : conducteurs de métro ou de RER, agents de manœuvre, animateurs agents mobiles (agents de recettes et agents de contrôle). Ces quatre métiers sont ceux regroupant le plus grand nombre d'agents en souterrain pratiquant des activités non génératrices de poussière et ne nécessitant pas le port d'équipement de protection individuelle.

La participation des agents a été volontaire et sans obligation. Les fumeurs ont accepté de ne pas fumer au cours des prélèvements. La campagne a été organisée en trois phases, le présent article exposant les résultats de la deuxième phase qui s'est orchestrée courant 2003-2004 et a concerné 5 lignes (1, 4, 5, 8, RER A). Ces lignes ont été choisies par tirage au sort avec une stratification suivant le type de matériel roulant (roues en fer, roues à pneus, RER). Les recettes étudiées ont, elles aussi, été tirées au sort avec une stratification selon la présence ou l'absence de correspondance. Les prélèvements d'air ont été effectués en semaine et à deux moments de la journée centrés sur les deux périodes de pointe d'activités : une période allant généralement de 5h30 à 12h30 et une période autre allant généralement de 12h30 à 19h30.

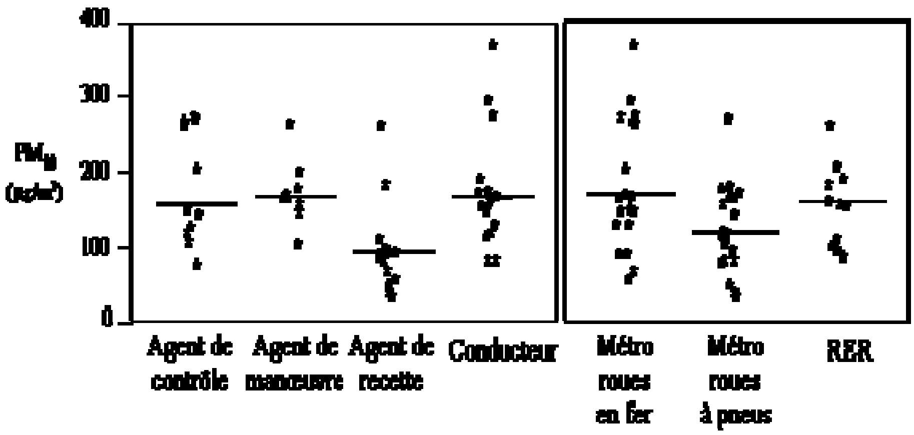
Les prélèvements ont été confiés pour analyse à des laboratoires extérieurs indépendants spécialisés : LHVP (Laboratoire d'Hygiène de la Ville de Paris), LEPI (Laboratoire d'Etudes des Particules Inhalées), CENBG (Centre d'Etudes Nucléaires de Bordeaux Gradignan).

Résultats

Les prélèvements réalisés concernent 54 batteries complètes d'indicateurs de qualité de l'air associés à 10 agents de manœuvre, 15 agents de recette, 12 agents de contrôle et 34 conducteurs de train (nombre de conducteurs doublé pour compléter une batterie d'indicateurs). Les résultats des niveaux de PM_{10} et de $PM_{2,5}$, et des éléments métalliques contenus dans ces particules, sont donnés dans le tableau 1. Les niveaux de PM_{10} sont différents suivant le métier (p=0,001, figure 1). Ils sont plus faibles pour les agents de recette en comparaison aux 3 autres métiers. Quant aux niveaux de $PM_{2.5,}$ il n'y a globalement pas de différence (p=0,08, figure 2) mais la comparaison des métiers 2 à 2 montre que les niveaux de $PM_{2.5}$ sont significativement plus faibles chez les agents de recette comparativement aux niveaux mesurés chez les conducteurs et les agents de contrôle. Pour 7 des 8 éléments métalliques constituant les PM_{10} et les $PM_{2,5}$ (montrés au tableau 1 et tous sauf le zinc), les différences de niveaux en fonction du métier sont toutes significatives au seuil de 5%. Les différences semblent plus importantes au sein des PM_{10} qu'au sein des $PM_{2.5}$. Les niveaux, ainsi que leur dispersion entre les valeurs minimales et maximales, sont les plus faibles pour les agents de recette.

Figure 1 : Distribution des niveaux de PM_{10} par métiers (p=0,001) et par type de ligne (p=0,05). Les lignes horizontales indiquent les niveaux médians. Phase 2.

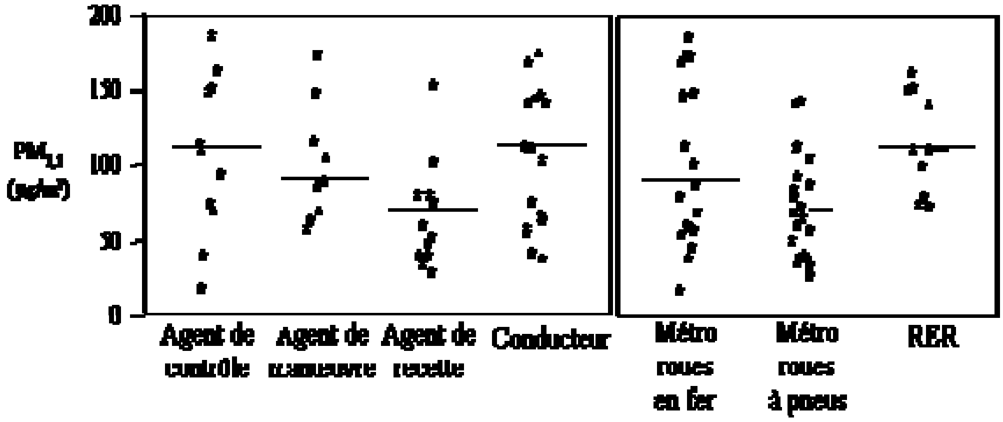
Figure 1: PM_{10} levels by occupations (p=0.001) and by underground lane types (p=0.05). Median levels are shown by horizontal bars. Phase 2.



En ce qui concerne les constituants métalliques des PM10, l'élément largement majoritaire est le fer (27% de la masse totale), puis par ordre d'importance décroissant, on trouve le manganèse (0,3%), le cuivre (0,3%), le zinc (0,2%), le baryum (0,2%), le chrome (0,1%), le plomb (0,05%) et le nickel (0,02%). Pour les PM2,5, le fer est également l'élément métallique majoritaire (environ 17,2% de la masse totale). Puis par ordre d'importance décroissante on trouve le manganèse (0,7%), le cuivre (0,6%), le baryum (0,19%), le zinc (0,17%), le chrome (0,09%), le plomb (0,07%), et le nickel (0,03%).

Figure 2: Distribution des niveaux de $PM_{2,5}$ par métiers (p=0,08) et par type de ligne (p=0,03). Les lignes horizontales indiquent les niveaux médians. Phase 2.

Figure 2: $PM_{2.5}$ levels by occupations (p=0.08) and by underground lane types (p=0.03). Median levels are shown by horizontal bars. Phase 2.



Pour quelques prélèvements, les analyses de certains métaux donnent des résultats inférieurs au seuil de détection. Dans ce cas nous avons choisi de prendre comme niveau mesuré le seuil de détection (choix pénalisant pour les indicateurs de qualité de l'air car il maximise les niveaux). En ce qui concerne l'arsenic et le cadmium, en revanche nous avons choisi de ne pas tenir compte de ces éléments car toutes les valeurs sont inférieures au seuil de détection.

Les niveaux de PM10 (p=0,047 ; figure 3) et de PM2.5 (p=0,032 ; résultats non montrés) diffèrent selon le type de ligne. Les niveaux sont plus faibles pour les métros qui ont des roues à pneus. En ce qui concerne les éléments métalliques constituant ces particules, les métros avec des roues à pneus ont les niveaux mesurés les plus faibles en nickel, en fer, en chrome, en manganèse, en zinc et en baryum. Ils ont les niveaux mesurés les plus élevés en cuivre.

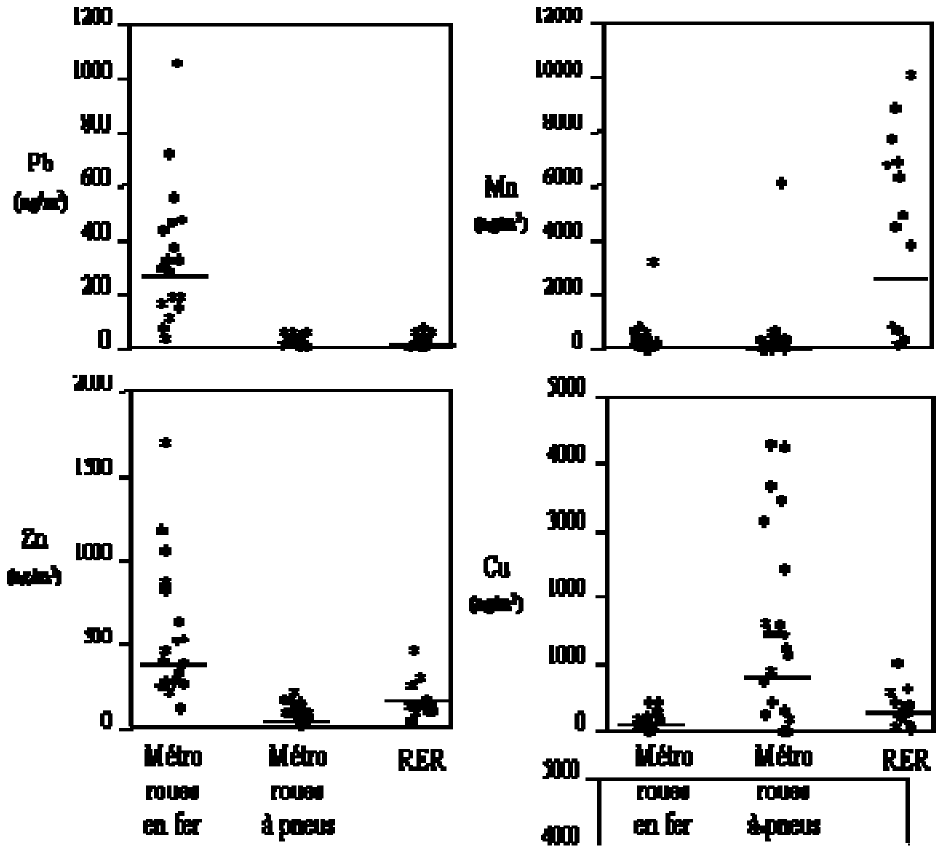
Les métros avec des roues en fer ont les niveaux mesurés les plus élevés en plomb, et les moins élevés en cuivre.

Les RER ont les niveaux mesurés les plus élevés en nickel, en fer, en chrome et en manganèse.

Les teneurs en carbone total, élémentaire et organique, figurent dans le tableau 1. Le carbone est un des éléments majoritaires (environ 34% de la masse totale des PM_{10}). Il n'y a pas de différence significative des niveaux en carbone (quelque soit sa forme) en fonction du métier. En revanche, les niveaux mesurés en carbone diffèrent en fonction du type de ligne. Les niveaux mesurés sont les plus faibles sur les lignes de métros avec des roues à pneus. Ils sont les plus élevés sur le RER.

Figure 3: Distribution des niveaux de certains éléments métalliques de la fraction granulométrique PM_{10} par type de ligne (p=0,001). Les lignes horizontales indiquent les niveaux médians. Phase 2.

Figure 3: PM_{10} metallic component levels by underground lane (p=0.001). Median levels are shown by horizontal bars. Phase 2.



Le tableau 1 indique les teneurs en HAM et en HAP (respectivement exprimés en µg/m^3 et en ng/m^3).

Les concentrations de HAP [Benzo (k) fluoranthène - Bkf et le diBenzo (a)anthracène - diBahA] ne figurent pas dans ce tableau car elles sont souvent inférieures à la limite de détection analytique (0,85 ng/m^3).

Tableau 1 : Niveaux d'indicateurs de qualité de l'air. Phase 2.
Table 1: Air quality indicator levels. Phase 2.

	Moyenne	(écart type)	Percentiles 25%	50%	75%
PM_{10} (µg/m^3)	157	(73)	99	155	184
Plomb (Pb) (ng/m^3)	154	(209)	34	63	185
Nickel (Ni) (ng/m^3)	48	(34)	23	37	60
Fer (Fe) (ng/m^3)	42292	(33301)	13186	41263	61368
Chrome (Cr) (ng/m^3)	187	(174)	47	162	251
Manganèse (Mn) (ng/m^3)	1602	(2657)	217	362	762
Cuivre (Cu (ng/m^3)	839	(1094)	203	339	1042
Zinc (Zn) (ng/m^3)	300	(324)	113	165	330

Barium (Ba) (ng/m^3)	509	(453)	78	422	787
PM$_{2.5}$ (□g/m^3)	96	(44)	62	86	131
Plomb (Pb) (ng/m^3)	77	(106)	20	38	98
Nickel (Ni) (ng/m^3)	32	(22)	15	25	46
Fer (Fe) (ng/m^3)	18740	(15688)	5508	14000	27930
Chrome (Cr) (ng/m^3)	90	(74)	21	80	130
Manganèse (Mn) (ng/m^3)	789	(1348)	76	188	519
Cuivre (Cu (ng/m^3)	605	(934)	131	222	538
Zinc (Zn) (ng/m^3)	166	(163)	74	104	187
Barium (Ba) (ng/m^3)	208	(206)	59	98	313
Carbone total **(□g/m^3)**	44,4	(11.8)	35,7	43,3	51,8
Carbone élémentaire (□g/m^3)	17,3	(5.4)	12,5	17,6	21,1
Carbone organique (□g/m^3)	27,5	(9.0)	21,6	26,2	32,1
Hydrocarbures aromatiques monocycliques (HAM)					
Benzene (□g/m^3)	4,9	(1,9)	3,4	5,1	6,0
Toluene (□g/m^3)	29,2	(12,4)	20,0	28,1	35,1
Ethylbenzene (□g/m^3)	4,6	(3,1)	3,0	4,2	5,2
m+p xylene (□g/m^3)	13,3	(10,9)	8,2	11,6	15,1
o-xylene (□g/m^3)	4,2	(3,2)	2,6	4,0	4,8
1,2,4 TIMbenzene (□g/m^3)	5,6	(4,1)	3,7	4,5	6,5
Hydrocarbures aromatiques polycycliques (HAP)					
BAA (ng/m^3)	1,1	(0,3)	0,85	0,85	1,1
Chrysène (ng/m^3)	3,1	(2,1)	0,85	2,5	4,4
Bdf (ng/m^3)	1,3	(0,5)	0,96	1,4	1,6
Bap (ng/m^3)	1,2	(0,6)	0,85	0,87	1,3
BghiP (ng/m^3)	4,0	(4,7)	1,30	2,1	2,5
IP (ng/m^3)	1,1	(0,4)	0,85	1	1,3

En ce qui concerne les niveaux de HAM, la différence des niveaux est significative entre les métiers pour le benzène, le toluène, l'ethylbenzène et le triméthylbenzène. Il n'y a pas de différence significative pour les m-p xylène et l'o-xylène. Les conducteurs et les agents de manœuvre sont les moins exposés au benzène, au toluène et à l'éthylbenzène et au TMB. Il n'y a pas de différence significative des niveaux de HAM entre les 3 types de ligne. Pour les HAP, il n'y a de différence en fonction du type de ligne que pour le chrysène : les niveaux sont plus élevés pour les lignes de métro à pneus. Aucune différence significative des niveaux de HAP en fonction du métier n'est relevée. Les médianes de PM$_{10}$ et PM$_{2.5}$ sont plus faibles chez les agents peu ou pas exposés au tabac (moins d'une heure) mais les différences des niveaux de PM$_{10}$ et de PM$_{2.5}$ en fonction de l'exposition au tabac ne sont pas significatives au seuil de 5% (résultats non montrés).

Discussion

Les niveaux mesurés dans le cadre de cette étude sont nettement inférieurs à ceux définies par le code du travail. A titre d'exemple, en ce qui concerne les particules, selon la directive 98/24/CE et l'article R232-5-5 du code du travail (décret du 7 décembre 1984), concernant les locaux à pollution spécifique, et les poussières sans effet spécifique (circulaire du 9 mai 1985), les valeurs limites pour les concentrations moyennes en poussières inhalables et alvéolaires de l'atmosphère respirée par une personne évaluée sur une période de 8 heures en hygiène du travail ne doivent pas dépasser 10 mg/m^3 pour la fraction inhalable et 5 mg/m^3 pour la fraction alvéolaire. Ces valeurs sont actuellement remises en cause du fait d'un risque de surcharge pulmonaire pouvant être observé à ces niveaux (Hervé-Bazin 2005). De plus, la présence de particules ultra fines définies par un diamètre inférieur à 0,05 µm dans l'aérosol n'est pas prise en compte. Les valeurs de 5 mg/m^3 pour la fraction inhalable et de 2 mg/m^3 pour la fraction alvéolaire à faible proportion de particules ultra fines sont proposées (Hervé-Bazin 2005). Les autres paramètres et notamment les métaux mesurés sont également très en deçà des valeurs limites professionnelles.

Les expositions aux HAP et HAM semblent proches des expositions retrouvées dans l'air extérieur. En revanche, pour les éléments métalliques, certaines valeurs affleurent ou dépassent les normes environnementales. La réglementation vise le plomb (500 ng/m^3), le nickel (20 ng/m^3), l'arsenic (6 ng/m^3) et le cadmium (5 ng/m^3). Si les teneurs mesurées en arsenic et en cadmium ne dépassent pas ces seuils, il n'en est pas de même pour le nickel, plus de 75% des prélèvements ayant une teneur supérieure à 20 ng/m^3 et pour quelques prélèvements concernant le plomb.

Conclusion

Ces résultats préliminaires constituent une première analyse de l'étude visant la caractérisation de l'air inhalé par les agents. Ils devront toutefois être confirmés et approfondis lors de la troisième phase de cette étude qui portera sur un nouvel échantillon de 280 prélèvements d'air ambiant répartis sur l'ensemble des lignes du réseau de façon à permettre d'établir une typologie des lignes.

L'interprétation de ces données reste délicate, les normes disponibles, professionnelles et environnementales n'étant pas directement transposables dans leur intégralité à la problématique souterraine. Le respect des normes professionnelles ne nous permet pas d'assurer l'absence d'effets sur la santé du personnel travaillant dans les enceintes ferroviaires.

Les visites systématiques annuelles en médecine du travail n'ont pas décelé de pathologies spécifiques. Pour s'en assurer, le Service de santé au travail de la RATP réalisera prochainement une enquête épidémiologique (prenant en compte les résultats des campagnes de métrologie) pour estimer le risque sanitaire potentiellement encouru par les agents travaillant dans les enceintes ferroviaires souterraines. Les participants à cette étude seront répartis en trois groupes distincts selon leur lieu de travail (en surface ou en souterrain) et du moyen de transport utilisé pour leurs trajets domicile – travail (métro/RER ou non). Le premier groupe sera constitué de 2000 agents des enceintes ferroviaires, le second de 1000 agents de surface usagers du réseau souterrain (groupe se rapprochant d'une population de voyageurs) et le troisième de 1000 agents de surface non usagers du réseau souterrain (groupe comparable de non voyageurs).

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Satellite-assisted estimation of transport particulate pollution and the associated risk on human health

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Résumé

Des travaux récents ont démontrés la faisabilité de fusionner des données satellitaires sur des indicateurs de pollution atmosphérique affin d'obtenir d'évaluations quantitatives de la charge des particules dans la couche limite planétaire. La méthode présentée ici, en conjonction avec des modèles de dispersion atmosphérique et de mesures prises par les réseaux de monitorage de la qualité de l'air peut nous fournir une estimation compréhensive de la pollution troposphérique des particules. La méthode de fusion des données ICAROS NET a été appliquée dans le cas du bassin d'Athènes en Grèce et du Munich en Allemagne et les résultats ont démontrés qu'elle est capable de calculer la concentration de particules fines et d'estimer le risque associé pour la santé publique avec très haute précision et a haute résolution spatiale.

Mots-clefs *: fusion de données, données satellitaires, pollution atmosphérique, particules*

Abstract

Recent work has demonstrated the feasibility of fusing satellite-derived data of atmospheric pollution indicators to draw quantitative maps of particulate loading within the planetary boundary layer. The method presented herein, when used in conjunction with atmospheric dispersion models and ground data, can provide a comprehensive estimate of tropospheric pollution from particulate matter. Information filtering techniques are used to reduce the error of the information fusion algorithm and, consequently, produce the best possible estimate of tropospheric *aerosol. The ICAROS NET fusion method was applied in the greater area of Athens, Greece and Munich, Germany over several days of observation. The results showed that the conceptual model for tropospheric aerosol formation and fate in the atmosphere that has been developed based on experimental analyses across different European sites allows highly accurate estimates of particulate pollution and their health effects at high spatial resolution.*

Keywords: *Keywords: data fusion, satellite data, air pollution, particulates, aerosol*

Introduction

Recent studies worldwide have revealed the relation between urban air pollution – particularly fine aerosols – and human health (Roemer et al, 2000; Medina et al, 2001). This, in turn, has created a pressing request from both environmental scientists and decision-makers for spatial, timely and comparable information on air pollution and associated indicators (WHO, 2000). Particles in the size range 0.1–3 µm are considered to be amongst the most important with regard to adverse health effects (Goldsmith and Kobzik, 1999; Brunekreef and Holgate, 2002). Childhood asthma prevalence and incidence have been associated with local variation in traffic patterns within communities in many studies (van Vliet et al, 1997; Zmirou et al, 2004) examining the impact of local traffic or traffic-related air pollutants near children's residences. Some studies have found increased asthma prevalence in children living within 100 m of a major road, and there is evidence that the risk increases dramatically within 75 m (van Vliet et al, 1997; Venn et al, 2001). A study by McConnell et al (2006) examined the relationship of local traffic-related exposure and asthma and wheeze in southern California school children. Residence within 75 m of a major road was associated with increased risk of lifetime asthma, prevalent asthma and wheeze. The higher risk of asthma near a major road decreased to background rates at 150 – 200 m from the road.

Current problems with air pollution monitoring are linked to the high costs of extensive and continuous measurement of atmospheric pollutants in a changing chemical composition of the lower atmosphere, which results in changing requirements for monitoring systems. An additional problem linked to this inherent difficulty is the increasing need for source apportionment of pollution (Lucarelli et al, 2004) in order to optimize environmental management aiming at abating the pollutant emission intensity via technology improvements and mitigating their effects on human and ecosystem health by limiting human exposure. Concentrations of pollutants including both volatile organic chemicals and particulate aerosol in fresh vehicular exhaust are high near roadways but decline sharply within a distance of 150 – 300 m from the road (Zhu et al, 2002). Accurate assessment of this large but local variation in actual human exposure may be important in identifying the associated health hazards.

To date ground measurements of particulate matter are not taken from dense enough monitoring networks around the world to permit a satisfactory analysis of the actual influence of fine urban aerosol on the health of vulnerable population groups, such as the elderly, children under the age of fifteen, asthmatics, people with cardiovascular problems. Recent attempts to improve our estimation of fine aerosol concentrations at the urban and regional scale from combining ground data with numerical modeling are hampered by the need for high quality and up-to-date emissions inventories, as well as for accurate estimates of initial and boundary conditions of the models. This information is not always available in many cities around the world. Satellite-derived information has been considered a possible solution to this impasse. Satellite data, however, although producing environmental data at various geographical scales (from 0.5 m to 10 km of resolution) are time-deficient due to generally long periods of revolution around the Earth.

Fusion of ground data, meteorological and pollution modeling with physical indicators of airborne pollutants to derive a computational assessment of particulate matter concentration is the solution to this apparent contrast. The method described in this paper permits the estimation of PM concentration at high spatial resolution (30 m) covering a domain as large as 80-100 x 80-100 km^2. The theoretical model developed in the frame of this work was applied in Athens, Munich and Milan in order to test it under various climatic and geographical conditions during different seasons. The results of the applications in Athens and Munich are presented herein in order to demonstrate the applicability of the method for accurately assessing actual exposure to transport-related particulate matter and for estimating the potential adverse health risk.

Methodology

The ICAROS NET project developed a novel methodology for monitoring and managing air quality and related health risks at the urban setting. The method, which has been applied in four cities in Europe, allows a quantitative evaluation of the link between different emission sources, air quality levels and, finally, the associated public health impact through the fusion of satellite-derived information with ground-based monitoring and numerical modeling. This results in quantitative maps of airborne particulate matter with very high spatial resolution (30 by 30 meters or less) covering a geographical extent of approximately 100 km × 100 km. A key advantage of this approach is the quantitative assessment of atmospheric aerosol both at microscale (single road, street canyon) and macroscale (metropolitan area) levels; this permits the precise assessment of the contribution traffic-related emissions have on total pollution levels and associated health effects.

The current state of the art in air quality assessment and management comprises analytical measurements and atmospheric transport modeling. Earth observation from satellites provides additional information through the calculation of synoptic air pollution indicators, such as aerosol optical thickness. The method presented herein integrates these three information sources through suitable data fusion techniques providing a comprehensive estimate of tropospheric pollution from particulate matter. Information filtering is used to reduce the error of the fusion algorithm and to produce the best possible estimate of tropospheric aerosol loading. Linking the latter with epidemiological data and activity modeling, allows reckoning the potential health risk form fine and ultra-fine particulate matter associated with traffic-related emissions. The key to the success of the data and model fusion approach described in this work is the combination of physical and chemical process modeling that allows linking physical (e.g. optical) properties of tropospheric aerosol with the atmospheric physical-chemical processes that determine the total mass concentration and size distribution and chemical composition of particulate matter. Assimilation of these data sources with ancillary data including classification of population vulnerability to the adverse health effects of particulate pollution in the ambient air integrates them into an optimally managed environmental information processing tool, which can be used for integrated air pollution monitoring and environmental health assessment in the urban and regional environment (Sarigiannis et al, 2002; 2004).

Atmospheric aerosols consist of particles from different sources and their properties depend on physical and chemical processes during atmospheric transport (Seinfeld and Pandis, 1998). The resulting aerosol has variable size distribution and chemical composition. It is often described as mixture of aerosol components with spectral dependent complex refractive index (Dubovik et al., 2002) , with the size distribution of each component given as log-normal distribution (D'Almeida et al., 1992, Hess et al., 1998). A single-mode distribution of mode i is:

$$n_i(r) = \frac{N_i}{\sqrt{2\pi}} \cdot \frac{e^{-\frac{(\ln r - \ln r_{0i})^2}{2(\ln \sigma_i)^2}}}{r \ln \sigma_i}$$

A multimodal distribution is obtained by simply summing the single-mode distributions *i*. In the particle distribution equation *r* is the radius of the aerosol particles, N_i is the total particle number density of mode *i* (e.g. in particles per cubic centimeter), r_i is the geometric mean radius of component *i* and σ_i its geometric standard deviation which measures the width of the radius range of the component. The particles of a certain component are assumed to have the same chemical properties resulting in a consistent particle density ρ (in gr/cm^3) and complex refractive index – which will depend on the wavelength. Particles of different components, e.g. water soluble, soot, mineral, or sea salt, can be combined to aerosol types (Hess et al., 1998; Dubovik et al., 2002). The radiative properties of aerosol particles which are of relevance for the data fusion approach in this work are: height dependent extinction σ_e (z); scattering coefficient, σ_s (z); and absorption coefficient, σ_a (z) and the aerosol optical depth, AOD.

$$AOD = \int_0^{\infty} \sigma_e(z)dz$$

Extinction, scattering and absorption coefficient usually are calculated after Mie-theory assuming spherical particles.

$$\sigma_e(r_{\min}, r_{\max}) = \int_{r_{\min}}^{r_{\max}} Q_e \cdot \pi r^2 \cdot \frac{dN}{dr} dr$$

The volume extinction coefficient attributable to particles with radius between r_{min} and r_{max} depends on the number of particles at a given radius in the volume element, the square of this radius and the extinction cross section Q_e, which again depends on the spectral refractive index and on the ratio between radius and wavelength. Similar definitions can be written for the scattering and absorption coefficients. These quantities are related as follows:

The particle mass in a given volume of air is given by the density ρ of the aerosol particles multiplied by their volume, taking into account the radius of the contributing particles.

$$M(r_{\min}, r_{\max}) = \int_{r_{\min}}^{r_{\max}} \rho \cdot \frac{dV}{dr} \cdot dr$$

Water soluble or any hydrophobic aerosol particles increase in size with relative humidity and the density of the particles changes towards the density of water (Waggoner et al, 1981). This effect, due to the uptake of water, changes the size and refractive index and, consequently, the radiative properties of the particles. Since size distribution gets an increased amount of larger particles with increasing relative humidity, but the maximum radius taken into account by the mass measurement is fixed, relative humidity affects particle mass even if aerosol number density remains constant. This behavior is particularly true for the fine fraction of the particulate matter (i.e. PM2.5 and PM1) (Sloane, 1984).

According to the ICAROS NET methodology satellite data in the visible are used to calculate the optical depth of atmospheric aerosol. Visible light is selected because this is where the difference between Mie and Rayleigh scattering is the largest. The method applied for the evaluation of aerosol optical depth combines two approaches that consider physically independent optical effects in the atmosphere (Sifakis and Soulakellis, 2000): (a) contrast reduction by scattering efficient airborne particles in short wavelengths (i.e., visible spectrum); and (b) radiation attenuation that particles engender in longer wavelengths (i.e., thermal infrared spectrum). The optical depth is the integral of the extinction coefficient due to scattering from the ground to the height of the satellite sensor. Its magnitude is linked to the columnar concentration of the optically effective aerosol. In order to calculate the scattering coefficient, σ_s, of the aerosol that lies close to the ground, AOD has to be divided by the appropriate scale length. Under well-mixed conditions such as the ones around noon (when the satellite vector passes over the areas of interest), almost 90% of fine aerosol stays within the lower half of the mixing layer of the atmosphere. Even accounting for measurement error, considering that the correct scaling height for the scattering coefficient is half the height of the mixing layer is a reliable approximation. In the visible domain optically effective aerosols are generally considered those comprising fine particles with diameters between 0.1 and 3μm (Seinfeld and Pandis, 1998). To date, in Europe, measurements of particulate matter close to the ground usually focus on PM10, as the ambient air level of only this aerosol fraction is regulated at Community level so far. Hence, in order to relate the satellite-derived scattering coefficient of fine particles with the ground-measured concentrations of PM10, due account has to be taken of the effect of relative humidity in the lower troposphere.

The physicochemical model that was found to best reflect the dependence of PM10 concentration on the ground with the scattering extinction coefficient and relative humidity is

$C_{PM10} = a \cdot \sigma_e + b \cdot RH$ for RH < RH_o

$C_{PM10} = a' \cdot \sigma_e + b' \cdot e^{K \cdot RH}$ for RH > RH_o

The exact value of the inflection point RH_o, depends on the prevalent meteorological conditions (in terms of atmospheric humidity and ground-level air temperature). Factors *a*, *b*, a' and b' depend on the relative ratio of fine to coarse particles; the latter depends on the chemical composition of tropospheric aerosol. *K* is the kinetic constant of fine particle growth due to atmospheric humidity.

Fusing the concentrations of tropospheric aerosol calculated on the basis of the values of its scattering coefficient with the PM concentrations on the ground as reckoned by atmospheric numerical modeling using Gaussian or Eulerian models on a cell-by-cell basis throughout the computational domain “corrected” maps of aerosol loading in the lower atmosphere are produced. Model fusion has been done following two alternative schemes: (a) a weighted-average of the PM concentrations resulting from the two computation models; and (b) a Kalman filter designed to account for the relative error introduced by each of the two computation models. This technique ensures that data singularities generated from point disturbances of the optical parameters field captured by the satellite sensor are eliminated. At the same time, atmospheric model weaknesses, such as the high dependence of result accuracy on the adequacy of the emissions inventory and boundary/initial values can be dealt with via fusion with the satellite-derived estimates. Both filters improve the overall performance of the ICAROS NET algorithm in estimating accurately ambient particulate concentrations. Kalman filter has a slightly better performance overall, and especially in areas of the computational domain where uncertainty in the results of the model is high due to an imprecise or out-of-date emissions inventory or abrupt changes in the air humidity profile occur.

3 - Results and discussion

The ICAROS NET method was applied in Athens and Munich over several days of observation. Results showed that it provides accurate and robust estimates of particulate pollution at very high spatial resolution in various environmental settings and across different seasonal and emission intensity features. The overall estimation error is reduced to less than 6% (ε = 5.66%).

1. The case study of Athens

The summer Olympic Games of 2004 gave us the opportunity to perform dedicated satellite retrievals of particulate matter in order to isolate the effect of traffic on the overall PM10 loading in the ambient air and the associated public health burden. Daily satellite retrievals in the visible were taken using the SPOT5 platform during the 15-day long Games. Some of these data were not used due to high cloud coverage; the remaining data were analyzed to provide the exact parameterization of the physicochemical model linking ambient air PM10 concentrations with the scattering coefficient and humidity as outlined above. A typical example is given in figure 1 below.

Figure 1: High resolution map of ambient air PM10 concentration in Athens during the 2004 Olympic Games (black dots denote the location of the ground monitors)

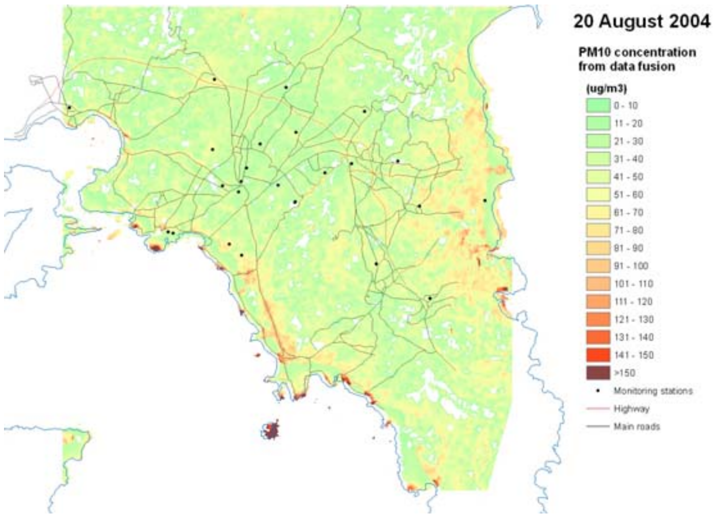
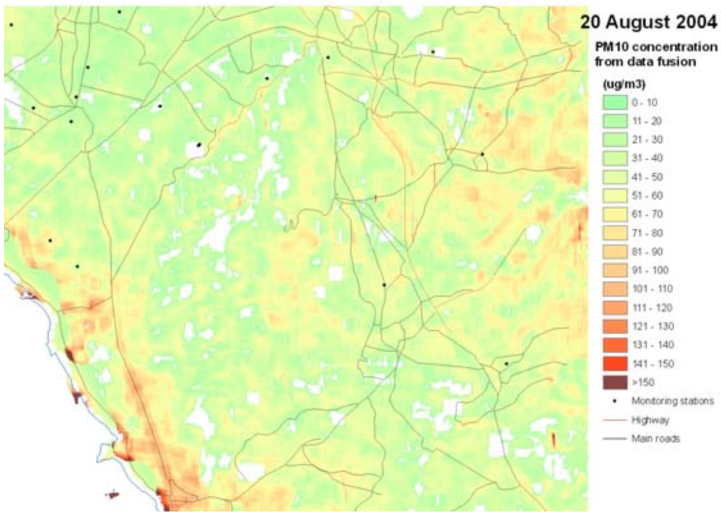
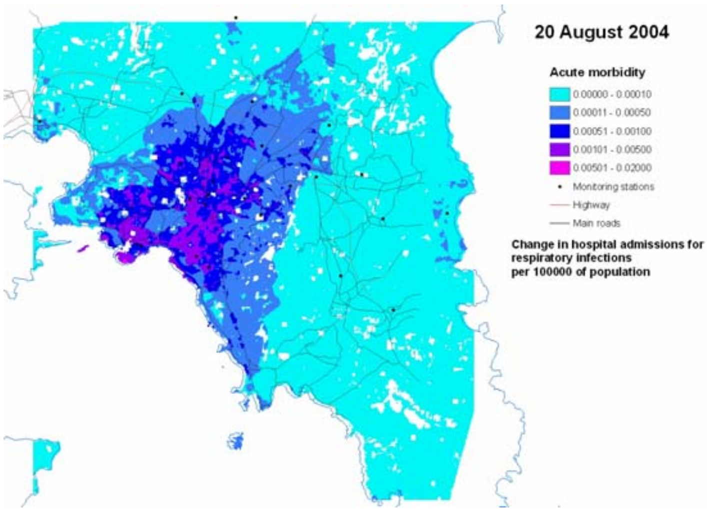


Figure 2: Traffic-related ambient air PM10 concentration close to the Athens coast



Zooming in the area close to the Saronikos Gulf coastline (see figure 2), the main two traffic arteries leading out of the downtown Athens area towards the seaside are characterized by high PM10 values ranging from 80 to 110 μg/m^3 close to the roads, with peak values reaching up to 170 μg/m^3. Ambient air concentration of PM10 away from the main roads (at distances over 200 m) takes values ranging from 20 to 40 μg/m^3, well below the concern threshold of 50 μg/m^3. This finding is reasonable when taking into account that during the Athens Olympic Games, special traffic control and traffic flow management measures had been taken aiming at ensuring easy access to all Olympic venues across the Attika peninsula. The combination of these measures with the provision of financial incentives and public awareness campaigns towards strengthening the use of public transport with special emphasis on the new Athens underground railway system and the corresponding tramway corridors has alleviated significantly the traffic load of the road network in the Athens basin. Key traffic arteries such as Vouliagmenis avenue and the Attika Road (the brand new Athens ring road) took up the remaining load concentrating the main particulate loading in the areas near them.

Figure 3: Estimated change in hospital admissions for traffic-related respiratory infections in Athens



Health indicators such as metrics of acute morbidity expressed in change in hospital admissions with respiratory problems can be estimated spatially on the basis of the exposure potential estimated by the concentration of particulate matter in the ambient air and the geographical extent of population density (daily average) as estimated by the implementation of a micro-activity pattern model for the local population taking into account the main traffic flow streams and types of land use in the greater area of the Athens basin and the region of Attica. Results are shown in figure 3.

2. The case study of Munich

In Munich satellite data retrievals were coupled with intensive measurements on the ground assessing the concentration, chemical composition and optical properties of the urban aerosol. The results showed that it is feasible to discern pollution related to the heavy traffic load characterizing the main roadways by zooming in the spatial pattern of ambient air aerosol as estimated from the comprehensive satellite imagery (figure 4).

The results in the vicinity of the Munich international airport (see figure 5) shows that the ambient air loading of particulate matter ranges between 35 and 65 μg/m^3 with isolated peaks reaching 105 μg/m^3 close to the main highways and other main roads in the area. The landing pathway and the fence road of the airport facility are characterized by increased values of particulates. In contrast, the ambient air loading in PM10 reaches level lower than 10 μg/m^3 in the areas away from the main transport-related emission sources such as the main roads and highways and the airport itself.

Figure 4: Ambient air concentration of PM10 in the greater area of Munich during spring 2003.

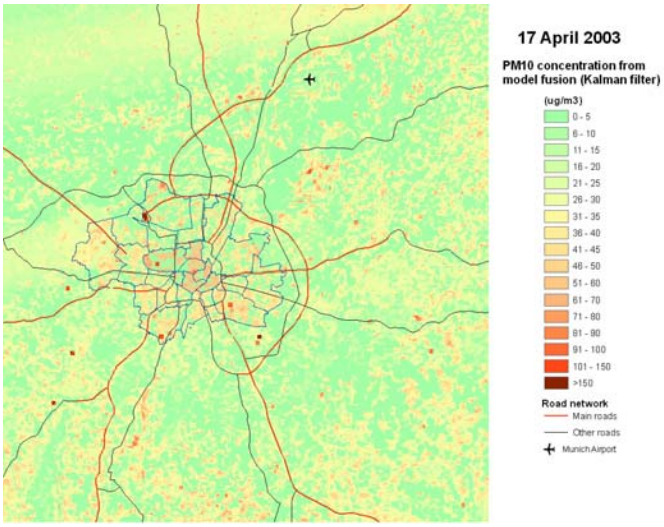
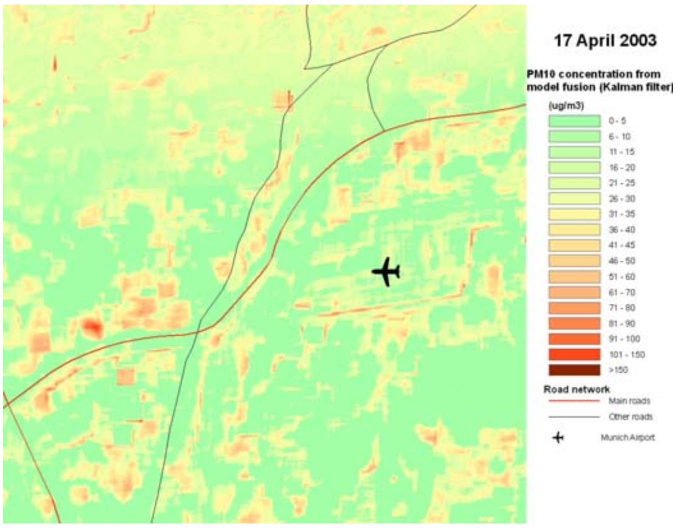


Figure 5: Ambient air concentration of PM10 in the vicinity of the Munich international airport.



Conclusions

In rural areas, such as in the vicinity of Munich airport, traffic-related atmospheric loading in PM10 is as much as five times higher than the background concentrations. In Munich, this value reaches consistently and sometimes exceeds the concern threshold of 50 µg/m^3. In the urban area of Athens and the semi-urban zones traversed by the Attika Road (the Athens outer ring), traffic related PM10 in the ambient air is one and a half to three times higher than the PM10 measured 200 m away from the main road network. Traffic-related particulate pollution reaches and slightly exceeds the 50 µg/m^3 threshold along the Attika Road and exceeds it by almost a factor of two in the proximity of the main urban traffic arteries. The estimated potential health risk that can be associated to traffic emissions follows the spatial pattern observed in other studies, i.e. decreases significantly with distance from the main roads and becomes negligible in areas beyond 250 m away from them.

The results shown in this work demonstrate not only the high accuracy (error lower than 6% in all three sites tested) and estimation power of the ICAROS NET method for deriving human exposure to particulate matter from satellite data. They show that combined use of high resolution satellite data on the pollution loading of the lower atmosphere with ground-based air quality data and meteorological / physicochemical modeling can provide a valid approach for overcoming the pitfalls of current atmospheric observation systems with a view to addressing coherently the different challenges presented by the scale difference characterizing the main emission processes in the urban and regional setting. High resolution (30 m) estimates of near-surface particulate matter concentration in the ambient air may provide the bridge that would allow the seamless integration of air pollution estimates from the single road and street canyon level to the greater metropolitan areas that are typical of modern day mega-cities.

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A multi-method approach to extracting objective and subjective accounts of road traffic related noise and air pollution

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Abstract

Densely populated congested urban areas are hotspots for both noise and air pollution, with the main source being road traffic. This paper presents a study exploring air and noise pollution within UK city centres. Pollutant concentrations are examined through environmental monitoring and also through the experiences of city centre dwellers living in these conditions. The work reported here aims to compare and explore the relationships between these varying accounts and approaches.

An innovative multi-method approach, combining qualitative and quantitative data collection techniques, has been developed and employed. Environmental monitoring of roadside air quality and noise levels (Leq, CO, TSP, $PM_{2.5}$) were undertaken alongside participant led photo surveys, sound walks and semi-structured interviews with city centre residents. The approach is explained within the paper and enabled the collection of both subjective and objective data. The environmental conditions varied considerably within the case study area. Objective and subjective accounts document this variation to differing extents. Both provide useful information on the environmental conditions and the impacts of road traffic on environmental quality and quality of life. The wider implications of this study to the concept of urban sustainability are also discussed. This method of analysis allows the trade-offs between the positive and negative effects of urban living to be considered at the same time.

***Key-words**: air quality; noise; road traffic; city centre; subjective and objective accounts*

Introduction

Noise and air pollution are extremely complicated, multi-dimensional environmental problems. They are the most widespread and unavoidable environmental issues in our society: inescapable and affecting everyone. Densely populated, congested urban areas are hotspots for pollution, with the main source of pollutants being road traffic. It has been argued that most urban environmental problems are now under control with the exception of the persistent problems of air and noise pollution (de Hollander and Staatsen 2003). Collecting measurements on these issues enables researchers and policy makers to have a variety of indicators that reflect present day conditions, however, to truly understand these complex subjects consideration needs to be given to the publics' perception and experiences. Surprisingly, both of these aspects (objective and subjective) have not been investigated in detail; however a vast amount of research has explored the actual levels of pollutants generated and the physical processes taking place (Kaur *et al.* 2005, Onuu 2000), whilst others have independently studied the publics' response to traffic and pollutants, exploring perceptions and attitudes (Klaeboe *et al.* 2000, Bickerstaff and Walker 2001). Due to the multifarious nature of these problems many researchers have recommended an integrated approach to research (van Kamp *et al.* 2003, Brown 2003). For instance, in the case of air quality, recent policy developments in the UK have urged local authorities to engage with the public on air quality management issues alongside physical monitoring to gain support and implement policy procedures. Involving the public is central to the delivery and success of air quality objectives, as personal responsibility and individual action will allow the necessary steps to be taken to ultimately improve air quality (Bickerstaff and Walker 2001).

This paper reports on an ongoing project exploring road traffic generated noise and air pollution within UK city centres via two different approaches - objectively, through environmental monitoring and subjectively through the experiences of city dwellers. It is essential to note that this is only one specific part of a wider project examining environmental quality, city centre living and urban sustainability within three UK city centres – London, Sheffield, Manchester. The project is titled 'Designing Environmental Quality into City Centre Living', and forms part of the EPSRC funded, multi-disciplinary VivaCity2020 project: 'VivaCity2020: urban sustainability for the 24-hour city' (www.vivacity2020.org). This paper presents some key aspects of the project so far, concentrating on the findings from a case study undertaken in London.

Methodology

An innovative multi-method approach combining qualitative and quantitative data collection techniques has been developed and employed in this project. Case studies have been undertaken in three UK city centre areas: London (Clerkenwell), Sheffield (Devonshire Quarter) and Manchester (City Centre). This paper focuses primarily on the London case study in Clerkenwell, located to the north east of central London within the borough of Islington. As the project concentrates upon the experiences of city centre living, the area targeted for the study, within Clerkenwell, was a mixed use area with housing located near the main daytime and night-time commercial and leisure activities.

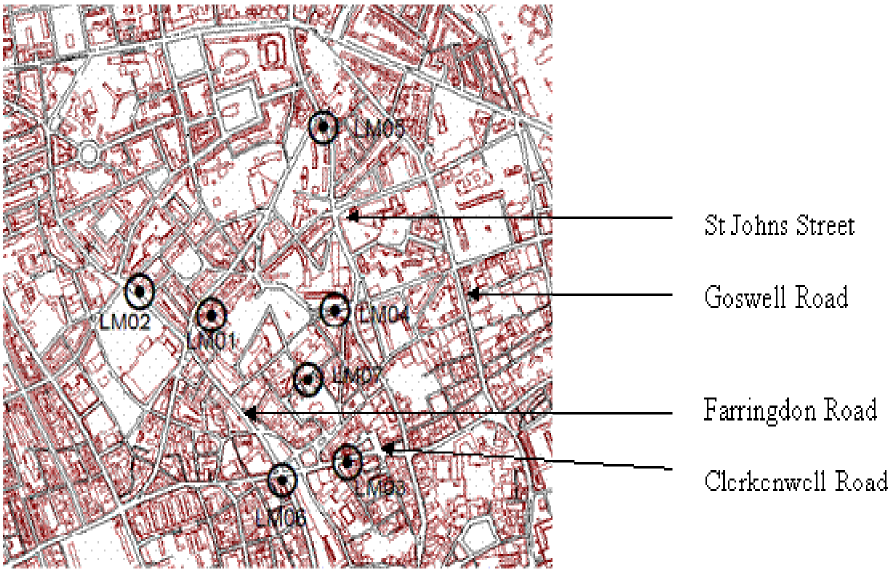
Table 1: London case study monitoring details

Site No.	Location	Variables	Monitoring season/period	Notes
LM01	Exmouth Market	TSP, CO, noise (15min)	Winter (6 weeks in total)	On balcony area, north facing, leeward side. Height approx 3.5m.
		PM 2.5, CO, noise (3min)	Summer (2 weeks)	Same as above
LM02	Farringdon Road	CO, noise (3min)	Winter (2 weeks)	Near Margery Street, east side of the road, windward side. Height approx 2.5m.
LM03	Clerkenwell Road	CO, noise (1min)	Winter (2 weeks)	Near Britton Street, south side of the road. Windward side. Height approx 2.5m.
		CO	Summer (2 weeks)	Same as above
LM04	Skinner Street	CO, noise (15min)	Winter (2 weeks)	Near St John Street, south side of the road, windward side. Height approx 2.5m.
LM05	St John Street	CO	Winter (1 week)	On first floor of balcony, east facing, leeward side. Height approx 3m.
LM06	Farringdon Road (2)	CO, noise (15min)	Winter (2 weeks)	On fourth floor balcony, east facing, leeward side. Height approx 3.5m.
LM07	Sans Walk	CO, noise (15min)	Winter (1 week)	On third floor balcony, south facing, windward side. Height approx 3.5m.

The project methodology can be divided into two fundamental components: (i) the environmental monitoring, (ii) the experiences of city centre dwellers. The environmental monitoring involved the intensive monitoring of an urban road system at a number of locations (kerb-side). Noise levels (dB(A)), carbon monoxide (CO, ppm), temperature (°C) and particulate matter (TSP, $PM_{2.5}$) were monitored at a number of sites within the case study area over a summer and winter period (the monitoring campaign is shown in Table 1 and Figure 1). The monitoring sites were purposely located near the residential premises of participants, to enable comparisons between the data. The sites provided a range of conditions (high/low traffic/pedestrians level and mixed land use) that are found within Clerkenwell. A specially designed noise and air quality monitor, the Streetbox (Learian, www.learian.co.uk) was developed and used; a sound level meter has been incorporated into a standard carbon monoxide Streetbox to enable the continuous monitoring of noise and air quality simultaneously (Figure 2, see Croxford and Penn 1998 for more information on the Streetbox). A Met One E-Sampler (light scatter aerosol monitor) was used to monitor particulate matter. Average temperature, CO and PM levels were collected at 15-minute intervals, however average noise levels

(L*eq*) were collected at differing intervals (15min, 3min or 1min), depending on the Streetbox used (details in Table 1).

Figure 1: Map of Clerkenwell and monitoring locations



A variety of qualitative methods were utilized in this study to understand residents' experiences of noise and air pollution. Thirty residents were involved in a photo survey, a sound walk and a semi-structured interview. Approximately two weeks before each scheduled interview date a disposable camera (27-exposure, 35mm film, 400 ISO with flash), a log sheet, prepaid envelope and instructions were sent to the participants. Participants were asked to take photographs of their local area, noting the time, date, location and a short description of the photograph on a log-sheet provided (the photo survey). We did not want to be too prescriptive in telling participants what to photograph, so the instructions simply stated: 'we would like you to take photos that record both the positive and negative aspects of your area'. They were given approximately one week to take photographs before sending the camera back to a researcher in the prepaid envelope provided. The photographs were then developed and numbered and brought along to the scheduled interview. Prior to the start of the interview participants were asked to mark a 5 to 10-minute walking route around their local area on a map supplied. This walk was undertaken by a researcher and the participant, and recorded with a DAT recorder. Participants were asked not to talk during the walk, but to listen and observe (the sound walk). On return to the participant's home a semi-structured interview was conducted. The interview was based upon a number of general questions about the urban environment made specific to the resident's locality. Questions were open to interpretation, they included; how would you describe your urban environment? What do you think the air is like outside your home? How would you describe the sound of the area you live in? How would you describe the environmental quality of this area? Participants were also asked to refer to their photographs and to the sound walk at any stage during the interview.

In summary a multi-method approach was employed in this study, combining environmental monitoring with participant-led photo surveys, sound walks and semi-structured interviews, with the aim to gauge both objective and subjective accounts of road traffic generated pollution.

Figures 2 & 3: Photograph of streetbox air and noise pollution monitor; example photograph taken by participant LP37 as part of their photo-survey titled "C in road".

Results

Table 2: Descriptive statistics from monitoring campaigns

Site No	Season/Round	CO *A* ppm	CO (*95 per*) (ppm)	CO Max (ppm)	Noise Level *A* dB(A)	Noise Max dB(A)	Noise Min dB(A)	PM *A* mg/m^3	*A* Temp (°C)	*Max* Temp (°C)	*Min* Temp (°C)
LM01	W1	0.27	0.51	1.36	48.6	67.8	28.7	0.026*	6.64	12.9	0.8
LM01	S2	0.25	0.42	0.86	33.8	52.8	22.9	0.023**	20.4	30.8	11.8
LM02	W1	0.45	0.95	1.44	48.6	64.2	39.2	-	10.8	26.3	1.6
LM03	W1	0.42	0.86	1.53	54.8	75.0	36.5	-	10.3	20.3	2.3
LM03	S2	0.49	1.16	2.39	-	-	-	-	19.9	29.9	11.9
LM04	W1	0.34	0.69	1.36	50.0	63.5	35.0	-	9.9	20.4	1.7
LM05	W1	0.32	0.87	1.69	-	-	-	-	12.5	21.1	5.9
LM06	W1	0.40	0.83	2.00	46.1	54.8	38.3	-	10.8	21.6	1.9
LM07	W1	0.29	0.55	4.50	36.7	62.6	25.4	-	4.8	14.6	0

W= Winter, S= Summer, 1= Round 1 monitoring campaign, 2= Round 2 monitoring campaign
A = Average
95 per = 95th percentile
Max = Maximum level recorded over monitoring period *Min* = Minimum level recorded over monitoring period
*= TSP, ** =$PM_{2.5}$

Firstly, the descriptive statistics on the parameters monitored are presented (see Table 2). Concentrations of CO measured over the monitoring periods are shown for each site. CO concentrations recorded in this study fall well within guideline levels (National Air Quality Objectives: 10ppm per 8hr running mean), with the maximum level of CO recorded being 4.5ppm (15-minute average) at Sans Walk, LM07. An overall average CO level can be calculated from the data collected over the different sites, for the different seasons this is 0.37ppm for summer and 0.36ppm for winter. Monitored noise levels are also presented in Table 2. It is important to note that these values are from differing L*eq* periods (15min, 1min and 3min), depending on the site and equipment used. In this format the data are not directly comparable, as the shorter recording period (L*eq*-1min) is likely to record higher/lower noise levels due to the shorter averaging period. Nevertheless, the levels illustrated in Table 2 do reflect the current situation in Clerkenwell. For all sites the average noise level was over 30dB(A), this is relatively high considering it encompasses data for 24hr periods. Maximum levels recorded reached 75dB(A) (L*eq*1min). The World Health Organisation (WHO) recommend the environmental noise guideline of 55dB(A); below this level people will be protected from becoming moderately or seriously annoyed. They also note how measurable effects of noise on sleep begin at levels of about 30dB(A); the more intense the background noise, the more disturbing is its effect on sleep (WHO 2001). So far this study indicates that people are exposed to levels bordering on these guidelines. Further necessary analysis separating the noise data into L_{day} and L_{night} levels is being carried out. Vehicle numbers and pedestrian numbers were also collected in Clerkenwell by a separate work package on the project (WP2: Land Use Diversity, Perdikogianni and Penn 2005). This data, shown in Table 3, can be utilised within this study to illustrate the varying conditions between each site.

Table 3: Pedestrian and Vehicle flows

Location	**Average daily pedestrians (Weekday)**	**Average daily vehicles (Weekday)**
Exmouth Market	588	48
Farringdon Road (S)	613	2519
Farringdon Road (N)	467	2042
Clerkenwell Road	688	3000
Sans Walk	147	4
St John Street	291	287
Skinner Street	155	583

Secondly, the experiences of the city centre residents have been explored. Participants ranged in age from 21-90 years old, with a mixture of housing occupations (social, private, owned), employment status (unemployed, retired, employed), and ethnic background, all thirty of them lived within the case study area. In Clerkenwell 680 photographs were taken by participants for the photo surveys, these were numbered and coded by a researcher via information provided by the participant on their log sheet and their interview. Content analysis showed that 39 of the 680 photographs were of road traffic related issues; nevertheless the subject matter was wide ranging, incorporating traffic congestion, public transport, parking and traffic control measures (see Figure 3). The sound walks recorded approximately 180-minutes of outdoor sound, the walks have been mapped in a geographical information system (GIS) together with the photographs. The interviews were transcribed and coded for analysis by a researcher. Not surprisingly all participants mentioned road traffic within their interview, many noted traffic as the dominant sound source in their area, and made direct references to traffic when discussing the quality of air outside their home. Interestingly, traffic noise and air pollution were often referred to simultaneously.

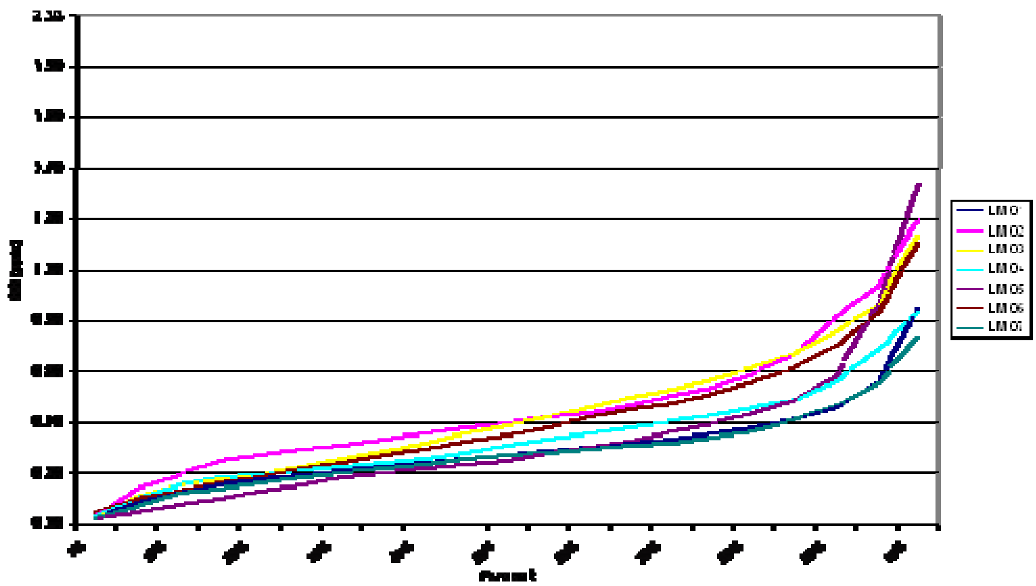
Discussion

The aim of this paper was to specifically explore and compare the relationships between two distinct accounts of noise and air pollution in an urban area. A multi-method approach was employed to collect both subjective and objective data; this resulted in the compilation of a multi-faceted set of data over different scales and perspectives, from the individual to area based. As interest lies in exploring the relationship between the two accounts a unifying approach to analysis was deemed necessary. A spatial analytical framework can be used to look at the data e.g. through comparison of the environmental monitoring sites, the coding of situational comments within the interview, and the mapping of the locations of photographs and sound walks.

Spatial variations in air and noise pollution are well recorded; Croxford *et al.*

(1996) and Kaur *et al.* (2004) have both reported marked variations in monitored levels within urban systems at the micro-level. The monitoring sites within this study offer an interesting comparison as they have relatively different characteristics, but are located within close proximity of each other. Figure 4 shows the levels of each pollutant over specific periods of time during the monitoring period, for each site. Similarities can be seen between the level of CO measured over the various sites, particularly Exmouth Market (LM01) and Sans Walk (LM07), both semi-pedestrianised streets. It was somewhat expected that Farringdon Road and Clerkenwell Road, would have higher CO levels than the other locations due to their higher traffic levels. Differences between the environmental conditions in Clerkenwell can also be seen through the comparison of the noise levels monitored. Looking at the average levels at each site there is a difference of over 20dB(A) between the highest and lowest average levels – a significant difference.

Figure 4: Percentile plot of CO over winter monitoring campaign



The interviews demonstrated that personal experiences of noise and air pollution were grounded spatially in local places and specific streets. Participants spoke of pollution occurrences in specific locations, particular roads and under contrasting conditions that they have experienced within their local area. The following quote by participant L17 expresses this clearly:

L17: “Yeah, and there are lots of little hidden areas like that which are really nice in Clerkenwell. But then you sort of, it almost makes it worse when you kind of emerge from those lovely quiet areas and go onto the main, big two big main roads, Clerkenwell Road and Farringdon Road, and then it’s sort of - oh God - and there’s all these lorries and buses”.

Spatial comparisons were particularly evident when participants talked about the environmental quality of the street they live on, many evaluated their street with other locations; be they local streets or previous residences. The quote below illustrates this; when asked about the air quality on their street participant L28 compared their street with a neighbouring, parallel road (Goswell Road). Their evaluation is based on a local environmental assessment undertaken either consciously or unconsciously:

I: “What’s the air like outside your home?”

L28: “Outside, it’s, I think it’s fine. For some bizarre reason, I’m sure it’s to do with the fact, I, I think one is more aesthetic than the other, I think the air on Goswell Road is worse than St John Street”.

I: “Is that right?”

L28: “Yeah. I don’t know why. I think it’s probably/ I think it’s cause there’s some quite big buses go up and down Goswell Road and it’s quite an arterial road into the city. If you continue going down you arrive at the Barbican, and there’s a lot of traffic goes down there, particularly cabs and the diesel engines. No I think the air’s fine.”

Conversely, the majority of participants remarked on the wider spatial connections of Clerkenwell, particularly its links to the rest of London, alongside the consequences of this accessibility:

I: “How would you describe the sound of the area you live in?”

L4: “Traffic mainly. Traffic is the main sound. Because it’s a main junction, Goswell Road and City Road are two of the main roads in London. There are other main roads, but in this area they are the two main roads.”

Accessibility was often seen as a positive aspect to living in Clerkenwell, participants frequently commented on the proximity to other areas within the city and the benefits this brought to them. Intriguingly, this emphasises that people do not see their local area in isolation, but part of a wider system.

It is worth noting that the interviews and photographs also yielded information concerning the perceived sources of the pollution. Participants did not only identify traffic as the main source, but also noted particular vehicles and behaviour by certain vehicles leading to noise incidents or increases in pollution; an aspect not covered by the monitoring. Participants L14 and L17 express this below:

L14: “And, and certainly the lorries around here when they come steaming down the road [Goswell Road] you, I mean it makes the house shake sometimes.”

L17: “And there’s traffic lights [on Clerkenwell Road], so um they’re, while they’re idling, I don’t know much about motoring, but if a car is idling I would imagine it giving out more fumes”

It is evident that public perception and understanding of air and noise pollution is embedded into people’s individual experiences, often from local knowledge and encounters.

The level and scale of information provided from each account (objective and subjective) is obviously different, from general to the local, from individual experiences to the measured. Residents identified main roads in their local area (e.g. Clerkenwell Road, Goswell Road, Farringdon Road), disparities they have experienced in the environment (e.g. quiet-loud areas) and explained daily encounters with these issues. These subjective accounts are very different to the monitored data which has objectively recorded the conditions in a number of sites. However combined together they provide detailed knowledge on the environmental conditions in the case study area, documenting the spatial nature of air and noise pollution to differing extents.

Conclusion

Multi-disciplinary projects, such as VivaCity2020, give rise to exciting research opportunities, innovations in methodologies and wide-ranging analytical approaches. Within this paper we present a project that is exploring environmental issues through combining different disciplinary approaches. The methodology employed in this study was effective in extracting subjective and objective accounts of noise and air pollution within a city centre location. We have found that through combining varying approaches and accounts a comprehensive knowledge base for certain environmental issues can be constructed. These differing accounts can compliment each other, for instance, the experiences of the people that live in urban areas can supplement monitored environmental data, providing information on the micro-scale, with details of specific sources and the personal effects of these conditions. Local knowledge can also be used to shape an environmental monitoring campaign. There is potential to develop the methods used in this study further; providing tools and techniques to enable and assist communication between city centre residents and professionals about environmental issues, or as even as a way to encourage further participation in research. However, restrictive time scales and budgets may limit widespread use.

The impact of certain environmental factors upon the overall perception of the quality of the environment should not be neglected. A holistic approach needs to be taken when looking at urban sustainability, focusing on the wider relationships and connections between the environment, society and the economy. Bickerstaff and Walker (2001) discuss Irwin's work on environmental perception, they explain how physical deterioration of other components of the local environment blend into how the public understand and respond to air quality around them. They state how a negative expression regarding the quality of the air could reflect a person's overall dissatisfaction with a place, alternatively, satisfaction with a residential area or neighbourhood may lead to a downplaying of any negative aspects to do with the quality of the air. Overall environmental quality was considered within this study. When asked if people liked the area they live in most stated that they really enjoyed living in Clerkenwell - the benefits of living in the city centre (e.g. accessibility to amenities, transport etc) outweigh the negatives (e.g. pollution, crime). These wider trades offs are also marked with a tolerance to the negative aspects of city centre living, the quote by participant L1 regarding traffic noise echoes what many said; "You get, I mean it's a terrible thing to say but you just get used to it".

Acknowledgements

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A method of building an aggregated indicator of air-pollution impacts

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Abstract

We intend to build a global environmental impact indicator of air pollution to assess transport infrastructures and technologies. This indicator should be simple and transparent to facilitate its use in decision-making. The intention is for the indicator to resemble the Global Warming Potential (GWP), which establishes a relationship between the emission of six greenhouse gases and the average temperature increase of the Earth. The indicator will therefore permit estimating the global environmental impact of transport-generated air pollution, while simultaneously conserving the value of the environmental impact of each type of air pollution and the emission assessment. This work is based on an impact typology, a set of indicators, and an aggregation architecture of atmospheric pollution.

The typology is established as a function of the specific and homogenous characteristics of each type of pollution in terms of pollutant, impact mechanism, target and environmental impact. To ensure exhaustiveness and non-redundancy, 10 types of air pollution impact are proposed: greenhouse effect, ozone depletion, direct ecotoxicity (this type of pollution excludes greenhouse effects on nature, ozone depletion, eutrophication, acidification and photochemical pollution), eutrophication, acidification, photochemical pollution, restricted direct health effects (not taking into account welfare, and excluding the effects on health of the greenhouse effect, ozone depletion, acidification and photochemical pollution), sensitive pollution (annoyance caused by odours and fumes), and degradation of common and historical man-made heritage.

Indicators similar to GWP can be identified in the literature for each type of atmospheric pollution, except for the degradation of common and historical man-made heritage, for which the financial cost of conservation could be used. However, these indicators do not seem to have achieved wide scientific consensus, except for GWP, which may make it necessary to continue research in this field.

Aggregating the different indicators is proposed by using an architecture composed of two structures that aggregate types of air pollution. One is based on the target affected, whereas the second has three dimensions, i.e. targets, space *and time. This architecture allows the indicator's users to establish a hierarchy of concerns for each type of atmospheric pollution.*

***Keys-words**: Air pollution, environmental impact, emissions, target, typology, aggregation, indicator.*

Introduction

Transport causes myriad environmental impacts that stir increasing concern among specialists, decision-makers and society as a whole. Given the diversity and complexity of the impacts, the procedures used for putting them into perspective vary considerably. A simple, transparent and synthetic evaluation of these impacts would facilitate decision-making. Assessing the environmental impacts of atmospheric pollutants is part of this process.

We propose to identify the different categories of environmental impacts caused by atmospheric pollution and characterise them by an indicator based on scientific knowledge, using as reference the global warming potential proposed by GIEC (2001). This assesses the environmental impacts of the greenhouse effect simply and synthetically, via the predictable increase of temperature due to the emission of six pollutants, without modelling concentrations, and then the chain of impacts (disturbances of climatic and ecological balances, etc.).

We then propose to aggregate these indicators in a global indicator in order to answer pertinent questions such as "is gasoline more pollutant than diesel fuel?", and "what is the progression of the environmental impacts of atmospheric pollutants emitted by road transport in France?" An attempt will be made to base this aggregation of indicators as much as possible on scientific knowledge, taking care not to step outside the legitimacy of the scientific community, for example, to weight impacts such as sensitive pollution, the greenhouse effect and health effects. Aggregating indicators should also mirror social concerns relating to air pollution.

Air pollution typology

A typology of air pollution will be formulated as a function of the different categories of environmental impact affected by atmospheric pollutants emitted by transport, before evaluating them with an indicator. This typology is based on physical, chemical, biological, ecological knowledge, etc. of atmospheric pollution. Each type of pollution corresponds to a category of homogenous environmental impacts defined by a specific combination of pollutants, impact mechanisms, targets and impacts. We seek to ensure that the typology is as accurate and exhaustive as possible, while at the same time reducing redundancies.

Three targets, defined as a group of homogenous receptors, are mainly affected by air quality: nature understood as ecosystems, i.e. the association between a physicochemical and abiotic (the biotope) environment and a living community characteristic of the latter (the biocenosis), humankind which we extract from nature and focus on its health as defined by the World Health Organisation (WHO, 1999), and man-made heritage for which a distinction is made between common buildings and historic ones. A fourth item is added to these three targets, i.e. the Earth, which is not a true target since it covers all the targets: the three previous targets and physical environments such as the atmosphere and the oceans. Therefore it is considered as a pseudo target.

An impact corresponds to the response of a target exposed to a condition of air quality. The impacts vary greatly and often form successions and chains of impacts. Thus greenhouse gas emissions lead first to global climatic warming, from which stem among other impacts such as the rise in sea levels in turn leading to flooding liable to cause the displacement of human populations and health effects, and modifications of ecosystems liable to cause loss of biodiversity.

Guinee et coll. (2002) distinguished several categories of environmental impact: the greenhouse effect, ozone depletion, photochemical pollution, acidification, eutrophication, health effects, and effects on nature. All these categories provide a relatively wide view of our air pollution typology, though leave out three categories of environmental impact to which atmospheric pollutants contribute: sensitive pollution, and the degradation of common and historic man-made heritage. However, a certain number of redundancies can be observed: health impacts due to photochemical pollution, ozone depletion and the greenhouse effect are, for example, redundant with health effects. To avoid redundancies while covering all the environmental impacts caused by the different pollutant gases emitted by transport, we redefine ten categories of impact (tab. 1):

- The greenhouse effect or more exactly the increase of greenhouse gases (GIEC, 2001).
- Ozone depletion: halogen compounds react with stratospheric ozone and lead to the depletion of the ozone layer. Although theoretically under control, this impact has not disappeared and is thus still of great interest (Académie des sciences, 1998; Solomon & Albriton, 1992).
- Photochemical pollution: nitrogen oxides and volatile organic compounds react to form tropospheric ozone outside urban centres, toxic for humankind and nature (Académie des sciences, 1993; Derwent & coll. 1998).
- Acidification: nitrogen oxide and sulphur dioxide are transformed into acid compounds that acidify the natural environment up to 1,000 km away from the point of emission (Potting & coll., 1998; Fuladi, 2002; Rey & Hermeline, 1994).
- Eutrophication: nitrogen oxides contribute towards increasing plant biomass whose excessive development leads to anoxia in aquatic environments, then harms fauna and flora (Finnveden & Östlund, 1997).
- Direct restricted health effects: effects on human health, which is restricted since it does not include harm to welfare and psychological aspects (integrated in sensitive pollution), and direct since it only considers effects due to exposure to primary pollutants. Health impacts due to secondary pollutants (acidification, photochemical pollution, etc.) are regulated by impact laws of different natures (Campagna & coll., 2002; Chiron & coll., 1996; Huijbregts, 2000a).

- Direct ecotoxicity: primary pollutants affecting human health can also affect nature (Labrot & coll., 1996; Huijbregts, 1999).
- Sensitive pollution: perceived by our senses, mainly sight and smell, it is composed of smoke, soiling and odours (Moch & coll., 2000; De Boer & coll., 1987; Joumard & coll., 1995).
- Degradation of common man-made heritage: this is mainly due to the affects of particles and corrosive products. It incorporates the impacts of photochemical pollution and acidification on buildings (Diren & coll., 2004).
- Degradation of historic man-made heritage: this is separated from the previous category as the impact is not chiefly sensitive or economic, but cultural and irreversible insofar as each work is unique and impossible to recreate identically. There is also the factor of loss of know-how in certain cases (Diren & coll., 2004)

We have decided not to take into account radioactive pollution in our typology since it is caused by nuclear power production and is not closely linked to the transport sector. The characteristics of these impacts on the environment are also of a very different nature.

Air pollution indicators

For each of the ten impact categories mentioned above, one or more impact indicators are required before their aggregation can be considered. These indicators must necessarily result from scientific knowledge of impact mechanisms and thus from the scientific discipline associated with each impact. However, specific communities are more or less well developed and structured vis-à-vis these impacts as a function of the progress made in obtaining knowledge and financial and human investment.

Thus a large number of scientists from a wide range of disciplines work on the greenhouse effect, aided by strong internal cooperation, particularly within GIEC. This organisation provides sound and synthetic information, in addition to an indicator known as potential global warming (PGW), which is the subject of widespread international agreement (GIEC, 2001). This indicator establishes a simple relation (weighted total) between the emission of six greenhouse gases and the average increase of the Earth’s temperature, which is the initial impact of the chain of impacts of the greenhouse effect. It permits evaluating the initial impact of any transport system or sub-system.

Table 1: Scientifically designed typology of air pollution.

Tableau 1 : Typologie de la pollution de l’air définie scientifiquement.

Targets		Types of air pollution	Pollutants	Characteristic impacts	Indicator	Position / impact chain
Earth	Physical environments Ecosystems Health Heritage	Greenhouse effect	**CO_2, HFC, N_2O et CH_4**	**Global warming**	**PRG**	**Start**
		O_3 depletion	**CFC, HFC, HCFC**	**Degradation of the ozone layer**	**PDO**	**Start**
Nature	Species Ecosystems	Direct ecotoxicity	**PM_{10}, NO_x, COVNM, SO_2, HAP et Cu**	**Mortality Morbidity Loss of biodiversity**	**PET**	**Start**
		Eutrophication	**NO_x**	**Anoxia of environments**	**PE**	**Start**
		Acidification	**NO_x et SO_2**	**Acidification of environments**	**PA**	**Start**
		Photochemical pollution	**NOx, COVNM, CO**	**Photochemical oxidation**	**PCOP**	**Start**
Humankind	Health in the strict meaning	Direct restricted health effects	**PM_{10}, NO_x, COVNM, SO_2, HAP, Cu**	**Mortality Morbidity**	**PT**	**Start**
	Welfare	Pollution sensible	**Visual polluants: particle emissions Odour pollutants : SO_2, VOC with low molecular weight**	**Annoyance**	**PO**	**Start**
Man-made heritage	Common	Degradation of common man-made heritage	**Particles, NOx, COVNM, Cu, SO_2.**	**Degradation**	**Economic cost of restoration**	**End**
	Historic	Degradation of historic man-made heritage	**Particles, NOx, COVNM, CO, SO_2.**	**Loss of heritage**		**End**

On the other hand, specialists in sensitive pollution seem to be fewer and more cut off from each other. The literature in the field is less abundant. What is more, there do not appear to be any indicators linking an annoyance level to a quantity of smoke, soiling or odour emitted. Nonetheless, chemists have developed a global potential odour indicator (PO), built in the same way as the PGW, that establishes a relation between an intensity of odour and a quantity of pollutant emitted (Guinee & coll., 2002). The global odour is given by the total emissions of pollutants weighted by a coefficient corresponding to an olfactive perception threshold. However, this indicator has not achieved consensus since many specialists underline the fact that sensitive pollution is characterised by annoyance, which is not directly related to the intensity of an odour, but far more to its variation through time. There is no similar indicator for sensitive visual pollution. Despite this we consider opting for odours as the indicator for sensitive pollution for want of a better one.

As for the degradation of common and historic man-made heritage, there is a lack of both specialists and literature. However, the diversity and extent of impacts are more limited, since they are essentially physicochemical impacts that affect materials, giving rise to general scientific consensus. Economic theories are moreover well-adapted for assessing the commercial goods and services linked to maintaining buildings, which explains the widespread use of economic indicators for evaluating environmental impacts of pollution on buildings (O'Connor & Spash, 1999). These indicators are very simple and represent final impacts rather than intermediate ones. Nonetheless they do not establish a link between the emission of pollutants and the intensity of degradation. They also make use of more or less disputed concepts such as agreement to pay, to give an economic expression to the value of a heritage that has no relation to any market, for example, a historic monument whose value is above all cultural (Spash & Hanley, 1993). Despite these criticisms, for the sake of simplicity we consider selecting the economic cost of maintaining man-made heritage for assessing the environmental impacts of degradation to common and cultural man-made heritage.

The field of health has many specialists, is well structured and is the source of high quality literature on indicators (Chiron & coll., 1996). However, the great heterogeneity of the impacts dealt with by different specialities has made it difficult to develop a synthetic indicator of health impacts. To our knowledge, there is only one global health effect indicator, i.e. potential toxicity (PT) which corresponds to the total pollutant emissions weighted by the toxicity of each pollutant (Huijbregts, 2000a). However, this indicator has inspired very little agreement. Given the importance of this impact, we consider calling on a college of specialists to build a new direct restricted health effect indicator.

The same observations apply to direct ecotoxicity, for which an indicator similar to PT exists, called potential ecotoxicity (PET) (Huijbregts, 1999), though this indicator is not subject to general consensus either.

Lastly, although the problems of eutrophication, acidification, photochemical pollution and ozone depletion are dealt with by a large number of specialists, mostly biologists, ecologists, chemists and health experts, these appear to be dispersed. The literature takes stock of the physicochemical mechanisms and ecological and health impacts of these four impacts. Works performed by chemists also permit proposing synthetic indicators, built in the same way as the PGW, i.e. potential eutrophication (PE) (Huijbregts, 2000b), potential acidification (PA) (Huijbregts, 2000b), the photochemical oxidant creation potential (POCP) (Anderson-Skold & coll., 1992; Derwent & coll., 1998), and the ozone depletion potential (ODP) (Solomon & Albritton, 1992). Once again, these indicators are not subject to widespread consensus.

Consequently the literature shows that an impact indicator exists for each type of pollution (tab. 1). The great majority of these indicators are built according to the same structure as that used for the PGW, with the exception of degradation of common and historic man-made heritage, which is assessed on the basis of an economic indicator. They generally establish a relationship between pollutant emissions and an impact characteristic of a type of pollution, which permits the assessment in a simple way of the contribution made by transport to each type of pollution. The impact chosen as characteristic is systematically situated at the start of the impact chain, implying the indirect hypothesis that the intensity of the final impacts is proportional to that of the initial impact, which could hardly be a more approximate approach. Despite this fact, they are much used today for analysing product lifecycles, and they appear to be the only tools available at present for performing an aggregated assessment of types of atmospheric pollution caused by transport.

Aggregation architecture used for air pollution

Due to the diversity of types of air pollution it is not possible to make a global assessment of all its impacts on the environment without aggregating the different types of pollution (Faucheux & O'Connor, 1998). However, aggregating or arbitrating between types of pollution as different as the greenhouse effect and the degradation of common and historic man-made heritage, or health impacts and sensitive pollution, depends on personal and collective preferences. To avoid making use of often personal, non-justified and opaque procedures for synthesising data, we propose formulating an architecture for aggregating types of atmospheric pollution, thus of indicators. This architecture should help the user of the indicator to aggregate the different types of pollution and identify the types of atmospheric pollution they consider the most important.

Indeed, only the society concerned or its representatives can perform such arbitration legitimately (Funtowicz & coll., 1997). Scientists can only facilitate the expression of such arbitrations and in no way can they pronounce as scientists on the predomination of such and such an impact. To do this society and those that represent it should be capable of distinguishing between the ten types of pollution chosen. On the other hand, although the four targets affected by atmospheric pollution (the Earth, nature, humankind, man-made heritage) are well distinguished socially, this is so for only seven types of pollution: the greenhouse effect, ozone depletion, effects on ecosystems, health effects, sensitive pollution, the degradation of common anthropic heritage, and the degradation of cultural man-made heritage. This distinction is also rather fragile since it is based exclusively on the knowledge of terms and not their specific meaning (Weber & Vanolli, 1986). Although we at INRETS hope to produce a more robust and socially pertinent typology by carrying out the nation-wide survey now in progress on the perception of the environment, we are obliged to conclude at present that this socially pertinent typology does not correspond to the typology based on ten types of impact established scientifically.

Consequently, a more complex aggregation architecture is required. We propose a direct aggregation structure by target, and an indirect structure with three dimensions well-known to all: the targets, space and time.

1. Target-based direct aggregation structure

In this case the aggregation of types of pollution is carried out according to the preference given to each target affected, to which correspond one to four types of pollution distinguished socially and two to four types of pollution defined scientifically (tab. 2). Therefore, the more a user (e.g. decision-maker) privileges the target Earth, the greater the importance they give to the greenhouse effect and ozone depletion. These two types of pollution can therefore be aggregated within the pollution that mainly affects the Earth.

Table 2: Direct aggregation structure of types of air pollution, according to targets affected.

Tableau 2 : Structure d'agrégation directe des types pollution de l'air par cible affectée.

Scientifically designed air pollution typology	Types of air pollution distinguished socially	Targets	
Greenhouse effect	Greenhouse effect	Globe	
Ozone layer depletion	Ozone layer depletion		
Direct ecotoxicity	Effects on nature	Nature	
Eutrophication			
Acidification			
Photochemical pollution			
	Health effects	Health in strict meaning	Humankind
Direct restricted health effects			
Sensitive pollution	Sensitive pollution	Well being	
Degradation of common man-made heritage	Degradation of common man-made heritage	Common heritage	Heritage
Degradation of cultural man-made heritage	Degradation of cultural man-made heritage	Cultural heritage	
	Aggregation by specialists	*Political aggregation*	*Political aggregation*

This aggregation can only be partial, since a type of pollution distinguished socially can sometimes correspond to several types of scientifically established pollution. This leads, very occasionally, to switching a social or political aggregation by target, with an aggregation established by the scientific community. Thus direct ecotoxicity, acidification, eutrophication and photochemical pollution must be aggregated using this process in effects on ecosystems.

This process, which lacks legitimacy, and the weakness of social distinction which relies only on the knowledge of terms, leads us to propose an additional aggregation structure, in order to make our aggregation architecture more robust.

2. Indirect aggregation structure

This second aggregation structure relies on the target dimension and two other very well-known dimensions: time and space, with the aim of distinguishing types of pollution that are not considered as social if viewed solely from the standpoint of targets. These three dimensions are pertinent for the aggregation since only one type of scientifically defined pollution out of the ten corresponds to a target, a time frame (short, medium or long term) and a geographic characteristic (global, regional, local). The only exceptions are:

- acidification and eutrophication (target: nature, regional, medium term),
- the greenhouse effect in the medium term and ozone layer depletion (global, medium term),

- sensitive pollution and direct restricted health effects in a short space-time frame (target: humankind),
- degradation of common and historic man-made heritage (buildings, local, medium and long term).

The first case can only be treated by the scientific community, since it involves pollutions that are socially inseparable, both directly and indirectly. The three other cases correspond to types of pollution distinguished socially which therefore do not lead to aggregation problems.

We also observe that ecotoxicity, direct restricted health effects and degradation to common and historic man-made heritage are not or hardly subject to differentiation through time. This amounts to de-aggregating certain scientifically defined types of pollution, which may contribute, for example, towards the development of a global indicator of direct restricted health effects.

Lastly, we propose to make use of surveys and other participatory methods such as forums to define the number of spatiotemporal classes and their limits (short, medium and long term, local, regional, global), and to know the level of social concern corresponding to each dimension. Finally, the levels of concern corresponding to indirect and direct aggregation structures will be combined in order to produce a mean result that will probably be more robust than either structure alone.

Conclusion

The air pollution typology proposed here contributes towards the scientific and transparent identification of the different types of pollution to be taken into account in an air pollution indicator. An indicator must correspond to each type of pollution characterised by a specific environmental impact category, but this indicator must result from collaboration between specialists in this category of impact. Consequently, indicators can be identified by type of pollution, although certain indicators do not enjoy widespread scientific consensus. We propose to aggregate these types of pollution and their corresponding indicators by using a typology considered as socially pertinent, less detailed than the typology defined scientifically, along with a direct method based on the differentiation of the major targets affected, and an indirect method that associates spatiotemporal dimensions. This architecture permits framing the aggregation of types of atmospheric pollution in a scientific process, by helping the user to assign a preference to each of the types of pollution defined scientifically, including in the case where they are not socially distinguished by direct and indirect aggregations. This method should permit building a global impact indicator for air-pollution.

Acknowledgements

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Hybrid Electric vs Conventional Vehicle: Life Cycle Assessment and External Costs

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Abstract

The environmental damage costs ("external costs") of the hybrid electric vehicle (HEV) are compared with those of a conventional gasoline vehicle, by carrying out a life cycle assessment (LCA) coupled with an impact pathway analysis based on the ExternE project series of the EC. The LCA data for vehicle production are from Delucchi et al of UC Davis and from studies by MIT, those for fuel production from the well-to-wheel software of Argonne National Laboratory. The HEV is a Toyota Prius whereas the Toyota Camry and Corolla are taken to specify the conventional alternative. The damage costs of the HEV are significantly lower than those of a conventional vehicle, especially in urban driving. The difference is proportional to the fuel savings since the damage costs from the vehicle production stage are nearly the same. The damage costs of the conventional vehicle and HEV are, respectively, 1.0 and 0.66 Eurocents/km.

Key words: *life cycle assessment, hybrid electric vehicle, conventional vehicle, external costs, ExternE, impact pathway analysis, greenhouse gases*

Introduction and Objectives

The hybrid electric vehicle (HEV) appears as an attractive option until fuel cell vehicles are sufficiently well developed to compete in the market. But whereas several manufacturers, especially Toyota and Honda, are already actively selling HEVs in the USA, that has not yet been the case in the EU. The objective of the present paper is therefore to quantify the environmental advantages of the HEV for European conditions.

For a correct evaluation and comparison of different vehicle options a life cycle assessment (LCA) is required because one could get a very misleading picture by just looking at the pollutants emitted during driving. Whereas numerous LCA (life cycle assessment) studies of cars have been carried out [e.g. Delucchi 2003, MacLeana & Lave 2003, Weiss et al 2000], their results for the environmental effects have been reported only in terms of the quantities of the emitted pollutants. That makes the interpretation and use of these studies difficult because it is not obvious how harmful the different pollutants are. The results would be much simpler to understand if the impacts of all the pollutants were expressed in terms of a single common metric. The most convenient metric is monetary, i.e. in terms of damage costs (such costs are often called “external costs” because they are not included in market prices). The monetary valuation of environmental damage has the great advantage of allowing a direct comparison with conventional costs and a cost-benefit analysis. For that one needs an impact pathway analysis (IPA).

In recent years enormous progress has been made in this regard, thanks to several major research projects in the USA and the EU. The most extensive and up-to-date research of the kind is the ExternE [1998, 2000, 2004] project series of the EU, a large multinational and interdisciplinary undertaking in which the present authors are active participants. Environmental cost-benefit analysis is now used extensively in the USA and the EU. In particular, the European Commission recently decided that the internalization of external costs is a key tool for the attainment of sustainable development (one of the main policy goals of the EU). This obliges industries that manufacture or sell their products in Europe to take into account their environmental damage costs.

Methodology

LCA

The life cycle of a vehicle can be broken down into the following stages:

- Production of the materials needed for making the vehicle,
- -Assembly of the materials,
- Fuel feedstock (e.g., extraction of petroleum from the ground),
- Fuel supply (refining of petroleum and transport of fuel),
- Utilization of the vehicle,
- Disposal of the vehicle at the end of its life.

In terms of damage costs the vehicle utilization accounts by far for the largest contribution, about two thirds of the total.

Disposal of the vehicle at the end of its life involves as major steps i) dismantling the vehicle, ii) recycling the recyclable fraction, and iii) disposing the rest fraction in a landfill or incinerator. For the present study we have found it most convenient to include the recycling in the first stage (Production of the materials), because the available data for materials include the emissions due to recycled materials. The impacts due to the dismantling of the vehicle are entirely negligible, as can be seen from the fact that the energy for dismantling is only about 1/3 of the energy for the assembly of the materials [Stodolsky et al., 1995] and that the impacts of the assembly are only a very small part of the total life cycle impact. The impacts from the disposal of the rest fraction are negligible if current environmental regulations for waste disposal are respected [see Rabl, Spadaro & McGavran 1998]. For these reasons we do not show explicit results for the disposal stage.

We have used the following sources of information for the LCA phase:

- A large and comprehensive review paper by MacLeana & Lave [2003],
- A major LCA of many vehicle and fuel technologies by MIT [Weiss et al 2000],
- The most comprehensive well-to-wheel analysis (the GREET software of ANL [2004]),
- The most comprehensive LCA of vehicle and fuel technologies [Delucchi 2003],
- For on-the-road emissions and fuel economy USEPA and independent test organizations.

1. IPA

The IPA is based on ExternE [2004]. The principal steps of this analysis can be grouped as follows:

- Emission: specification of the relevant technologies and the environmental burdens they impose (e. g. kg of NO_x per km emitted by vehicle);
- Dispersion: calculation of increased pollutant concentrations in all affected regions (e. g. incremental concentration of ozone, using models of atmospheric dispersion and chemistry for ozone formation due to NO_x);
- Impact: calculation of the dose from the increased exposure and calculation of impacts (damage in physical units) from this dose, using a dose-response function (e. g. number of cases of asthma due to this increase in ozone);
- Cost: the monetary valuation of these impacts (e. g. multiplication by the cost of a case of asthma).

A more detailed description of the IPA methodology can be found in the reports of ExternE [www.externe.info].

We have taken into account all the impacts on health and environment that have been quantified by ExternE, in particular

- Health,
- Agricultural losses,
- Damage to materials and buildings, and
- Global warming.

There are of course additional types of impact, for instance acidification and eutrophication of ecosystems, but in monetary terms they turn out to be far less important for the pollutants emitted during the life cycle of vehicles.

Since the present analysis has been concerned only with damages due to pollutants, we have not considered noise. Preliminary estimates by the ExternE team suggest that in European cities noise can impose external costs of a magnitude comparable to those of pollution. Since the HEV is much more quiet it enjoys another major advantage in terms of external costs.

The damage costs per kg of pollutant are listed in Table 2; they are typical values for central Europe. Based on numerous site-specific calculations for ExternE we estimate that the cost per kg of $PM_{2.5}$ for urban driving is a factor of ten higher than for rural driving. However, for NO_x and SO_2 the difference between urban and rural emissions is negligible because their impacts are believed to arise from secondary pollutants (aerosols and ozone) that are created gradually and further away from the emission site. For CO and Pb the variation with local population density can also be neglected because the dispersion of these pollutants in the environment covers very large distances (Pb through food chain pathway, CO because of small deposition velocity). We do not show the damage cost of CO because it is so small that its contribution to the total would not even be visible in the graphs.

Table 2: Damage costs in € per kg of pollutant. Based on ExternE [2004]. The number for PM_{10} is appropriate for particles emitted by industrial sources, the one for $PM_{2.5}$ is for particles emitted by cars (much higher because emission at ground level and mostly in cities; also the particles are smaller and more toxic).

Tableau 2 : Coûts de dommage par kg de polluant, basé sur ExternE [2004]. Le chiffre pour PM_{10} est pertinent pour des sources industrielles, celui pour $PM_{2.5}$ pour les voitures (plus important parce l'émission est au niveau du sol, et les particules sont plus petites et plus toxiques).

NO_x	SO_2	NMVOC	CO	Pb	Ni	PM_{10}	$PM_{2.5}$	CO_{2equiv}
2.4	4.6	1.14	0.0016	1600	3.8	22.4	373	0.019

NMVOC = non-methane volatile organic compounds

The cost of the greenhouse gases (CO_2, CH_4 and N_2O) does not depend on the emission site. For greenhouse gases other than CO_2 the cost are be expressed in terms of equivalent emissions of CO_{2equiv}, using the global warming potential (GWP). The damage cost per kg of CO_2 is uncertain and controversial. The estimates by various experts fall in the range of about 0.002 to 0.050 €/kg of CO_2. Here we follow ExternE [2004] in using 0.019 €/kg of CO_2. Not only is this a very reasonable choice in view of the various estimates but it equals the abatement cost in the EU that is implied by the acceptance of the Kyoto protocol. Because of the growing development of a worldwide market for trading CO_2 emissions, the effective cost per kg of CO_2 permits is fairly close to this value.

Assumptions and Data

1. Reference vehicle

Ideally, the HEV should be compared to a gasoline version that is identical except for the drive train. However, since the difference in damage costs is crucially dependent on the fuel consumption, we have chosen the Toyota Prius as the HEV since it was the first to enter the market and its consumption data are more reliable than for more recent models that are offered in both versions. Unfortunately for the present analysis there is no exact conventional equivalent. The Toyota Camry comes closest, although it is slightly larger; for that reason we have also analyzed the data for the Toyota Corolla, not shown here.

2. Production of vehicles

We have taken the inventory data of Delucchi [2003], Weiss et al [2000] and AMM [2003] (they are very similar) and scaled them in proportion to the actual mass of the cars under consideration. The first two of these sources provide separate inventories for conventional and for hybrid cars.

The vehicle disposal stage involves the dismantling of the vehicle and recovery of any materials that can be reused, and the treatment of the remainder. Metals are easy to recycle, and at the present time already 90% to 95% of the metals are reused, either in the same or in different sectors of the economy. The quantity of glass in a car is small and so are the associated impacts. Plastics are much more difficult to recycle, and it is not clear what fraction will find another use. Fluids in cars are either water or petroleum based. Most of the latter are already recycled in some form or other (including thermal recycling via incineration). In any case the recycled fractions are bound to increase under the growing pressure from governments, especially in the EU, to increase the recycling of waste. The MIT study [Weiss et al 2000] assumes as target for 2020 that 50% of the plastics are recycled; Delucchi [2003] does not try to estimate a percentage. In the present study we take 5% for plastics and rubber.

For this analysis it does not matter whether the materials are reused in vehicles or in other sectors, because the emissions avoided by recycling are essentially independent of the sector in which the production of virgin materials is avoided. We assume that an average passenger car is driven 126,000 miles during its lifetime.

3. Emissions during use of vehicle

The assumptions regarding fuel consumption and tailpipe emissions are summarized in Table 1. Test results by the EPA are based on standard and supplemental federal testing procedures (http://www.fueleconomy.gov), but are not representative of real-world driving conditions, due to such factors as personal driving habits, on-road driving conditions, on-board equipment malfunction, air-condition use, and extra load. We have therefore looked at the results of actual driving tests by independent testing organizations such as Consumer Reports (http://www.ConsumerReports.org), Consumer Guide (http://www.auto.consumerguide.com) and the US Department of Energy (advance Vehicle Testing Activity, http://www.eere.energy.gov/), and we have chosen the mean of the tests of Consumer Reports as a representative value. The consumption of the Corolla is 8.2 L/100 km (city and highway combined), somewhat lower than the Camry.

Table 1: Vehicle data, consumption and tailpipe emissions.

Tableau 1: Les données des véhicules, consommation et émissions.

	Camry	Prius
Engine	2.4L, 4-cyl	1.5L, 4-cyl
Vehicle weight (kg)	1,435	1,311
Fuel consumption (L/100km)		
EPA (city/highway) combined	(10.2/7.4) 8.9	(3.9/4.6) 4.3
Independent tests – combined	10.1	5.5
Tailpipe emissions (g/km)		
$PM_{2.5}$ (urban emissions)	0.0034	0.0034
$PM_{2.5}$ (rural emissions)	0.0028	0.0028
NO_x	0.062	0.019
SO_2 (at 120 ppmS)	0.018	0.0098
CO	2.61	1.31
NMVOC	0.124	0.076
$CO_{2,equiv}$ (CO_2, CH_4, N_2O)	239	131

Since we have no measured emissions data for these vehicles, we assume the regulatory limit values of the national Tier 2 emission standards of the USA. They are to be phased-in over a five year period, beginning with 2004 car models. That the actual emissions are expected to be lower introduces some uncertainty, but that turns out not to matter very much because the external costs are dominated by CO_2 emissions which are determined by the fuel consumption. We do not show emissions data for the Corolla because the "bin" of the TIER 2 regulation to which it has been assigned allows some of the emissions to be higher than the Camry, even though the vehicle is smaller. The ideal conventional reference vehicle for the Prius would have lower emissions than the Camry.

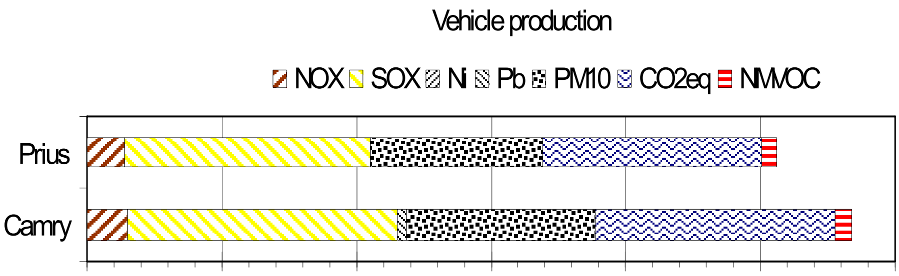
Results

1. Vehicle production

The damage cost due to the vehicle production stage is shown in Figure 1, with breakdown by pollutant. It represents very roughly one percent of the price of the car. Greater detail for the contribution of the individual materials and of vehicle assembly is provided in Figure 2 for the Camry. The corresponding numbers for the Prius are very similar, slightly smaller in proportion to the vehicle mass; if the rest of the car were identical the only difference would come from the battery (Pb for the Camry, a larger quantity of the much less toxic Ni for the Prius). Compared to the total cost of this stage the contribution of vehicle assembly is only about a quarter. NMVOCs are relatively more important for assembly than for the individual materials because of emissions during painting and coating. In terms of damage costs, the most important materials are iron and steel, aluminum and plastics.

Figure 1: Damage cost due to the vehicle production stage.

Figure 1 : French translation: Coût des dommages dus à la fabrication du véhicule.



2. Well-to-tank and tank-to-wheel

For the present study, the GREET model (version 1.5a, dated April 2001) was used for the well-to-wheel stages. This software was developed by ANL [2004]. Since its first release in 1996, GREET has been used extensively by industry, government and academia. The default input data and assumptions in GREET apply to the United States (e.g.., electricity mix for feedstock production, fuel specifications, upstream boiler emission factors, etc.); however, the technologies are sufficiently similar in the EU. Figure 3 shows the breakdown by pollutant for the stages of the well-to-wheel analysis. Only the Camry is shown, because these impacts are proportional to fuel consumption.

Figure 2:Contribution of the individual materials and of vehicle assembly to damage cost of vehicle production, for the Camry. For the Prius the contributions from Pb and Ni are negligible.

Figure 2 : Contribution des matériaux et de l'assemblage.

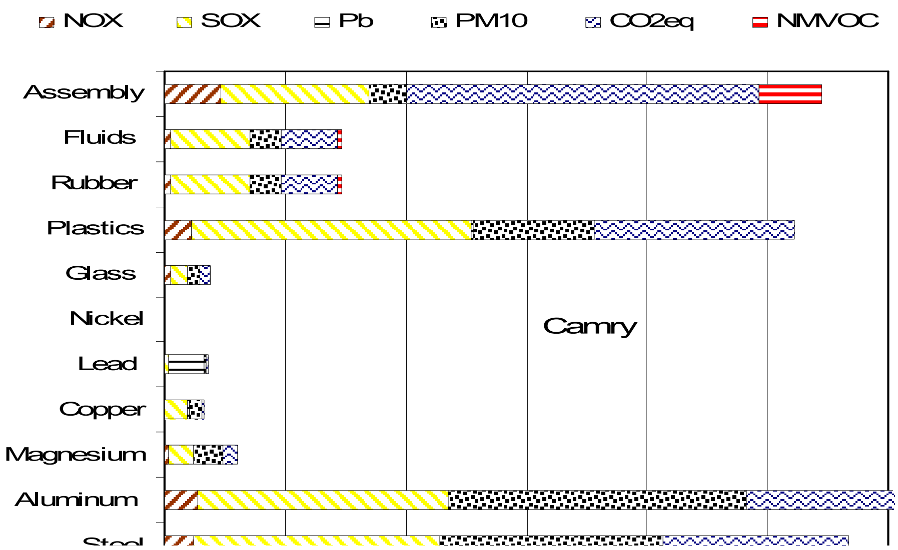
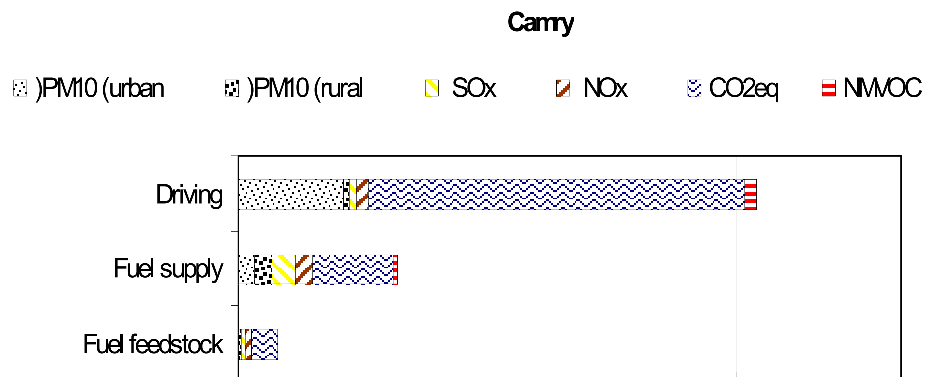


Figure 3. Contribution of the pollutants to the well-to-wheel stages, for the Camry.

Figure 3 : Contribution des polluants dus au carburant.

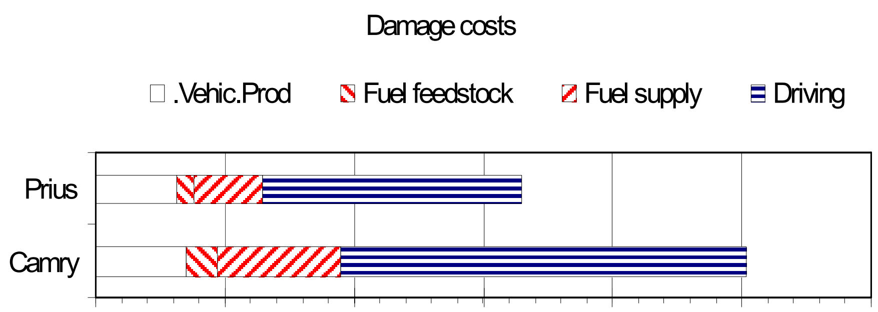


3. Comparison of total cost per mile

The results for the total damage cost per mile are shown in Figure 4. Of course, the hybrid vehicles cause by far the lowest damage. Damages from vehicle production are essentially proportional to vehicle mass since the relative composition does not vary much.

Figure 4: Total damage cost Eurocents/km.

Figure 4 : Coût total des dommages.



Conclusion

In urban driving the HEV has much lower damage costs than the conventional alternative (0.66 vs 1.0 Eurocents/km), the difference being proportional to the fuel savings. The damage cost would be even lower if the engine were a diesel with particulate filter, a device that can reduce the PM emissions to the level of gasoline engines. A further advantage can be gained from the plug-in option (charging the HEV with baseload electricity at night); this option is strongly dependent on the driving pattern.

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Aide multicritère à l'évaluation de l'impact des transports sur l'environnement

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Résumé

Ce travail vise à contribuer à la conception d'un outil informatique pour des responsables publics qui souhaiteraient s'appuyer sur un modèle afin de gérer de manière durable les problèmes environnementaux dûs aux transports et cela dans un contexte de démocratie participative. Nous proposons donc une « aide multicritère à l'évaluation de l'impact sur l'environnement » qui consiste à structurer hiérarchiquement un ensemble d'objectifs, puis à appréhender leur degré d'atteinte à l'aide de critères dans un but d'établir un diagnostic, de donner l'alerte, de réaliser une analyse tendancielle et non systématiquement d'envisager une prise de décision. Compte tenu de la complexité du problème et pour faire face au conflit qu'il existe entre la simplicité et la transparente d'un résultat d'évaluation, nous proposons, à tous les niveaux de la hiérarchie d'objectifs, une agrégation multicritère qui s'inspire fortement des méthodes Electre et de l'indice Atmo. Nous illustrons cela par une modélisation du problème et un prototype.

Mots-clefs : *évaluation, multicritère, agrégation, environnement, transports.*

Abstract

Multicriteria evaluation aiding of the impacts of transports on environment.

This work leads to contribute to the conception of a computer science tool for the public in charge who would wish to use a model in order to manage environmental problems due to transports in a sustainable way, and in the context of participative democracy. Thus we propose a "multicriteria evaluation aiding of environmental impact of transports" which consists of structuring a set of objectives as a hierarchy, then to apprehend their degree of achievement using criteria, with the goals of: establishing a diagnosis, giving alarm or carrying out a trend analysis and not systematically considering a decision problem. Taking into account the complexity of the problem and to face the conflict between the simplicity and the transparency of an evaluation's result, we propose, at all the levels of the hierarchy of objectives, a multicriteria aggregation which is strongly inspired by the Electre Tri method and the

Atmo index. We illustrate that by a modelisation of the problem, a functional description of such a tool and a prototype.

***Keys-words**: evaluation, multicriteria, aggregation, environment, transports.*

Introduction

L'activité de transport a des conséquences tant économiques que sociales et environnementales. Si l'on s'intéresse aux effets à moyen et long terme des transports sur l'environnement, pour préserver la situation pour les générations futures en assurant le développement économique et social, on est exactement dans le cadre du développement durable. En conséquence, les nombreuses prises de décisions publiques concernant les transports entrent pleinement dans ce cadre. Face à cela, une demande de plus en plus forte provenant des acteurs entrant en jeu dans une démocratie participative (les élus ou responsables politiques, l'opinion publique et les représentants associatifs) témoigne du besoin d'outils permettant d'appréhender au mieux l'impact des transports sur l'environnement afin de considérer ces aspects conjointement aux aspects économiques et sociaux et répondre ainsi aux enjeux d'un développement durable des transports. Il s'agit avant tout d'une demande d'instrumentation afin d'aider à évaluer, de façon globale, l'impact des transports sur l'environnement.

Répondre à cette demande est une tâche très ambitieuse qui a donné, entre autre, naissance à un projet de recherche, le projet « Prospectives et Indicateurs Environnementaux » (PIE) [Pujol, 2000, Maurin, 2000 a] mené par le Laboratoire Transports et Environnement de l'Institut National de Recherche sur les Transports et leur Sécurité (Inrets-Lte).

Une des raisons à l'origine de la demande d'instrumentalisation dont il est question dans le projet PIE est que, dans le domaine des transports et de l'environnement, les décideurs sont bien souvent confrontés à l'analyse de longs rapports d'experts mélangeant à la fois des informations descriptives, qualitatives et quantitatives. Ils ont alors une grande difficulté, d'une part à considérer chaque aspect dont il est question dans ces rapports, d'autre part à se forger une opinion globale. L'intelligibilité et l'agrégation de l'information sont donc souhaitées.

Les paniers d'indicateurs environnementaux sont nombreux. Ils diffèrent tant dans le nombre d'indicateurs qu'ils retiennent, que dans les buts pour lesquels ils sont construits. Pas ou peu d'applications proposent une agrégation des indicateurs, sauf dans le cas particulier ou l'on souhaite synthétiser l'information en vue d'aider décider entre plusieurs projets ou alternatives : elles entrent alors dans le cadre théorique de l'« aide multicritères à la décision ».

Cependant, on ne souhaite pas, ici, concevoir un outil dédié à un seul problème décisionnel, mais un outil permettant d'aider à évaluer une ou des situations. Un enjeu théorique est donc d'utiliser les différents concepts fondateurs des méthodes d'aide multicritère à la décision pour tenter de proposer une aide à l'évaluation qui permette l'agrégation multicritère. C'est ce que nous entendrons par « aide multicritère à l'évaluation ».

Aide multicritère à l'évaluation

1. L'évaluation

Dans [Rousval, 2005], la définition suivante de l'évaluation est proposée : « L'évaluation est un processus qui vise à quantifier et/ou qualifier un système grâce à toute information nécessaire à la construction de critères permettant d'appréhender au mieux l'atteinte de l'ensemble des objectifs concernant ce système et jugés pertinents dans le cadre d'une activité plus large mais préalablement identifiée ». Ce processus, représenté dans la Figure 1, englobe les cinq étapes suivantes, qui font intervenir différents acteurs :

- définir le système à évaluer,
- décliner un système de valeurs à travers des objectifs,
- sélectionner les critères mesurant l'atteinte des objectifs,
- évaluer les performances sur les critères,
- consulter les résultats de l'évaluation.

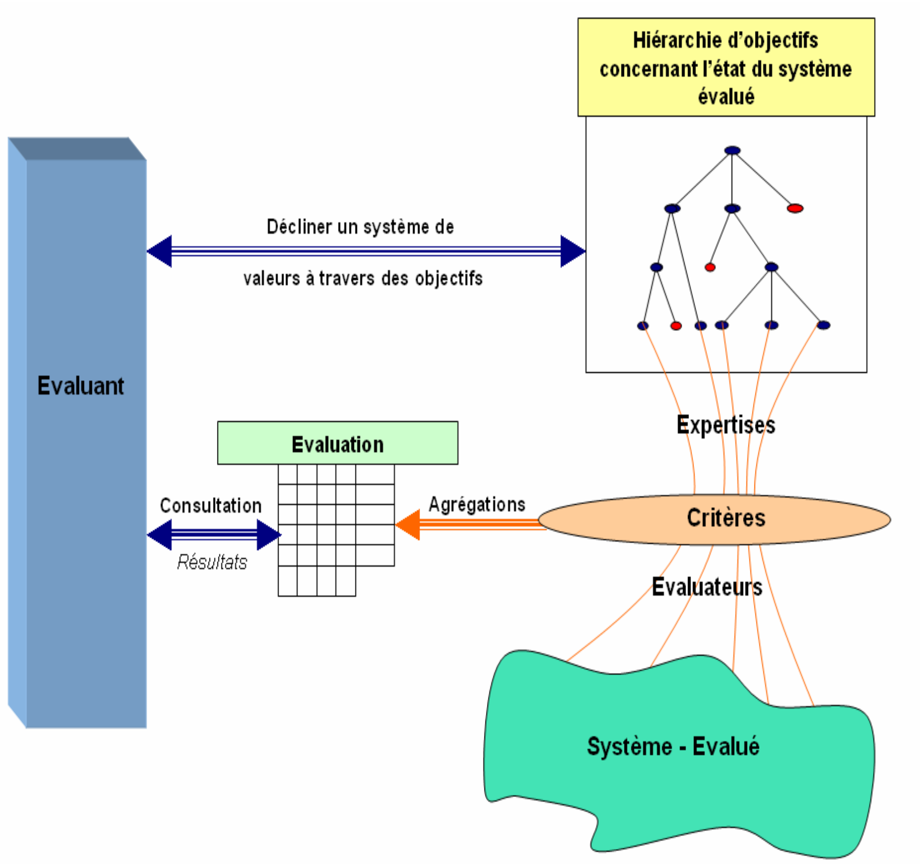
Cette façon d'envisager l'évaluation prendre place dans le modèle de pensée « évaluation-gestion ». Elle est une approche de rationalisation par les objectifs qui tient compte de la distinction fin-moyen pour structurer les objectifs. La seconde étape du processus d'aide à l'évaluation consiste à décliner le système de valeurs du décideur à travers des objectifs afin de construire le référentiel de l'évaluation. C'est cette étape que nous détaillons dans la section suivante.

2. Structuration des objectifs

Dans la littérature sur les méthodes d'aide à la décision, notons tout particulièrement les travaux de [Keeney, 1992]. Dans « Value-focused thinking » l'auteur propose une approche des problèmes d'aide à la décision en se focalisant sur les valeurs du (ou des) décideur(s). Cette approche s'appuie sur la déclinaison des objectifs reflétant le système de valeurs du décideur et parait donc utilisable dans notre contexte

Figure 1 :Le processus d'aide à l'évaluation [Rousval, 2005]

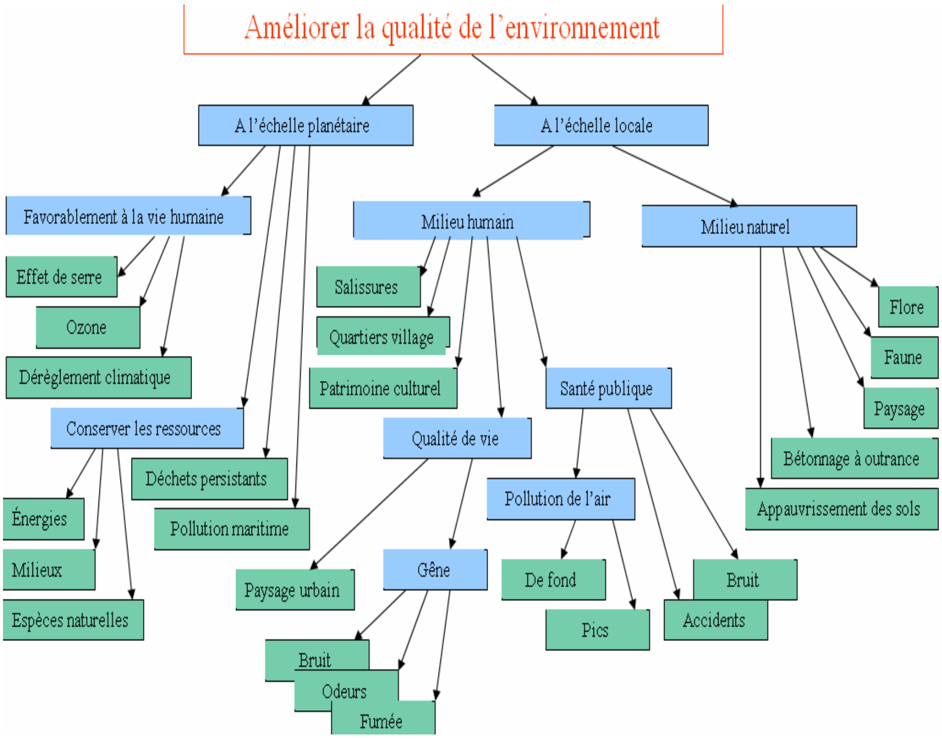
Figure 1:Evaluation aiding process [Rousval, 2005]



Pour diverses raisons, [Rousval, 2005] propose, dans un premier temps, de ne considérer uniquement que les objectifs de fin pour construire le référentiel de l'évaluation d'un système. Selon [Keeney, 1992], un objectif de fin est décomposable. En effet, « limiter l'encombrement du trafic routier » peut se décomposer par exemple en deux sous-objectifs : « limiter l'encombrement du trafic routier dans les villes » et « limiter l'encombrement du trafic routier en dehors des villes ». Ces deux objectifs sont deux facettes de l'objectif plus général « limiter l'encombrement du trafic routier ». Ils sont aussi objectifs de fin mais d'un niveau plus bas. Ces nouveaux objectifs de fin de plus bas niveau peuvent à leur tour être décomposés en objectifs de fin de plus bas niveau, jusqu'au moment où ils ne seront plus décomposables. A l'inverse, plusieurs objectifs de fin peuvent être regroupés au sein d'un même objectif s'ils sont diverses facettes d'un même objet plus général. Après un travail d'interview auprès de décideurs en charge au niveau urbain, [Rousval, 2005] obtient une arborescence synthèse représentée de manière simplifiée dans la Figure 2. Cette arborescence présente un ensemble d'objectifs qui se décomposent en sous-objectifs. Pour plus de détail, le lecteur peut se référer à [Rousval, 2005].

Figure 2 :Arborescence d'objectifs de fin : référentiel pour l'évaluation [Rousval, 2005]

Figure 2: Means objectives tree: evaluation referential



3. But de l'aide multicritère à l'évaluation

Ne perdons pas de vue qu'il s'agit ici de contribuer à la conception d'un outil permettant d'appréhender l'impact des transports sur l'environnement. La diversité des nuisances, l'hétérogénéité de la nature des données qui représentent ces nuisances, la diversité des échelles d'observation (spatiales et temporelles), le besoin d'une information synthétique, font que l'outil ne pourra pas se passer d'utiliser une méthode d'agrégation multicritère. Les principales méthodes multicritère permettant d'agréger l'information ont été développées pour faire de l'aide à la décision. Les préoccupations de ce cadre théorique sont de mettre en avant la ou les meilleures décisions à prendre parmi celles identifiées et cela par rapport aux préférences des décideurs. Mais, ici, la problématique est différente. On ne se préoccupe pas des décisions possibles. Il s'agit d'évaluer une ou des situations environnementales relativement à un ensemble d'objectifs concernant l'état environnemental. A partir d'un ensemble d'objectifs prédéfinis, d'un référentiel, les buts de l'aide multicritère à l'évaluation d'un système peuvent être finalement de :

- faire le constat, le diagnostic de l'état d'un système,

- communiquer avec transparence sur l'état d'un système,
- instaurer le dialogue et aider à la concertation,
- faciliter la négociation concernant les objectifs à retenir,
- sur une base commune d'objectifs, comparer un système avec un autre,
- réaliser le suivi de l'évolution d'un système au cours du temps,
- contrôler les dérives par rapport aux objectifs,
- identifier les faiblesses d'un système,
- alerter,
- révéler certains problèmes,
- -faire apparaître de nouveaux enjeux décisionnels,
- prévoir l'évolution de l'état du système,
- anticiper les conséquences de certaines prises de décisions,
- -comparer plusieurs scénarii.

Principe d'agrégation

Un enjeu théorique est d'utiliser certains concepts fondateurs des méthodes d'aide multicritère à la décision pour tenter de concevoir une « aide multicritère à l'évaluation » qui permette l'agrégation. L'agrégation apparaît donc comme centrale. Or, en aide multicritère à la décision, il existe deux grandes familles de méthodes d'agrégation [Roy & Bouyssou, 1993] :

- méthodes se ramenant à un critère unique de synthèse (monétarisation, analyse cout-bénéfices, utilité, utilité espérée, somme pondérée, …)
- méthodes ne se ramenant pas à un critère unique de synthèse (principalement les méthodes de type Electre [Roy & Bouyssou, 1993], mais aussi PROMETHE [Brans et coll., 1984]…).

Dans la pratique, les méthodes les plus répandues utilisent un critère unique de synthèse. Mais cela est très critiquable (totale compensation, pas de prise en compte de l'incertitude et l'imprécision des données…). Les résultats obtenus avec les méthodes de type Electre semblent plus pauvres. Cependant, ces dernières permettent de rendre compte de certaines situations apparemment ambiguës (incomparabilité, préférences et indifférences intransitives) mais qui sont la traduction de cas auxquels peuvent être confrontés les décideurs. Un autre avantage de la famille des méthodes de type Electre est le caractère non totalement compensatoire de l'agrégation multicritère. De plus, à aucun moment il n'y a de transformation d'échelle permettant de ramener à un critère unique de synthèse, une famille de critères prenant des valeurs de nature a priori hétérogènes. Les poids attribués à chaque critère sont complètement indépendants de l'échelle du critère en question. La présence de seuil de veto est un moyen supplémentaire de maîtriser le danger de la compensation en sanctionnant des écarts trop importants sur des critères dont les aspects peuvent être plus sensibles. Enfin, les méthodes de type Electre permettent (notamment grâce à l'introduction des seuils d'indifférence et de préférence) d'introduire la notion de flou afin de distinguer dans les différentes échelles du modèle, les écarts de préférence pertinents et significatifs de ceux qui ne le sont pas tout en prenant en compte le degré d'incertitude et d'imprécision des données. Or, les « indicateurs d'impacts » sont bien souvent entachés d'incertitude et/ou d'imprécision. Cela renforce donc l'intérêt d'utiliser une méthode de type Electre dans notre contexte de l'évaluation de l'impact des transports sur l'environnement.

Concernant la modélisation, les méthodes de type Electre nécessitent de fixer un nombre important de paramètres tandis que les méthodes d'agrégation en critère unique de synthèse nécessitent « simplement » la construction des fonctions de transformation servant à convertir toutes les données sur une échelle unique. Cependant, il est illusoire de croire en cette « simplicité » si tant est que l'on souhaite faire une modélisation rigoureuse car la tâche de construction de ces fonctions de transformations doit alors être extrêmement fine. Il faut en effet tenir compte à la fois de considérations liées à la nature de chaque critère mais aussi liées à la famille de critère. Séparer les étapes de modélisations en augmentant le nombre de paramètres (cas des méthodes de type Electre) peut simplifier la tâche de modélisation même s'il paraît alors difficile pour des personnes non averties de bien comprendre et interpréter le sens de ces paramètres. Notons que, dans un contexte d'aide à l'évaluation ou plusieurs acteurs sont amenés à participer à la modélisation avec différents rôles (expert ou évaluant), augmenter le nombre de paramètres peut alors permettre de bien séparer les rôles de chacun à condition de bien identifier « qui doit faire quoi ».

Pour toutes ces raisons, et sous certaines condition, nous préconisons d'agréger les critères en utilisant Electre (ici, dans sa version « Tri », qui affecte à une catégorie prédéfinie) pour ce contexte d'aide multicritère à l'évaluation de l'impact des transports sur l'environnement.

Mais, l'agrégation peut contribuer tant à introduire de la clarté dans la présentation des résultats, rendus de fait plus condensés et synthétiques, qu'à introduire de l'opacité concernant les règles, méthodes, paramètres, que l'agrégation utilise pour synthétiser les résultats. Agréger contribue donc à deux phénomènes antagonistes :

- simplifier la lisibilité des résultats,
- nuire à la transparence de ces mêmes résultats.

[Rousval, 2005] propose de réaliser autant d'agrégations que d'objectifs. Autrement dit, à chaque objectif de l'arborescence, il s'agit d'affecter le système évalué à une catégorie et cette affectation sera réalisée en prenant en compte tous les critères issus de la décomposition de cet objectif. Cette façon d'évaluer à tous les nœuds de l'arborescence offre la possibilité de parcourir la hiérarchie d'objectifs en consultant, à chaque niveau, le résultat agrégeant tous les critères directement ou indirectement rattachés à l'objectif en question. En partant par exemple de la racine de l'arborescence, l'évaluant pourra alors naviguer dans la hiérarchie pour descendre jusqu'aux objectifs non décomposables et remonter à sa guise tout en consultant à chaque niveau un résultat agrégé. Une telle fonction semble pallier l'antagonisme entre simplicité du résultat et transparence en rendant possible l'alternance de phases d'agrégation de l'information (remontée dans l'arborescence d'objectifs) et de désagrégation (descente dans l'arborescence) lors de la consultation des résultats.

Contexte de mise en œuvre d'un tel outil

[Rousval, 2005] propose d'organiser la navigation dans l'outil autour de sept processus :

- définition du système évalué,
- définition de la structure des objectifs,
- sélection et pondération des objectifs,
- définition des critères,
- définition des catégories,
- évaluation des performances,
- consultation des résultats.

Tous ces processus ne seront pas forcément disponibles selon le contexte. Trois modes d'utilisation sont donc proposés selon le contexte :

- mode figé, « indicateur environnemental global » : construire un indicateur global concernant les transports dont la vocation est de qualifier la situation environnementale d'une ville, d'une région ou d'un pays afin de comparer différentes villes, régions ou pays, afin de communiquer.
- mode semi-ouvert, « démocratie participative au niveau de l'agglomération » : disposer d'un outil permettant d'appréhender la situation environnementale en support à la mise en place et au suivi d'une politique de transports au niveau d'une agglomération dans un contexte de démocratie participative (élus, public et acteurs économiques),
- mode ouvert « suivi local et personnalisé » : disposer d'un outil de suivi afin que l'utilisateur soit à même de se forger une opinion personnelle concernant l'impact des transports sur l'environnement d'un sous-ensemble géographique.

Si l'on souhaite, par exemple, construire un indicateur environnemental global permettant la comparaison de villes (l'exemple de l'indice Atmo si on restreint l'environnement à la qualité sanitaire de l'air), il ne faut pas permettre aux responsables des différentes agglomérations de modifier les paramètres du modèle d'évaluation (critères, seuils, poids…). Si tel était le cas, une même valeur de l'indicateur n'aurait pas la même signification d'une ville à l'autre. Chaque mode d'utilisation implique alors différents niveaux de disponibilité des fonctions. Le tableau 1 propose des combinaisons et rappelle quels sont les acteurs concernés par le processus en question. Bien entendu, les personnes qui joueront le rôle de l'évaluant ne seront pas les mêmes si on souhaite construire un indicateur global ou si on souhaite disposer d'un outil pour un suivi local et personnalisé. Il en est de même pour le rôle des experts qui ne sera pas représenté par les mêmes personnes selon le contexte. Le tableau 2 illustre comment le mode d'utilisation permet d'adapter l'aide à l'évaluation aux trois contextes que nous avons proposés et donne un exemple de personnes pour chaque catégorie d’acteurs.

Tableau 1 : Trois contextes illustratifs [Rousval, 2005]

Table 1: three example contexts

Contexte	Evaluant	Experts	Mode d'utilisation
Indicateur environnemental global	Collège d'experts, vision globale	Collège international d'experts, par thématique	Figé
Démocratie participative au niveau de l’agglomération	Elus, public, acteurs économiques	Collège national d'experts, par thématique	Semi-ouvert
Suivi local et personnalisé	Un élu	Conseillers techniques de l'élu	Ouvert

Pour le contexte « indicateur environnemental global », en mode figé, l’évaluant pourra être un collège d’experts ayant une vision globale des problèmes environnementaux liés à l’activité des transports. En revanche, le rôle de l’expert pourra être endossé par un ensemble de personnes chacune spécialisée dans une thématique environnementale. Dans ce contexte, les experts et l’évaluant contribueront chacun au paramétrage de l’outil en définissant le référentiel servant à l’évaluation (objectifs, poids, critères, seuils, ...). Le mode d’utilisation figé semble alors convenir le mieux pour ce genre d’outil, afin de ne pas permettre aux utilisateurs de modifier tout ou partie de ce référentiel. Avec un référentiel (objectifs, poids, critères, catégories, seuils…) invariant (ou variant peu et peu souvent), les comparaisons de situations auront davantage de signification (on peut par exemple imaginer pour ces faibles variations, de mettre en ligne sur Internet, des mises à jour à télécharger). Ainsi, mis en œuvre dans une ville, cet outil permettra la définition du système évalué, la saisie des performances et la consultation des résultats. Les utilisateurs pourront finalement être très peu experts en environnement, l’expertise étant déjà incluse dans l’outil par le biais de divers paramètres. De plus, pour ces mêmes raisons, ce genre d’outil pourrait être mis à la disposition de personnes non familiarisées avec la modélisation multicritère. Ainsi, l’accompagnateur, l’expert en modélisation que nous nommerons l’ « homme d’étude » en référence à [Roy & Bouyssou, 1993], interviendra principalement dans la phase de paramétrage avec l’expert et l’évaluant.

Tableau 2 : Disponibilité des processus, modes d'utilisation, acteurs [Rousval, 2005]

Table 2: Process availability, use cases, actors

Processus	Modes d'utilisation			Acteurs
	Ouvert	Semi-ouvert	Figé	
Définition du système évalué	Disponible	Disponible	Disponible	Evaluant
Définition de la structure d'objectifs	Disponible	Non disponible	Non disponible	Evaluant et experts
Sélection et pondération des objectifs	Disponible	Disponible	Non disponible	Evaluant
Définition des critères	Disponible	Non disponible	Non disponible	Experts
Définition des catégories	Disponible	Non disponible	Non disponible	Experts
Evaluation des performances	Disponible	Disponible	Disponible	Automatisé/Experts
Consultation des résultats	Disponible	Disponible	Disponible	Evaluant

Au contraire, dans le cadre d'une « démocratie participative au niveau de l'agglomération », en mode semi-ouvert, il est souhaitable de laisser plus de liberté aux l'utilisateurs. Seuls les processus de « définition de la structure d'objectifs », de « définition des critères » et de « définition des catégories » ne seront pas disponibles : ils feront l'objet d'un paramétrage préalable. Ces paramètres seront fixés par l'expert qui pourra, dans ce cas, être un collège d'experts spécialisés dans les différentes thématiques environnementales. En mode semi-ouvert, en plus des processus de « définition du système évalué » de « saisie des performances » et de « consultation des résultats », les utilisateurs pourront donc « sélectionner et pondérer les objectifs » à leur disposition dans la hiérarchie. Ainsi, la concertation et la communication autour des objectifs retenus et de leurs poids pourront avoir lieu en mettant en scène les différents protagonistes jouant le rôle de l'évaluant qui pourront en l'occurrence être les élus, le public et les acteurs économiques (ceux de la démocratie participative). La concertation et la communication pourront s'appuyer sur les résultats de l'évaluation de la situation en question. Les utilisateurs auront donc une certaine liberté pour faire varier le référentiel de l'évaluation (objectifs et poids), sans pour autant avoir de connaissances d'experts ni être trop familiarisés avec la modélisation multicritère (pas besoin de savoir construire un critère, définir une catégorie, déterminer les valeurs des seuils). Ainsi, l'homme d'étude aura un rôle non négligeable lors de l'utilisation de l'outil dans ce contexte.

Enfin, pour un suivi local et personnalisé, le dernier mode proposé (mode ouvert), laisse libre les utilisateurs de définir le référentiel de l'évaluation (objectifs, critères, seuils, poids, catégories…) à leur guise. Les résultats produits par un tel outil d'aide à l'évaluation n'engagent donc qu'eux. Une bonne connaissance des thématiques est alors requise pour les personnes jouant le rôle de l'expert, comme par exemple les conseillers techniques d'un élu. L'homme d'étude doit alors superviser toutes les phases de la modélisation multicritère qui sont des fonctions disponibles. L'évaluant, l'élu est de son coté parfaitement libre de créer ses propres objectifs. En définitive, un tel mode d'utilisation d'outil n'est pas destiné à la communication mais est plutôt le support au suivi personnalisé d'une situation par un élu.

Conclusion

Le projet « Prospectives et Indicateurs Environnementaux » mené par l'Inrets est ambitieux. Nous proposons certaines orientations méthodologiques et techniques pour la mise en œuvre d'un outil permettant l'agrégation multicritère de l'impact des transports sur l'environnement. Les perspectives d'utilisation de l'outil dont il est question dans ce texte semblent assez ouvertes et nous avons envisagé trois modes d'utilisation. Notons finalement, que l'aide multicritère à la décision proposée ici, peut tout à fait être envisagée dans des domaines complètement différents comme la gestion, le suivi de politiques, le contrôle.

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Dépréciation immobilière, polarisation sociale et inégalités environnementales liées de bruit des avions. Application de la Méthode des Prix Hédoniquesà proximité de l'aéroport d'Orly

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Résumé

La conciliation entre les grands équipements de transport et leurs territoires d'accueil passe par l'analyse de leurs effets environnementaux et territoriaux. Nous évaluons ici les effets du bruit des avions sur les valeurs immobilières et la mobilité résidentielle des ménages, en appliquant la méthode des prix hédoniques (MPH) aux transactions observées entre 1995 et 2003 dans huit communes proches de l'aéroport d'Orly. Ce bruit déprécie la valeur des logements, et surtout le taux de décote croît depuis 1995 alors que les niveaux de bruit sont restés stables, révélant la sensibilité croissante des ménages à leur environnement dans les choix résidentiels. D'autre part, le renouvellement des populations ne s'opère pas à l'identique : les arrivants sont plus jeunes et plus modestes que les partants. Des inégalités environnementales émergent du croisement de ces résultats, puisque des ménages plus modestes supporteront des décotes plus importantes. Toutefois, pour être réellement comprises, ces inégalités devront donner lieu à l'observation de la gêne sonore ressentie par les ménages résidents ou anticipée par des ménages en quête de logements.

***Mots clés** : bruit des avions, dépréciations immobilières, Méthode des Prix Hédoniques, gêne, ségrégation sociale, inégalités environnementales.*

Abstract

Trying to find conciliations between large infrastructures of transport and territories nearby emerges as an important goal of sustainable development strategies. And, this goal passes through a better knowledge of environmental and territorial effects of such infrastructures. We report here an Hedonic Price Method (HPM) application, so a measurement process based on revealed preferences, to the property values concerning eight communities situated near Orly Airport and exposed to aircraft noise. This work follows several works already done for Orly Airport case by University Paris XII: evaluation of the social cost of aircraft noise by using the contingent evaluation method, annoyance survey and meetings with groups of neighbouring residents etc.),

First of all, this work shows that aircraft noise causes property values depreciation, all else equal. This is not a real surprise regarding the 35 applications of the HPM performed abroad on this noise source. Above all, even if the level of noise remained stable (because of a slots cap at Orly airport since 1995), the Noise Depreciation Index – NDI increases since this date. This validates the hypothesis according which the choice of buyers is affected more by annoyance and its anticipation than acoustical levels alone. It allows to understand the role of the households environmental sensitivity in their residential choices. Lastly, thanks to socio-economic information concerning buyers and sellers, it appears that the moving of populations is not identical: the newcomers in communities for which an NDI had been established are younger and of lower social level than movers out. This indicates a process of polarisation and even spatialized social segregation.

Hence, the crossing of these results allows to have a glimpse of increasing environmental inequities, because it will be more low income households which will have in the future to endure more important depreciations, based on social sensitivity about noise.. And, pursuing sustainable development goals, the political treatment of those specific inequities should be more based on annoyance (actual and anticipated), in addition to actions exclusively focused on noise levels. It seems to be time to take better account of the direct and indirect effects of aircraft noise.

Key-words : *Aircraft noise, property values depreciation, Hedonic Price Method, annoyance, social segregation, environmental inequities.*

Introduction

Le maintien ou la restauration du bien-être environnemental s'affirme comme des enjeux inhérents au développement urbain durable. Parmi les facteurs d'inconfort environnemental des populations, le bruit des transports figure en bonne place[1], et la sensibilité individuelle au bruit des avions est indéniablement la plus forte, en France comme à l'étranger. Comme nous avons pu le montré par l'analyse comparée de plusieurs conflits aéroportuaires (Faburel, 2003a), la problématique des effets du bruit des avions sur les populations (affections sanitaires, troubles du sommeil, gêne…) et sur les territoires (contraintes urbanistiques, décotes immobilières, modification des dynamiques locales…) s'affirme de nos jours comme un des sujets forts des revendications locales et des attentes d'éclairages dans les

[1] Cf. sondages d'opinions et quelques enquêtes de gênes depuis CREDOC (1989) jusqu'à INSEE (2003).

aires aéroportuaires. Les débats, parfois tendus, entre les plates-formes et les localités qui les accueillent s'ouvrent donc sur des considérations plus ancrées dans les territoires.

Dans ce registre, les dépréciations des biens immobiliers susceptibles d'être causées par le bruit des avions tiennent partout une place prépondérante. Toutefois, très peu de travaux empiriques ont, en France, abordé de tels impacts. Jusqu'en 2004, nous ne trouvions, pour les aéroports français, et plus particulièrement ceux de la région parisienne, qu'une évaluation économétrique remontant à la fin des années 1980 à Orly (SEDES, 1978), et surtout des évaluations descriptives tirées de dires d'agents immobiliers (Lambert et *al.*, 1997 pour Orly ; Martinez, 2001 pour Roissy CDG). A l'inverse, plus de 35 travaux de recherche étrangers renseignent les dépréciations immobilières pour cause de bruit des avions (Navrud, 2002 ; Button, 2003 ; Nelson, 2004). Or, de récents travaux empiriques menés autour d'Orly, concernant les ambitions de déménagement des ménages exposés au bruit des avions (Faburel et Maleyre, 2002), ou autour de Roissy, concernant les dynamiques spatiales des communes proches de l'aéroport (Martinez, *op. cit.* ; Faburel et Barraqué, 2002), tendent à montrer que les nuisances provoquées par les trafics aériens auraient des effets tangibles sur la mobilité résidentielle des ménages.

Qu'en est-il concrètement des dépréciations immobilières imputables au bruit des avions, de la mobilité résidentielle des ménages dont elles peuvent découler et plus largement de l'évolution des tissus urbains des espaces concernés ? Prolongeant l'effort engagé depuis plusieurs années sur Orly et Roissy CDG sur le coût social de la gêne sonore (Faburel, 2001), et les conflits aéroportuaires en France (Faburel et Mikiki, 2004) comme à l'étranger (Faburel, 2003b), ces décotes ont fait l'objet de l'évaluation livrée ici, aux alentours de l'aéroport d'Orly (2ème aéroport français et 18ème européen en mppa). Après un bref rappel des principes de la méthode des prix hédoniques (MPH) et de l'échantillon de travail, nous présentons et discutons les résultats obtenus, puis les resituons dans leur contexte territorial (section 2), pour en conclusion, aborder la question des inégalités environnementales dont les aires aéroportuaires seraient progressivement le siège, et alors nourrir le débat sur le développement durable de ces lieux (section 3). L'ensemble renvoie à Faburel, Maleyre et Peixoto (2004).

Présentation de la Méthode des Prix Hédoniques (MPH) et de sa mise en œuvre à proximité d'Orly

La MPH est le procédé d'évaluation dit de « préférences révélées » de référence pour mesurer les décotes immobilières imputables à des charges environnementales liées aux transports. Destinée à l'analyse des biens, tels les logements, différenciés par leur qualité, et décrits par une suite de caractéristiques, la MPH se donne deux objectifs, auxquels sont associées les deux étapes de sa mise en œuvre (Rosen, 1974 ; Maleyre, 1997).

La première étape a pour but d'analyser statistiquement les déterminants du prix du bien étudié, en l'occurrence ici le logement. La fonction des prix hédoniques est obtenue en régressant le prix du bien sur l'ensemble des caractéristiques internes (nombre de pièces, étage, garage, jardin…) et externes (proximité de moyens de transport, sécurité du quartier, offre scolaire, qualité de l'environnement…) qui le décrivent. Or, par construction, cette fonction de prix résulte de la rencontre, sur le marché, de l'ensemble des offres et demandes individuelles qui s'y expriment. Elle représente donc un équilibre (Rosen, *op. cit.*). La seconde étape vise alors à rapprocher ces prix marginaux, des caractéristiques des agents, offreurs ou demandeurs. Les fonctions dès lors obtenues, généralement présentées sous la forme inverse, sont les fonctions d'enchère ou de consentement à payer (côté demande) et les fonctions d'acceptation (côté offre). Elle permet notamment de calculer les variations de bien-être individuel associées à une modification marginale d'une des caractéristiques du logement (niveau d'exposition sonore, par exemple). L'évaluation du « coût » ou du « bénéfice social » est alors obtenue par agrégation de ces variations, pour les différentes caractéristiques du logements sur l'échantillon observé.

Toutefois, précisons qu'en raison de l'imperfection de l'information sur les caractéristiques du bien échangé, de la viscosité des ajustements… le marché du logement ne reflète qu'incomplètement l'effet des aménités environnementales sur le prix : il ne « capitalise » pas complètement la valeur de ces aménités (Van Praag et Baarsma, 2000). Aussi considère-t-on en général que l'évaluation tirée de la seconde étape constitue une borne inférieure du « coût social vrai » (Bartik et Smith, 1987), des nuisances sonores par exemple.

Le travail présenté se limite à la première étape. Nous verrons que l'analyse des déterminants des prix immobiliers suffit à fournir des indications intéressantes sur le fonctionnement du marché, sur les stratégies de localisation des ménages et alors des informations fiables sur les effets du bruit sur les valeurs immobilières, dans un contexte de différenciation forte de la qualité de l'environnement des logements.

La disponibilité depuis peu en France d'informations plus exhaustives et fiables sur les valeurs de transaction immobilière (Base de données proposée par la Chambre des Notaires de Paris) a permis de renseigner les prix des logements et certains des facteurs participant de leur explication (localisation précise du bien immobilier et certains éléments concernant le type de logement ainsi que le profil social-démographique des acquéreurs/vendeurs). Par ailleurs, les débats relatifs aux nuisances sonores ont conduit les autorités aéroportuaires à étoffer leurs systèmes de mesures sonométriques, autant qu'ils ont incité de plus en plus de collectivités locales a entreprendre la réalisation de diagnostics environnementaux, et singulièrement acoustiques. Nous avons opté pour celles exprimées en Lmax, ou niveau de pression acoustique maximal, pondérée par le nombre de survols (Béture, 1996). Ces données ont été jugées plus pertinentes que des expressions moyennées (en Leq par exemple), en regard de la particularité des phénomènes sonores considérés (émergences au passage des avions), et ce dans des espaces à dominante résidentielle, c'est-à-dire dans des lieux où de telles émergences sont distinctement perceptibles, et alors observables.

Toutefois, pour garantir l'imputabilité de nos résultats au seul bruit des avions, les sites de multi-exposition ont été retirés de la base (logements situés le long de routes nationales et départementales présentées comme bruyantes par les services de l'Etat). De même, pour disposer de situations sonores suffisamment contrastées, seuls les logements situés au centre de chacune des 3 zones définies par le travail du Béture ont été pris en compte. Enfin, pour ne pas attribuer cette fois-ci au seul bruit la responsabilité d'éventuelles décotes liées à des dégradations du cadre de vie général, les autres facteurs d'environnement ont été neutralisés ; il est maintenant courant d'en résumer les effets par une variable communale, représentative de l'ensemble des facteurs d'environnement (Kaufman et Espey, 1997).

Au final, partant d'un échantillon de plus de 10 000 valeurs, l'analyse économétrique a porté sur 688 observations ainsi triées : distribuées de manière homogène du 1er janvier 1995 (date d'entrée en application du plafonnement à 250 000 créneaux horaires annuels à Orly) au 30 septembre 2003 (date d'acquisition des valeurs immobilières) ; et réparties dans les huit communes du Val-de-Marne les plus exposées au bruit des avions : Ablon-sur-Seine, Boissy-Saint-Léger, Limeil-Brévannes, Marolles, Sucy-en-Brie, Valenton, Villeneuve-le-roi et Villeneuve-Saint-Georges.

Dépréciations immobilières liées au bruit des avions et polarisation sociale progressive des territoires dépréciés

Comme dans un grand nombre de travaux portant sur la première étape de la méthode des prix hédoniques (Soguel, 1994), la forme semi-logarithmique a donné les meilleurs résultats. La variable endogène est donc le logarithme (base 10 ici) du prix des logements, les variables exogènes étant introduites sous leur forme naturelle. Seules les variables statistiquement significatives figurent dans le tableau de résultats (cf. tableau 1).

Tableau 1 : Modèle explicatif du prix des logements de 8 communes du Val-de-Marne exposées au bruit des avions (n=688)

Table 1: Econometric model of dwellings price function in eight cities exposed to aircraft noise from Orly Airport

Variable	Estimations	T-Student
Constante	4,8432	38,076***
SUPERF	0,0025	8,546***
NPIECE	0,0381	5,53***
ETAGE	0,0108	2,077**
MAISON	0,1347	7,907***
BOISSY	0,0591	3,544***
LIMEIL	0,0483	2,84***
VALENTON	- 0,0942	- 3,841***
VILLEROI (Villeneuve-le-Roi)	- 0,0558	- 2,637***
VILSAINT (Villeneuve-Saint-Georges)	- 0,071	- 3,497***
MUT97	0,0454	2,739***
MUT02	0,0436	2,722***

MUT03	0,1041	5,037***
BRUIT	- 0,0042	- 2,416**
R^2	0,726289	
Test de FISCHER	137,51***	

Les traitements et tests statistiques réalisés indiquent tout d'abord que le modèle explicatif est robuste économétriquement. L'ensemble des variables retenues explique 72 % de la variance du prix du logement ce qui est tout à fait satisfaisant, au regard des résultats habituellement obtenus pour ce type d'analyse. Le test de Fisher montre d'autre part que le modèle est globalement significatif. Surtout, les résultats avancés attestent d'une cohérence d'un point de vue économique. La surface et le nombre de pièces influent positivement, et conjointement, sur le prix. Comme attendu, l'étage exerce une influence positive : toutes choses égales par ailleurs, il augmente de 2,5 % la valeur d'un logement, sur la base d'un échantillon constitué à près de 50 % d'appartements. Dans ce registre, les maisons sont valorisées de 36 %, par rapport à un appartement de caractéristiques équivalentes. Les variables désignant la date de mutation (MUT…) permettent de tenir compte de l'incidence de la conjoncture immobilière sur le prix des logements, la reprise franche n'étant intervenue qu'à la fin des années 90 dans ces lieux. Les coefficients associés aux variables communales doivent être interprétés de la manière suivante : toutes choses égales par ailleurs (surface, nombre de pièces, étage, année de mutation… niveau d'exposition sonore), les prix des logements sont, du fait des caractéristiques communales (desserte, services…), plus élevés à Boissy-St-Léger et Limeil-Brévannes, qu'à Valenton, Villeneuve-le-Roi et Villeneuve St George.

Enfin et surtout, toutes choses égales, plus le niveau d'exposition est élevé, plus la valeur du logement est faible. Partant de ce dernier résultat, le taux de dépréciation ou NDI (*Noise Depreciation Index*, Walter, 1975) mesure la variation en % du prix du logement pour une variation d'1 dB(A) de l'exposition sonore. Outre sa commodité d'interprétation, cet indicateur présente l'avantage de permettre le rapprochement avec les résultats obtenus par les études précédentes.

Pour une fonction de prix semi-logarithmique, le NDI s'obtient en appliquant la formule :

$$NDI=\frac{P^B}{P^A}-1=\frac{P^B-P^A}{P^A}=\frac{\Delta P}{P}=10^{(a_1)}-1$$

où P^A et P^B représentent les prix des logements A et B, identiques pour toutes leurs caractéristiques sauf pour la caractéristique 1 : $x_1^B = x_1^A + 1$ (détail des calculs en annexe A.2)

Nous obtenons ici : $\boxed{\textbf{NDI = - 0,0096 = - 0,96 \%}}$

Ce résultat s'interprète ainsi : pour les huit communes de la zone d'étude, chaque unité de Lmax dB(A) supplémentaire dévalorise le logement de 0,96 %. Cette estimation correspond à une hypothèse basse car Boissy-Saint-Léger, commune la moins exposée au vue des données acoustiques utilisées, n'est pas totalement évitée par les survols.

Ce NDI est conforme à ce qu'indique l'abondante littérature sur le sujet essentiellement d'origine anglo-saxonne. Certes, les taux de dépréciation relayés par cette littérature varient parfois grandement d'un environnement à l'autre.

Différentes méta-analyses (ex : Schipper et *al.*, 1998 ; et Button, 2003), ou simplement des recensions (Bateman et al., 2000 ; Navrud, 2002 ; MacMillen, 2004) indiquent que les contextes spatiaux et temporels, les types de marchés immobiliers observés, leur degré de segmentation, l'indice acoustique retenu, la spécification des fonctions explicatives… n'y sont pas étrangers. Mais, la très grande majorité fait apparaître une décote moyenne située entre 0,6 % à 0,9 % (cf. moyenne de 0,83 % proposée par Schipper et *al.*, *op. cit.*), cette décote tendant, d'après les résultats produits sur ce sujet, à augmenter sur les 20 dernières années.

Le NDI établi ci-dessus nous permet d'évaluer les pertes en capital immobilier supportées par les communes les plus exposées au bruit des avions, par rapport à celles qui en sont – relativement – préservées (Boissy-Saint-Léger et Limeil-Brévannes). Le calcul de la perte en capital immobilier pour les trois communes identifiées par le modèle économétrique comme impactées (Valenton, Villeneuve-le-Roi et Villeneuve-Saint-Georges) s'effectue comme suit:

Dépréciation du capital immobilier	=	Stock de capital	x	NDI	x	Différence des Lmax entre Boissy et les 3 communes pour lesquelles le modèle économétrique permet d'appliquer le NDI

La décote ainsi calculée est estimée à 4,4 % du prix (soit 4 535 euros par logement) à Valenton, 5,5 % (5 229 euros par logement) à Villeneuve-le-Roi et 6,5 % (5 132 euros par logement) à Villeneuve-Saint-Georges (cf. tableau 2). L'hypothèse haute (commune témoin totalement exempte de bruit aéronautique) correspondrait à une décote de l'ordre de 10 % de la valeur des logements de ces communes (soit plus de 10 000 euros sur la base de la valeur moyenne des logements de notre échantillon), elle-même conforme à la littérature sur le sujet.

De plus, alors que le bruit est demeuré stable, voire à légèrement décliné dans notre espace d'investigation (source : Aéroports de Paris), en grande partie du fait du plafonnement des créneaux horaires annuels intervenu au 1er janvier 1995 à Orly, le NDI calculé en moyenne traduit en fait une augmentation sur l'ensemble de la période considérée (1995 à 2003), passant de 0,86 % à 1,48 % du prix du logement par décibel. Ce qui, rapporté aux montants de capital immobilier, correspond à une perte moyenne de plus de 3 207 euros par ménage ayant acheté son logement entre 1995 et 2000 à Villeneuve-le-Roi, Villeneuve-Saint-Georges ou Valenton, et revendu depuis. Ces ménages, s'ils devaient revendre leur bien aujourd'hui, perdraient cette somme pour une valeur moyenne d'acquisition de 90 000 € (base comprenant appartements et maisons individuelles), perte non compensée par la décote d'origine.

Tableau 2 : Perte moyenne de capital immobilier par logement des communes subissant une décote pour cause de bruit de savions (en % de la valeur des logements et en €)

Table 2: Mean depreciation amount for a dwelling located in one of the cities concerned by NDI (in % of total value and in Euros)

Commune	Capital immobilier moyen tiré de l'échantillon	Différence de niveaux de bruit / Boissy (Lmax)	Dépréciation en % de la valeur des logements	Dépréciation du stock de capital immobilier	Dépréciation moyenne par logement
Valenton	4 464 131€	+ 4,55	4,4 %	194 993 €	4 535 €
Villeneuve-le-Roi	8 555 940€	+ 5,73	5,5 %	470 645 €	5 229 €
Villeneuve-Saint-Georges	10 169 216€	+ 6,73	6,5 %	657 012 €	5 132 €

Or, parmi les communes étudiées, celles affectées par la dévalorisation immobilière sont aussi le siège d'une polarisation sociale. En fait, l'analyse des profils socioprofessionnels des acquéreurs de logement (Base de données proposée par la Chambre des Notaires de Paris) indique, sur la période étudiée :

- que la proportion des acquéreurs appartenant aux catégories sociales supérieures est en baisse sensible dans les communes pour lesquelles la décote a été identifiée, alors que cette proportion demeurait stable ou diminue très légèrement dans les autres communes de l'échantillon,
- et que la proportion cumulée des ménages acquéreurs de condition plus modeste y est en hausse bien plus sensible que dans celles « épargnées » par la décote.

En outre, concernant cette fois-ci la vente, la proportion des vendeurs de catégorie plus aisée augmente substantiellement depuis 1995 dans les trois communes dépréciées, alors qu'elle diminue dans les autres communes. Plus largement, pendant que, sur l'ensemble des populations communales (et non plus seulement les vendeurs ou acquéreurs de notre échantillon d'analyse), ce rang social voyait sa proportion augmenter de plus de 42 % depuis le début des années 1980 à l'échelle du Val-de-Marne, de près de 25 % à Boissy-Saint-Léger, et de plus de 20 % à Limeil-Brévannes (communes témoins de notre échantillon), sa proportion n'augmentait que de 9 % à Villeneuve-le-Roi, et de 12,5 % à Villeneuve St Georges (INSEE, RGP 1982, 1990, 1999). Et, toujours à partir de ces données de recensement, le cumul des catégories modestes indique une baisse en proportion de près de 23 % à Limeil-Brévannes, de près de 13 % à Boissy, avec une diminution de plus de 20 % à l'échelle du Val-de-Marne, alors qu'il stipule par exemple une baisse de seulement un peu plus de 7 % à Villeneuve St Georges. Enfin, l'âge moyen des acquéreurs (tirés de notre échantillon de mutations) a diminué depuis 1995 de plus de deux ans à Villeneuve St Georges, de un an et demi à Villeneuve-le-Roi et à Valenton, tandis qu'il baissait de moins d'un an à Boissy-Saint-Léger et augmentait de trois ans à Limeil-Brévannes.

Ces différentes informations convergent pour annoncer un phénomène lui-même seulement pointé dans certaines communes fortement exposées au bruit des avions de Roissy CDG (Martinez, 2001). Il s'agit d'une polarisation sociale progressive de l'espace par l'arrivée de jeunes couples et disposant de peu de moyens, attirés par la possibilité d'une accession plus rapide à la propriété du fait des décotes identifiées. Et, ce phénomène de polarisation ne ferrait, là-aussi, qu'attester de mécanismes observés de longue date dans le domaine des industries ou d'autres équipements dits externalisant : « *Toute décision en faveur d'un équipement nuisible à l'environnement entraîne une baisse des valeurs foncières et immobilières, ce qui favorise l'attraction de populations pauvres.* » (Been Vicki, 1994, cite in Ghorra-Gobin, 2000, p. 156).

Discussion : le vécu du bruit comme objet de compréhension des dynamiques sociales et spatiales dans les pourtours aéroportuaires

A l'issue de ce travail, quatre résultats principaux méritent d'être soulignés :

- sans grande surprise en regard des 35 applications de la MPH réalisées à l'étranger sur ce seul bruit, celui des avions déprécie la valeur des logements dans les communes les plus proches de la plate-forme aéroportuaire d'Orly ;
- le taux de dépréciation augmente depuis 1995, alors même le bruit est demeuré stable du fait notamment du plafonnement du nombre de créneaux dans cet aéroport ;
- les ménages acquéreurs dans les communes exposées au bruit sont plus jeunes et de rang social plus modeste, laissant entrevoir un processus de polarisation sociale de l'espace.

Et, dans le prolongement, un quatrième résultat a aussi été plus indirectement livré. Puisque c'est davantage la gêne sonore vécue (pour les vendeurs) ou sa crainte (pour les acquéreurs) qui fonde à ce jour la décote immobilière, la sensibilité croissante des ménages au bruit des transports pourrait amplifier ce phénomène, et les efforts fournis pour diminuer les seules émissions sonores pourraient ne pas restaurer la valeur vénale des logements. En outre, puisque le renouvellement des populations dans les communes affectées par une telle dévalorisation immobilière ne se fait pas socialement à l'identique. Dès lors, des ménages modestes achetant, en nombre, à moindre coût pourraient perdre encore plus d'argent au moment de la revente du logement.

Aussi, se trouve posée ici la question de la justice environnementale et de son traitement politique au nom des actions promues par le développement durable, notamment dans le domaine des transports (WHO, 2004). Toutefois, l'approche conventionnelle des inégalités environnementales ne saurait suffire. Cherchant à caractériser les différentes catégories sociales selon leur seul niveau moyen d'exposition sonore, elle pourrait obérer la compréhension de mécanismes ségrégatifs qui, bien que complexes, semblent alimentés par des inégalités reposant sur le ressenti de gêne ou son anticipation. Il y a donc lieu de mieux comprendre le rôle joué par les nuisances (qui correspond à la définition même du bruit : « Ensemble de sons, ressentis comme désagréables » - Robert, « Phénomène acoustique produisant une sensation auditive considérée comme désagréable ou gênante » - Afnor - NFS 30105) et vécus du bruit dans la composante environnementale de telles dynamiques spatiales inégalitaires, en complément des lectures strictement acoustiques (i.e. caractérisation par les doses sonores).

Et, dans la logique de l'analyse hédonique, le rôle de cette sensibilité sonore accrue est décelable dans l'accroissement du coût implicite du bruit sur le marché immobilier. A l'identique de ce qui est admis concernant la gêne déclarée (elle n'est que très partiellement expliquée par les caractéristiques physiques des sons appréhendées par l'acoustique) ; et dans le prolongement de ce que nous avions pu montré dans un passé récent concernant les motivations résidentielles des ménages (l'ambition de déménager est, à proximité d'Orly, bien plus reliée à la gêne sonore qu'aux intensités de décibels, in Faburel et Maleyre, 2002) ; un autre effet du bruit n'est donc pas non plus uniquement lié aux caractéristiques physiques du bruit.

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New Capabilities for On-road Emissions Remote Sensing

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Abstract

On-road remote sensors measure CO, HC, NO and smoke opacity as a ratio to CO_2 emissions. From these ratios we calculate mass emissions per unit of fuel burned. At a typical location 5000 to 10,000 vehicles can be measured per day. Recently we have added the capability to monitor sulfur dioxide (SO_2) and ammonia (NH_3) emissions by extending our capability in the ultraviolet from 227nm (NO) down to 220nm for SO_2 and 213nm for NH_3. Ammonia emissions are the first on-road pollutant which we have measured in which the oldest vehicles are the lowest emitters. Ammonia emissions are very transient and are associated with fully functioning catalysts. By contrast, SO_2 emissions are a simpler function of fuel chemistry and can be used as a tool to detect fuel tax avoidance. A separate spectrometer monitors nitrogen dioxide (NO_2) emissions. Results from automobiles and light-duty trucks are presented.

Key-words: *Motor vehicle emissions, CO, HC, SO_2, NH_3, NO_2. Emissions des autos, émissions des camions, CO, HC, SO_2, NH_3, NO_2.*

Résumé

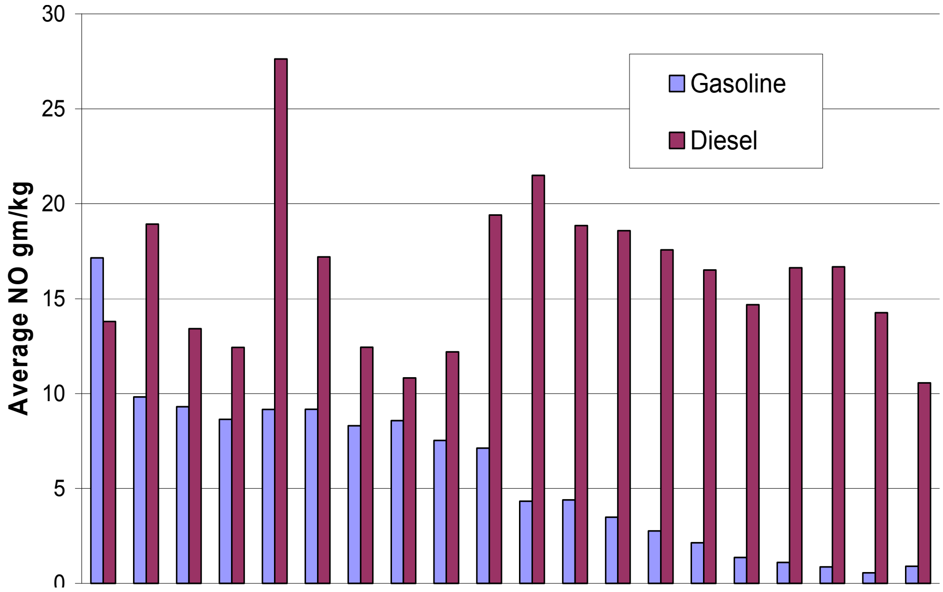
Des détecteurs éloigneés sur la route mesurent CO, HC, NO et l'opacité de la fumée en proportion avec les émissions de CO_2. Avec ces proportions on peut calculer les émissions masses par unité d'essence consommée. Aux environs typiques on peut receuiller entre 5000 et 10,000 résultats par jour. Récemment nous avons incorporé la capabilité de surveiller les émissions de SO_2 et NH_3 en ameliorant notre capabilité dans la domaine ultraviolet de 227nm (NO) jusqu'à 220nm pour SO_2 et à 213 pour NH_3. L'émission d'ammoniaque est le premier pollutant que nous avons mesuré où les véhicules les plus âgés sont ceux qui ont émis le moins. Les émissions d'ammoniaque sont très transitoires et sont associées avec des catalyses en plein function. Par contre, les émissions de SO_2 sont une fonction plus simple de la chimie des carburants et peuvent être utilisées pour détecter ceux qui cherchent a éviter les impôts. Un autre spectroscope mesure les émissions de NO_2. Les résultats des voitures et des camionettes sont ci-inclus.

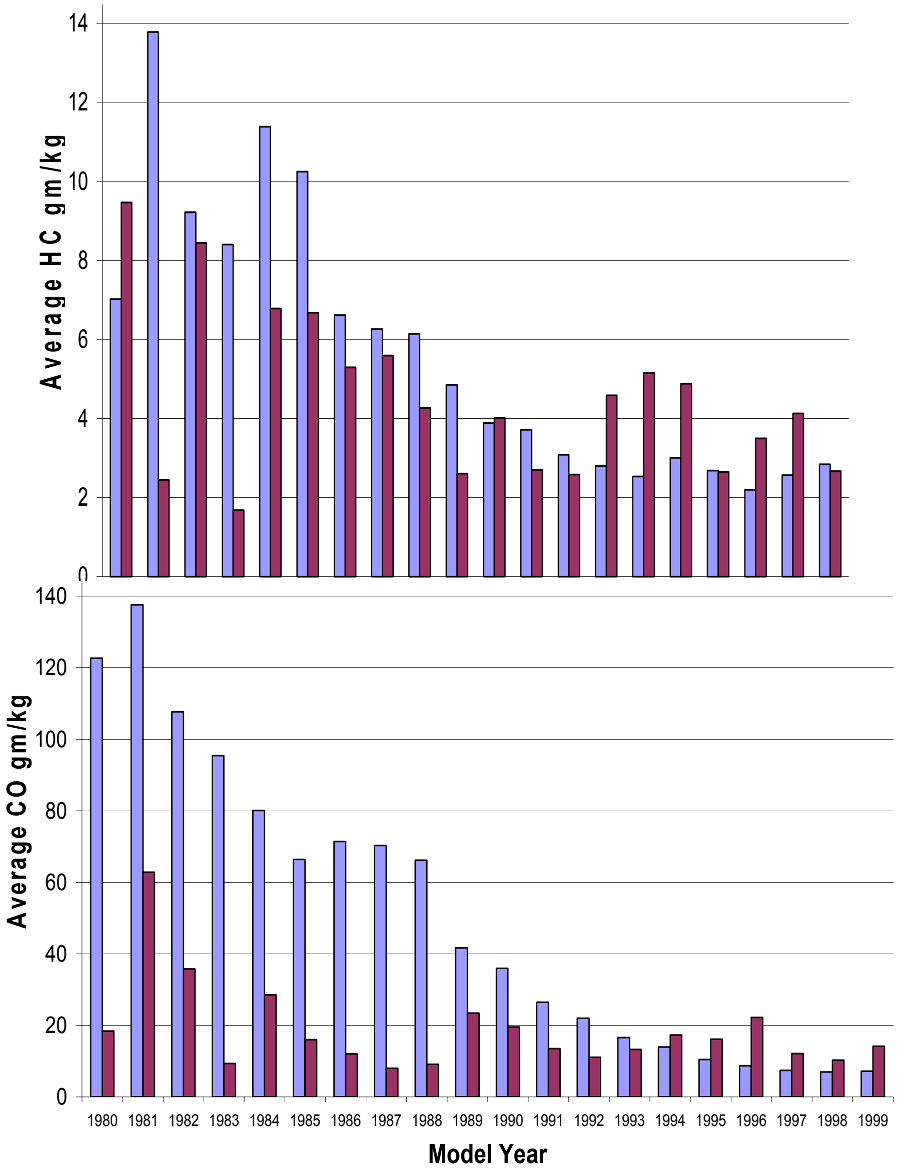
Introduction

On-road remote sensing has been under development since the late 1980s. Many publications and reports can be found at our web site www.feat.biochem.du.edu. The basic instrument uses selectively filtered infra-red (IR) radiation to monitor CO, HC CO_2 and an IR reference channel. An ultraviolet (UV) spectrometer measuring NO at its absorption maximum at 227nm and smoke at 230 nm was added in the mid 1990s. The system reports in about 0.8 seconds the pollutant load in the emissions of a passing vehicle as the ratio of the pollutant to CO_2. From these reported ratios one can determine the fuel-based emissions gm/kg of fuel or one can determine the readings which would be monitored by a tailpipe probe (after the probe readings are corrected for excess water and any excess oxygen not participating in combustion).

Figure 1: Light duty diesel and gasoline emissions versus model year, Phoenix AZ 2000.

Figure 1 : Emissions avec essence et diesel par rapport á l'année de véhicule.



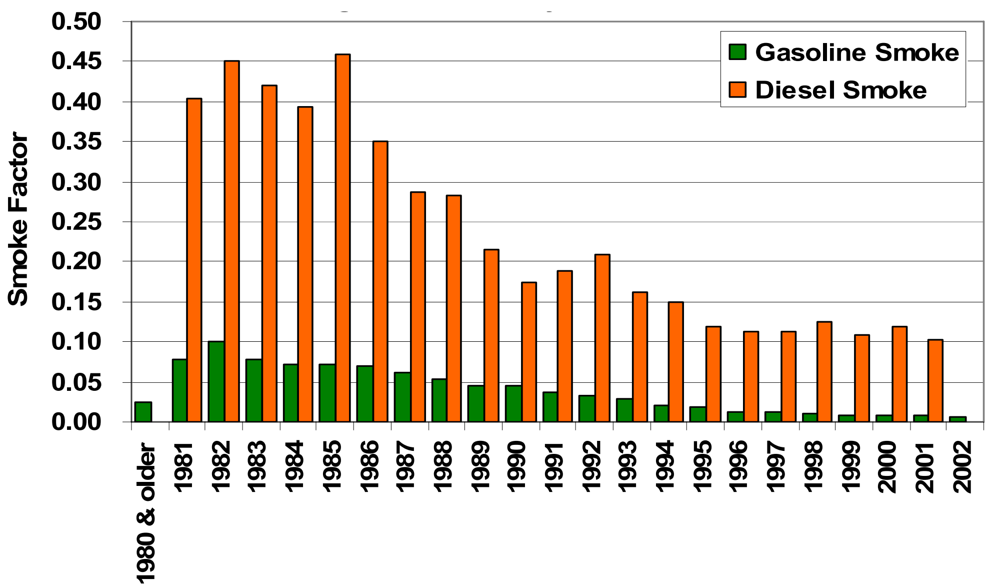


These latter readings are possibly somewhat misleading because we report values for CO, HC, NO and CO_2 however, these are derived from only three ratios thus the fourth reported concentration, namely CO_2, has no added information content. For details see Bishop & Stedman, (1996) and Burgard et al. (2006). The instrument has been used to monitor emissions from automobiles, snowmobiles, locomotives, heavy duty trucks and airplanes in the taxi and takeoff modes.

Recent results illustrating the capability of the traditional instrument are shown in Figures 1-3. Figure 1 compares emissions by model year for CO, HC and NO from two fuel types, diesel and gasoline, for about 20,000 vehicles measured in Phoenix, Arizona in 2004. In the USA only about 2% of the on-road fleet are light duty diesel vehicles, thus the diesel data binned by model year are expected to show significantly more scatter. The top panel compares NO emissions between the two fuel types. As expected, the diesel powered vehicles emit significantly more NO per kg of fuel than the gasoline until one reaches the oldest model years when the two fuels are not significantly different. If light duty diesel NO emissions have declined in the most recent model years, the decline is certainly not very large. The second panel compares the hydrocarbon emissions. While the newest vehicles show similar and low emissions, the older gasoline powered vehicles certainly have higher HC emissions than the diesel. The lowest panel shows CO. In this case, the much higher emissions expected from the older gasoline powered vehicles is apparent, what was a surprise to us was that the newest gasoline powered vehicles have controlled CO emissions to such low values that the newest diesel vehicles are now statistically significantly higher emitters per kg of fuel. Studies in Chicago, Denver and Los Angeles show very similar results.

Figure 2: Average smoke emissions versus model year for two fuels. Virginia 2003 Remote Sensing Device Study Addendum – Vehicle Opacity, February 2003 (full report available at http://www.deq.virginia.gov/air/pdf/air/opacityAdd.pdf)

Figure 2 : Les mesures de la fumée moyennes pour essence et diesel en fonction de l'année de véhicule.



Smoke opacity measurements are best performed in the UV region of the spectrum and require a very stable UV light source. The UV source used by the University of Denver instruments is a 75w xenon arc lamp. It has very high UV brightness over a wide wavelength range allowing monitoring of NH_3 at 205nm and NO_2 at 430nm and providing excellent signal/noise. The commercial on-road remote sensing units use a deuterium lamp. This has lower brightness and less flicker, thus providing improved smoke opacity data. Figure 2 shows the results of a 250,000 vehicle study in Virginia with this instrument, again comparing the diesel with the gasoline fleet. As expected, the diesel fleet shows larger smoke opacity readings and significant effect of increasing smoke emissions from the older model years.

Proportionately the gasoline powered vehicles show an even larger increase as the model years get older because the newest vehicles have negligible average smoke readings. Nevertheless, the diesel smoke values are always higher than the gasoline regardless of model year. Similar results have been observed in a study in British Columbia.

The smoke factor numbers on the y-axis are approximately the carbon soot emissions in mass%. Thus, a smoke reading of 0.1 (very high for gasoline and low for diesel) amounts to about 1 gm soot/kg of fuel. This is approximate because the conversion depends upon a fixed scattering plus absorption coefficient at 230 nm of 18 m^2/g for the soot emissions whereas we know that this coefficient actually varies with the composition, size, size distribution and morphology of the emitted particulate matter. Application of the same opacity to mass conversion coefficient to the average gasoline fleet is certainly inappropriate, however the comparisons shown in Figures 2 and 3 are correct as they stand, because they compare measured fuel based opacity at 230nm from a single fuel.

Figure 3: Average smoke emissions versus model year for vehicles subject and not subject to emission testing.

Figure 3 : Les mesures de la fumée moyennes en fonction de l'année de l'auto pour autos qui avaient été inspectés et qui n'avaient jamais été inspectés.

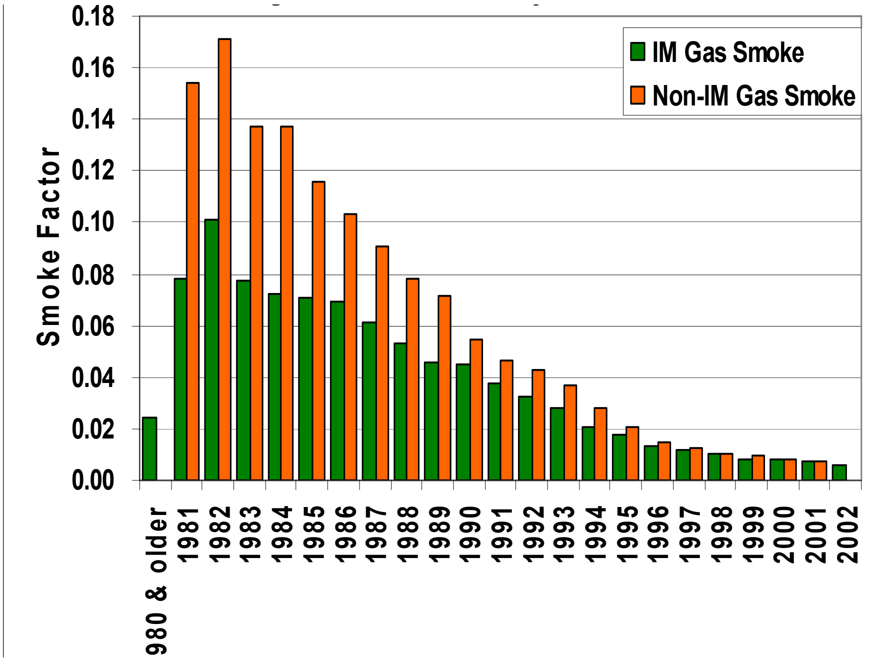


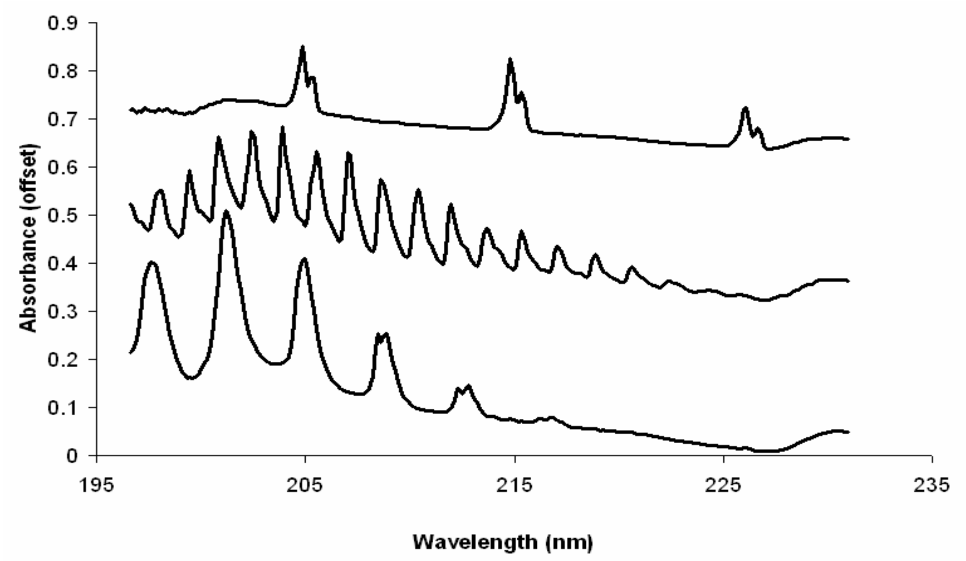
Figure 3 is included because it expands the y-axis and compares only gasoline powered vehicles measured in Virginia. The comparison is between two fleets of vehicles, the one registered in a region where an emission test (I/M) including a smoke inspection is mandatory and the second registered outside that region. Figure 3 shows that as older and older model years are observed on-road, there is a steadily increasing difference in smoke factor between the lower smoke from the I/M fleet and the higher smoke from the non-I/M fleet. Furthermore, these statistically significant differences are on the order of 0.01 RSD 4000 smoke factor units. This impressive discrimination cannot be the result of chance, or of instrument noise, because the model year determination and the I/M status determination are totally independent of the smoke opacity readings.

Objectives

The objectives of our newer developments were to add the capability to monitor other exhaust gases of interest. Ammonia (NH_3) and sulphur dioxide (SO_2) both absorb in the UV at shorter wavelengths than the NO band at 227nm. Significant optical upgrade was necessary to enable the instrument to observe these molecules with good signal/noise at these short wavelengths. The instrumental advances are described in Burgard et al. (2006). The optimum wavelength at which to monitor nitrogen dioxide (NO_2) is 430 nm. The 256 pixel diode array spectrometer which we use to monitor in the 200-230 nm region does not have the spectral range to include bands at 430nm. We therefore reconfigured a second monochromator for the purpose of monitoring NO_2.

Figure 4: Calibration spectra for NO (top), SO_2 (centre) and NH_3 Approximately 3200, 1600 and 920 ppm.cm each respectively.

Figure 4 : Spectres de NO (en haut), SO_2 (centre) et NH_3. A peu près 3200, 1600 et 920 ppm/cm respectivement.



Ammonia is interesting as a pollutant in its own right because it is the major counter ion in secondary particulate matter as ammonium sulphate and ammonium nitrate. Future diesel NOx emissions control systems are expected to use either ammonia or urea as a specific reducing agent for NO. It is quite possible that under some circumstances these systems will not perfectly titrate the NO with the ammonia source and excess ammonia will be emitted.

Sulfur dioxide emissions are a result of sulphur in the fuel. Diesel fuel has traditionally contained more sulphur than gasoline. With the continued installation of catalytic emissions systems for diesel vehicles, on-road diesel fuel sulphur standards have been, and will further be progressively tightened. This processing, amongst other factors tends to increase the price of on-road diesel fuel relative to less regulated or unregulated kerosene fuel for off-road use. To the extent that vehicles are actually using this illegal fuel on the road, then the ability to measure sulphur dioxide emissions becomes a potential revenue source for the authorities who enforce these regulations.

Nitrogen dioxide is formed in the atmosphere from NO. It is a direct photochemical precursor to photochemical ozone formation. NOx is emitted almost entirely as NO in the exhaust of gasoline fuelled vehicles. Diesel vehicle exhaust has been reported to contain approximately 10% by volume of the NO as NO_2. For instance, studies by Carslaw and Beevers 2004 estimate that the NO_2 fraction of NO_x is 12.7 % by volume for diesel vehicles. Tang et al. (2004) made measurements of a series of heavy duty diesel trucks and transit buses, and found that emissions of NO_2 as a fraction of NO_x range between 6.4 -14 % on volume basis. One of the strategies for removing soot from diesel exhaust is a self-regenerating particle trap and intentional catalytic conversion of NO to NO_2 before the trap is a favoured regeneration technique. Because of its importance to photochemical smog, the State of California has promulgated a regulation concerning the amount of NO_2 which a diesel vehicle would be allowed to emit. A remote sensor for NO_2 would enable one to rapidly determine the extent to which this regulation was (or was not) being obeyed.

Figure 4 shows calibration spectra from the remote sensor when used on-road, illustrating that the spectra of NO, SO_2 and NH_3 in the 200-230 nm region can be clearly distinguished.

Results

Table 1 shows on-road emission measurements for over 2000 vehicles from Denver Colorado in June 2005. There are a number of interesting contrasts between the relatively small light duty diesel fleet and the larger gasoline powered fleet. As has been observed previously, the average mass emissions per kg of fuel are comparable for CO and HC and about four times lower for NO when the gasoline fleet are compared to the diesel. For the newly measured pollutants note that the small light duty diesel fleet emits significant NO_2 and SO_2 while all the ammonia emissions come from the gasoline powered fleet. The small negative average ammonia reading from the diesel fleet is the result of software which has not completely eliminated the cross talk between the large NO peaks from the diesel fleet and the currently negligible ammonia emissions. Recent studies of 1500 heavy duty diesel vehicles confirm the above observations. In particular the average ratio of NO/NO2 by mass remains about ten in the driving mode monitored which is acceleration at low speed.

Table 1: Denver, June 2005, on-road, light duty vehicle emission measurements.

Table 1: Mesures d'émissions à Denver Colorado des autos et camionnettes ; juin, 2005.

	Gasoline	n=1945	Diesel	n=67
	Mean	Std err	Mean	Std err
speed ms-1	10.44	0.08	6.01	0.68
accel ms-2	0.28	0.01	0.10	0.05
Average Model Year	1998.62	0.12	2000.39	0.50
CO g/kg	43.88	2.15	23.66	3.44
HC g/kg	2.85	0.25	2.91	1.35
NO g/kg	2.83	0.13	13.57	0.72
SO2 g/kg	0.23	0.01	0.51	0.03
NH3 g/kg	0.45	0.02	-0.04	0.01
NO2 g/kg	0.05	0.03	1.28	0.26

Figure 5 : Cumulative emissions of NO and NH_3 compared to cumulative vehicle fraction. 2016 vehicles ; Denver Colorado, 2005.

Figure 5 : Emissions accumulés de ammoniaque et e monoxyde d'azote comparées á la fraction de véhicules. 2016 véhicules, Denver Colorado 2005.

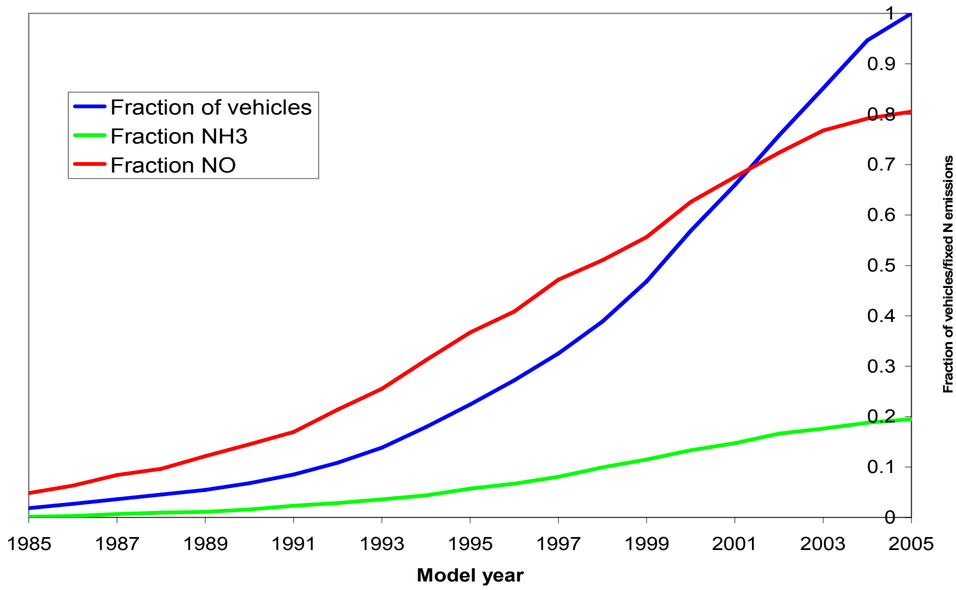


Figure 5 shows cumulative emissions and cumulative vehicle count for NO and NH_3 emissions measured in Denver Colorado in June 2005. Ammonia is the first species that we have investigated in which the emissions are negligible not only for the newest but also for the oldest vehicles. On average the NH_3 accounts for 20% of the fixed nitrogen, The fraction is constant for the newest vehicles, however their

NO emissions increase with age. As a result of this phenomenon, the maximum actual on-road NH_3 emissions comes from vehicles about ten years old. Older vehicles the NH_3 emissions drop off in ratio to NO, presumably because the catalysts become less effective at reducing NO.

Conclusions

Novel configurations of on-road remote sensors has provided the capability not only to monitor the usual regulated pollutants but also smoke opacity, ammonia, sulphur dioxide and nitrogen dioxide from individual passing vehicles in real time. Ammonia emissions might increase as selective catalytic reduction of nitrogen oxides is introduced, Nitrogen dioxide emissions, previously regulated only as a NOx component now have a separate regulation in California because of their smog forming potential and sulphur dioxide emission monitoring assumes ever greater importance as the temptation to use illegally high sulphur fuels increases.

Acknowledgments

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On the Definition of Two Wheelers Exhaust Emission Factors

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Abstract

The paper presents the results of different investigations performed at the Internal Combustion Engines Group (ICEG) of the University of Genoa on the environmental impact of two wheel vehicles in real urban operation. As a first item, an experimental analysis, currently in progress in the city of Genoa, has been aimed at the definition of the main driving kinematic characteristics for two wheelers considering typical urban driving trips on which speed measurements are developed in different traffic situations. The first results are discussed focusing on the comparison between driving cycles and measured mean speed levels. A second aspect was deepened by processing emission data measured in several experimental investigations: real use hot emission factors are proposed in the paper, referring to normalised pollutants (CO, HC, NO_x), particulate matter and carbon dioxide, for different classes of motorcycles. These emission factors are now included in Progress, a code for road vehicles emissions evaluation developed by ICEG.

***Keywords**: motorcycles, driving characteristics, emission factors, hot start conditions.*

Résumé

Il s’agit des résultats de recherches qui ont été faites au ICEG (Internal Combustion Engines Group) de l’Université de Gênes sur l’impact environnemental des véhicules à deux roues en ville. Il y a d’abord une analyse expérimentale qui est en train de s’effectuer dans la ville de Gênes visant à la définition des principales caractéristiques de conduite concernant les deux roues à propos des comportements typiques de conduite en ville dont on a fait des mesures de vitesse dans différentes situations de circulation. Les premiers résultats mettent l’accent sur la convenance de faibles niveaux de vitesse habituellement appliqués à la situation réelle. Une seconde étude a été basée sur les données du processus d émission mesurée au cours de plusieurs investigations expérimentales : les facteurs *d'émissions à chaud dans le conditions réelles de conduite sont proposés faisant référence à des polluants normalisés (CO, HC, NO_x), du particules et du dioxyde de carbone pour différentes sortes de motocycles. Ces facteurs d'émission sont maintenant inclus dans Progress, un code d'évaluation des émissions de véhicules routiers entrepris par ICEG.*

Mots-clefs : motocycles, caractéristiques de conduite, facteurs d'émission, départ à chaud.

Introduction

In spite of a substantial enhancement of vehicle powertrain systems, fuels quality and aftertreatment devices, the environmental impact of road transport remains still high, especially in congested city areas, as reported in Capobianco et ali (2002). Among passengers transport in urban environment, a significant role is played by two wheel vehicles: the trend to a wider use of motorcycles and mopeds in central zones is typical of large urban areas, due to traffic congestion and parking difficulty. The share of two wheel vehicles is quite dissimilar in different European countries, depending on several aspects, such as climate conditions and the structure of large urban areas: data from European Commission (2005), referred to EU15 in the year 2002, account for a contribute of two wheelers ranging from 5% to over 20% of the total passenger vehicles fleet, with an average level of about 13% and a general trend to a significant increase of this share. This tendency is stronger in Italy, where the territory characteristics and the climate conditions allow for an extensive use of these vehicles all over the year. The Italian two wheel fleet is about 23% of the total, but in some congested urban areas this share approaches 35%. It is also interesting to note that Italian mopeds and motorcycles are respectively above 40 and 25% of the total European fleet: data are referred to the motorcycles registration and to an estimation of mopeds number, since no information on this category are included in the vehicles National Register (at least in Italy).

Two wheelers contribution to air pollution is generally significant: with reference to the urban situation in the city of Genoa in the year 2005, motorcycles and mopeds overall contribution to CO, HC and PM road transport total emissions are respectively around 43, 70 and 28 per cent. NO_x amount is negligible (around 4%), while a rough estimate of CO_2 share is between 15 and 20%; the number of two wheel vehicles and the relevant urban mileage are slightly under 33 and 40% of the total, respectively. These results are strictly related to the total emission factors (averaged on the category fleet composition and mileage): mopeds show the highest HC emissions, while CO and PM are comparable to SI and Diesel passenger cars, respectively; the same considerations can be developed for motorcycles CO and HC mean total emission factors.

The paper presents the results of a study performed by the authors on the environmental impact of two wheel vehicles in real urban operation. An experimental analysis, jointly developed by ICEG and the Environment Department of the Province of Genoa, is currently in progress in the city of Genoa aimed at the definition of the main driving kinematic characteristics for two wheelers in urban environment. To this purpose, typical trips were selected and several measurements were developed in different traffic situations. The first results of this investigation are discussed in the paper in order to compare several driving cycles mean speed levels with the real situation and to correlate emission factors to different parameters.

The definition of appropriate emission factors in real driving conditions for this vehicle category represents a basic requirement of any model aimed at the evaluation of road transport impact in urban areas. In the paper real use emission factors referred to hot conditions are proposed, considering normalised pollutants (CO, HC, NO_x), particulate matter and carbon dioxide, for different classes of motorcycles (pre Euro, Euro I and Euro II, fitted with two and four stroke engine). The study developed at ICEG was based on the processing of emission data measured in several experimental investigations performed within the Artemis (Assessment and Reliability of Transport Emission Models and Inventory Systems) European FP5 project, for the development of the Worldwide harmonised Motorcycle emissions certification/Test proCedure (WMTC) and by different Italian research institutes and companies with reference to urban environment. These emission factors are now included in Progress, a code for road vehicles emissions evaluation developed by ICEG, presented in Capobianco and Zamboni (2003).

Experimental activity on typical urban driving trips

The Progress (computer PROGramme for Road vehicles EmiSSions evaluation) code allows the assessment of pollutants (CO, HC, NO_x and PM) from different road vehicle categories and classes, taking into account basic input information, such as the composition of circulating fleet, the related urban mileage for each vehicle class and the definition of typical driving conditions, as reported in Capobianco and Zamboni (2003). Both cold and hot exhaust emissions from the considered vehicle categories and classes are estimated, referring to the whole urban area (or a specific portion of it) and to different time intervals. Since its development and applications are mainly focused on the city of Genoa, and in general to Italian urban environments, particular attention is dedicated to two wheeler categories (motorcycles and mopeds), due to their importance in terms of vehicles number and mileage, focusing both on typical driving patterns and on emission factors evaluation.

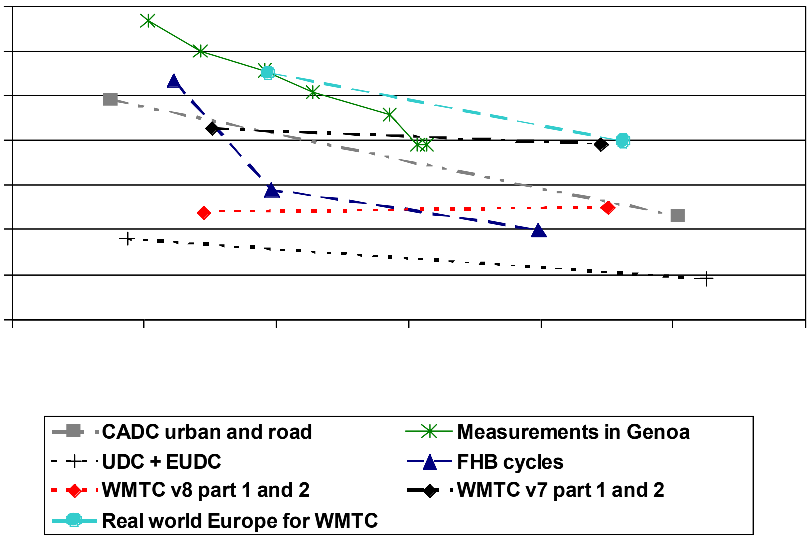
With reference to the first aspect, a dedicated investigation is being performed: a number of urban trips were selected, taking into account the physical peculiarities of Genoa territory, which force the traffic to flow through a few routes connecting the western and eastern zones to the city centre, where most of the offices and commercial activities are located. Other main traffic directions are related to the link of two narrow valleys (which are densely populated and host a number of industrial activities) with the city centre. On this basis, four different trips were selected, respectively related to the central area and to the connections of western, eastern and Bisagno valley residential area to the downtown. Excluding the central trip, referred to an homogeneous zone, from a more general point of view each route can be divided in different parts, related to suburban and central zones and to the use of two fast flowing roads.

The experimental activity is related to the acquisition of instantaneous speed of instrumented vehicles, equipped with a Racelogic VBOX data logger, performing

GPS measurements of position and speed. Acquisitions are related to four different hourly intervals, with particular reference to the morning and afternoon rush hours. In a first phase, covering the period June ÷ November 2005, one passenger car and a scooter with an engine displacement of 150 cm^3 were considered; in a second step, starting from May 2006, two passenger cars and two motorcycles will be instrumented. Data processing allows to evaluate a number of kinematic parameters (mean speed, positive and negative acceleration, idling time, relative positive acceleration, frequency distribution of instantaneous speed classes, etc). The first results on motorcycles show interesting hints in order to compare and analyse information on traffic conditions, real world driving cycle and emission factors.

Figure 1: Comparison of RPA and average speed values as measured for different driving cycles and in real world motorcycle behaviour.

Figure 1 : Comparaison entre RPA et vitesse dans différentes cycles de conduite et dans les conditions réelles de conduite des motocycles.

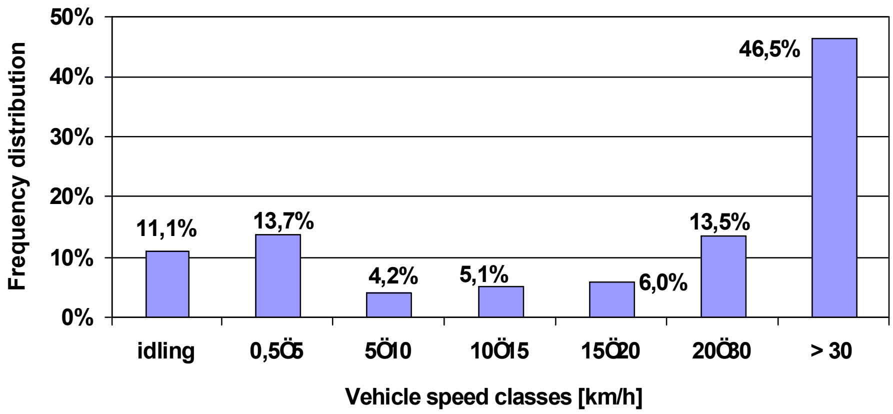


Average values of vehicle speed and relative positive acceleration (RPA), defined in Steven (2001), resulting from ICEG experimental activity are shown in fig.1, together with data referred to real world driving cycles, such as the Common Artemis Driving Cycles, presented in Joumard (2001) and in André (2004), and the FHB cycles, whose data were found in Gense and Elst (2003a), to normalised cycles (ECE 40 and EUDC) and to the WMTC, presented in Steven (2002), considering different versions of the driving cycles and the real world information referred to Europe used for their definition. Average speed from real world cycles are included in two intervals, 17 ÷ 30 km/h and 50 ÷ 62 km/h since, as regards the CADC urban and road, the overall levels were considered, due to the fact that, at this moment, available emission factors are related to the complete cycles; on the other hand, experimental data (which are referred to the central route and to different portions of one of the trips connecting the suburban zones to the downtown) show also levels near 40 km/h: when more detailed results obtained within the Artemis project will be published, it will be possible to consider the emission behaviour related to different CADC sub-cycles, with average speed values ranging from 8 to 65 km/h and more, which will allow a complete characterisation of motorcycles urban driving modes. As regards RPA, it is well known that normalised cycles values are significantly lower than real world ones; this is also true for the final version of WMTC cycles, since a reduction of acceleration and deceleration levels was applied in order to limit the wheel slip when driving on the roller bench, as reported in Steven (2002); anyway, emission data presented in Steven (2002) were obtained considering the previous version. Measurements in Genoa show the same trend reported in Gense and Elst (2003b) for small bikes.

Other interesting considerations can be derived from experimental data: fig.2 shows the frequency distribution of instantaneous speed classes, ranging from idling to > 30 km/h, referred to the central route: it can be seen that only the extreme classes (i.e., idling and 0.5 ÷ 5 km/h on one side, 20 ÷ 30 and > 30 km/h on the other) present a frequency higher than 10%; the average speed of the > 30 km/h class is around 40 km/h, while the overall average speed on this trip is 24.3 km/h. A similar behaviour is shown in the central parts of the other routes and it is even more pronounced for suburban zones, where the fastest class is generally over a 65% frequency, while the idling phase accounts only for a 5 – 7%.

Figure 2:Frequency distribution of motorcycle speed classes in a central route in Genoa.

Figure 2 : Fréquence de distribution de classes de vitesse des motocycles - axe central à Gênes.



Further work is planned, in order to enlarge the available experimental database with reference to the considered routes and to new significant trips, to apply statistical analysis to the considered samples and to process instantaneous speed and acceleration data in order to define more refined classes. As a conclusion, it can be observed that since available information for the definition of emission factors based on average speed (par.2) are concentrated in two ranges, the intermediate interval from 30 to 50 km/h, which may be of a certain interest for urban fast flowing roads, requires, in this phase of the work, a suitable interpolation. If a different calculation procedure is applied, for example taking into account the frequency distribution of vehicle speed and associating a specific emission factors to the selected classes, the lack of information is wider, in particular for the lowest speed and the idling phase, therefore more detailed data have to be considered.

Definition of motorcycles hot emission factors

In this paragraph an overview of the work performed in order to upgrade hot emission factors is presented. Starting from values defined for the first release of the Progress code, presented in Capobianco and Zamboni (2003), a wide bibliographic research allowed to find more information, with particular reference to motorcycles. To enlarge the emission factors database two general rules were, if possible, followed: new data had to concern European vehicles, in order to consider homogeneous engine layout and setting, and tests had to be performed on real world driving schedules, since normalised cycles show very different characteristics in terms of acceleration and required energy (fig.1), leading the engine to operate far from real conditions. A further rule regards vehicle characteristics (engine type and displacement, legislation class, aftertreatment system, etc) which had to be known to allow a proper processing of available information. Different sources were therefore compared for gaseous pollutants, such as Gorgerino and Graziano (2002), Steven (2002), Gense and Elst (2003a), Kis and Pollak (2003), and a number of studies was found for particulate matter, since information are reported in Santino et ali (2001), Gorgerino and Graziano (2002), Prati (2003), Picini et ali (2005), Rijkeboer et ali (2005), even if a reduced number of vehicles is generally considered and data are referred to the standard ECE 40 cycle. Tab.1 presents a summary of the collected information: data were organised according to the three classes of average vehicle speed considered in Progress calculation procedure (< 10, 10 ÷ 40, > 40 km/h), but no information is available for the lower speed class.

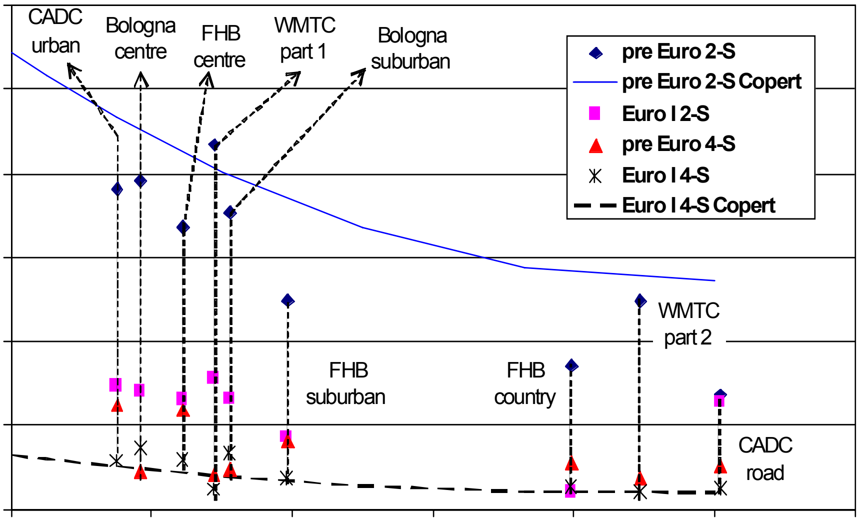
Table 1: Available data for motorcycles in hot real world conditions.

Tableau 1 :Données disponibles pour motocycles dans les conditions réelles de conduite à chaud.

	Number of considered vehicles/overall number of tests				
	CO, HC and NO_x		PM	CO_2	
	Average speed class [km/h]				
Vehicle class	10 ÷ 40	> 40	10 ÷ 40	10 ÷ 40	> 40
pre Euro 2-S	18/44	13/25	4/6	6/8	1/1
Euro I 2-S	10/23	6/12	2/3	4/5	-
pre Euro 4-S	46/122	43/80	14/16	9/11	6/6
Euro I 4-S	82/205	87/147	12/15	25/34	30/33

Figure 3: Hot real world motorcycle HC emission factors.

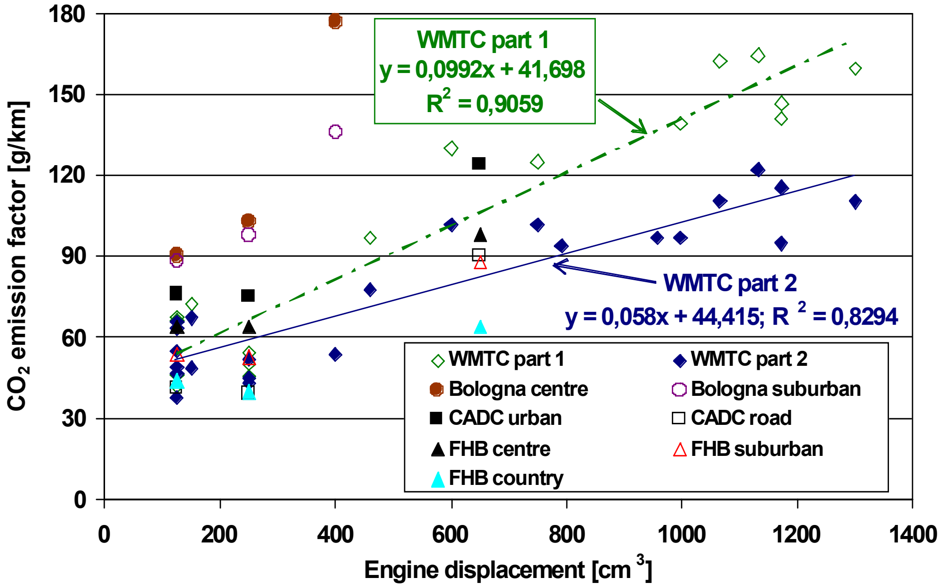
Figure 3 : Facteur d'émission de HC des motocycles (conditions réelles de conduite à chaud).



Tab.1 outlines a number of interesting remarks: results are generally concentrated in the middle speed class, while fewer data are related to the upper interval, for which no tests on PM emissions were found. As regards legislation classes, Euro I vehicles fitted with 4-stroke (4-S) engines were commonly investigated, while less information was available for Euro I 2-stroke (2-S); this is in line with the evolution of the engine technology in order to comply with the limits on exhaust emissions and, consequently, of the market and the estimated fleet, as shown in table 2 with reference to the Italian situation. It is interesting to note that no information was found on Euro II and III classes: this is not surprising for the most recent class, since the relevant phase for registration will come into force at the beginning of 2007, while the lack of information for Euro II vehicles puts some problems for the estimation of emission inventories concerning the short and medium term. Finally, also carbon dioxide emission factors are quite difficult to evaluate, since the number of available data is low and the influence of engine displacement and average vehicle speed would require further analysis (see fig. 4).

Figure 4: Influence of engine capacity on CO_2 emission factors for Euro I 4-stroke motorcycles.

Figure 4 : Influence du cylindrée du moteur sur les facteurs d'émission du CO_2 pour les motocycles Euro I 4 temps.



As a typical example, fig.3 shows HC emission factors, as a function of average vehicle speed, grouped according to the four legislation classes. The reference to the considered cycles is also reported: most of them (CADC, FHB and WMTC) are commonly known, while Bologna cycles were developed within a dedicated study performed by Emilia Romagna Regional Administration, presented in Gorgerino and Graziano (2002). Taking into account the relationship between data and speed values, only in the case of factors related to the Artemis project (measured on CADC and FHB cycles) it is possible to identify, for each vehicle class and pollutant, a decreasing trend in the whole speed range, with the exception of the Euro I 2-S class, which shows an increase for the highest speed. The other data don't confirm them, presenting in some cases high differences, in particular for the pre Euro 4-S class: emission factors to be applied in the Progress code (see table 2) were therefore evaluated as average levels of all the collected results, weighted on the number of vehicles, since in each considered study there was no evidence of wrong figures or of a lower overall reliability. Fig.3 also presents a comparison of available data with the Copert III emission curves, reported in Ntziachristos and Samaras (2000), for which the most widespread classes in Italy (pre Euro 2-S and Euro I 4-S) were considered: the curve for the pre Euro 2-S class seems to be far from the experimental values, while the Euro I 4-S class function fits available data. In general, Copert approach seems to lead to low regression coefficients of polynomial functions, while it is possible to verify cases in which expected speed influence doesn't match the real trends (CO emissions for pre Euro 2-S class, NO_x for 4-S classes).

Considering carbon dioxide emissions, in the case of 2-S vehicles, for which a limited number of experimental values were found (tab.1), the comparison between available sources (including Copert III levels derived from fuel consumption) outlines a substantial scattering of factors. Similar considerations can be developed for pre Euro 4-S class: while data referred to the Artemis project, presented in Kis and Pollak (2003), outline an influence of average vehicle speed and may suggest that Copert III levels are overestimated, especially at lower speed values, results from the other considered investigations lead to increase the estimated emission factors, while reducing the confidence in the observed speed influence. A further aspect to be considered for CO_2 emissions of Euro I 4-S vehicles is the influence of engine displacement, shown in fig.4: though for the lowest capacities (125 ÷ 250 cm^3) values are highly scattered and the relevant ranges are overlapped, a clear effect is apparent, independently of the considered driving cycle, even if in some cases few data are available. This is confirmed by the linear regressions on WMTC data (fig.4), each showing a high correlation coefficient. Moreover, the influence is generally more remarkable for the cycles with lower average speed, which may suggest a relationship also with RPA.

Table 2: Calculated hot emission factors for motorcycles.

Tableau 2 : Facteur d'émission à chaud calculé pour les motocycles.

Vehicle class	Share on the total Italian fleet[1] [%]	CO		HC		NO_x		PM	CO_2	
		[g/km]								
		Average speed class [km/h]		Average speed class [km/h]		Average speed class [km/h]		For all speed classes	Average speed class [km/h]	
		10 ÷ 40	> 40	10 ÷ 40	> 40	10 ÷ 40	> 40		10 ÷ 40	> 40
pre Euro 2-S	22.0	22.0	18.2	10.1	4.8	0.03	0.04	0.190	48.2	55.0
Euro I 2-S	4.0	9.3	11.2	3.8	2.2	0.04	0.07	0.018	61.9	-
pre Euro 4-S	15.0	23.9	21.7	3.1	1.6	0.15	0.18	0.014	65.0	56.6
Euro I 4-S	44.0	10.0	9.2	1.5	0.8	0.22	0.26	0.004	92.9	71.6
Euro II[2]	13.0	7.1	3.9	0.7	0.3	0.42	0.26	-	-	-
Euro III[2]	-	2.6	1.4	0.4	0.1	0.21	0.13	-	-	-

[1]: estimation referred to 2005, based on processing of data from vehicles National Register and Trade association of Italian two and three-wheeled vehicles manufacturers;

[2]: emission factors for Euro II and III classes were obtained from Euro I 4-S values by comparing legislation limits on exhaust emissions (Euro I 4-S, II and III phases).

On the basis of collected data, average emission factors for the different motorcycle classes were calculated with reference to two vehicle speed classes (10 ÷ 40 and > 40 km/h): the relevant results are reported in table 2. As regards the effect of vehicle speed, it can be outlined that, together with the expected reduction of CO and HC emission factors, an increase of NO_x values is apparent, probably due to the fact that CO and HC reduction requires, among the different solutions, engine operating conditions with air-fuel ratios closer to the stoichiometric value, thus increasing NO_x. As expected, 2-S vehicles emit more HC and less NO_x than the corresponding 4-S vehicles, while CO emissions are, quite surprisingly, higher for 4-

S motorcycles (with the exception of Euro I vehicles for average speed > 40 km/h); anyway, data show an overlap for the relevant CO levels, and the result may be related to the number of available values, which is much higher for 4-S motorcycles.

Conclusion

The wide investigation activities performed at ICEG on two wheelers air pollution allowed to define emission factors referred to hot operating conditions and to deepen motorcycle driving behaviour in urban environment, but also to highlight a number of items requiring specific studies in order to enlarge the knowledge on this theme which, taking into account the significant number of motorcycles and mopeds and the related mileage in several European Union countries, surely influences the total pollution due to road transport.

From the analysis of available emission data, different aspects appear to be further investigated, such as the evaluation of carbon dioxide motorcycles emissions, the definition of mopeds emission factors (for which very little real world data were found), the cold start effect (emitted quantities, transient duration, travelled distance, influence of ambient temperature) and the mileage influence on emissions deterioration, particularly for catalysed vehicles. With reference to this last item, only an investigation on prototype catalysts aging in 2-stroke vehicles, presented in Coultas and ali (2002), and an Inspection and Maintenance programme developed within the Artemis project are known, but specific studies should be considered for a better focusing of the problem. It must be also remarked that, with reference to the European legislation on exhaust emissions, data on the most recent vehicle classes (Euro II and III) are not available and the relevant emission factors can be defined only through reduction coefficients obtained from the comparison of normalised limits.

As regards future developments of ICEG activities in this field, further bibliographic research will be performed, starting from the analysis of Artemis project final reports, as soon as they will become available. The experimental work on the definition of motorcycles driving kinematic characteristics in typical urban driving trips will continue, in order to enlarge the available database and to develop emission calculation procedures referred not only to average vehicle speed but also to different information, such as the instantaneous speed frequency distribution. Finally, the Progress code will be applied in order to evaluate two wheelers contribution to road transport pollution in selected urban environment.

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How to get realistic emission factors from vehicles with active operating after treatment systems?

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Abstract

Active operating exhaust after treatment systems are increasingly employed in passenger cars for meeting the given legislative pollutant emission levels. These systems include pollutant storage units that have to be regenerated on occasion. Their strategy for regeneration depending on the drive pattern, the resultant emission levels and their share on the emission level during normal operation mode are key issues for determining realistic overall emission factors of these cars.

In order to investigate these topics, test series with four cars featuring different types of such after treatment systems have been carried out. Its emission performance in legislative and real-world cycles has been detected as well as at constant speed levels. The extra emissions determined in these measurements are presented together with the methodology applied for obtaining them.

The main conclusions to be drawn are that regenerative exhaust after treatment systems cause notable overall extra emissions. These extra emissions can be taken into account either by conducting sufficient repetitions of emission measurements with the single car or by enlarging the sample of cars with comparable after treatment systems.

***Keys-words**: regeneration, NO_x storage catalytic converter, diesel particle filter, real-world emissions, extra emissions, passenger car.*

Introduction

Recently active operating vehicle exhaust after treatment systems have come on the market in order to meet the subsequently tightened emission limits for the homologation of new passenger car models. The different after treatment systems have in common that they work in a non-continuous way, i.e. they store a certain pollutant in a trap during normal operation mode of the engine and disintegrate it chemically in a special procedure denominated as regeneration. The behavior of some of these systems employed as retrofits is known (Ntziachristos et ali, 2005), however the operation and performance of OEM-systems in real world situations is not known yet. The scope of this study is to investigate these topics for being able to derive methods for determining the emission level of pollutants caused by cars with such after treatment systems. Therefore a test series has been carried out with four Euro 4 passenger cars that feature different designs of these after treatment systems. The emission levels of the cars have been determined in single test series on a chassis dynamometer test bench including several repetitions of legislative cycles and so-called real-world cycles. Measurements at different constant speed levels have also been considered to examine the possible effect of regenerations on the emission level of the car at static operation mode.

From the given experimental campaign, one can basically deduce that the single strategies employed for operating the active exhaust after treatment system differ from each other. The resulting extra emissions due to a regeneration of the pollutant storage unit strongly depend on the driving situation at which the regeneration occurs, the regeneration strategy applied as well as on the frequency of these regenerations. The latter varies with the driving history too. But the overall extra emissions that are obtained by taking into account the incidence of appearance of extra emissions in a certain driving situation are mostly less pronounced. Thus, the effect on the pollutant emission factors of such a car type can be included by adjusting its experimental program, either by increasing the measured number of driving cycles with a single car or the investigated car sample.

Experimental program

One naturally-aspirated gasoline fired and three turbocharged diesel passenger cars of the legislative period Euro 4 have been used for the experimental campaign, see Table 1. The gasoline car B4-01 features direct fuel injection and is able to run in lean combustion mode. Therefore a NO_x storage catalytic converter is employed for preventing its extra emissions. One diesel car, namely D4-03, is also equipped with a NO_x storage catalytic converter in combination with a particle filter and exhibits the possibility of exhaust port fuel injection. The two other diesel cars dispose of a diesel particle filter (DPF), whereas they differ in their proceeding for filter regeneration: D4-02 provides the heat required for a particle burn out by late fuel injection in the combustion chamber. D4-01 employs a fuel borne catalyst, as its unit-injector fuel-injection system does not offer wide flexibility in fuel injection regarding crank angle position. Besides this particular car, a common rail fuel injection system is applied for the other diesel cars. All four cars feature cooled exhaust gas recuperation and an oxidation catalytic converter applied as a precatalytic converter except for car D4-03, where it is located after the main after treatment unit.

Table 1: Main characteristics of the passenger cars employed in the experimental campaign

denotation		B4-01	D4-01	D4-02	D4-03
manufacturer		Volkswagen	Volkswagen	Opel	Toyota
model		Touran 1.6 FSI	Passat Variant 2.0 TDI DPF	Vectra Caravan 1.9 16V CDTI	Avensis D-Cat
fuel		gasoline	diesel	diesel	diesel
empty weight	[kg]	1585	1649	1605	1530
displacement	[cm^3]	1598	1968	1910	1995
power	[kW]	85	100	110	85
gearbox		manual 6	manual 6	manual 6	manual 5
mileage	[km]	7966	3057	3093	3091
legislative period		Euro 4	Euro 4	Euro 4	Euro 4

For each car a test series has been conducted on a chassis dynamometer test bench carrying out several repetitions of the real-world cycle CACD, which includes as cycle parts the urban, rural and highway cycles derived for the ARTEMIS research project (André, 2004). Repetitions of the real-world cycle IUFC15 (André et ali, 1999) and the so-called cycle L2, a combination of the legislative cycle NEDC and the cycle BAB, have also been considered. In addition, test runs at different constant speed levels have been accomplished in order to investigate the influence of regenerations of the after treatment system on the emission level excluding distortion caused by changing operation mode of the car. Note that the settings of the test bench, its ambient conditions, the test procedure as well as the exhaust gas sampling and analyzing procedure have been applied according to the guidelines 70/220/EEC of the Economic Commission for Europe for passenger cars as far as included. Time resolved pollutant emissions have been recorded too at tailpipe and before the main after treatment unit. Additional signals such as the pressure drop over the pollutant storage unit, the exhaust gas temperature and single gas components at selected duct points have also been detected for gaining more information on the behavior of the after treatment systems. Furthermore, the number of emitted particles PM_N has been measured with a condensation particle counter (CPC) that is preceded by a unit for thermal desorption of volatile compounds on the particle's surface (Mohr et ali, 2006). The fuel used in the experimental campaign corresponds to standard diesel and gasoline with an octane number of 98, both featuring a sulphur content of less than 10 ppm.

The method applied for identifying the regenerations of the cars basically consists of detecting any anomalous course of the relevant signal traces and to determine in each case if a regeneration takes place or not, either controlled or spontaneously. In order to estimate the resulting effect on pollutant emissions, the overall relative extra emissions $ee_{cp,tot,pol}$ of a pollutant *pol* in a certain cycle part *cp* during the test series of the single car can be calculated, cf. Equation (1). There, the average of the extra emissions of a pollutant discharged per unit distance relative to its usual emission level in the single affected cycle parts *i* is weighted with the relative incidence of appearance of extra emissions in that particular cycle part $x_{cp,ee}$. The latter is obtained by relating the total distance of the same affected cycle parts of a single car to the total distance of the respective cycle repetitions conducted.

$$ee_{cp,tot,pol} = x_{cp,ee} \cdot \overline{ee_{cp,i,pol}} \qquad (1)$$

Experimental results

1. General emission performance

An overview over the emission performance of the cars can be obtained by comparing the emission levels in the legislative cycle NEDC to their respective limit values, see Table 2. One can see that the gasoline car clearly fulfils the limits given, although the good performance slightly decreases when it operates partly in lean combustion mode, especially for HC and CO emissions. This occurrence indicates a possible incomplete combustion in combination with insufficient fuel-mixture generation in that operation mode. Note that given the data of the measured cycles, no definable criteria for switching the combustion mode of the engine have been found. Also no remarkable NO_x peaks after finishing the NECD cycle measurements have been detected like in other experimental campaigns with similarly equipped cars (Mittermaier et ali, 2003)

Table 2: Pollutant emission levels in the cycle NEDC relative to its limit values

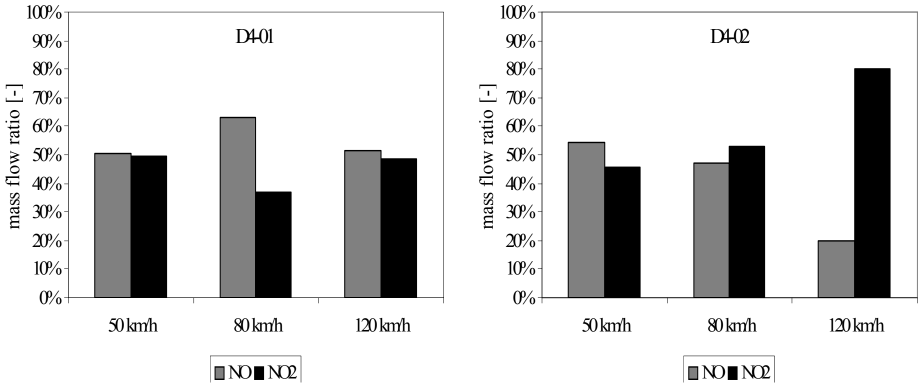
substance	unit	B4-01 stoich.	B4-01 lean	D4-01	D4-02	D4-03
CO	[%]	34.2	59.1	7.3	43.0	42.3
HC	[%]	40.2	52.4	-	-	-
NOx	[%]	15.9	17.4	96.1	113.9	61.3
HC+NOx	[%]	-	-	86.9	103.9	60.4
PM_m	[%]	-	-	6.5	4.8	5.3

All three diesel cars show a rather good performance in CO emissions too, whereas the car D4-01 exhibits a remarkable underbidding of the limit value. The same can also be noticed for the three diesel cars with respect to the gravimetrically determined emissions of particulate matter PM_m, as the measured emission level by far falls below the respective limit value. But regarding NO_x emissions, the cars differ considerably: car D4-03 features a quite low emission level for nitrogen oxides, thus one can assume that its NO_x storage catalytic converter valuably contributes to lower the emissions of this pollutant. In contrast to that, D4-01 remains below the limit value only with a small margin and D4-02 even exceeds the emission level by almost 14%. As these two cars inhibit particle emission with a DPF, their discharge of nitrogen oxides is supposed to be lowered by adjusting the engine management system in order to minimize its formation during the combustion process. However this strategy has apparently not been entirely implemented in the present case, at least not for car D4-02. The HC emissions of the diesel cars do not represent a critical issue, as the emission levels improve when comparing the resulting values of

NO_x to the respective figures for the combined HC and NO_x emissions.

For some cars during single measurements at constant speed level, the NO_2-converter of the CLD analyzer has been bypassed for being able to measure NO alternatively to NO_x. From these measured concentration levels, the mass flow rates of NO and NO_2 have been derived in order to quantify their respective ratios, cf. Figure 1. One can see that car D4-01 features typical ratios of 50-60% NO of the total at all measured speed levels. Car D4-02 shows an increasing share of NO_2 for higher speed levels, whereas at 120 km/h its appearance is predominant with a ratio of about 80%.

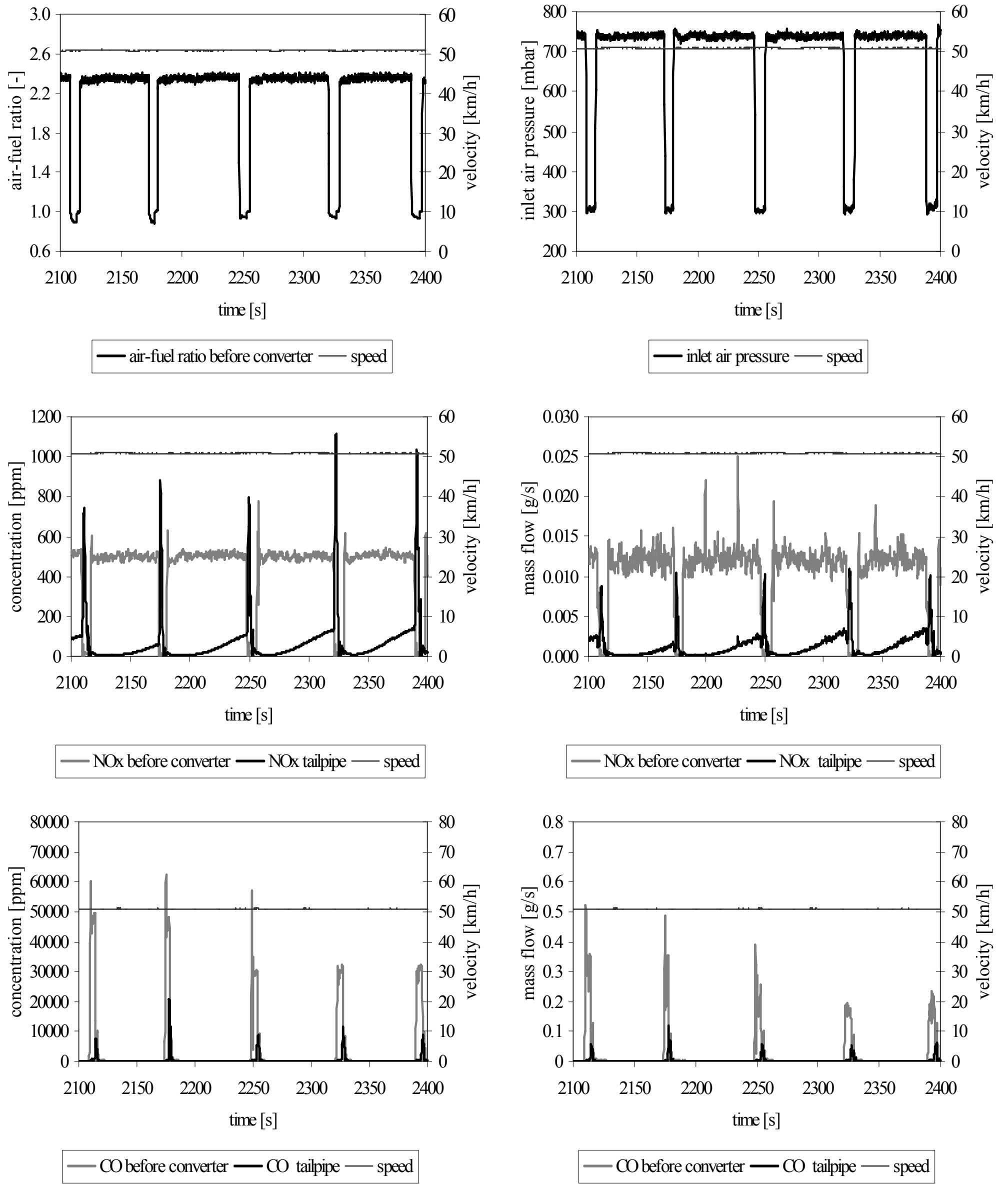
Figure 1: NO- and NO_2-ratio (mass) at different constant speed levels for car D4-01 and D4-02



2. Effect of regenerations

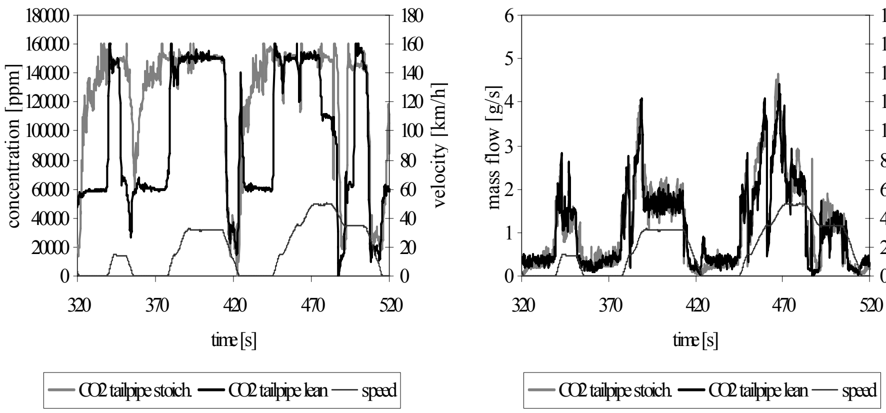
For car B4-01, measurements at constant speed level are most suitable to investigate the effect on pollutant emissions while the NO_x storage catalytic converter regenerates. In that driving regime, the engine usually runs in lean combustion mode with periodical short phases of air fuel mixture enrichment for reducing the stored nitrogen oxides. The time intervals of these phases are 80 s, 60 s and 110 s for 30 km/h, 50 km/h and 80 km/h car speed respectively and last around 10 s. At 120 km/h the car runs exclusively in stoichiometric combustion mode and no regeneration occurs.

Figure 2: Course of relevant signal traces during a regeneration of car B4-01 at constant speed level



A more detailed analysis on the emission behavior during the regeneration can be obtained from a single test run at 50 km/h, see Figure 2. The course of the air-fuel ratio signal clearly indicates the two different operation states. At normal operation, a high discharge level of NO_x before the converter can be detected, that is stored in the converter unit as the tailpipe emissions are much lower. The Latter rise slowly in the time interval between two regenerations, i.e. the storage ability of the converter decreases. Interestingly, this rise in NO_x tailpipe emissions starts earlier with increasing number of regenerations, thus the storage efficiency seems to lower progressively. While regenerating, the air fuel mixture is drastically enriched, which causes peak emissions of NO_x and CO that are not completely compensated by the after treatment system. But the resulting discharge of pollutants per unit mass flow are less pronounced, especially for HC emissions (not shown), due to the fact that the mentioned mixture enrichment is basically achieved by throttling the intake air, see Figure 2. This effect acts on CO_2 concentrations too, even in cycle test runs, cf. Figure 3. Besides fuel cut of in overrun situations, B4-01 shows lean combustion mode at idle and some constant speed levels, but the CO_2 discharge per unit mass flow exhibits no particular improvement.

Figure 3: Course of CO_2 concentration and mass flow in a cycle part for car B4-01



Also note that within the test series of this car, only in two cases definable extra emissions of particles have been detected in measurements with driving cycles, cf. Table 3. In one case, the extra emissions determined from the CPC recording is much higher than the one derived from the gravimetric particle measurement, but the other pollutants show no particular deviation. These extra emissions are rather supposed to result from a somehow disturbed combustion of the directly injected fuel (Mohr et ali, 2003) than from a regeneration of the NO_x storage catalytic converter considering the fact that the car partly operates in lean combustion mode in the affected cycles.

In the test series involving car D4-01 not a single controlled regeneration has been observed. But the analysis of the measurement data indicates that the fuel borne catalyst employed for regenerating the DPF allows particle burn out at the usual exhaust gas temperatures and thus regenerates consecutively (Alvarez et ali, 2006). In fact, single time resolved CPC measurements at constant speed levels reflect peak particle emissions at the beginning of the test that decline before the exhaust gas reaches normal operating temperature. However an additional initial partial blow-out of stored particles cannot be excluded too, because measurement results from tests with driving cycles also show that the particle number emission in the first section of the cycle is significantly higher than in the following, even in cycles with warm engine start.

Table 3: Overall relative extra emissions in cycle parts due to regenerations (diesel cars) or lean combustion mode (gasoline car) weighted with their relative occurrence in test series

			$ee_{cp,tot,pol}$					
car	cycle part cp	$x_{cp,ee}$ [-]	CO [-]	HC [-]	NO_x [-]	CO_2 [-]	PM_m [-]	PM_N [-]
B4-01	L2 ECE	0.050	-0.004	0.008	0.014	0.001	0.038	0.025
	L2 EUDC	0.084	-0.012	-0.019	-0.052	0.002	-0.002	2.234
D4-02	L2 ECE	0.050	0.008	0.004	-0.001	0.000	-0.007	0.190
	L2 EUDC	0.084	0.028	0.070	0.000	0.004	-0.026	0.311
	post-cycle NEDC	0.030	0.004	0.032	0.016	0.031	-	-
D4-03	L2 BAB	0.172	0.036	-0.019	0.009	0.011	0.087	0.757
	CADC urban	0.017	-0.004	-0.005	0.040	0.004	0.015	0.027
	CADC rural	0.063	-0.018	-0.025	0.004	-0.001	0.005	0.069
	CADC highway	0.094	0.004	-0.030	0.028	0.009	0.039	0.367
	constant 80 km/h	0.084	7.982	2.797	0.053	0.031	-	0.823
	constant 120 km/h	0.286	6.738	-0.114	0.180	0.049	-	2.174

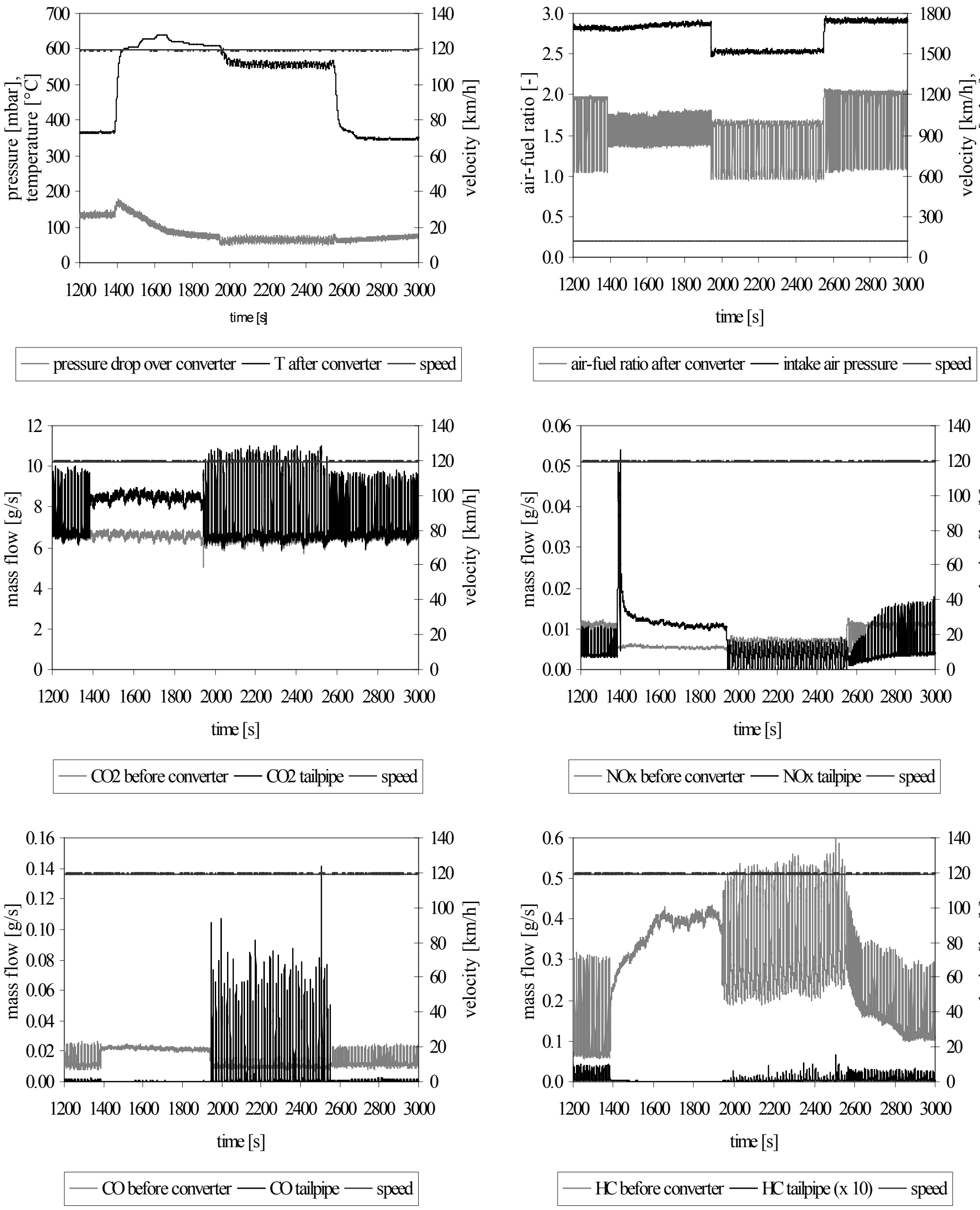
Car D4-02 features a similar behavior regarding the incidence of regenerations. Within its test series of 1300 minutes duration, only a single controlled regeneration has been detected with increased exhaust gas temperature at the end of a cycle. There CO and NO_x emissions do not vary particularly from the respective usual values, but HC and CO_2 emissions rise remarkably. Besides, a higher number of particles has been recorded during the two cycle parts preceding the regeneration, cf. Table 3, and in one of these cases increased HC emissions can be noticed too. There however the exhaust gas temperature does not exceed typical values.

The after treatment system of car D4-03 with its combined NO_x storage catalytic converter and particle filter exhibits an interesting regenerative ability. Again measurements at constant speed level deliver most insight, cf. Figure 4. In normal operation mode, i.e. when the exhaust gas temperature reaches usual values, a similar strategy than in car B4-01 is applied for reducing NO_x with short air fuel mixture enrichment phases. The time intervals between these enrichment phases (e.g. 25 s at 80 km/h) and its duration (e.g. 5 s at 80 km/h) are in any case shorter compared to the respective operation states of car B4-01. In regeneration mode of the particle filter, a two-stage regeneration strategy is visible. The first stage serves to recover the loaded particle filter and the second to desulfurize the NO_x strorage catalytic converter. The time intervals of these regeneration stages are 3.6 h and 0.8 h for 80 km/h and 120 km/h respectively and last around 20 min.

In the first stage of the regeneration, highest exhaust gas temperature appears together with a notable decrease of the counter pressure of the converter. The temperature rise is achieved by injecting fuel in the exhaust port as indicated by the increase of CO_2 tailpipe emissions with respect to the discharge level before the converter. This fuel injection generates the heat required for particle burnout in its filter, but causes considerable NO_x extra emissions too, especially at the beginning of the regeneration phase. Pronounced engine out CO and HC emissions occur as well, but they are almost completely reduced in the converter. In the second stage, periodical mixture enrichment phases occur at high exhaust gas temperature to regenerate and possibly desulfurize the NO_x storage unit. These enrichment phases are also caused by fuel injection in the exhaust port, as the intake air pressure remains constant during that period but CO_2 tailpipe emissions feature peak amplitudes. The resulting CO peaks are not completely compensated by the after treatment system and peak NO_x emissions at the end of a single enrichment phase can be detected too. HC peak emissions are strongly reduced, though its resulting extra emissions are still considerable as the usual HC discharge is even lower. Obviously the sudden change of the air fuel mixture in the exhaust gas prevents the proper functioning of the converter.

The overall extra emissions of car D4-03 caused by regenerations in the test runs at constant speed level are calculated for the combined regeneration phases mentioned above, cf. Table 3. The resulting extra emissions of $NO_{x,}$ PM_m, and CO_2 are remarkable, but the additional discharge of CO, HC and PM_N is striking. Another kind of regeneration observed in the first part of a cycle CADC shows just considerable extra emissions for nitrogen oxides. Besides, a notable increase of particle number emissions has been detected in some parts of different driving cycles. The Latter corresponds with the respective gravimetrically determined particle discharge. In these cycle parts, the temperature of the exhaust gas after the converter reaches the level at which particle burn out in the filter is supposed to occur, but no significant decrease of the pressure drop over the latter can be observed. Thus, one cannot assume a regeneration taking place in the sense of filter recovery, however it represents a process conducted by the control unit of the after treatment system. Interestingly such operation phases of car D4-03 have all in common that they appear at rather high engine load, as the rated power of the engine is moderate and the gearbox only features 5 gears.

Figure 4: Course of relevant signal traces during a regeneration of car D4-03 at constant speed level



Discussion and conclusions

The present experimental campaign carried out with four Euro 4 cars featuring different active operating exhaust after treatment systems provides varied insight into its functioning and emission behavior. Measurements at constant speed level on chassis dynamometer test benches are most suitable to understand the particular regeneration procedure and its effect on the discharge of pollutants. Series of driving cycles, especially real-world cycles, allow estimating their frequency of occurrence at normal driving conditions and the resulting change in the emission level.

Within the given experimental campaign, the two NO_x storage catalytic converters installed in different cars exhibit very short regeneration intervals, so that the resulting extra emissions are included in the emission levels determined with the usual measurement effort. Overall NO_x emissions are lowered, but the strategy employed for regenerating the unit, consisting of air fuel mixture enrichment phases combined with rather high exhaust gas temperature, causes additional HC, CO and also NO_x peak emissions that vary depending on its quality of implementation.

The regeneration strategy of the particle filters applied in three different cars delivers varied perceptions. The car employing a fuel borne catalyst featured not a single regeneration conducted by the respective control unit, another car only one. Therefore no particular overall extra emissions can be accounted to the regeneration process. But the car with the combined particle filter and NO_x storage catalytic converter exhibits an interesting behavior: the overall extra emissions due to regenerations carried out in driving cycles are notable. At constant high speed regime though, the rise in emissions is impressive and to be attributed to both the extra emissions of a single regeneration and the high frequency of regenerations in that driving mode.

Thus, the implemented regeneration strategy and the occurrence of regeneration phases in the different driving situations of a single car valuably influence its emission behavior. In fact, both the extra emissions caused by regenerations and the time interval between two regenerations strongly depend on the driving pattern conducted. Consequently, with regard to determining real-world emission factors of a sample including cars with such active operating after treatment systems, one has to considerably increase either the number of measured driving cycles or the number of cars to be measured in order to take into account the mentioned variations.

Acknowledgments

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Modelling of cold start excess emissions for passenger cars in Artemis model

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Abstract

One of the subtasks of the Artemis study was to develop a cold start excess emission model for passenger cars. The present model, developed empirically with available data in Europe, consists in modelling the cold start impact on road vehicle's emissions as function of the pollutant (regulated or not), the vehicle type (i.e. the emission standard and fuel type), ambient temperature and average running speed. The final model consists in fact in three different models. The first one is to calculate the cold start excess emission for one start and a pollutant. The two others allow calculating the emission for a fleet. Each one of these two last models is appropriate to the available data of the user.

Keys-words: *cold start, modelling, Artemis, passenger cars.*

Introduction

The cold start excess emission for a vehicle and a driving cycle is defined as the excess emission for a vehicle starting at the ambient temperature compared to the emission of the same vehicle running in hot (engine) conditions. So, a vehicle starting in ambient conditions and running a driving cycle, will take a time t_{cold} to reach hot running conditions. This time t_{cold} is linked to the distance d_{cold} by the mean speed of the driving cycle during the cold period. The total emission E_{tot} during a driving cycle of a vehicle which does not start in hot conditions can be calculated by the sum of the hot emission E_{hot} and the cold start excess emission EE_{cold}:

$$E_{tot}=E_{hot}+EE_{cold}.$$

EE_{cold}, which is the aim of this paper, is the absolute cold start excess emission (in gram) defined as the additional emission value obtained under cold conditions compared to the emissions values that have been recorded for the same period (cycle) under hot conditions.

Three methods are till now available in Europe to model excess emission at start:

- The Handbook, applied mainly in Germany and Switzerland (Keller et al., 1995)
- The MEET approach, based on a synthesis of the available cold emission data in Europe (Joumard and Serié, 1999)
- The Copert III approach (Ntziachristos and Samaras, 2000), which is a mixture of the former Copert and MEET approaches.

Samaras et al. (2001) evaluated the values of excess emissions for various situations in Europe, by using the three approaches. He found that, due to the differences between the methodologies of Copert III and MEET, there are differences between the models' results. These effects however are mostly exhibited at very low values of the speed and ambient temperature and become negligible when intermediate values of these parameters are approached. In general, the difference between the results obtained by Copert and those by MEET are reduced for temperatures between 15°C and 25°C and also for high vehicle speeds. The agreement between the results of Copert III and those of the model suggested in the German handbook is very good, especially in the case of Euro I vehicles, even though the two models exhibit several differences with respect to the methodology. All these calculations show that the excess emissions depend of course on the methodology used and on the emission data used.

The present study (André & Joumard, 2005) is using cold measurements collected during the MEET project, together with measurements collected recently, and measurements carried out specifically for the Artemis study. The aim of this study consists in modelling the cold start impact on road vehicle emissions as functions of the pollutant and the vehicle type, using all the existing data in Europe. This model is developed empirically, considering the available data for passenger cars: excess emissions indeed, but also ambient temperature, and driving behaviour statistics.

Such model is necessary for large-scale applications - national inventories for instance - but could also be used for smaller scale applications.

New method to calculate EE_{cold}

At the beginning of the work, two methods were available to calculate cold start excess emissions on the basis of repeated or successive driving cycles.

The first method (Joumard & Sérié, 1999) consists in calculating the standard deviation on the last two data, then on the three last … of a cycle repeated as long as enough to be sure that the vehicle is hot at the end. As long as the vehicle runs (back) under hot conditions, the emissions are stable and so the variations occur around a mean (hot emission value). As soon as cold start conditions appear, the standard deviations increase rapidly. The cold start distance d_{cold} ends where the minimum of the standard deviation appears. The cold start excess emission is the difference between the emissions during hot and cold conditions (i.e. resp. before and after d_{cold}).

The second method (Weillenman, 2001) consists in calculating the continuous cumulative emissions from the start. The linear regression of this curve is then calculated on the basis of the hot part alone, which is arbitrary chosen. The value of the regression at distance 0 gives the cold start excess emission value. The cold start distance is the distance where the cumulative emission falls completely between two straight lines with the same slope than the regression but which have 95 % and 105 % of the cold start excess emission.

These two first methods show that there are quite differences in cold start excess emission calculation and, above all, in cold start distance. In the first method, the calculated cold distance is overestimated. In the second method, the hot conditions are not determined rigorously.

We developed a new method based on the advantages of both methods (André et ali, 2005). We first calculate a rough cold start distance by using the first method. Then we calculate the hot emission, the standard deviation and the linear regression of the hot emission. The value of the regression at distance 0 gives the cold start excess emission. The exact cold start distance is determined by looking at the distance where the emission falls entirely between two straight lines which are the hot emission ± 2 standard deviations.

We used data recorded with 'short Inrets' and 'Artemis' driving cycles (André, 2004), and with legislative cycles. These last cycles do not reflect the reality, but they represent the main part of the data. Joumard et ali (1995b) showed that theses cycles were not hot at the end. So we had to introduce a slight adjustment for each vehicle category and pollutant, to obtain the full excess emission, especially for ECE-15 cycle.

The method is used for the regulated pollutants. For the unregulated pollutants, as the emissions were not measured on successive cycles, we apply the cold start distance calculated for total hydrocarbons (HC). The cold start excess emission is calculated by the difference of the value for a cycle beginning in cold conditions to the value of the same cycle beginning in hot conditions.

Data base

To model cold start excess emission, we used data available in Europe. These data come from the MEET project, from national programs and from measurement made within the Artemis project. All these data come from 14 laboratories in Europe. After selection of usable data, we kept 35 941 emission factors (all vehicle categories and pollutants merged) measured with 1 766 different vehicles over 5 different driving cycles (see table 1 for the main characteristics of the cycles). The vehicles emission standards are Euro 0 to Euro 4 for diesel and petrol fuel type. It was so possible to work with 10 vehicle categories.

Table 1: Main characteristics of used driving cycles.

Tableau 1 : Principales caractéristiques des cycles de conduite utilisés.

Type	Name	Short name	Duration (s)	Distance (m)	Average speed (km/h)
Legislative	FTP72-1		505	5779	41.2
	ECE-15		780	4052	18.7
Inrets	Urbain fluide court	IUFC	189	999	19.0
	Route court	IRC	126	1439	41.1
Artemis	Urban	Art. Urban	921	4472	17.5

Cold excess emission for a start

The collected data allow us to express the cold start excess emission (EE) for a start and a vehicle type (i.e. emission standard and fuel type) and a regulated pollutant as a function of the ambient temperature (T), the mean speed during the cold period (V), the distance (d) and the parking time duration (t) before starting. So EE could be expressed as:

$$EE(T,V,\delta,t) = \omega_{20\,°C,20\,km/h}.f(T,V).h(\delta).g(t) \qquad (1)$$

With $\omega_{20\,°C,\,20\,km/h}$ is the total excess emission at 20°C and 20 km/h, $f(T,V)= \omega(T,V)/\ \omega_{20\,°C,\,20\,km/h}$ is the correction function depending on ambient temperature T and mean speed V, $\delta=d/d_c$ is the adimensional distance, i.e. the distance d divided by the cold distance d_c, $h(\delta)$ is the correction function depending on adimensional distance and g(t) is the correction function depending on parking time duration t.

$\omega(T,V)$ (and so f(T,V)) and $h(\delta)$ are computed by using the method described in paragraph 1.

f(T,V) is a 3D linear regression which depends of T and V with the condition that f must tend toward 0 when T increases.

$h(\delta)$ is an exponential function of δ which can be expressed as $h(\delta) = \dfrac{1-e^{a.\delta}}{1-e^{a}}$ where a is deduced from data. In this function $\delta=d/d_c(T,V)$, where cold distance $d_c(T,V)$ is a 3D linear regression depending on T and V.

g(t) was computed from rare data available in Europe. Only 3 laboratories (VTI, EMPA and TUG) were able to provide such data. We then calculated an average table of the parking time duration influence for each pollutant and vehicle category and plot the best fit. It was so possible to give a polynomial function for each case.

Here is an example for CO and Euro 2 petrol cars:

$\omega_{20\,°C,20\,km/h} = 17.060$ g

$f(T,V) = 1.927-0.043*T+0.003*V$

a = -9.007

$d_c(T,V) = 4.409-0.002*T+0.024*V$

$g(t) = -2.916*10^{-7}*t^3-1.941*10^{-4}*t^2+4.315*10^{-1}*t$

So EE is expressed as:

$$\underset{CO,Euro2,Gasoline}{EE(T,V,\delta,t)} = \left| \begin{array}{l} \underbrace{17.060}_{\omega_{20°C,20km/h}} \cdot \underbrace{[1.927-0.043\cdot T+0.003\cdot V]}_{f(T,V)} \cdot \underbrace{\left[\frac{1-e^{-9.007\frac{.d}{4.409-0.002 T+0.024 V}}}{1-e^{-9.007}}\right]}_{h(\delta)} \\ \cdot \underbrace{\left(-2.916\cdot 10^{-7}\cdot t^3 - 1.941\cdot 10^{-4}\cdot t^2 + 4.315\cdot 10^{-1}\cdot t\right)}_{g(t)} \end{array} \right.$$

(2)

We also introduce in the above model, the possibility to compute excess emission for near future vehicles by using the cold excess emission rates given by Samaras and Geivanidis (2005). These rates are only for CO, HC, NOx and particulates for Euro 4 and Euro 5 emission standards. To obtain the cold start excess emission for a near future vehicle, we apply the rate to both cold start distance and cold start excess emission of a diesel Euro 3 or petrol Euro 4 vehicle without DISI (Direct Injection Spark Ignition). We do this choice because a decrease of cold start excess emission can come either from a decrease of cold start excess emission along the same cold start distance, or from only a decrease of the cold distance. It could be easily shown that if we have a rate α for a near future vehicle which is applied to both cold start distance and cold start excess emission, the total decrease of the cold start excess emission would be α^2.

The different cold start Artemis models

The final Artemis model is provided for different users which have not the same information to compute emissions. So we propose three different models depending on the available information that the user could have.

The first model gives an excess emission per start (i.e. per trip) in mass unit for a vehicle type i and a given pollutant p as a function of the ambient temperature T, the mean speed V during the cold period, the travelled distance d and the parking time t. It is the equation (1) but with EE, ω, f, h, δ and g depending on vehicle category i and pollutant p (regulated or unregulated one).

$h(i,p,\delta(i,p,T,V,\delta,t))$ and $g(i,p,t)$ are not available for the unregulated hydrocarbons (URHC). For theses components, the functions h and g used are the specific ones for the total hydrocarbons (THC):

$$h(i,URHC,\delta(i,URHC,T,V,\delta,t))= h(i,THC,\delta(i,THC,T,V,\delta,t)) \text{ and } g(i,URHC,t)= g(i,THC,t) \qquad (3)$$

The second model is the full one for the excess emission of a given traffic. It should be usually used to compute emission inventories for the whole traffic characterised by a number of parameters such as vehicle flow, average hot speed and environmental conditions (hour, temperature…).

The first model, initially applied to a single trip, must be extended to the whole traffic by using the available statistical data relative to traffic parameters. The excess emission of a traffic due to cold starts is therefore the product of the unit excess emission for a trip EE (first model), by the number of trips cold starting N_{tcs}:

$$E_c = N_{tcs}.EE$$

N_{tcs} is expressed globally as the ratio of the total distance started with cold start $L_{cold}Total$ by the mean distance of the trips started in cold conditions $L_{cold}Mean$:

$$N_{tcs} = L_{cold}Total / L_{cold}Mean$$

$L_{cold}Total$ is the product of the traffic flow tf_i expressed in veh.km by the percentage of mileage cm(s,i) started at cold start. This last parameter depends of the season s and the mean trip speed v_i.

$$L_{cold}Total = tf_i.cm(s,v_i)$$

If we consider only the cold started trips of length d_m, their number is expressed as:

$$N_{tcs}(dm) = L_{cold}Total.p_m/d_m$$

Where p_m is the share of total distance started with a cold start corresponding to trips of length d_m. In the same way, if we consider the cold started trips with an average speed v_i, these trips correspond to a cold distance of average cold speed v_j:

$$N_{tcs}(d_m,v_i) = \sum_j L_{cold}Total.p_{m,j}.p_{i,j}/d_m$$

Where $p_{i,j}$ is the distribution (%) of the cold started distance with an average trip speed v_i among the different speeds v_j during the cold distance. At the same time p_m has to be related to the speed v_j and expressed as $p_{m,j}$. In the same way, if we consider a stop or a parking time t_n, it corresponds to a distance share p_n:

$$N_{tcs}(d_m,v_i,t_n) = \sum_j L_{cold}Total.p_{m,j}.p_{i,j}.p_n/d_m$$

As we must take into account all the cold started trip length d_m and all parking time duration t_n, the number of trips cold starting is therefore the summations over m and n of the above expression. In addition we would like to take into account the influence of the hour (in the day) on the start number and on the parking time. Therefore the traffic flow tf_i and the parking time share p_n are functions of the hour and are transformed into $tf_{i,h}$ and $p_{n,h}$. At the same time, the cold starts must be distributed along the day by introducing the relative number of cold starts $ptf_{i,h}$ of the hour h (relative to the average hourly cold start number) p_h. Moreover, all the distributions depend hardly on the season s. So the equation of the model 2 could be expressed as:

$$E_c(p)=\sum_i \frac{cm(s,v_i)}{100}\cdot \omega_i(p)\cdot\left(\sum_h tf_{i,h}\cdot\frac{p_h(s)}{ptf_{i,h}}\cdot\left\{\sum_n \frac{p_{n,h}(s)}{100}\cdot\left[\sum_j \frac{p_{i,j}(s)}{100}\cdot\right.\right.\right.$$

$$\left.\left.\left.\left(\sum_m \frac{p_{m,j}(s)}{100\cdot d_m}\cdot f\left(p,v_j,T\right)\cdot h\left(p,\delta\left(p,T,v_j,d_m\right)\right)\cdot g\left(p,t_n\right)\right)\right]\right\}\right) \tag{4}$$

This equation gives an excess emission of a traffic in g as a function of the traffic flow, the season, the average speed, the ambient temperature and the hour in the day. It is a very complex formulae which cannot be simplified. It is not composed as a product of distributions, but as sums fitted into each others. So no distribution extraction is possible. Among all the parameters of this equation, we can distinguish three types:

- Some ones are purely internal and should not be modified by the user: $\omega_i(p)$, $f(p,v_j,T)$, $\delta(p,v_j,T,d_m)$ and $g(p,t_n)$
- Some ones are input parameters: i, s, v_i, h, $tf_{i,h}$, $ptf_{i,h}$ and T
- Some ones are internal parameters but could be modified by an advanced user: $cm(s,v_i)$, p_h, $p_{i,j}$, $p_{m,j}$, $p_{h,n}$, d_m and v_j. The default values of these distributions used in model 3 come from the Modem-Hyzem campaign (André et ali, 1994).

According to Duboudin and Crozat (2002), the taking into account of the average speed in the above equation is problematic, because the difference between the average speed during the cold period and the average speed during the whole trip. A trip with an average trip speed v_i is subdivided into a cold and a hot phase. The cold one can have an average speed v_j different from the global speed v_i. To calculate the global emission, we add a hot emission calculated with v_i and a cold excess emission calculated with v_j:

$$E_{total}(trip) = EE_{cold}(v_j)+E_{hot}(v_i)$$

It is not really coherent: if the distance travelled during the cold phase d_c corresponds to an average speed v_j different from the speed of the whole traffic v_i, the travelled distance in hot conditions cannot have an average speed v_i, and the global emission should be calculated with the formulae:

$$E_{total}(d_c+d_{hot}) = EE_{cold}(v_j,d_c) + E_{hot}(v_j,d_c) + E_{hot}(v_{hot},d_{hot})$$

where v_{hot} is the average speed of the hot distance d_{hot}. Therefore, when we calculate the traffic emission, we should use the equation of the second model, but add $(E_{hot}(v_j,d_c)-E_{hot}(v_i,d_c))$. As the difference should be quite small, we do not apply this correction.

The third model is in fact the second model with default values. We developed such model because a lot of statistical data are not really common. The using of this model allows us to take into account the distribution of the cold starts along the day. But the calculation of the third model needs a specific assumption on the relative traffic distribution along the day ($ptf_{i,h}$): we used a base distribution which is the average of 2 European traffic distribution (Belgium and Switzerland).

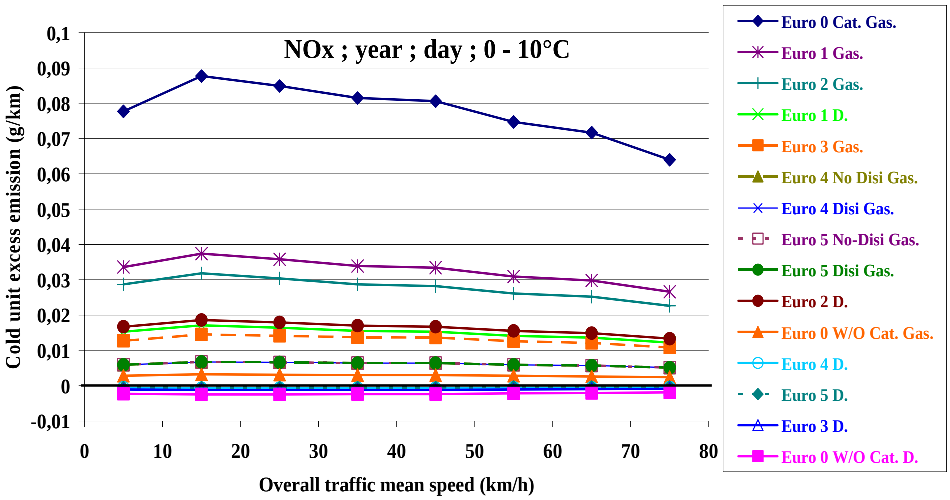
But when applying this model, if the actual traffic distribution is very different from the base distribution, the overall emission calculated during the day can be wrong. For instance for average traffic distributions representative of Belgium and Switzerland, the using of the third model introduces an error for the whole day between 3 and 7 %. In this case, we recommend not to use the third model hour per hour, but:

- Either to use the second model: the calculation will be very precise, with a detailed distribution of the cold excess emission along the day and an accurate summation over the day,
- Or to use the third model for the whole day (hour = whole day): the summation over the day of the hourly cold excess emissions will be accurate, but its distribution among the hours will not be accurate.

This model gives for a given vehicle type and an atmospheric pollutant an excess unit emission of a traffic in g/km, according to the season s, the ambient temperature T, the average speed v_i and the hour h in the day. It is a combined table for 4 seasons (winter, summer, intermediate, whole year), 8 speed classes (5 to 75 km/h), 7 temperature classes (-25°C to 35°C) and 25 hours (24 hours and the whole day).

Figure 1: NOx cold start unit excess emission vs. speed for the different vehicle categories.

Figure 1 : Surémission unitaire à froid du NOx versus la vitesse pour les différentes catégories de véhicule.



The figure 1 shows the influence of the vehicle category on the cold start excess emission for NOx pollutant as a function of speed. We could see that increasing the speed from 15 to 75 km/h decreases the excess emission by 30 %. It corresponds to a increasing of ambient temperature of 10 °C (figure 2). We could see also negative excess emission on figure 1. It corresponds to measured cold emission lower than hot emission. The season influence is very small, but could be up to 30

% at very low speed. The figure 3 shows that hour has a biggest influence. The shape of this figure comes from the shape of the distribution of the parking time t_n at the hour h. The maximum excess emissions at 8 am and 17 pm are the results of the long parking time during the night and the work day. The excess emissions along the other hours in the day come from short time parking and so to almost hot engine. The influence of all these parameters depends on the pollutant considered.

Figure 2: NOx cold start unit excess emission vs. temperature for the different speed classes.

Figure 2 : Surémission unitaire à froid du NOx versus la température pour les différentes classes de vitesses.

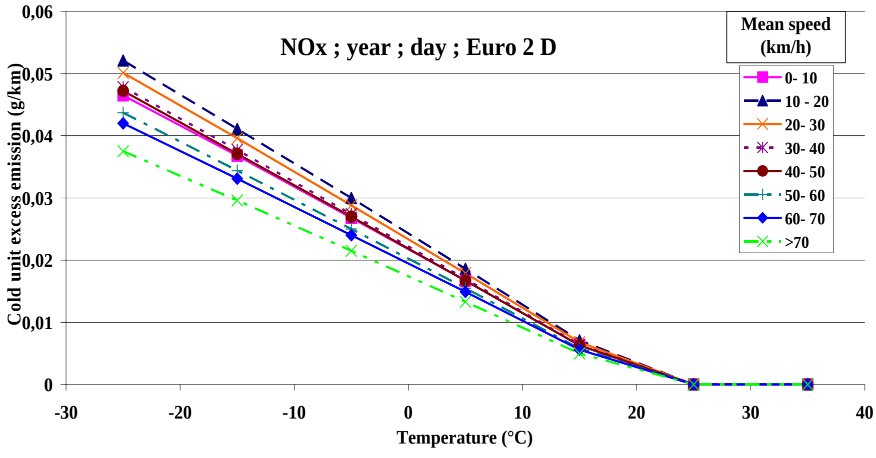
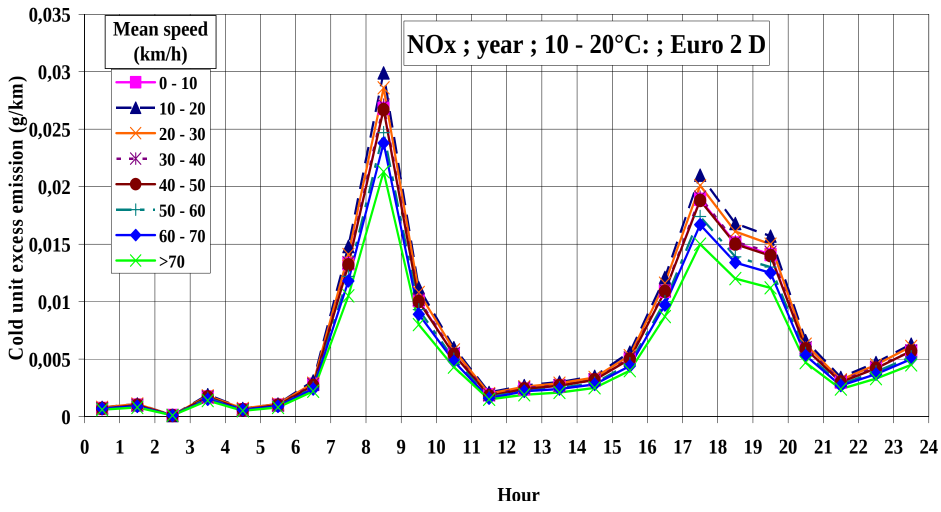


Figure 3: NOx cold start unit excess emission vs. hour in the day for the different speed classes.

Figure 3 : Surémission unitaire à froid du NOx en fonction de l'heure de la journée pour les différentes classes de vitesses.



Conclusion

This modelling of excess emission under cold start conditions for passenger cars results from data provided by various European research organisations as part of the MEET and Artemis projects. The models take into account the average speed, ambient temperature and travelled distance, among other parameters. They are based on measurements made over 5 driving cycles at different ambient temperatures. The average speeds of these cycles range from 17.5 km/h to 41.5 km/h, and the temperatures range from -20 °C to 28 °C. The modelling counts in fact three models.

The models can be applied at different geographic scales: at a macroscopic scale (national inventories) using road traffic indicators and temperature statistics, or at a microscopic scale for a vehicle and a trip. If a model user could not access to necessary statistics, it would be recommended to use the most aggregated model, i.e. the third one, which is parallel to the hot emission modelling, with the same shape.

All these models are improved versions of the former MEET model. The third one should replace the Copert III cold start model within the new inventorying tool Artemis. Theses models are presented for petrol and diesel vehicles, the present and the near future ones.

This study corresponds to the state-of-the-art at the present time. In the future, this model could be improved by different ways:

- By updating this model using new data when available, either for the most recent passenger cars, or the light duty vehicles, or the heavy duty vehicles
- It would be much more precise to have crossed distributions for different speeds and ambient temperatures
- The number of analysed data has to be increased, especially for different speeds, low or high temperatures, and for unregulated pollutants.

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Cold start extra emissions as a function of engine stop time: The evolution during the last 10 years

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Abstract

Cars with catalysts show a significant increase of exhaust emissions at engine start. These extra emissions are expressed as the difference, over a particular driving cycle, between emissions generated when the vehicle is started and when the engine is stably warm. Experimental data, suitable for the assessment of cold start emissions, are usually available for completely cooled out engines. Most results originate from tests at ambient temperature of 23°C and an engine stop time of at least 12 hours. On the other hand, data including shorter stop times are very rare.

This present work investigates on the influence of exhaust emissions at shorter stop times, i.e. 0.5, 1, 2 and 4 hours. The main goal consists in the comparison of emissions exhausted by recent car models (Euro-4) against emissions assessed in the framework of a similar campaign 10 years ago (FAV1/Euro-1 vehicles).

In this paper a short survey of the current extra emission estimation methods is presented. It is shown that some methods are not suited to provide correct estimations in all cases. We discuss the fact that different estimation methods can show either similar or completely different results depending on the evolution behaviour of the hot emissions.

Due to new technologies, e.g. the catalyst and improved engine control algorithms, emissions have been considerably reduced over the last 10 years. In this study it is determined how the relative extra emissions expressed as a function of stop time have changed. We may claim with caution that for middle stop times of 0.5 to 4 hours the average relative extra emissions of Euro-4 vehicles are distinctly below the average extra emissions of Euro-1 vehicles.

Keys-words*: relative cold start extra (excess) emissions, stop (parking) time, long term evolution, extra emission estimation, gasoline passenger cars.*

Introduction

The analysis of additional emissions during the cold phase, called the cold start extra (or excess) emissions, has gradually gained in importance in order to improve emission models and thus emission inventories. An exhaustive survey of the researches carried out in the past concerning cold start emissions is given in the report by (André and Joumard, 2005). We can sum up the past research by stating that they focus on the characterisation of cold start emissions as a function of following 5 parameters: i) technology or emission standard (FAV1/Euro-1 … Euro-4), ii) average vehicle speed, iii) ambient temperature, iv) travelled distance and v) engine stop time (also called parking time). Many data and research results are available for the first 4 parameters, while suitable data for stop time analysis are very rare since for such investigations the required measurement procedures are time and cost consuming. Former stop time investigation results have been presented in (Hammarström, 2002), in (Schweizer et ali, 1997), in (Hausberger, 1997) and in (Sabate, 1996). In the present work we focus on stop times of 0, 0.5, 1, 2, 4 and >12 hours by using 15 repetitive IUFC (Inrets urbain fluide court, i.e. short free-flow urban) cycles.

In this paper a short survey of the current extra emissions estimation methods is presented. We show that some methods are not suited to provide correct estimations in all cases. It turns out that we have to distinguish between different hot phase emission classes which have to be considered for the choice of the estimation method. We discuss the fact that different estimation methods can show either similar or completely different results depending on the hot phase emission class.

The main goal of this work is to describe the relative cold start extra emissions as a function stop times. We assume therefore, as in (André and Joumard, 2005), that a stop time of >12 hours is sufficient to cool down the engine and the catalyst to an ambient temperature of 23 °C. Thus, the cycle after a stop time >12 hours is considered as the fully cold started cycle. The emissions of cycles with smaller stop times are normalized by dividing by the emissions of the fully cold started cycle. This results in a relative extra emission as function of stop time. We present a comparison of relative extra emissions of a recent campaign with Euro-4 gasoline passenger vehicles against relative extra emissions assessed in the framework of a similar campaign 10 years ago with FAV1/Euro-1 gasoline passenger vehicles (Schweizer et ali, 1997). Moreover, we compare the relative extra emissions of both campaigns with the INRETS model proposed by (André and Joumard, 2005).

Preliminaries

1. Description of the considered Euro-1 and Euro-4 campaign vehicles and cycles

- Euro-1 campaign (in fact it was a FAV1 campaign, but since the Swiss legislation FAV1 corresponds to the Euro-1 legislation we denote it as Euro-1 campaign)
    - The considered gasoline vehicles are listed in Table 1. All 4 cars are equipped with a three-way catalyst.

- The considered cycle is the T50 cycle which has a length of 6.5 km, a duration of 830 s and an average velocity of 28 km/h. This cycle is a city-centre cycle which is not repetitive.
- Ambient temperature: 25 °C
- Ambient relative atmospheric humidity: 50 %

Remarks: According to the report of (André and Joumard, 2005), the T50 cycle could be too short to catch the whole cold phase emissions. This statement seems to contradict results of the same report: for Euro-1 vehicles with an ambient temperature of 25 °C and an average velocity of 28 km/h cold distances (i.e. the distance of the cold phase) of 5.11 km for CO, 6.63 km for HC, 3.87 km for NOx and 1.83 km for CO_2 are given. Thus, these suggested cold distances are below the cycle length of 6.5 km, except for HC for which it is slightly above the cycle length. Of course these are averaged cold distances and some cars could evidently show longer cold distances. But there is another fact that validates the use of T50 cycles for cold start emission estimations. The results of our Euro-4 campaign show that almost all cold distances are about 1 km with an absolute cold distance maximum of 3 km (see Figure 3, where the cold distances are about one IUFC cycle long). On the other hand, the averaged cold distances for Euro-4 vehicles suggested in (André and Joumard, 2005) are clearly above 1 km, i.e. with similar values as for the Euro-1 vehicles. This leads to the conclusion that the T50 cycle seems to be appropriated to catch to whole cold phase emissions.

- Euro-4 campaign
  - The considered gasoline vehicles are listed in Table 2.
  - The considered cycle is the IFCU15 cycle which is a repetition of 15 IUFC cycles. One IUFC cycle has a length of 0.999 km, a duration of 189 s and an average velocity of 19 km/h. In what follows the IUFC cycle is referred to as the subcycle of the IUFC15 cycle.
  - Ambient temperature: 23 °C
  - Ambient relative atmospheric humidity: 50 %

Table 1: The 4 gasoline vehicles which are considered for the Euro-1 campaign

vehicle No.	make	model	eng. capacity cm^3	power kW	mileage km	gearbox type
1-1	BMW	325i	2493	125	133264	manual 5
2-1	BMW	320i	1989	95	154662	autom. 4
3-1	Opel	Vectra GT	1997	85	65912	manual 5
4-1	VW	Golf CL	1595	51	31134	manual 5

Table 2: The 6 gasoline vehicles which are considered for the Euro-4 campaign

vehicle No.	make	model	eng. capacity cm^3	power kW	mileage km	gearbox type
1-4	Opel	Agila 1.2	1199	55	46492	manual 5
2-4	Fiat	Stilo 1.6	1596	76	77801	manual 5
3-4	VW	Sharan 1.8	1781	110	101065	manual 6
4-4	Toyota	Avensis 2.0	1998	108	37700	autom. 4
5-4	Ford	Mondeo 2.5	2495	125	42412	manual 5
6-4	Audi	A4 3.0	2976	162	49938	ECVT

1. Cold start extra emissions estimation methods

For this work we consider following 4 cold start extra emissions estimation methods.

1.1 Subcycle analysis method

To apply this method a repetitive cycle, as e.g. the IUFC15, is required. We have to separate the cycle into a cold phase and a hot phase. Thus, we first have to determine which subcycles belong to the cold phase. This separation can be effectuated graphically “by hand” by analysing the total subcycle emissions as a function of the chronological succeeding subcycles which corresponds to a discretized time line. Figure 1 illustrates a cycle with 15 subcycles, as e.g. the IUFC15 cycle. In this case the 8^{th} subcycle can be considered as the last subcycle of the cold phase referred to as n_c. A method developed at INRETS (see (André and Joumard, 2005)), the so-called standard deviation method, permits to compute n_c. During this present work it turned out that his method does not work in several cases. This is probably due to the fact that the cold start extra emissions are not emphasized enough after short stop times to permit an accurate detection. We therefore had to develop a more robust method to determine n_c. Once n_c is known the cold start extra emission is given by

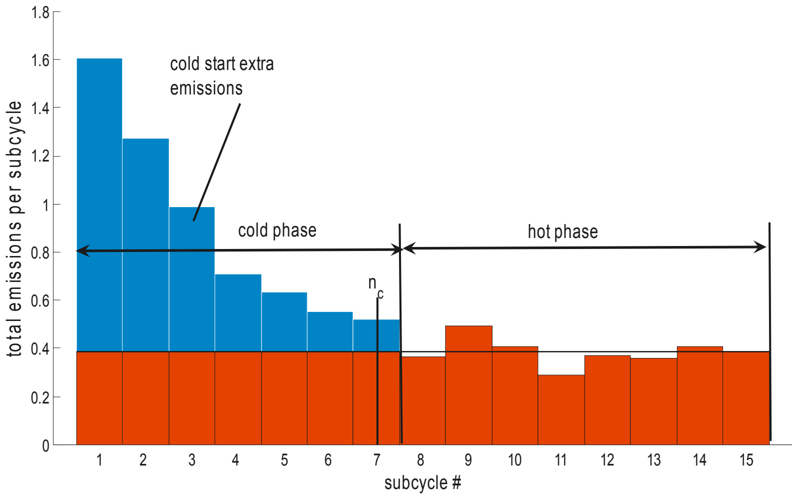
$$EE_{cold} = E_{cyc} - E_{hot} = \sum_{i=1}^{15} E(i) - \frac{15}{15-n_c} \sum_{i=n_c+1}^{15} E(i)\,,$$

where $E(i)$ is the total emission of subcycle i, E_{cyc} is the total emission of the cycle and E_{hot} is the sum of the hot emission part of the cycle.

1.2. Subcycle analysis with linear regression method

The second subcycle method consists of determining a linear regression of the cumulative emissions during the hot phase. A detailed discussion of this method is provided in (Weilenmann, 2001) and in (André and Joumard, 2005). As for the first method it requires to determine n_c as described above.

Figure 1: Emissions evolution as a function of subcycle. Separation of the cycle into a cold and a hot phase.



1.3.Bag analysis method

Some laboratories measure emission factors by means of bags only. At EMPA a complete cycle is subdivided into 3 bag measurements. In the case of an IUFC15 the total emission of subcycles 1-5, 6-10 and 11-15 correspond to the bags 1, 2 and 3, respectively. By assuming that the cold phase ends before the start of the third bag (first 10 subcycles) the emission is given by

1.4.Cycle analysis method

If only the total emission of a whole cycle, referred to as E_{cyc}, is available, we additionally need to measure the emissions of a fully hot cycle, referred to as E_{hotcyc}. A fully hot cycle starts with a hot engine and a catalyst above light-off temperature. A cycle after zero stop time can typically be considered as a fully hot cycle. The cold start extra emission is given by

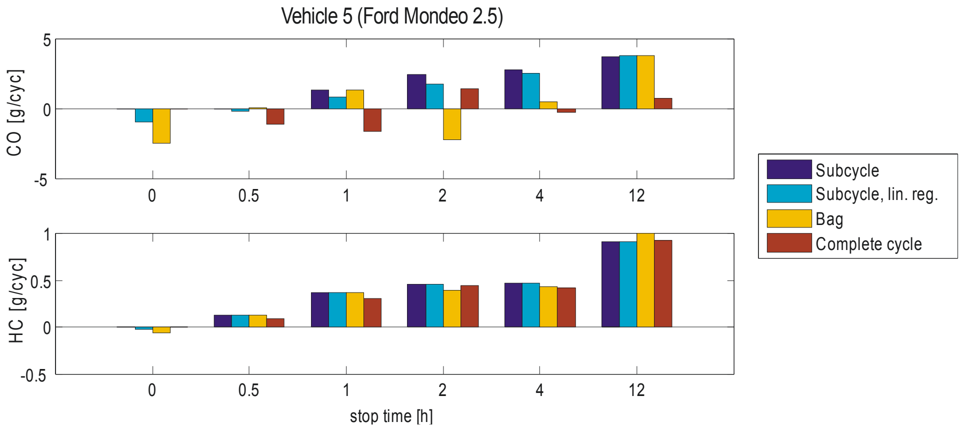
$$EE_{cold} = E_{cyc} - E_{hotcyc} .$$

This is the sole method which can be applied to non-repetitive cycles like the T50 cycle.

Comparison of the different extra emission estimation methods

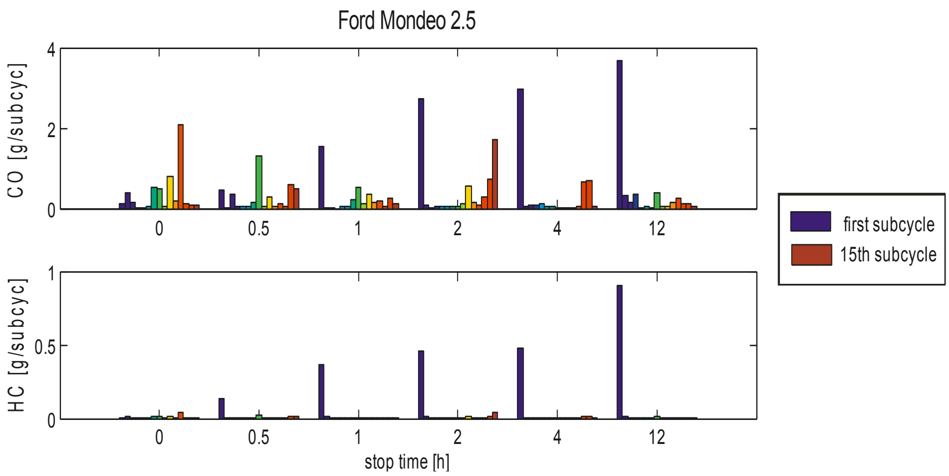
Figure 2 illustrates the estimated extra emissions of vehicle 5-4 (Ford Mondeo). The extra emissions are expressed as a function of stop time for the 4 different estimations methods. The extra emission of HC shows similar values for all methods. While for CO and NOx only the subcycle methods provide similar extra emissions which qualitatively correspond to the evolution that one can expect. On the other hand, the bag and cycle methods provide strongly varying and oscillating extra emission evolutions.

Figure 2: Extra emission as a function of stop time estimated with 4 different methods



These strange results can be explained by analysing the evolution of the subcycle emissions. Figure 3 depicts the total subcycle emissions as a function of the chronological succeeding subcycles for each stop time. With the focus on the CO emissions solely the first subcycle can be assigned to the total cold start extra emissions. The cycles corresponding to stop times of 0 and 2 hours show a considerable excess of emissions in the last part of the their hot phases, i.e. subcycles 10-15 (third bag). This excess leads therefore to negative emission estimations for the bag analysis method. A similar observation can be established for the cycle analysis method: Due to the excess emission during the last 5 subcycles of the zero stop time cycle (fully hot cycle) the hot phase emission E_{hotcyc} is overestimated compared to the hot phase emissions of the other cycles. This inevitably leads to an underestimation of the extra emissions particularly for the cycles corresponding to stop times of 2 and 4 hours.

Figure 3: Total subcycle emissions as a function of chronological succeeding subcycles for each stop time.



Both subcycle analysis methods are in general more robust, especially for cycles (as the considered zero stop time cycle) for which no cold phase can be determined, i.e. n_c = 0. This robustness results from the fact that the hot phase emissions are averaged over several subcycles. In our considered case the cold phases end in subcycle one, and thus the hot phase emissions are averaged over 14 subcycles providing representative hot phase emissions. We conclude that the estimation accuracy can be improved by increasing the hot phase length: either by appending supplementary subcycles or by measuring a cycle with the same stop time several times.

Similar considerations hold for the NOx emissions. On the other hand, the hot phase emissions of HC show less variation (Figure 3). Thus, the 4 methods provide similar extra emissions estimations (Figure 2).

By analysing the 6 vehicles of the recent Euro-4 campaign we can distinguish between 3 different hot phase classes:

1.Homogeneous distributed hot phase emissions:

For this class the hot phase emissions evolve almost flatly with only small variations. Thus, all 4 extra emission estimation methods provide meaningful results. Vehicles 1-4, 3-4 and 6-4 belong to this class regarding the pollutants CO, HC and NOx. Moreover, the HC pollutant for all vehicles belongs to this class too.

2.Heterogeneous distributed hot phase emissions

For this class the hot phase emissions oscillate and vary greatly. Thus, only the two subcycle analysis methods are accurate enough. The CO and NOx pollutant of vehicles 2-4 and 5-4 belong to this class.

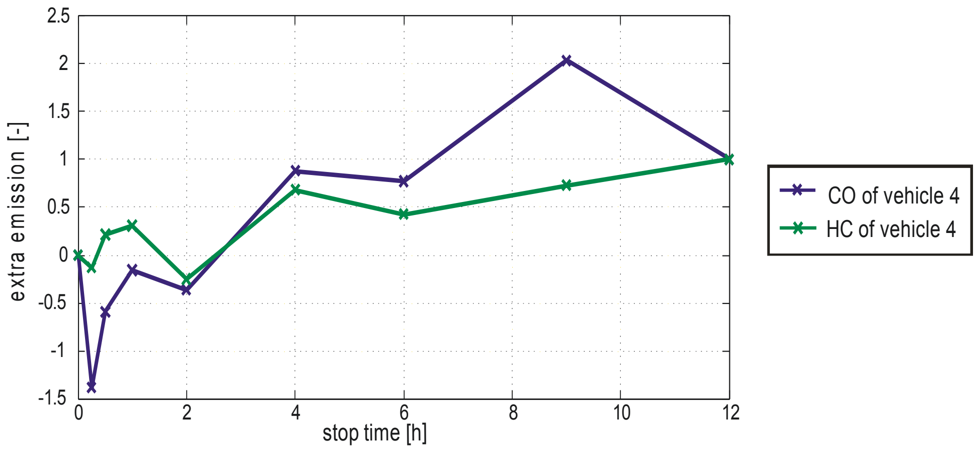
3.Non-determinable hot phase

The CO and NOx pollutants of vehicle 4-4 show emission evolutions with following characteristics: i) extreme subcycle emission variations, ii) huge variations of averaged hot phase emissions between cycles and iii) for unusual many hot phase subcycles the emissions are larger than for cold phase subcycles. The analysis of the air/fuel ratio of vehicle 4-4 reveals that different control strategies occur during the hot phases. Occasional air/full ratio drop peaks ($\Delta\lambda$ = -0.15...-0.2) and increase peaks ($\Delta\lambda$ = +0.01…+0.03) lead to a considerable augmentation of CO and NOx emissions respectively. The rule of these control strategies could not be established so far. Since the evolutions of the emissions show quasi-chaotic behaviours we conclude that for this type of vehicles a meaningful estimation derived from any proposed method can not be expected.

Comparison of relative cold start extra emissions

According to the last section the estimated cold start extra emissions can be very erroneous depending on the hot phase class and on the applied estimation method. Thus, it has first to be determined which samples provide too erroneous estimations in order to exclude these data for the comparison. For the Euro-4 campaign we obviously use the subcycle analysis method since it provides the most accurate estimations. Thus, we only have to exclude the data of vehicle 4-4 since it belongs to the 'non-determinable hot phase' class. For the Euro-1 campaign only the cycle analysis method is applicable since cycle T50 is non-repetitive. The analysis of the normalised extra emission evolution as a function of stop time reveals that two samples, i.e. HC and CO of vehicle 4-1, have to be excluded. As illustrated in Figure 4, these two samples oscillate with large amplitude variations and reach for several stop times negative values.

Figure 4: Evolution of the normalised extra emission of the two Euro-1 samples excluded for the comparison.



In general, as proposed in (André and Joumard, 2005), the cold start extra emission can be expressed as a function of temperature *T*, averaged velocity *V*, travelled distance *δ* and the stop time *t*, i.e.

$$EE(T,V,\delta,t) = E_0 \cdot f(T,V) \cdot h(\delta) \cdot g(t),$$

where E_0 is the reference extra emission at T = 20 °C, V = 20 km/h, $\delta = d_c$ and t = 12 h. The function $f(T,V)$ expresses the influence of the temperature and averaged velocity, while $h(\delta)$ is the distance influence function and $g(t)$ is the stop time influence function. The extra emission function is normalized by dividing by the extra emission of the fully cold started cycle (stop time of t = 12 h). Thus, since for the considered campaigns the temperature, the velocity and the travelled distance are constant and since $g(12) = 1$, we obtain a relative extra emission function as a function of stop time which is equal to

$$EE_{rel}(t) = \frac{EE(T,V,\delta,t)}{EE(T,V,\delta,12)} = \frac{E_0 \cdot f(T,V) \cdot h(\delta) \cdot g(t)}{E_0 \cdot f(T,V) \cdot h(\delta) \cdot g(12)} = g(t).$$

In what follows we compare the relative extra emissions $EE_{rel}(t)$ of both campaigns and the INRETS (André and Joumard, 2005) model against each other. Notice that the INRETS model consists of polynomials as a function of stop time.

- **CO** (Figure 5): The standard deviations point out that the CO mean variations of Euro-1 vehicles are much larger as for Euro-4 vehicles. The model fits the Euro-1 data well, except at *t* = 1 h. This result is not astonishing since the model is based on older vehicle data. For stop times longer than 1 h the relative extra emissions of the Euro-4 vehicles are distinctly below the modelled and averaged Euro-1 extra emissions.
- **HC** (Figure 6): In contrast to CO, the HC mean variations are much large for the Euro-4 vehicles. Surprisingly, the model fits the Euro-4 vehicles much better than the Euro-1 vehicles. The HC relative extra emissions of the Euro-4 vehicles are distinctly below the Euro-1 emissions in the stop time interval of 0.5 … 4 h.
- **NOx** (Figure 7): In the stop time interval of 0.5…4 h the Euro-4 campaign fleet exhausts lower NOx relative extra emissions as the Euro-1 fleet. But since the standard deviations are large for both campaigns this establishment should be taken with caution. By taking into consideration the large standard deviations the model can be qualified as appropriate for representing both, Euro-1 and Euro-4 vehicles.
- **CO_2**: Concerning the Euro-4 vehicles, potential relative extra emissions can only be detected for 12h stop time cycles. Thus, no particular statements concerning the relative extra emissions for the relevant stop times (≤ 4 hours) can be deduced, apart the fact that they seem to be approximately zero.

Figure 5: CO extra emission comparison of the Euro-1, Euro-4 campaigns and the Inrets model.

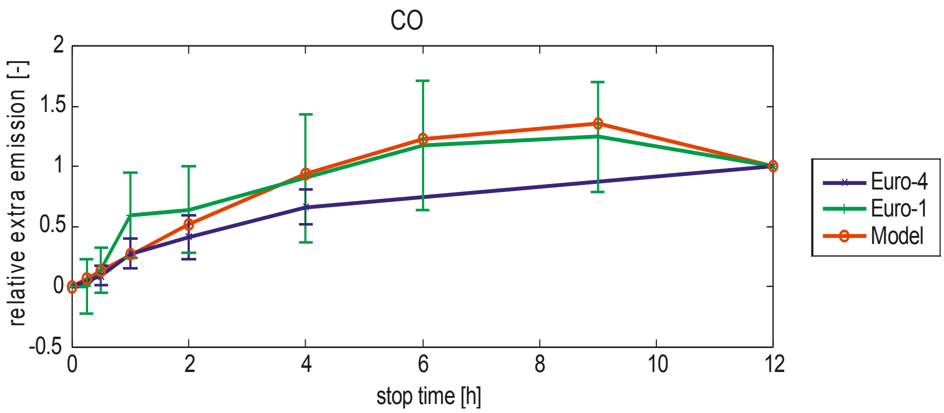


Figure 6: HC extra emission comparison of the Euro-1, Euro-4 campaigns and the Inrets model.

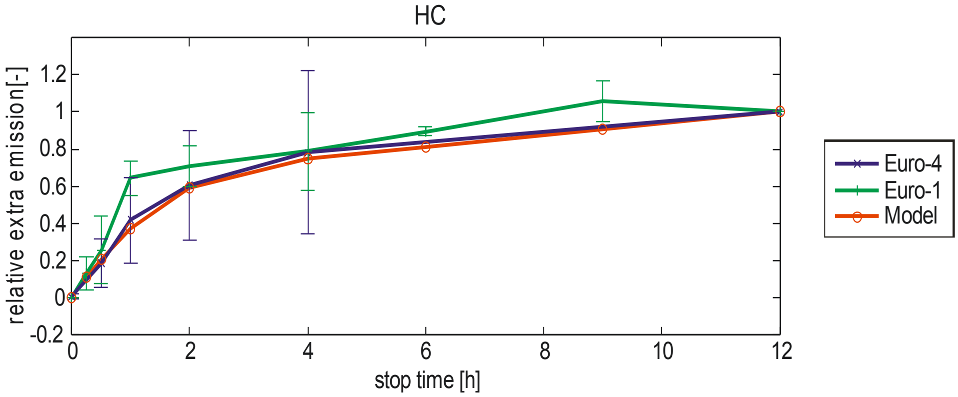
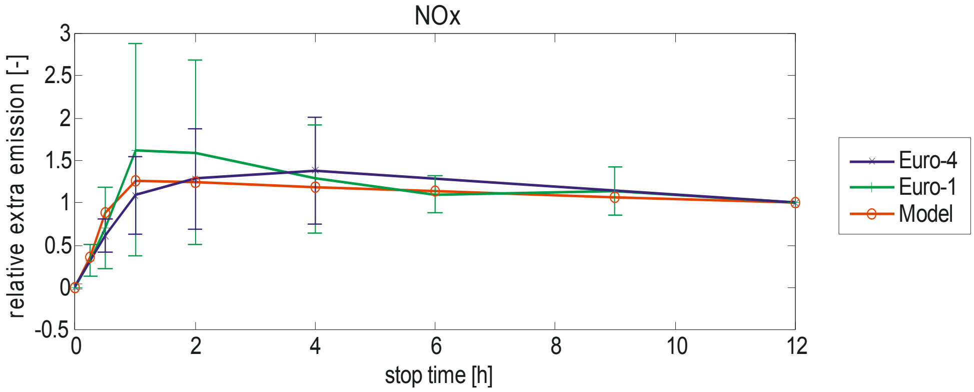


Figure 7: NOx extra emission comparison of the Euro-1, Euro-4 campaigns and the Inrets model.



Discussion and conclusion

The comparison of the different extra emission estimation methods clearly points out that the bag and cycle methods often fail when the hot phase emissions varies greatly. Thus, in general, one of the two distinctly more robust subcycle methods should be applied by preference in order to ensure meaningful estimation results. Nevertheless, there are vehicles for which the extra emissions cannot be estimated with any proposed methods. In such cases an improvement can probably only be achieved by considerably increasing the number of tests in order to obtain more accurate averaged hot phase emission estimations. Since the estimation methods may fail it is important to examine the resulting estimations in order to exclude senseless data for interpretation and modelling purposes.

Currently we are not in the position to explain the large emission variations during the hot phase. Therefore, we consider that the next steps should be the determination and the characterisation of the causes of this phenomenon.

For the relative extra emission comparison it is difficult to state tendencies with considerable certainty since only few vehicles are considered, i.e. 3 vehicles for Euro-1 and 5 for Euro-4. Nevertheless, we may claim with caution that for middle stop times of 0.5 to 4 hours the averaged relative cold start extra emissions of recent vehicles (Euro-4) are distinctly below the averaged extra emissions of 10 years old vehicles. This new result may prove useful for expanding the INRETS and other models with stop time influences of Euro-4 vehicles.

In this paper we incidentally mentioned that the estimated cold distance of the 6 examined Euro-4 vehicles are about 1 km, while in (André and Joumard, 2005) the cold distances evaluated from data of 14 Euro-4 vehicles are given in the range of 2.5 to 6 km. Thus, this large discrepancy has to be investigated in order to be able to provide a consistent model.

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Accuracy of exhaust emissions measurements on vehicle bench

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Abstract

Ten European laboratories worked together to study the influence of a lot of parameters of the measurement of light vehicle emission factors on vehicle bench, in order to improve the accuracy, reliability and representativeness of emission factors: driving patterns (driving cycles, gear choice behaviour, driver and cycle following), vehicle related parameters (technical characteristics of the vehicle, emission stability, emission degradation, fuel properties, vehicle cooling and preconditioning), vehicle sampling (method, sample size), and laboratory related parameters (ambient temperature and humidity, dynamometer setting, dilution ratio, heated line sampling temperature, PM filter preconditioning, response time, dilution air). The results are based on literature synthesis, on about 2700 specific tests with 183 vehicles and on the reprocessing of more than 900 tests. These tests concern the regulated atmospheric pollutants and pre-Euro to Euro 4 vehicles. We did not find any influence of 7 parameters, and find only a qualitative influence for 7 other parameters. 6 parameters have a clear and quantifiable influence and 5 among them allow us to design correction factors to normalise emission measurements: gearshift strategy, vehicle mileage, ambient temperature and humidity, dilution ratio. The sixth influencing parameter is the driving cycle, sometimes more significant than *the fuel or the emission standard. The results allow us to design recommendations or guidelines for the emission factor measurement method.*

***Keys-words**: emission factor, light vehicle, model, inventory, regulated pollutant, guidelines, measurement conditions, method.*

Introduction

Calculation of emissions has therefore gained institutional importance in the European Community, particularly with the development of the CAFÉ (EC, 2005a) and ECCP (EC, 2005b) programmes. Reliable and credible emission estimates are a central prerequisite, but comparisons of the results from emission models such as COPERT (Ntziachristos & Samaras, 2000), FOREMOVE (Samaras et ali, 1993), TREMOVE (De Ceuster et ali, 2005), RAINS (Amann et ali, 2004), Handbook (Keller, 2004) and national models have shown substantial differences. This causes doubts about the credibility of the underlying data and methodologies and might mislead the political discussions.

The European MEET (Methodologies for Estimating air pollutant Emissions from Transport) project (Hickman et ali, 1999), the COST 319 action (Joumard, 1999) and other research programmes raised a main question in relation to passenger car emissions, summarised as follows: large differences in measured emission levels occurred between the different laboratories in Europe; these differences appeared to be more pronounced for more recent (at this time) vehicle technologies (i.e. Euro 1), irrespectively of the way the emissions modelling is conducted (i.e. average speed dependency approach, traffic situation approach).

In order to be able to produce accurate emission factors for current and near-future technology, taking into consideration the aforementioned observations for modern car categories, a two-fold strategy is proposed in the present study: i) investigating and reducing the measurement differences between laboratories, ii) investigating, understanding and modelling the emission differences among comparable vehicles. The first aim is to study the sensitivity of pollutant emissions to the key parameters. The second aim is to develop methods that allow the harmonisation of any European emission measurements.

This study, detailed in Joumard et ali (2006) is a part of the Artemis project "Assessment and Reliability of Transport Emission Models and Inventory Systems", whose purpose is to arrive at a harmonised methodology and to develop a software that calculates emissions of any transport mode, at local, national and international level.

Methodology

The influence of all the potential parameters on the exhaust emission level and accuracy is studied first with a literature review and then by laboratory tests on vehicles. Four types of parameters of the measurement conditions are studied:

- Driving patterns: driving cycles, gear choice behaviour, influence of the driver and cycle following

- Vehicle related parameters: technical characteristics of the vehicle**s**, short term emission stability, long term emission degradation (mileage), fuel properties, vehicle cooling, vehicle preconditioning
- Vehicle sampling method: method of vehicle sampling, number of vehicles
- Laboratory related parameters: ambient temperature, ambient humidity, dynamometer setting, dilution ratio, heated line sampling temperature, PM filter preconditioning, response time, instantaneous vs. bag value, dilution air conditions

Parallel to the study of the impact of different parameters on emissions, we compare the roller test bench laboratories to each other by performing a round robin test with reference gases on the common fuels basis.

A specific test programme was built-up for each parameter studied, excepted the vehicle running conditions and the method of vehicle sampling, where only literature review or inquiries were performed. Emissions of CO, CO_2, HC, NOx, and PM are considered. Although a wide variety of driving cycles were tested for the whole study (65 cycles), most of them have been used either to look at the influence of the driving patterns, or when reprocessing existing data (case of the minimum size of a vehicle sample). For the influence of the vehicle and laboratory related parameters, the 3 Artemis driving cycles (André, 2004) have been generally tested with hot start, but in a few cases without the rural or motorway cycles. In many cases cold and/or hot NEDC have been tested in addition. All the tested driving cycles are described and analysed by (André et ali, 2006a; André and Rapone, 2006). The tests have been spred out in the different partner laboratories.

Globally 2753 tests are carried out (1 test = 1vehicle x 1 driving cycle), i.e. 537 tests to look at the influence of the driving patterns (48 vehicles), 1334 tests to look at the influence of the vehicle parameters (70 vehicles), 672 tests to look at the influence of the laboratory related parameters (64 vehicles), and 210 tests are part of the round robin test conducted within 9 laboratories with a petrol passenger car. In addition at least 910 tests (81 vehicles) from the European Artemis data base but not carried out within the project are processed in order to look at the influence of the driver and mainly the vehicle sample size.

Results

According to the outputs of the above studies and in the conditions of the tests, some parameters have no influence on the emission measurements, or have a qualitative, or a quantitative influence.

1. Not influencing parameters

We did not find any statistically significant influence on emission measurement for some parameters. It does not mean that these parameters have no influence on the emission measurements, but only that we cannot prove any influence, taking into account the small data sample or the contradictory results.

1.1 Vehicle related parameters

- Short term emission stability or driving cycle repetition
- Inspection-maintenance
- Fuel properties. The results confirm the influence of fuel on exhaust emissions, but in spite of observing significant differences, especially for PM emissions with diesel vehicle, it was not possible to propose an explanation based on the today knowledge of fuel effect
- Vehicle cooling. The open and close bonnet, the height of a small blower have no influence on the emissions measured. The cooling power, i.e. the flow of the cooling air, hasn't a clear influence on the measured emissions

1.2 Laboratory related parameters

- Heated line temperature, because the observed emission change contradicts what is expected from the physico-chemical properties of the diluted emissions
- PM filter conditions
- Dilution air condition

2. Parameters with qualitative influence

Some parameters have a qualitative influence. Therefore recommendations are made concerning these parameters:

2.1 Driving patterns

- The driver can be a human driver or a robot. Only the CO_2 emission was significantly higher by +4 % with human driver than with a robot driver, but the difference cannot be explained by the driving characteristics.

2.2 Vehicle related parameters

- The vehicle classification, through the type approval category (Euro 1 to 4) and the fuel, has a clear influence on the emissions, together with the engine capacity in some cases. But no correlation between emission behaviour and emission control technologies were found as long as the cars belong to the same type approval category. Therefore the additional introduction of technological characteristics won't improve the accuracy of emission data bases of conventional cars up to Euro 4.
- The vehicle preconditioning conditions have an influence in some cases, but very few for modern close loop vehicles. A 10 minute cycle at a constant speed of 80 km/h can be considered as the most suitable preconditioning cycle. It resulted in the lowest emission levels and the lowest standard deviation for the majority of the measurements.

2.3 Vehicle sampling method

- The sample characteristics influence the emission levels: the vehicle classes given above, but also the size and engine power at the maximum power of the vehicle, which influence a lot the CO_2 emission and fuel consumption.

- Minimum size of vehicle sample. Usually 10 to 15 vehicles are required for all the pollutants, in order to build-up an emission model which is representative of an average vehicle behaviour. Below these prescribed numbers, the weight of the individual behaviour of some vehicles is too significant to obtain a mean, which is representative of an average behaviour.

2.4 Laboratory related parameters

- The dynamometer setting has a clear influence on all emissions, but significantly only on CO_2 and fuel consumption, and on NOx for diesel vehicles. It cannot be excluded, however, that altered settings might affect these other pollutants too.
- Response time including instantaneous versus bag value. The measured instantaneous emission level must be corrected using specific functions, before building an instantaneous emission model (Zallinger et ali, 2005).

3. Influencing parameters

6 parameters have a clear and statistically significant influence on the emissions measured. The influence of 5 parameters can be quantified and quantitative correction factors are available in order to standardise emission measurements for the parameters gearshift strategy, vehicle mileage, ambient air temperature, ambient air humidity and exhaust gas dilution ratio.

3.1 Driving patterns

- Driving cycle. The variation induced by the driving type or cycle was more significant than the variation induced by the fuel type (for HC, CO_2), or by the emission standard (NOx, CO_2), or even between the vehicles (CO_2), with quite contrasted behaviour between diesel (rather sensitive to speed and stop parameters) and petrol cars (rather sensitive to accelerations). However, it was not possible to design a satisfying correction function, but an harmonisation approach was then developed, based on the similarities between cycles from a kinematic point of view (André and Rapone, 2006).
- Gearshift strategy. It is possible to classify the gearshift strategies according to their CO_2 emission (the only pollutant always influenced by the strategy). The most polluting strategy is the gear change at given engine speeds whatever the cycle. The less polluting strategy seems to be the gear change at given vehicle speeds (defined in the NEDC cycle). The ratio between these two strategies is around 15 %. For urban cycle, the strategy depending on the vehicle power-to-mass ratio and on the 3rd gear ratio (part of the Artemis cycles) pollutes as the gear change at given vehicle speeds. For rural cycle, the Artemis strategy pollutes less than given engine speed strategy (9 %) but more than the given vehicle speed strategy (6 %).

3.2 Vehicle related parameters

- The mileage has no influence on the CO_2 emission neither on the emissions of diesel vehicles, but increases a lot CO, HC and NOx emissions of petrol cars: between 0 and 100 000 km, these emissions increase by a factor 3.6 in average for Euro 1 and 2 vehicles, and by 15 % for Euro 3 and 4 vehicles (see an example Figure 1).

Figure 1: NO_x degradation in urban driving behaviour for petrol vehicles.

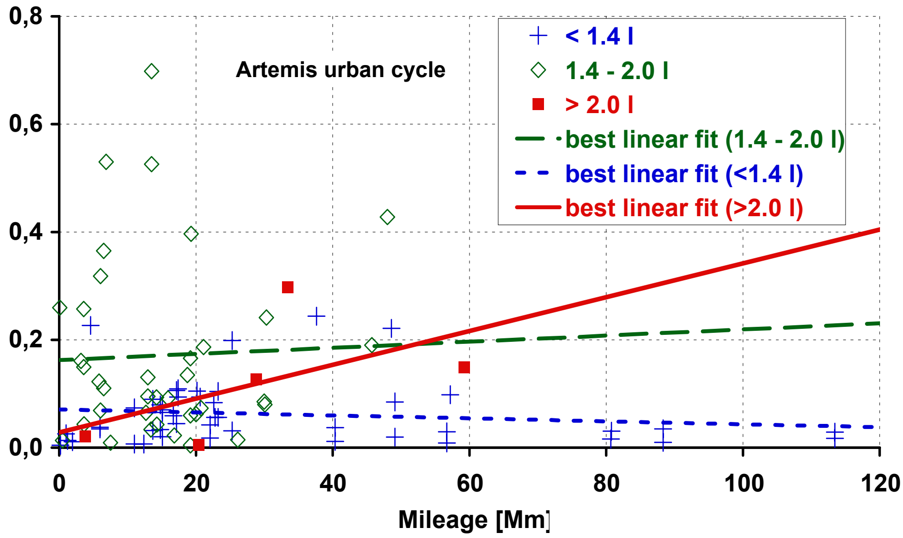
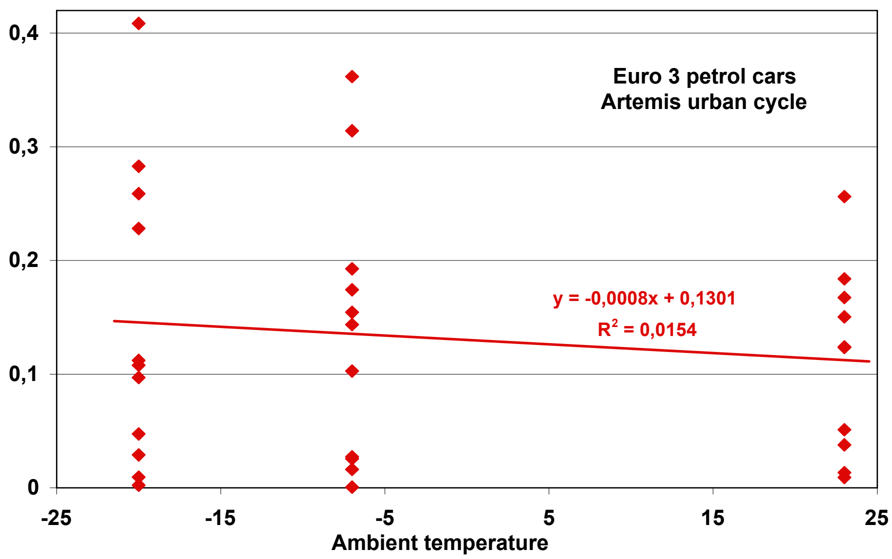


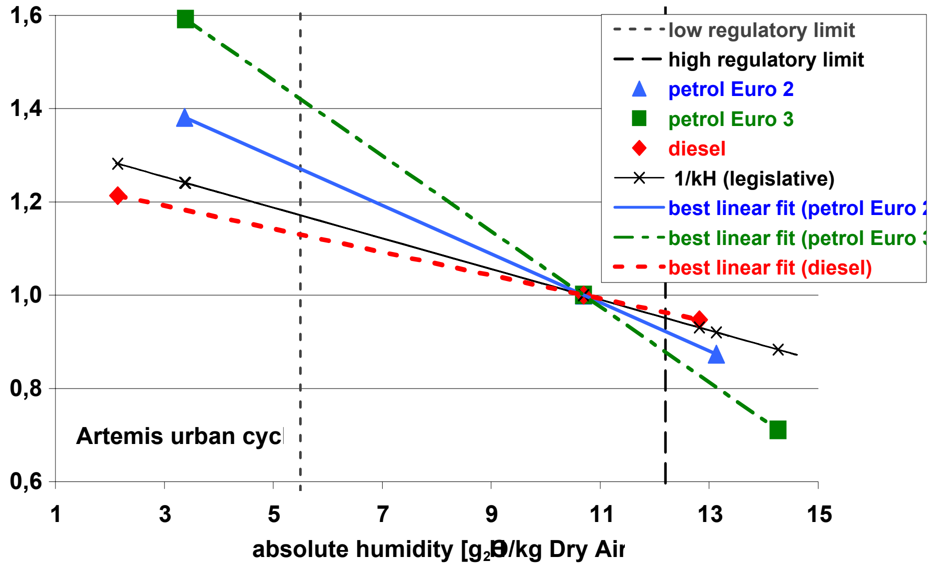
Figure 2: Influence of the ambient temperature [°C] on the NOx emissions [g/km] of Euro 3 petrol cars over the Artemis urban driving cycle.



3.3 Laboratory related parameters

- Ambient air temperature. The hot emissions decrease with increasing temperature for petrol cars but mainly for diesel ones. Between 10 and 20°C, the CO and HC emissions varies by 15-20 %, the NOx and CO_2 emissions by 2 %, and PM is constant. The influence of the ambient temperature is usually a linear function (see an example Figure 2) and sometimes an exponential one.
- The influence of the ambient humidity exists only for NOx and for some vehicle classes. It is a linear function. An increase in ambient humidity lowers the NOx emissions, which is also the expected general trend according to the humidity correction established in legislative testing (EEC, 1991). Figure 3 shows that in urban test cycle the standard correction is nearly valid for diesel cars with less than 5 % deviation from the now-established model. However, both groups of petrol cars would need much stronger correction, as the relative change over the allowed humidity range is about 35 % for the Euro 2 to and over 55 % for the Euro 3 test fleet, and the normative factor corrects only by some 20 % within the same range of humidity.
- Exhaust gas dilution ratio. A higher dilution ratio increases only the diesel PM emission measurement.

Figure 3: Linear models of (uncorrected) NOx emissions measured in Artemis urban driving cycle, fitted in average values for high, medium and low humidity, and correction factor according to legislative test protocol (as 1/kH).



4. Round robin test

The best accuracy (i.e. lowest spread in results) was encountered for CO_2, where the average coefficient of variation was around 5 %. This latter is around

40 % for CO, below 40 % for NOx, and around 60 % for HC. When comparing these variations to those values calculated on the basis of the repeated tests at the begin and at the end of the whole round robin test in a same laboratory, we see that the overall variability recorded for CO in the round robin test was roughly at the same order of magnitude than the “basic” repeatability combining the repeatability of the laboratory and fluctuations in the car performance. However, with HC the overall spread of results over the whole round robin test was higher, suggesting that some external factors, like the change in fuel quality, affected and lowered the repeatability. In terms of NOx, the overall round robin test variability was also somewhat higher than the basic value obtained from one laboratory alone, but we made no speculations over the probable reasons to this.

Guidelines

The knowledge of the sensitivity of vehicle pollutant emissions to the key parameters identified above allows us to design a best practice for measuring emissions of the European passenger car traffic. These guidelines can be displayed into four directions: Which cars to measure? In which conditions to test the cars? How to sample and analyse the pollutants? How to manage the data?

1. Vehicle sampling

We recommend to choose as far as possible a vehicle sample with similar distributions than the in-use fleet of the fuels, emission standard, vehicle size, maximum engine power. At least the means or medians of the cubic capacity, maximum power and mileage should be similar.

The variability between vehicles is also identified as a significant and preponderant factor, together with the emitter status (high/ or normal emitter). It is not possible to know a priori the emitter status before measuring, but the high variability between vehicles of a same category obliges to choose the cars randomly within a category and to sample a minimum number of vehicles. The minimum sample size per vehicle category, with the aim to calculate only an emission average per vehicle category, seems to be not less than 10 vehicles. We recommend to carry out only a limited number of repetition tests on these cars instead of taking a smaller sample tested many times. The vehicles to test should be chosen the most possible randomly in a list created by an official body as government, because it will give results closest to the fleet representativeness. If an official list cannot be obtained, the list created in laboratories should be completed by vehicles owners, which the profession is not in relation with the pollution, like the laboratory staff.

2. Usage conditions of the vehicles

The vehicle conditions in the measuring laboratory should correspond to the range of traffic conditions observed in Europe: it concerns not only the driving patterns, but also the environmental conditions, the vehicle load, the fuel used...

Driving cycle: It is highly recommended to test the passenger cars with real-world driving cycles. A lot of such driving cycles are available in Europe. We recommend the so-called Artemis driving cycles now widely used in Europe to measure passenger cars emissions (André, 2004), or vehicle-specific driving cycles (André et ali, 2006b) to measure actual European pollutant emission factors.

Gearshift strategy : The gearshift strategy depending on the vehicle power-to-mass ratio and on the 3rd gear ratio, i.e. foreseen in the Artemis and vehicle-adapted driving cycles, seems to be the most appropriate. But the strategy impact remains nevertheless relatively low as soon as realistic patterns are selected.

Vehicle preconditioning : We propose as preconditioning cycle a constant speed cycle with a reasonable vehicle speed level, especially for petrol cars: a 10 minutes cycle at a constant speed of 80 km/h for instance.

Driver: The robot does not give more stable emissions and some driving cycles are too aggressive for it. Therefore it is no reason to prefer robot than a human driver. We recommend that a cycle following should be in the tolerance band (± 2 km/h and ± 1 s) for more than 99% of time and with a driven distance within 1 % to the reference distance. A test is accepted with remark if it fails these values due to insufficient power, wheel slip, difficult gear box, in NEDC if deceleration is steeper than reference or if the engine stalls or does not activate immediately at test start. In all other cases a test should be rejected.

Fuel characteristics: Both diesel and petrol fuels influence a lot the emissions, but not CO_2. Therefore it is recommended to use common fuels rather than laboratory fuels.

Ambient air temperature and humidity: It is recommended to measure the emissions close to the average ambient temperature and humidity rather than at "standard" one when this one is far from the reality.

Vehicle cooling: We recommend to use a high power cooling system, in order to reproduce as far as possible the real-world cooling.

Dynamometer setting: Although only few effects were found significant, the chassis dynamometer settings should lead to a load applied to the driving wheels of a vehicle that is equivalent to the load experienced on the road at all speeds and accelerations. For the testing to be performed for the determination of real world emission factors, it is therefore primarily recommended to use road load information derived from the coast down method performed by the laboratory, and an inertia setting as close to the actual on road inertia as possible, which is also determined by the laboratory.

Conclusion

The study was designed to look at the influence of a lot of parameters of the measurement of light vehicle emission factors: driving patterns, vehicle related parameters, vehicle sampling method, and laboratory related parameters.

In the conditions of the tests, we did not find any influence of some parameters. For some other parameters we showed a qualitative influence we are not able to quantify. Finally some parameters have a clear and quantifiable influence and can be used to normalise emission measurements when the level of these parameters during the experiment is known, by using correction factors: gearshift strategy, vehicle mileage, ambient air temperature and humidity, exhaust gas dilution ratio. The results allow us to design recommendations or guidelines for the real-world emission factor measurement method. All these outputs have been used to design the Artemis emission inventorying tools for light vehicles, on a better basis than the previous European models.

The outputs of this study are nevertheless not fully positive, mainly because of the too small number of tests performed to look at the influence of some parameters, which did not allows us to find any significant influence. Some parameters could therefore be studied again.

Acknowledgments

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Application of a Scenario Based Modeling System to Evaluate the Air Quality Impacts of Future Growth

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Abstract

The structure and design of urban developments can have significant adverse effects on pollutant emissions as well as other ecological factors. When considering the future impact of growth on mobile source emissions, we generally scale the increase in vehicle kilometers traveled (VKT) on population growth. However, diverse and poorly planned urban development (i.e., urban sprawl) can force higher rates of motor vehicle use and in return increase levels of pollutant emissions than alternative land-use scenarios. The objective of this study is to develop and implement an air quality assessment tool that takes into account the influence of alternative growth and development scenarios.

Scenario techniques in land use planning have been around since the late 40's and been tested in many different applications to help the authorities in decision making. In this study we introduce the development of an advanced interactive scenario-based land use and atmospheric chemistry modeling system coupled with a GIS (Geographical Information System) framework. The modeling system is designed to be modular and includes land use/land cover information, transportation, meteorological, emissions, and photochemical modeling components. The methods and modularity of the developed system allow its application to a broad region of interest.

To investigate the impact of possible land use change and urbanization, we evaluated a set of alternative future patterns of land use developed for the southwestern region of California. Four land use and two population variants (increases of 500K and 1M) were considered. Overall the Regional Low-Density Future was seen to have the highest pollutant emissions, largest increase in VKT, and the greatest impact on air quality. On the other hand, the Three-Centers Future appeared to be the most beneficial alternative future in terms of air quality. For all cases, the increase in population was the main factor leading to the change on predicted pollutant levels.

***Key words**: Urban growth, land use scenarios, air quality modeling, transportation modeling, emissions modeling.*

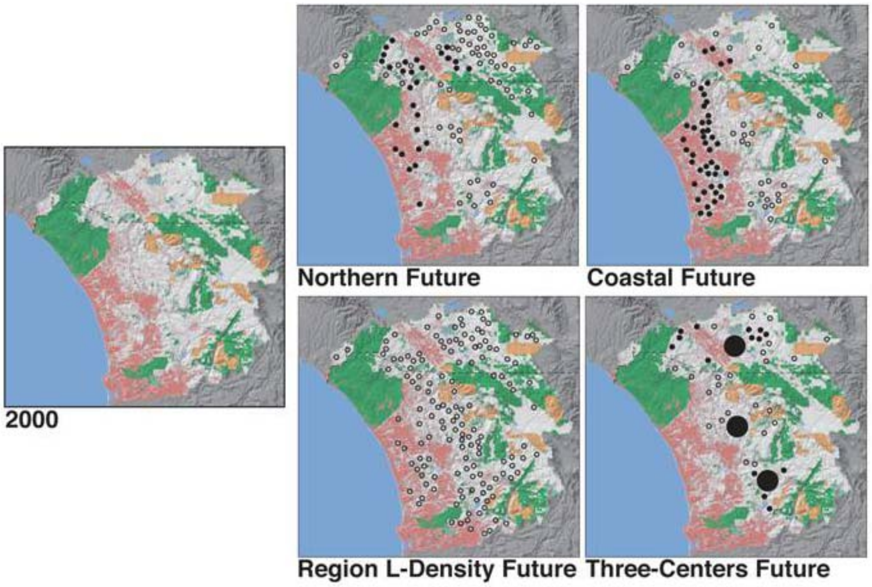
Introduction

Air pollution, one of the biggest environmental issues we face, is often associated with urbanization and industrialization. Starting from the early 20th century, a series of regulations targeted the reduction of air pollutant emissions from industrial sources; however, urbanization still remains an issue on both local and regional scales. The structure and design of urban developments can have significant adverse effects on the budget of these pollutants as well as other ecological factors (Southerland, 2004; Kepner et al., 2004). Diverse and poorly planned urban development, urban sprawl, forces higher rates of motor vehicle use and in return increases levels of pollutant emissions. Given the diversity and complexity of all these issues that adversely affect environment and air quality, there is a need to develop advanced tools which can predict the impact of future urban growth in all scales and recommend optimum approaches for achieving more sustainable environments. One effective method of assessing the future impact of urban growth is known as *Alternative Futures*, which is based on scenario analysis. Scenario techniques in land use planning have been around since the late 40's and been tested in many different applications to help the authorities in decision making (Schwartz, 1996; Steinitz, 1990; Shearer et al., 2004). Among the several advantages of this method are the flexibility to include broader aspects of related concerns and the inclusiveness to provide a linkage between them showing the feedback and possible mitigation aspects. In this paper we introduce the development of an advanced interactive scenario-based land use and atmospheric chemistry modeling system coupled with a GIS (Geographical Information System) framework. The modeling system is designed to be modular and includes land use/land cover information, transportation, meteorological, emissions, and photochemical modeling components. The methods and modularity of the developed system allow its application to a broad region of interest. This paper describes the development and application of the modeling system to the rapidly developing area north of San Diego, CA, to address the issues discussed above.

Land Use Scenarios

To investigate the impact of possible land use change and urbanization, a set of alternative future patterns of land use were developed concerning the northern San Diego area and parts of Riverside and Orange Counties, CA (Shearer et al., 2004). As discussed in Shearer et al. (2004), the set of alternative future land use scenarios or the *Alternative Futures* were based on a large spectrum of critical uncertainties representing the possible futures which are both difficult to predict and likely to have significant impact on social, economic, political, technological, and environmental trends. Issues were identified by looking at the critical uncertainties in the study area and addressed topics such as water, energy and possible changes in social and environmental regulations. To address these questions, four land use scenarios were developed each having two variants: a 500k population increase and a 1,000k population increase. The existing land use and land cover patterns were compiled using a Landsat Enhanced Thematic Mapper (ETM) image and available information obtained from different sources. Land use maps for the future growth patterns including the existing land use are shown in Figure 1 (from Shearer et al., 2004). In all of the *Alternative Futures* the existing development (built) was left intact (Shearer et al., 2004).

Figure 1: Land use map for the study area. (a) Existing land use (b) The *Coastal Future* (c) The *Northern Future* (d) The *Regional Low-Density Future* (e) The *Three-Centers Future.*



The *Coastal Future* is built upon a scenario that encourages the conservation of future resources such as water and energy. As a result, high-density urban residential development is concentrated west of the Interstate Highway 15 (I-15) close to the coast and the amount of low-density housing in more rural locations is reduced. Eighty eight percent of the new residential areas are located in San Diego County and ten and two percent in Riverside and Orange Counties, respectively.

The *Northern Future* represents a development plan that supports low density housing concentrated in the northern portion of the study area. Hence, the new suburban and rural residential development is concentrated in western Riverside County (44%) with the remaining development distributed in San Diego (55%) and in Orange (1%) Counties. Overall, the majority of housing is placed in subdivisions that are relatively close to incorporated cities and their associated infrastructure.

The *Regional Low-Density Future* best emulates the urban sprawl pattern of development present in the Western U.S. In this future scenario, the entire urban development is spread throughout the study area with new housing being predominantly developed on large lots. The majority of new housing is located in rural and ex-urban areas within San Diego (69%), Riverside (30%) and Orange (1%) Counties.

The *Three-Centers Future* concentrates development and assists in the conservation of some habitat. Much of the future housing is located close to existing development near the cities of Temecula, Vista, and Ramona. This lessens the amount of rural development sprawled throughout the southeastern part of the study area but adds some more rural residential development in the north. Percent distribution of the houses to the counties in the study area is the same as in the Regional Low-Density Future.

Modeling System

In this section, an overview of the approach used to develop, test, and apply a modeling system to assess the impact of future scenarios on regional air quality is presented. A GIS based land cover and infrastructure system was coupled with pollutant emissions, meteorology, and air chemistry models. The modeling framework included a number of individual models involving future land use, emissions, air quality, and their subsequent linkages. The overall framework may be envisioned as a series of loosely coupled models with outputs and inputs shared among the models. The backbone of the framework is a GIS capable of operating at multiple spatial and temporal scales. Thus, each individual model encapsulated within the framework contains a spatial allocation, which can be mapped.

To create input scenarios for the emissions components, two models were constructed which describe the land use and transportation infrastructure in the region. The land use predictions, termed the development model, and the transportation model were linked to assist computation of commuting routes, which provided estimates of future vehicle miles traveled (VMT). Future emission assessments require knowledge on how an area may change and these rely on the development and transportation models for input. Air quality modeling requires inputs from all aspects of an emissions assessment. Any future alterations in population, technology, or laws will factor in as inputs into scenarios that have the potential to alter the outputs of any one model through cascading linkages. Thus, the modeling framework linkages were constructed to account for potential scenario-based changes.

Brief descriptions of the three primary components of the modeling system (Transportation, Emissions, and Air Quality) are described below:

Transportation Modeling: Transportation models attempt to describe the flow of traffic between locations to allow for forecasting and analyzing future passenger and/or freight movement. The transportation model developed for this system accommodated all the basic conditions that are required (Beimborn et al., 1996), with a minimal level of complexity. The primary intent of this modeling approach was to route present and future passenger cars in the study region from their home locations to their destinations along the quickest path. The model was structured in a GIS platform to coordinate all the spatial aspects and linkages of spatial information with the future land use data using Tiger line transportation files and the associated attribute information as a foundation for determining the VKT between home and work (USCB, 2002a). Based on the results of the development model, starting points were identified as new housing units defined in a 30x30 m^2 cell within a GIS layer. The ending locations were determined as thirteen major commuting points or work centers within the study area or as locations where commuters would exit the study area heading mainly north to Los Angeles or to northern Riverside County. Although substantially more work centers exist in the region, this assumption was based on the fact that many of these are clumped into the represented commercial-industrial centers. Homes were randomly assigned a path to a commercial-industrial work center as a percentage derived from U.S. Census county commuter information (USCB, 2002b).

Emissions Modeling: The emissions component of the model incorporated emissions models for biogenic, area-wide, and mobile sources. Emissions were computed for five pollutants: NO_x, sulfur dioxide (SO_2), volatile organic carbon (VOCs), carbon monoxide (CO), and particulate matter (PM). Existing emissions estimates served as a base for estimating future emissions. The final product of the emissions modeling was a 5x5 km^2 gridded hourly day specific emissions inventory for the high O_3 period chosen for the modeling episode (July 7 – 11, 2003).

Spatially and temporally variable biogenic emissions were estimated for the photochemical modeling domain. Estimated biogenic emission species consisted of biogenic volatile organic carbons (BVOC) such as isoprene, monoterpenes, and methylbutenol. The California Air Resources Board's (CARB) biogenic emissions model, BEIGIS, was used as the basis for the biogenic emissions component of the model. BEIGIS is a GIS based biogenic emissions model that is built upon biomass and emissions studies performed in Southern California (Horie et al., 1991; Benjamin et al., 1996). Default emission rates given in BEIGIS come mostly from Horie et al. (1991) and Benjamin et al. (1996, 1997).

Area-wide emissions of the future land uses were estimated based on the use of consumer products, residential natural gas consumption, dry cleaning, and residential and commercial lawn maintenance. These categories were determined according to the given details in the future scenarios. Methodologies used for area source emission estimates were based on the CARB's Emissions Inventory Procedure Manual (CARB, 1997). Total emission rates were allocated spatially and temporally for each land use scenario. In contrast to the biogenic emissions estimates, area-wide emissions of the future scenarios were superimposed on the current emissions layer due to the fact that existing land use was preserved in the scenarios.

To estimate future on-road mobile source emissions, the GIS-based travel simulation algorithm described in the transportation model section was utilized in conjunction with EMFAC2002, the on-road emissions model specific to California (CARB, 2002). This model calculates emission factors and/or emission rates for the vehicle fleet in California as categorized in 13 vehicle classes and accounts for six criteria pollutant types. Total VKT served as the major input for EMFAC2002 as calculated by the transportation model.

Meteorological and Photochemical Modeling: In order to evaluate the impact of the future scenarios on the formation of secondary pollutants, the spatially and temporally resolved output from the emissions model was coupled with an air quality modeling system that can operate over multiple domains. This took place in two steps: meteorological and photochemical air quality modeling. As part of this study, simulations were performed for an episode from July 7 through July 11, 2003. During this period, the San Diego area experienced high levels of air pollution and so it provided an opportunity to investigate the possible highest impact of the future land use change on air quality.

A prognostic forecast model, the Fifth Generation Penn State/NCAR Mesoscale Model (MM5) (Grell et al., 1995), was used to generate all required field variables and parameters for the emissions model and the air quality model. MM5 is a well-known mesoscale, nonhydrostatic, terrain-following sigma coordinate model that is used in predictions of mesoscale and regional air circulation. MM5 provided the photochemical model with 3-dimensional field variables such as horizontal wind, temperature, pressure and other parameters, which are used by the photochemical model for the atmospheric transport and dispersion calculations. Temperature and ground level shortwave radiation variables used in biogenic emissions model were also predicted by MM5.

To address the formation of secondary species (e.g., O_3)and transport/dispersion of emissions, the Comprehensive Air Quality Model with extensions (CAMx) was employed (Environ, 2003). CAMx is a photochemical Eulerian dispersion “one-atmosphere” modeling system with multi pollutants and scaling that can be applied to regional or local domains to predict all phases of air chemistry. The model requires a variety of input variables including meteorological fields, photochemical reaction rates, gridded and/or point emissions, surface characteristics, initial conditions (IC), and boundary conditions (BC). For all the simulations, IC and BC were set according to the U.S. EPA’s standard profiles.

Results and Discussion

1. Emissions

Total future emission estimates were generated on a daily basis for the same time period as the air quality model was run (July 7-11, 2003). Within the study time period, the greatest change in emissions occurred on July 10 and was due to changes in biogenic emissions driven by the ambient temperatures. In terms of the other anthropogenic emission sources it was assumed that the modeling period extended throughout the weekdays with the same activity rate and therefore no changes were predicted on a daily basis. Table 1 compares the emission values by source category between the base case and eight *Alternative Futures* (four scenarios and two populations) for CO, NO_x, SO_2, VOCs, and total PM on July 10. Also included in Table 1 are the incremental differences between the base case and the *Alternative Futures*. Land use differences had the greatest influence on two categories: mobile and biogenic sources. Area wide emissions showed a linear dependence on population, and stationary/industrial sources were assumed unchanged in this study. Although the difference between the four scenarios was small in terms of the mobile source contribution, the *Northern Future* generated the highest emissions and the *Three-Centers Future* generated the smallest values. This was attributed to the longer commuting paths and associated higher VKT rates for the *Northern Future*.

Table 1: Emission estimates (tn day^{-1}) for the base case and the *Alternative Futures* with 500k and 1,000 increases in population within the study area. Change shows the difference between the base case and the future scenarios.

Species	Source	Base Case	Coastal		Northern		Reg. Low-Density		Three-Centers	
			500k	1,000k	**500k**	1,000k	500k	1,000k	500k	1,000k
CO	Area	6.92	7.25	7.50	7.25	7.50	7.25	7.50	7.25	7.50
	Change		0.26	0.51	0.26	0.51	0.26	0.51	0.26	0.51
	Mobile	1817.14	1968.83	2152.46	1941.83	2165.20	1941.75	2145.28	1958.69	2144.09
	Change		151.69	335.32	124.69	348.06	124.60	328.14	141.55	326.95
	Stationary	17.31	17.31	17.31	17.31	17.31	17.31	17.31	17.31	17.31
	Change		0	0	0	0	0	0	0	0
	Biogenic	0	0	0	0	0	0	0	0	0
	Change		0	0	0	0	0	0	0	0
NO_x	Area	2.63	3.26	3.89	3.26	3.89	3.26	3.89	3.26	3.89
	Change		0.63	1.26	0.63	1.26	0.63	1.26	0.63	1.26
	Mobile	220.21	233.00	247.27	230.64	249.21	230.77	247.33	228.61	247.00
	Change		12.80	27.07	10.43	29.00	10.56	27.13	8.41	26.79
	Stationary	18.47	18.47	18.47	18.47	18.47	18.47	18.47	18.47	18.47
	Change		0	0	0	0	0	0	0	0
	Biogenic	0	0	0	0	0	0	0	0	0
	Change		0	0	0	0	0	0	0	0
SO_2	Area	0.029	0.033	0.037	0.033	0.037	0.033	0.037	0.033	0.037
	Change		0.004	0.008	0.004	0.008	0.004	0.008	0.004	0.008
	Mobile	12.078	12.161	12.266	12.148	12.273	12.149	12.264	12.157	12.263
	Change		0.083	0.188	0.070	0.196	0.071	0.186	0.079	0.185
	Stationary	1.43	1.43	1.43	1.43	1.43	1.43	1.43	1.43	1.43
	Change		0	0	0	0	0	0	0	0
	Biogenic	0	0	0	0	0	0	0	0	0
	Change		0	0	0	0	0	0	0	0
VOC	Area	82.47	87.84	93.21	87.84	93.21	87.84	93.21	87.84	93.21
	Change		5.37	10.74	5.37	10.74	5.37	10.74	5.37	10.74
	Mobile	193.79	207.61	220.32	207.60	222.44	208.15	223.89	207.88	221.36
	Change		13.82	26.53	13.80	30.01	14.36	28.65	14.09	27.57
	Stationary	345.68	345.68	345.68	345.68	345.68	345.68	345.68	345.68	345.68
	Change		0	0	0	0	0	0	0	0
	Biogenic	165.58	172.96	184.07	183.05	193.99	192.42	210.52	177.35	181.29
	Change		7.38	18.49	17.47	28.41	26.84	44.94	11.77	15.71
PM (Total)	Area	287.08	287.16	287.24	287.16	287.24	287.16	287.24	287.16	287.24
	Change		0.08	0.17	0.08	0.17	0.08	0.17	0.08	0.17
	Mobile	6.06	6.99	8.04	6.98	8.26	7.03	8.38	7.01	8.14
	Change		0.93	1.98	0.92	2.20	0.97	2.32	0.95	2.08
	Stationary	20.17	20.17	20.17	20.17	20.17	20.17	20.17	20.17	20.17
	Change		0	0	0	0	0	0	0	0
	Biogenic	0	0	0	0	0	0	0	0	0
	Change		0	0	0	0	0	0	0	0

VOC emissions from biogenic sources, on the other hand, demonstrated completely different characteristics and have very distinct differences among the scenarios. This change was by virtue of the quantity and quality of the altered land in the future scenarios. In some of the scenarios, the amount of rural housing built on large lots that leads to higher emission rates (Benjamin et al., 1997) was larger than for the other scenarios and was the major factor for higher emission estimates. The *Regional Low-Density Future*, for example, had twice as much land allocated as new residential areas than the other scenarios and 80% of it was classified as rural. This, and the fact that most of the low emitting or non emitting land (e.g., barren, grassland, etc.) was converted into residential vegetation uses, led to the highest amount of biogenic VOC emissions, an increase of 44 tons from the base case. In contrast, the lowest biogenic emissions occurred in the *Three-Centers Future*, which has the smallest percentage of higher-emitting rural lots. Overall biogenic emissions were greater than or nearly the same as the total estimated anthropogenic VOCs. This highlights the importance and magnitude of biogenic emissions from urban areas.

Table 2: One-hour average predicted peak O_3 concentrations (ppb) over the study area and San Diego County for the entire simulation period. *Max O_3* column designates the predicted peak O_3 concentration and *Change* (ppb) column designates the difference with respect to the base case.

Simulations	July 8		July 9		July 10		July 11	
	Max O_3	Change	Max O_3	Change	Max O_3	Change	Max O_3	Change
BASE CASE	100		114		106		117	
Coastal Future (500k)	102	2	119	5	110	4	120	3
Coastal Future (1,000k)	105	5	123	9	115	9	124	7
Northern Future (500k)	103	3	119	5	111	5	121	4
Northern Future (1,000k)	105	5	122	8	114	8	124	7
Reg. Low-Density Future (500k)	104	4	121	7	113	7	124	7
Reg. Low-Density Future (1,000k)	108	8	126	12	118	12	128	11
Three-Centers Future (500k)	102	2	118	4	109	3	120	3
Three-Centers Future (1,000k)	103	3	121	7	112	6	122	5

2. CAMx Simulations

The air quality simulations using CAMx were performed for the period of July 7 through July 11, 2003 and covered an observed O_3 episode in the region. The first day of the simulations, July 7, was excluded from the analysis and was used as the model's spin-up time.

For all the simulations, the maximum predicted O_3 concentrations occurred on July 11. This situation was associated with low daytime wind speeds and relatively stagnant conditions. The location and time of the peak concentration over San Diego and Riverside Counties were approximately the same for all days. Table 2 presents the simulated peak O_3 concentrations for the base case and scenarios on a daily basis. It is seen from this table that the peak O_3 increases 10 ppb, on average, for the *Regional-Low Density Future* and ranges from 2 to 9 ppb for the other scenarios, suggesting that the most of the impact is local. These results show that both the regional and local impact is the least for the *Three-Centers Future*, which also had the smallest incremental changes in emissions.

The results shown in Table 2 indicate that while the base case is below the one-hour average federal standard for O_3 on July 9 and 11, all scenarios with 1,000k new residents are likely to exceed this standard. Unless the future growth plans include some mitigation measures, the area will be classified as non-attainment. Such measures could include newer and cleaner technologies in motor vehicles and the planting of selected plant species to reduce BVOCs (Benjamin and Winer 1998; Pun et al., 2002; Mendoza-Dominguez et al., 2000).

Summary and conclusions

In this paper we described the development and application of a scenario-based modeling system that couples a GIS based land cover and infrastructure system with pollutant emissions, meteorological and air chemistry models to predict the impact of growth on future emissions and air quality. Four land use and land cover scenarios with each having a 500k and 1,000k population increase were developed within a GIS system to depict the possible future growth and its consequences within the study area (Shearer et al., 2004). Subsequently, we linked this GIS system with transportation, emissions, meteorological and air quality models to predict air quality impacts for each scenario.

Emission estimates and air quality simulations were performed for an observed O_3 episode in the southwestern part of California from July 7 to 11, 2003. Estimates of future emissions were distinctly different for the four scenarios. Changes in BVOC emissions were comparable to the changes in the total anthropogenic VOC emissions. Overall the *Regional Low-Density Future* was seen to have the highest pollutant emissions and the greatest impact on air quality. On the other hand, the *Three-Centers Future* tended to minimize emissions from mobile and biogenic sources and appeared to be the most beneficial alternative future in terms of air quality. For all cases, the increase in population was the main factor leading to the change on predicted pollutant levels.

As standards for air quality become more stringent, the need for predictive tools that can assess the impact from future growth and developments is critical. The modeling system and simulation results presented in this paper were aimed to answer the question of what and how *Alternative Futures* should be designed if we are to implement effective strategies to reduce future air quality impacts. Since the system is fully modular and capable of integrating new sub-systems, it can be modified to include additional features as desired and applied a variety of regions where future development and growth plans are needed.

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Urban development scenarios and their impact on road transport and air pollution

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Abstract

Two urban development scenarios were evaluated for their impact on road transport and air pollution using the BUGS methodology. An extreme case of urban sprawl was compared to a case where population and jobs were re-distributed over five satellite cities, created from existing smaller towns in the central Ruhr area in Germany. For the urban-sprawl situation, the urban land use area in the study domain increases by almost 75 % compared to the reference case. For the satellite-city scenario, urban land use changes are around 9 %. The total traffic kilometres and associated emissions increased by up to almost 17 %. A detailed analysis shows that the urban-sprawl scenario results in an PM_{10} exposure reduction of 5.7 %, and a reduction of 1.4 % for the satellite-city case. The dominant driver of these exposure changes appears to be the movement of people from the relatively polluted conurbation to less-polluted areas.

Key-words*: air quality modelling, air pollution policy and abatement strategies, PM_{10}, ozone, exposure, land use changes, urban development*

Résumé

En utilisant la méthodologie BUGS, deux scénarios de développement urbain ont été évalués quant à leur impact sur le trafic routier et la pollution atmosphérique. Un cas extrême d'expansion urbaine a été comparé à un cas où la population et le travail étaient répartis dans cinq cités satellites, créées à partir de petites villes situées dans la région centrale de la Ruhr en Allemagne. La surface urbaine dans le domaine étudié augmente d'environ 75 % dans le cas de la situation d'expansion urbaine. Dans le scénario des cités satellites, l'expansion urbaine est de l'ordre de 9 %. Le kilométrage total parcouru et les émissions qui y sont associées augmentent jusqu'à environ 17 %. Une analyse détaillée montre une diminution de *l'exposition à PM_{10} de 5,7 % dans le scénario d'expansion urbaine et de 1,4 % dans le cas des cités satellites. L'élément principal responsable de ces changements est la migration de personnes de zones d'agglomération relativement polluées vers des zones moins polluées.*

***Mots clés**: modélisation de la qualité de l'air, stratégies contre la pollution de l'air, PM_{10}, MES, ozone, exposition, utilisation des terres, développement urbain*

Introduction

Cities and towns, being home to 80 % of Europe's citizens (European Commission, 2004), experience increasing signs of environmental stress, notably in the form of poor air quality and excessive noise. Road traffic is one of the most important sources of air pollution inside urbanized areas. It is greatly determined by the distribution of population and working places inside and around the urban centre. In the past, abatement strategies towards traffic-related air pollution and noise were based on technological progress, including the improvement of fuels, the development of catalytic converters, and the development of silent road pavements. Although these elements have induced significant improvements of environmental quality, increasing car numbers has offset their benefits. Among the main causes for increased car use is the enhanced transport demand induced by urban sprawl.

With respect to air quality, ground-level ozone and fine particulate matter are the main pollutants in terms of their health effects. In "Europe's environment: the third assessment" (EEA, 2003), the European Environment Agency (EEA) observes that the EU target value for ground level ozone is exceeded in many European cities. The World Health Organisation (WHO) attributes several thousand hospital admissions and premature deaths each year to the long-term exposure to particulate matter.

The EEA recognises urban planning as a tool for managing, protecting, and enhancing the environment, and promotes spatial planning systems that integrate urban land use management and environmental issues. In "Towards a thematic strategy for the urban environment" (European Commission, 2004), sustainable urban design is mentioned as a priority, and urban sprawl the most urgent of the urban design issues. The EU promotes compact and polycentric city forms, as expressed for instance in the European Spatial Development Perspective (ESDP).

In this study, a series of numerical simulations was performed to evaluate the impact of two urban development scenarios on air quality and associated human exposure. The methodology is described in section 2. Results are presented and discussed in section 3.

Methodology

Within BUGS (BUGS, 2004), a methodology has been developed to assess and compare different land use development scenarios - in particular those related to urban sprawl - in terms of air quality and exposure (De Ridder *et al.*, 2004). Starting from an analyses of the existing socio-economic as well as the surface parameters (derived from satellite data processing), scenarios are created for which simulations are performed using a traffic flow model, a traffic emission model, and a regional air quality model. In the end, an economic evaluation of the various air pollution-related health impacts is done by means of the ExternE-methodology (Friedrich and Bickel, 2001).

1. Urban development scenarios

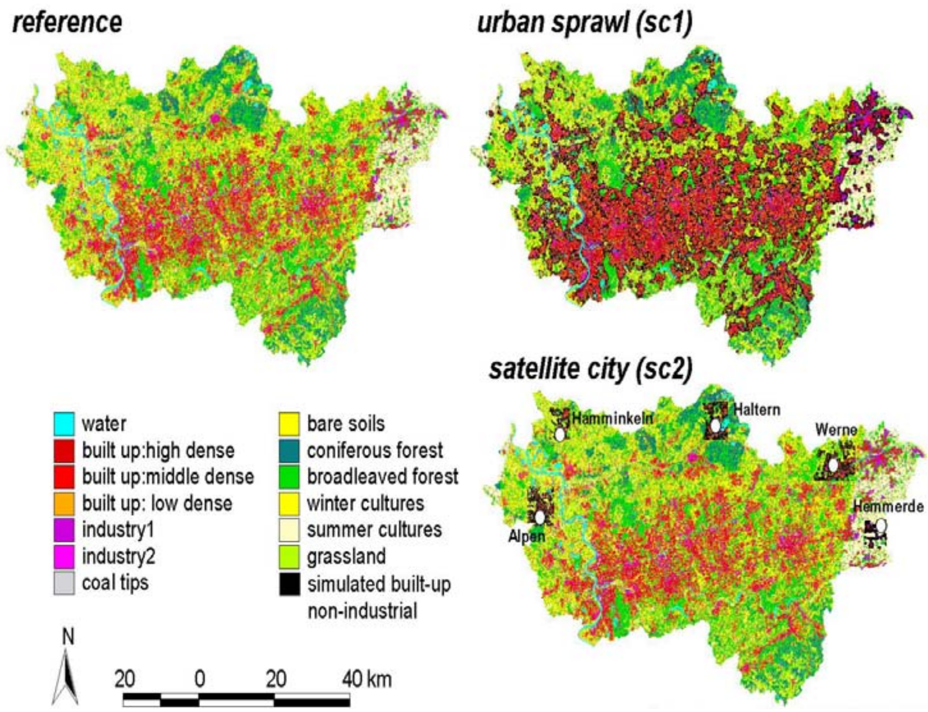
The area that was selected for the implementation and assessment of the scenarios is the highly urbanised region in the Ruhr area (administratively defined by the *Kommunalverband Ruhrgebiet*), located in the north-western part of Germany with a total population in excess of 5.5 million (Regionalverband Ruhr, 2004). The choice for this particular area was mainly motivated by its size and importance, as well as its conversion potential. Two distinct scenarios were selected for subsequent environmental assessment. The first is referred to as ‘urban sprawl’ and is characterized by a significant increase in built-up surface. This scenario supposes a continuation of the current process of people leaving the highly occupied central part of the study area to settle in the greener surroundings. In the second scenario, referred to as ‘satellite cities’, persons and jobs were displaced to five existing towns located near the core of the urban area. This scenario is an example of a ‘decentralised concentration’ strategy.

In a first step, so-called reference maps were created to characterise the conditions as they are today, using locally available information (population and jobs) together with satellite-based land use data. In a next step, spatial modelling techniques were applied to translate the qualitative planning rules concerning the scenarios into quantitative information, i.e., maps containing the required parameters for the two scenarios. Land use maps for the current situation were derived from satellite imagery of the Landsat Thematic Mapper (TM) instrument, using the ‘stepwise discriminant analysis’ image classification technique (Sabins, 1997) together with ground truth data.

After the establishment of the maps for the reference case, spatial modelling techniques were applied to simulate changes in land use, population, and job density, according to the urban-sprawl and the satellite-city scenarios outlined above. As far as the re-distribution of land use is concerned, spatial simulations were done using the so-called ‘potential model’, which models the potential for transition to a given land use type of all the grid cells in the domain (Weber and Puissant, 2003). The results of this procedure are contained in Figure 1. The main result here is that, for the urban-sprawl situation, the urbanised area in the study domain increases by almost 75 %, hence land consumption is rather drastic. For the satellite-city scenario, urban land use changes are much lower, around 9 %.

Figure 1: Land use categories of the reference state and the two scenarios.

Figure 1: Catégories d'utilisation des terres de la situation de référence et des deux scénarios.



The resulting land use maps for each scenario were then used to model the corresponding spatial distribution of people and jobs. As far as the population distribution is concerned, the procedure followed a two-step approach. First, the number of people per grid cell containing a non-industrial built-up land use type was taken constant per zone, using the same value as in the reference case. In the second step, a normalisation was carried through in the whole area to ensure that the total number of people remained the same as in the reference situation, as required. The re-allocation of jobs for the scenarios was treated in a similar way, the main difference being that, within each zone, jobs were distributed uniformly over both built-up categories, i.e., industrial and non-industrial, the latter corresponding to jobs in the services sector. Again, the data on population and jobs were interpolated towards the zones used in the traffic model.

2. Traffic flow modelling

The first step in the traffic modelling activities was the creation of a simplified network for the study domain, consisting of highways and other major roads. Apart from using this network, the traffic model also subdivides the study area into different zones (typically a few hundred). For each zone the following data have to be specified: the number of inhabitants and jobs, the area in square metres and the percentage of built-up non industrial, industrial and green land use. The number of trips to and from each zone (i.e., the traffic volume) was calculated for a 24-hour period. For the reference situation, the number of car trips was calculated from the number of inhabitants and jobs as well as the number of day car trips (one way) per inhabitant or per job.

As a next step, origin and destination matrices were created containing the traffic volumes between zones in the area. This was done using a so-called gravity model, which calculates the number of trips between any two zones following the number of trips produced in each zone as well as the number of trips attracted to each zone, the probability of travel between two zones decreasing with their mutual distance. Information regarding the traffic relations between individual zones was then used to calculate the traffic relations between individual network nodes, which constitute the actual traffic origins and destinations in the study domain. In order to obtain realistic spatial distribution of traffic loads for the reference case, a calibration procedure was carried out using data from the most recent traffic census.

In order to simulate the effect of the scenarios on traffic volumes and their spatial distribution, the traffic origins and destinations for each zone were recalculated based on the changes in land use, number of inhabitants and number of jobs. The network, including the characteristics of the individual segments and the spatial distribution of the zones were taken identical as for the reference state. Furthermore, in the urban-sprawl scenario, the number of daily car trips per inhabitant were increased compared to the reference case to simulate the reduced use of public transport, a phenomenon which is commonly observed when the number of inhabitants per hectare drops significantly, which is the case here. As was the case for the reference calculations, the new traffic volumes between any two zones were calculated with the help of the gravity model. Traffic intensities calculated for the scenarios differ from the reference case because of the changed patterns of land use, inhabitants and jobs. The main result here is that passenger car traffic increased by 17 % for the urban-sprawl scenario, and by 15 % for the satellite-city scenario.

3. Air quality modelling

Regional air quality was simulated using the AURORA modelling system (Mensink *et al.*, 2001). It employs the meteorological fields (wind vectors, diffusion coefficient, radiation, temperature and humidity) produced by the meso-scale atmospheric model ARPS (Xue *et al.*, 2000, 2001) to calculate dispersion of pollutants emitted by traffic, industry, and building heating. The physical and chemical processes are treated in a modular way. The most important modules are those representing physical transport phenomena (advection and turbulent diffusion), photochemical processes and chemical reactions (Carbon-Bond IV-model) This last module has been upgraded to include the effect of biogenic isoprene emissions from forests, which affect ozone levels.

Apart from calculating the dispersion and transformation of regionally emitted pollutants, the model accounts for large-scale meteorological influences as well as for remote pollutant emissions. An advanced land surface scheme (De Ridder and Schayes, 1997) was incorporated in AURORA to study the impact of land use changes on atmospheric circulations and pollutant dispersion. The surface scheme calculates the interactions between the land surface and the atmosphere, including the effects of vegetation and soils on the surface energy balance, and was adapted to better represent urban surfaces. Land use type and vegetation cover fraction are required as input parameters, both of which were derived from the same Landsat imagery as employed to derive land use for the traffic model, thus ensuring consistency between the different models constituting the integrated approach.

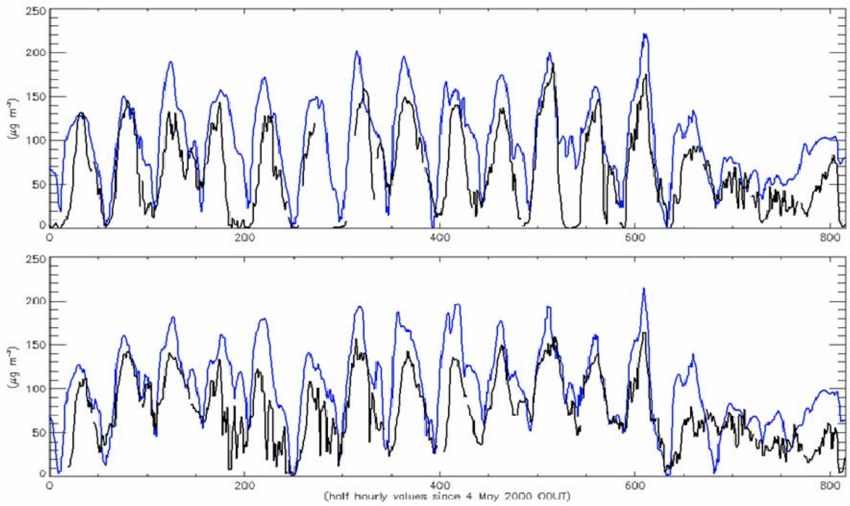
A three-week period, 1-20 May 2000, was selected to perform the AURORA simulations on. This period was characterized by the presence of a blocking anticyclone over southern Scandinavia, producing weak south-easterly winds, clear skies, and moderately high temperatures over the Ruhr area. The nice weather ended abruptly on the 17^{th}, when a cold front swept over the area. Emissions from industry, shipping and building heating were obtained from the 'Landesumweltamt Nordrhein-Westfalen', the local environmental administration. Traffic-related emissions were calculated using the MIMOSA model (Mensink et al., 2000; Lewyckyj et al., 2004), which uses the COPPERT III methodology to calculate geographically and temporally distributed traffic emissions using traffic information (including fluxes of vehicles and their speeds) from the traffic flow model. Apart from the above-mentioned anthropogenic emissions, biogenic emissions from forests (isoprene) were also calculated. The simulations carried out here focused on ground-level ozone and fine particulate matter, both pollutants being recognised as having major effects on human health.

Results and discussion

1. Model validation

Figure 2: Simulated (blue line) as compared to observed (black line) ground-level ozone concentrations for the stations Bottrop (upper panel) and Essen (lower panel).

Figure 2: Concentrations simulées (ligne bleue) et observées (ligne noire) d'ozone au sol aux stations Bottrop (en haut) et Essen (en bas).



Air quality simulations performed by means of the AURORA model were validated by comparing model results with available observations from two stations (Essen and Bottrop) in the area that measured ozone (Figure 2). The Bottrop station is characterised by intense traffic, whereas the monitoring station in Essen is more typical for an urban background situation. Even though the simulations overestimate the ozone peak concentrations rather systematically by up to a few ten percent, the diurnal cycle as well as the behaviour of the model over the entire three-week period is rather satisfactory, and well in line with results produced by models of this type. In particular the difference of night-time concentrations between the two locations, which is due to the titration effect (reduction of ozone by traffic-related NO emissions) caused by the more intensive traffic at Bottrop, is well captured by the model, meaning that the spatial distribution of traffic emissions as well as the chemical processes accounted for in the model perform correctly.

2. Air quality modelling results

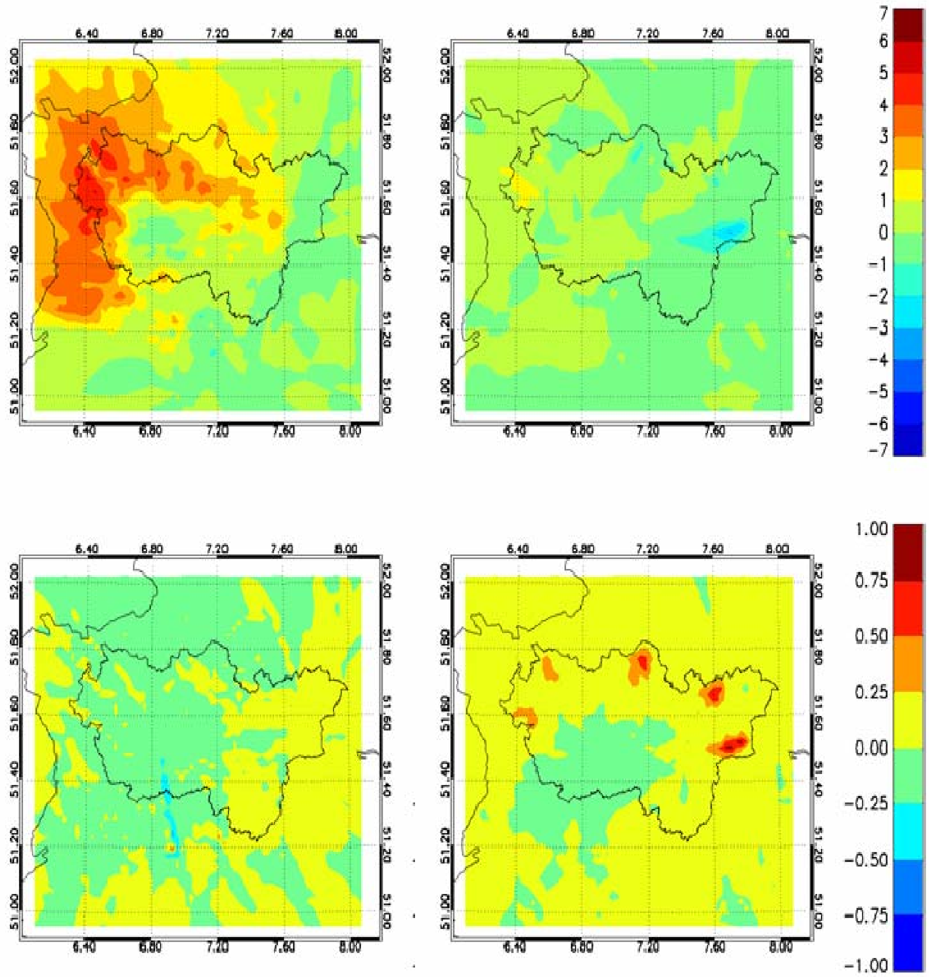
After the successful completion of the simulations for the reference case, the AURORA model was run on the urban-sprawl and satellite-city scenarios established previously, using the modified land use characteristics as well as the correspondingly modified traffic flows as inputs. The calculated pollutant emissions, largely traffic-related in this area, underwent increases of the same order as the increases of the traffic flows themselves. For instance, CO_2 – a greenhouse gas – saw its concentration increased by approximately 12 % for both scenarios.

The simulated change of ground-level ozone and of primary particulate matter is shown in Figure 3. With respect to ozone, the largest changes are seen to occur for the urban-sprawl scenario. Owing to the dominating south-easterly wind direction during this episode, an increased ozone plume is simulated north-west of the agglomeration. The titration effect, on the other hand, which is the consequence of increased traffic emissions, slightly depresses ozone concentrations in the central portion of the domain, i.e., where the highest population densities occur. As a result, the average exposure to ozone pollutants (calculated as the average of the concentrations, weighted with population density) remained almost unchanged (+ 0,3%) between the reference case and the urban-sprawl scenario. Also in the satellite-city scenario the changes are minimal (- 0,5%), despite the increased domain-average emissions.

With respect to fine particulate matter, the effect of the scenarios is perhaps not so clear. Whereas the satellite-city scenario clearly exhibits local spots of (a very modest) increase of this pollutant, the concentration patterns in the urban-sprawl case appear almost unaltered. A detailed analysis shows that there is a slight overall increase of domain-average concentration. However, the effects on human exposure to this pollutant are not so straightforward: whereas one would intuitively associate increased emissions and the ensuing increased domain-average concentrations with increased human exposure values, the contrary is seen to occur. Indeed, a detailed analysis shows that the urban-sprawl scenario results in an exposure *reduction* of 5,7 %, and a reduction of 1,4 % for the satellite-city case. The dominant driver of these exposure changes appears to be the movement of people from locations with high to locations with low particulate matter concentrations. This effect is enhanced by the fact that the decrease in these concentrations in densely populated areas is not completely offset by their rise in less densely populated areas. Stated otherwise, the global exposure decreases when a portion of the population moves from the relatively polluted conurbation to less-polluted areas.

Figure 3: Concentration change (in µg m^{-3}) of ozone (upper panels) and PM_{10} (lower panels) for scenario 1 (left panels) and scenario 2 (right panels).

Figure 3: Changement dans les concentrations (en µg m^{-3}) d'ozone (en haut) et de PM_{10} (en bas) pour les scénarios 1 (à gauche) et 2 (à droite).



The air pollution-related public health damage, together with changes in CO_2 emissions, were used as for the calculation of the damage costs using the ExternE methodology (Friedrich and Bickel, 2001). The emissions in CO_2 increase in both scenarios, though slightly more so for the urban sprawl scenario, when compared to the reference state. This results in higher damage costs related to global change. The changes in damage costs of public health related to primary particulate matter and ground-level ozone are determined by the changes in exposure which reflect the combined effect of changes in concentrations and changes in population at a specific location in the study area. Because of the dominant effect of relocating people to less polluted areas, the urban development patterns presented in both scenarios result in a positive effect in the public health damage costs, especially for the urban sprawl scenario. In the calculation of the total damage costs, the effects of exposure changes to particulate matter were dominant, owing to the severe health impacts attributed to this pollutant. As a result, the total avoided damage costs range between 5.6 M€ and 15.0 M€ for, respectively, the satellite city scenario and the urban sprawl scenario when compared to the reference state.

The main findings of the present study are summarised in Table 1. The figures are expressed in terms of changes induced by the spatial developments of the urban-sprawl and the satellite-city scenarios as compared to the reference case.

Table 1: Overview of the changes induced by the urban-sprawl and satellite-city scenarios. All changes are expressed as a percentage increase compared to the reference case, except the damage cost, which is expressed in M€.

Tableau 1: Aperçu des changements causés par le scénario d'expansion urbaine et le scénario des cités satellites. Ces changements sont tous exprimés en pourcentage d'augmentation par rapport à la situation de référence, excepté pour les coûts induits, exprimés en M€.

SCENARIO	Traffic kilometers	Built-up area	CO2 emissions	PM10 exposure	O3 exposure	Damage cost
	%	%	%	%	%	M€
Urban sprawl	17	75	12	-5,7	0,3	-15,0
Satellite cities	15	9	12	-1,4	-0,5	-5,6

Conclusions

Two urban development scenarios implemented for the German Ruhr area were assessed in terms of road transport and air pollution exposure. The first scenario represents an extreme case of urban sprawl in which the relative size of the urbanised area increases from 26 to 44 %. Population and jobs are re-distributed accordingly. In the second scenario, the same amount of people and jobs are re-distributed in five satellite cities created from existing smaller towns around the central Ruhr area. Results for the two scenario's show that, for the urban-sprawl situation, the urbanised area in the study domain increases by almost 75 %. For the satellite-city scenario, urban land use changes are much lower, around 9 %. The total traffic kilometres and associated emissions increased by up to almost 17 %. As a consequence, the domain-average pollutant concentrations of ozone and PM_{10} also increased, though by a smaller amount. A detailed analysis shows that the urban-sprawl scenario results in an exposure *reduction* of 5.7 %, and a reduction of

1.4 % for the satellite-city case. The dominant driver of these exposure changes appears to be the movement of people from locations with high to locations with lower particulate matter concentrations. Stated otherwise, the global exposure decreases when a portion of the population moves from the relatively polluted conurbation to less-polluted areas.

Acknowledgments

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Evaluation of a Wide Area Traffic Management Strategy in the City of Leeds (UK) using a Probe Vehicle equipped with On-board emission monitoring instrumentation.

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Abstract

The timing and co-ordination of the traffic signals on the Leeds (UK) inner loop road have been up-graded in 2005 to encourage the smooth progression of vehicles, and thereby reduce emissions generated in congested (or stop-start) driving conditions.

This paper presents a 'before' and 'after' environmental evaluation of the traffic management strategy using a highly instrumented probe vehicle equipped with: On-Board emission monitoring (Horiba OBS-1300), GPS, CAN message data-logging, thermocouples, digital imaging and driver/vehicle interface sensors. Validation work, comparing the On-Board Emission measurements with a certified chassis dynamometer/ certified laboratory, is also summarised.

In excess of 25 circuits of the Leeds inner loop road circuit were conducted for both the 'before' and 'after' scenarios. Analysis indicated that the traffic management strategy has the potential to reduce vehicle journey times, fuel consumption and CO_2 emissions. However, significant variability in emissions of CO, NO_X and HCs, were observed between individual runs and scenarios.

***Keys-words**: probe vehicle, traffic management, real-world emissions.*

Introduction

The majority of Air Quality problems in European cities are closely linked to traffic-related emissions. Air quality in the worst effected areas, or 'hotspots', are being studied in greater detail with sophisticated modelling tools and monitoring equipment. Where initial monitoring and/or modelling has indicated that levels will approach or exceed air quality objectives, a plan of action has to be developed to combat the problem. In the development and assessment of action plans/ traffic management policies, complex traffic and air pollution modelling tools are commonly used to simulate the dispersion of vehicular emissions within confined street (canyon) environments. However, the majority of published studies (Vardoulakis et al, 2003) adopt a simplified representation of traffic activity, vehicular emissions and vehicle type, e.g. road transport/ emission models typically assume the average emissions over a journey (trip) vary according to the mean speed (Pronello & Andre, 2000). Average speed emission models are thought to be representative when considering large spatial areas, if the drive cycles used in their development (e.g. EUDC and FTP-75) characterise local traffic (Ntziachristos & Samaras, 2000). However, such a modelling framework lacks the temporal and spatial resolution to assess real-world changes in tail-pipe emissions caused by local signal control and traffic management strategies.

Recent improvements in the reliability, response times and accuracy of in-flight (or on-board) transient emission monitoring instrumentation (see e.g. Nakamura et al, 2003) mean that 'probe' vehicles now have an increased value as a research tool, both in the development and evaluation of traffic management strategies. In this study the timing and co-ordination of the traffic signals on the Leeds (UK) inner loop road during off-peak periods were up-graded in 2005 to encourage the smooth progression of vehicles. This Urban Traffic Management and Control (UTMC) strategy was implemented to not only improve journey times and trip reliability, but also reduce the emissions generated by vehicles in heavily congested (or stop-start) driving conditions in the vicinity of the city centre. The probe vehicle deployed to conduct the '*before*' and '*after*' evaluation of the traffic management strategy was equipped with: GPS, On-board emission monitoring (Horiba OBS-1300), CAN message data-logging, thermocouples, front/ rear facing digital cameras and sensors at the driver/vehicle interface. In excess of 25 circuits of the Leeds inner loop road circuit were conducted for both the '*before*' and '*after*' scenario. The '*before*' and '*after*' surveys were conducted consecutively in Autumn 2005 in an attempt to minimise any seasonal and long term variability in traffic demand effects.

This paper firstly describes the study site, research methodology and instrumentation used. A summary of validation work carried out with the on-board emission instrumentation is then presented. The results of the '*before*' and '*after*' experimental work are illustrated before conclusions are drawn.

Methodology And Materials

1. Study Site

The Leeds inner loop road provides a clockwise gyratory route around the City centre's pedestrianised area (see Figure 1 - Site Schematic). It is 3.25 km in length, controlled by traffic signals at 15 junctions and 14 pedestrian crossings. The Leeds inner loop road is a very heavily trafficked section of the City's road network that has historically become congested during and outside peak periods. Queuing traffic also develops within environmentally sensitive areas, where high pedestrian volumes are found within confined urban streets with restricted dispersive air flows, commonly referred to as street canyons. In such areas, unfavourably high levels of human exposure to traffic related air pollutants are commonly experienced. This traffic management scheme is part of a raft of policies Leeds City Council (LCC) are developing in an attempt to improve air quality.

Figure 1. Study Site Schematic

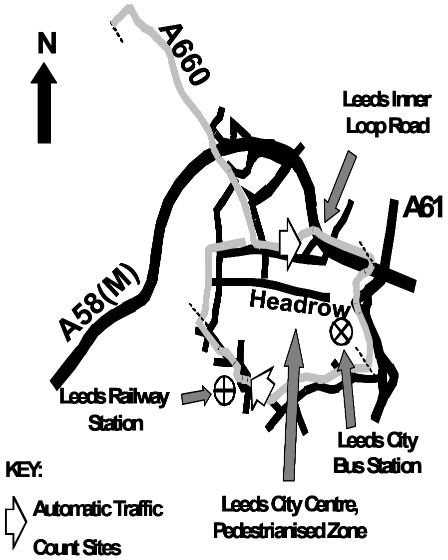
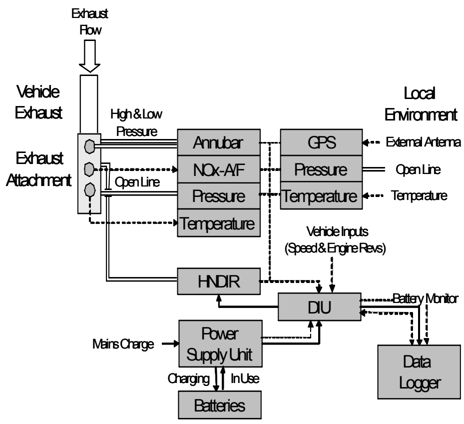


Figure 2. Horiba OBS-1300 System Schematic



The high density of signal controlled junctions and pedestrian crossings offer an opportunity to improve traffic flow progression around the City centre. The traffic management schemes improved signal timing settings/ co-ordination 'offsets' were established experimentally/ fine-tuned by Leeds City Council Traffic Engineers. By 'smoothing' the traffic flow, the hypothesis of the designed scheme is that fewer high polluting vehicle stop-start events will occur, resulting in improved network level journey times, fuel consumption and tail-pipe emissions of CO, CO_2, NO_X and HCs. As the strategy was only implemented over a short period, longer-term influences such an increase in traffic (induced) demand was shown to not have materialised, by comparing the *before* and *after* automatic traffic count data (see section 1.3).

2. Probe Vehicle

The 'platform' vehicle is a 2003 Ford Mondeo 2.0 litre petrol hatchback. This vehicle, the Institute for Transport Studies (ITS) instrumented car, has been specifically developed as a flexible research tool to study driver behaviour and evaluate in-vehicle information systems (IVIS). It monitors:

- the driver/ vehicle interaction (throttle, clutch and brake position, steering angle and speed);
- the driver and vehicle's interaction with surrounding vehicles and the road environment, e.g. distance to neighbouring vehicles (headways both front and rear), lateral lane position; and

- the vehicle and engine's operation e.g. vehicle and engine speed, spatial location.

Many factory installed and custom fitted sensors provide measurements to a central data acquisition PC. The design and development philosophy of the ITS instrumented car was to where-ever possible use factory installed sensors accessed via the vehicle's CAN (Controller Area Network). Where factory installed sensors could not deliver desired monitoring parameters, commercially available sensors were integrated into the vehicle and data acquisition system. These include:

- Front and rear facing radar rangefinders (two Autocruise® AC10 Adaptive Cruise Control radar sensors manufactured by TRW);
- Differential GPS Data Acquisition (VBOX II) System to accurately measure the speed and position of the instrumented car at 10Hz;
- Network of thermocouples measuring the exhaust skin and gas temperatures up-stream and down-stream of the catalytic converter; and
- Co-ordinated front and rear digital image collection system to assist off-line analysis.

The probe vehicle was also instrumented with the Horiba On Board Emissions Measurement System (OBS 1300), comprising a Data Integration Unit (DIU), heated Non Dispersive Infra Red analyser (HNDIR), zirconia type NO_X and Air/Fuel ratio analyser (NO_X A/F), Annubar (modified Pitot style) flow meter, additional in vehicle and ambient air monitoring, data logger (laptop) and software, global position system (GPS), power supply unit and dedicated batteries. A schematic of the OBS-1300 is illustrated in figure 2. All exhaust sampling/monitoring was conducted via a purpose built exhaust attachment fitted to the end of the vehicle exhaust pipe. CO, CO_2 and HC concentrations were sampled using a heated sampling line and measured by HNDIR, with both sampling line and analyser cells operated at ca. 87°C. NO_X concentrations and A/F were measured by NO_X A/F probe, exhaust flow rate was measured by Annubar, exhaust temperature monitored by thermocouple probe, all mounted on the exhaust attachment. Exhaust pressure was measured by open line connection and DIU mounted pressure sensor. Vehicle speed and engine revolutions were also logged from external (vehicle) inputs. In addition, ambient conditions (local temperature, pressure and humidity) were monitored using dedicated vehicle mounted sensors. For further details of this system, see development work of Nakamura et al (2002). A summary of validation work, comparing OBS-1300 measurements with those from a 'certified' chassis dynamometer emission laboratory, is presented in section 2.

3. Experimental Work

The probe vehicle was deployed during off-peak day-time periods (1000hrs – 1530hrs) to collect sample journey time and real-world emission measurements. The '*before*' and '*after*' comparison took place during the weeks commencing 10th October 2005 and 26th September 2005 respectively. As the traffic management scheme was already in operation, the '*after*' scenario was monitored first, before reverting to the old signal settings. The probe vehicle was only deployed Tuesday – Thursday to negate variability associated with atypical traffic flows on Mondays and Fridays. The analysis of data collected from the two automatic traffic count sites located on the Leeds inner loop road during the survey period (see figure 1) confirmed that the traffic flows levels in the two scenarios were statistically similar at the 98th percentile level of confidence (*t*-test).

The probe vehicle was always run in a 'hot running' state. Prior to testing the vehicle was run at idle for 10 minutes, then driven over a 2 mile route before joining the test circuit. This was not only to remove the large variability and proportionally high levels of emissions associated with cold-starts, but also because the accuracy of the zirconia NO_X sensor can be compromised when CO concentrations are more than 2.5% (Nakamura et al, 2003), as often observed during cold-starts. The OBS-1300 emission monitoring instrumentation was operated in accordance with the manufacturer's operating procedures, which includes:

- Analyser and heated sample line at operational temperature;
- Zero-point (baseline) calibrations of the HNDIR analysers carried out at hourly intervals, with Span calibrations performed daily; and
- Data is quality assured and checked (QA/QC) by trained personnel.

Figure 3:Comparison Target (FTP75) and Actual Driving Profile (a) Time Series (b) Scatter

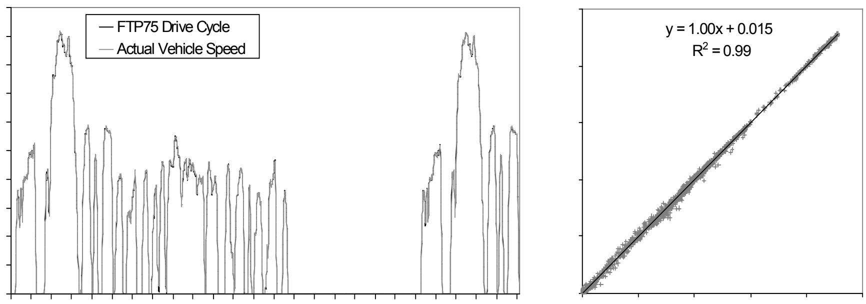
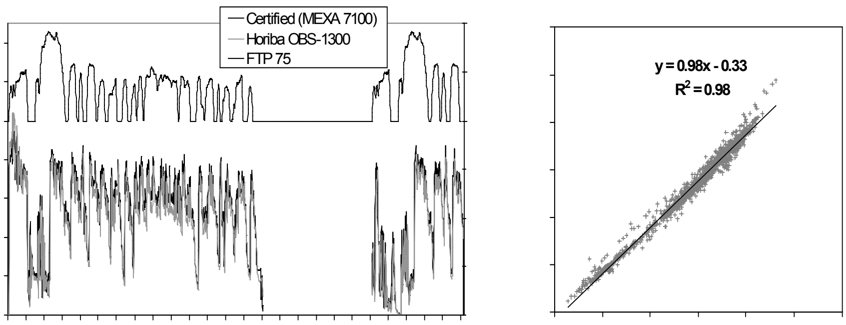


Figure 4: Comparison Certified and OBS-1300 CO Concentration (a) Time Series (b) Scatter

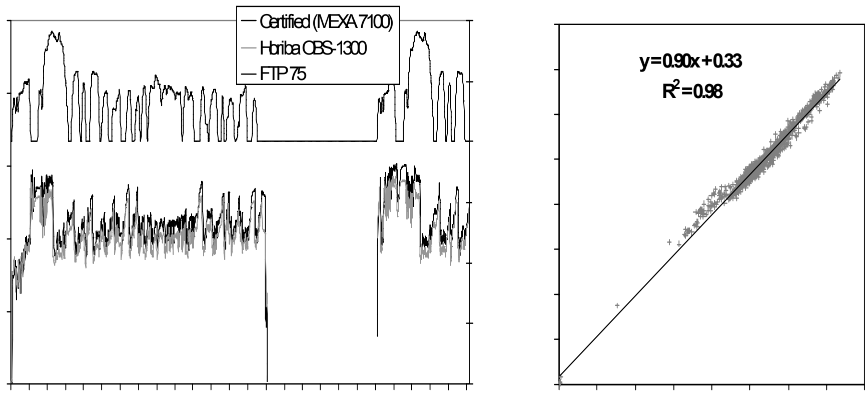


Validation of the on-board emission monitoring system

The desire to obtain transient emission measurements in real-world driving conditions, rather than from potentially unrepresentative legislative cycles, has given rise to the development of sophisticated ‘on-board’ emission-monitoring instrumentation. However the accuracy of on-board systems, in relation to certified measurement methodologies and instrumentation, is often discussed.

Researchers at the University of Leeds are undertaking a project to study REal-world Traffic Emissions Measurement and Modelling (RETEMM). Key components of this project are two on-board emission-monitoring instruments, namely: the Horiba OBS-1300 and a portable Fourier Transform Infrared (FTIR) spectrometer (Temet Gasmet CR 2000). Complementing existing literature (Nakamura et al, 2002 & 2003), a comprehensive validation study of the on-board systems was carried out to quantify the performance and limitations of the complementary instruments. The validation study was conducted at the University of Bath, Powertrain and Vehicle Research Centre (PVRC), Chassis Dynamometer (Zöllner 2 independent 126kW DC machines), Emission sampling (dilute flow CVS) and measurement (Horiba MEXA 7000 series analytical bench) facility.

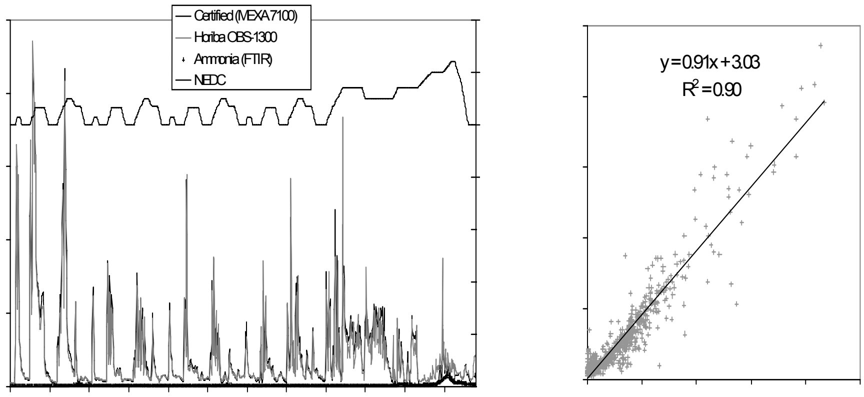
Figure 5: Comparison Certified and OBS-1300 CO_2 Concentration (a) Time Series (b) Scatter



A 1993 (EURO I) Ford Mondeo 1.8 litre (petrol) test vehicle was used in the validation study to ensure a range of concentrations would be observed. The vehicle was repeatedly driven over:

- US EPA Federal Test Procedure [FTP 75] including cold-start phase I;
- EU New European Drive Cycle [NEDC];
- Simple 2km test cycle emulating real-world urban free-flow driving conditions; and
- Complex 6km test cycle based on a single real-world journey, including periods of free-flow, congested and high speed/ power operation conditions.

Figure 6: Comparison Certified and OBS-1300 NO_X Concentration in 'Normal' Operating Conditions (a) Time Series (b) Scatter



Both 'standard' and 'real-world' drive cycles were used to validate the on-board instrumentation in a variety of operating conditions. Only a summary of the validation results, particularly focusing on the FTP 75 drive cycle tests are presented in this paper. Figure 3 illustrates the match of target and actual driving profile for a single FTP 75 (cold-start) test cycle. Figures 4 to 5 present comparisons of certified (MEXA 7400) and on-board (OBS-1300) exhaust gas concentration measurements for the pollutants CO and CO_2, respectively. A good correlation between instruments is observed for CO and CO_2, including during the cold-start phase (typically R^2>0.95). The correlation for HC was less precise (R^2 0.6-0.9, not shown here). However, this was not unexpected as the OBS-1300 and MEXA 7400 use different HC measurement methods (HNDIR and Flame Ionisation Detection, respectively). Non-linear relationships are often observed when comparing these techniques, especially over a range of driving conditions. As expected (Nakamura et al, 2003) the accuracy of the on-board NO_X sensor was significantly degraded when the vehicle was in cold or 'running rich' conditions. This is illustrated in figures 6 and 7 where NO_X concentration traces of measurements taken during a NEDC are compared with the vehicle in 'normal' (hot-running) and 'fuel-rich' operating conditions respectively. Encouragingly, in the 'normal' test, the NO_X probe performed well. During the 'fuel-rich', when in the presence of in excess of 2.5% CO, little correlation between the certified (MEXA 7400) and OBS-1300 NO_X probe measurements, was observed. Elsewhere, (e.g. Cadle et al, 1979; Karlsson, 2004) exhaust emission of ammonia (NH_3) has been attributed to the reduction of NO_X in the three-way catalyst under fuel-rich conditions. Clearly, the OBS-1300 NO_X probe has a strong cross-sensitivity with NH_3, as demonstrated by simultaneous NH_3 measurements obtained by FTIR. With these limitations in mind, during the traffic management scheme evaluation, the test vehicle was always operated in a 'hot-running' state. Finally, figure 8 illustrates the performance of novel OBS-1300 (Pitot) exhaust flow measurement, compared to the CO_2 tracer method.

Figure 7: Comparison Certified and OBS-1300 NO_X Concentration in 'Rich' Operating Conditions (a) Time Series (b) Scatter

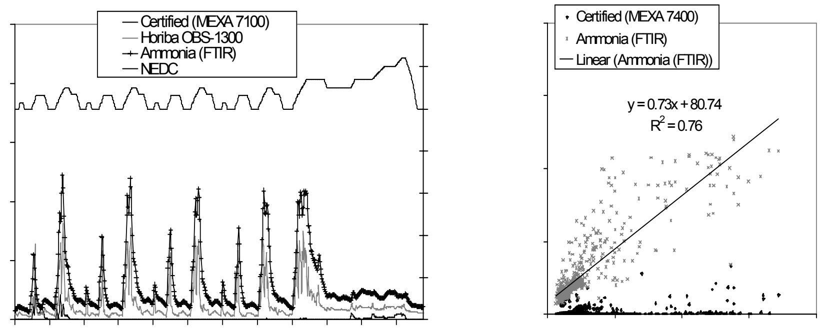
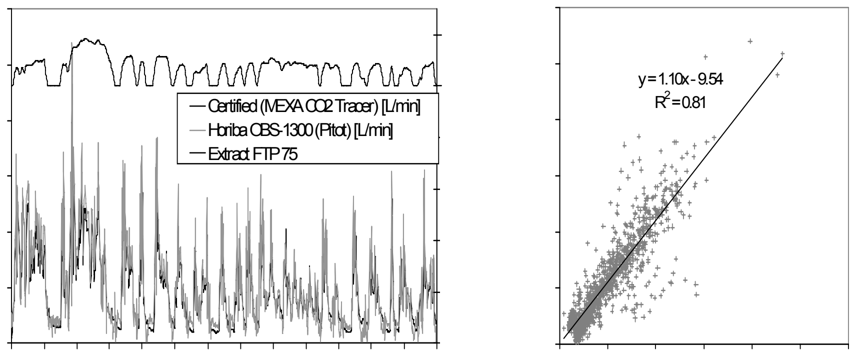


Figure 8: Comparison Certified (CO_2 tracer) and OBS-1300 (Pitot) Exhaust Flow Measurement (a) Time Series (b) Scatter



Results

The time-distance diagram (figure 9) summarises the driving profiles of the repeated test circuits in both the '*before*' and '*after*' scenarios. A key feature to note when the traffic management is in place, is that during four circuits the probe vehicle progressed smoothly around the loop road with minimal delay at junctions. This is demonstrated by the noticeably shorter journey times for these runs. This indicates that the signal timing 'offsets'/ vehicle progression settings, do allow a vehicle to travel quickly and smoothly around the loop road, but this was not always achievable. A comparison of the distance-emission traces for CO, HC and NO_X in figures 10, 12 and 13 respectively illustrates that there is significant variability in tail-pipe emissions of these pollutants. This variability is inherent of real-world conditions, as individual journeys are subject to changes in driver behaviour, other drivers' behaviour, local driving conditions, subtle changes in the vehicle/ emission control operating conditions, etc. This 'real-world' variability 'masked' the intermittent benefits of the traffic management strategy in relation to emissions of CO, HC and NO_X. However the four 'quick' circuits can be clearly identified in the CO_2 distance-emission diagram (figure 11) as the lower traces. CO_2 emissions can be broadly associated with fuel consumption. The faster and smoother traces therefore illustrate the potential of such a scheme to provide a more efficient transport service in terms of time, fuel consumption and greenhouse gas emissions. Figure 14 presents the measured mean journey time and tail-pipe emissions in both the '*before*' and '*after*' scenarios.

Figure 9. Time-distance comparison

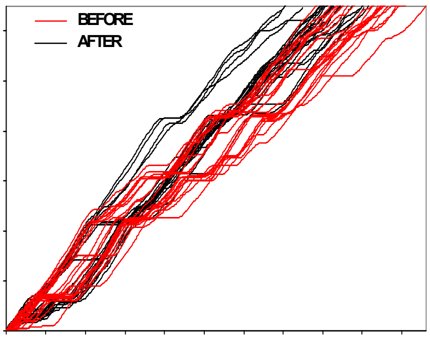


Figure 10. Comparison CO Emissions

Figure 11. Comparison CO_2 Emissions

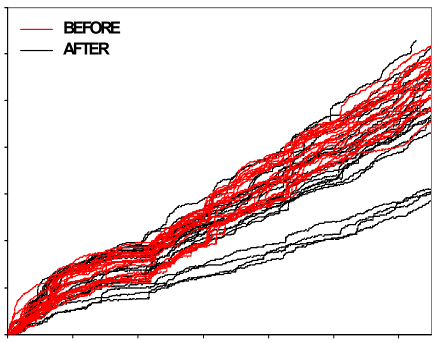
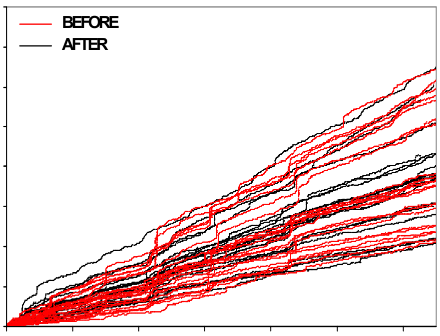


Figure 12. Comparison HC Emissions



Conclusions and future work

Traffic, emission and air pollution modelling frameworks commonly adopted lack the resolution/ capability to assess changes in tail-pipe emissions caused by local signal control and traffic management strategies;

Tools are needed by traffic and air quality practioners to allow action plans/ traffic management strategies to be developed and optimised to combat air quality in the worst effected areas;

Figure 13. Comparison NOX Emissions

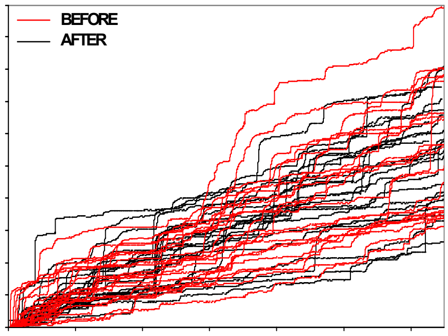
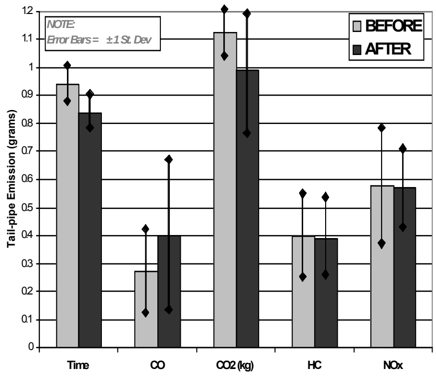


Figure 14. Comparison Aggregate Emissions



In the absence of suitable modelling tools, the Leeds inner loop road traffic management scheme was evaluated experimentally using a highly instrumented probe vehicle. Although the results reported here are only based on real-world measurements from one probe vehicle, it is hoped this study can be used to infer the potential of traffic management policies to help solve local air quality problems and advise the development of models;

The 'progression' signal timing policy was shown to inconsistently lower journey times, emissions of CO_2 (and hence fuel consumption). Future strategy improvements, possibly with real-time monitoring and intervention, may allow a higher proportion of vehicles to maintain a consistent progression around the loop road; and

Encouragingly the on-board emission instrumentation performed well in evaluation tests. Real-world emission monitoring instruments now look set to compliment traditional (chassis dynamometer/ CVS/ drive cycles) testing programmes and advise the development of vehicle, engine and emission control technologies.

Acknowledgments

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Changing Travel Behavior in Environmental Strategies: A New Research Approach

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Abstract

While there is consensus that present patterns of auto dependence are inconsistent with established goals for sustainability and climate change, there is little agreement on which policies and actions could lower overall auto dependence. This paper concerns the relationship between new mobility products/services and the propensity to change travel behavior. In a survey of 501 respondents, the Theory of Planned Behavior was first applied to establish base case conditions for key variables. Respondents were exposed to seven possible improvements to transit services. A follow-up application of the theory was undertaken to look for shifts in key attitudes. New products/services may influence 1) the traveler's personal inclination to change modal behavior, 2) her belief that a change in modal behavior would be socially acceptable, and 3) her belief that she actually could change the behavior. These three attitudinal categories were examined for four market segments for changed travel behavior. Research results suggest that new products and technologies dealing with a latent fear of being lost, abandoned or needing more information might contribute to an increased social acceptance of a lifestyle more dependent on transit and walking.

***Keys-words**: travel behavioral change, Theory of Planned Behavior, environmental strategies*

Introduction

It has been well documented in the professional literature that making significant changes in people's travel behavior in the direction of a more sustainable pattern will be difficult (Garling, 2005). A recent study undertaken in the United States by the Transit Cooperative Research Program (TCRP) addressed the issue of potential change in modal behavior in two innovative ways (TranSystems *et. al.,* 2006). First, the study examined the factors which influence one's choice of neighborhood in the same research project as the factors which influence the choice of mode once the neighborhood is held constant. Second, the study utilized research methods previously not applied to the issue of changed travel behavior in the United States. The project called for the use of theories developed largely in the field of public health intervention to be applied to the questions of selection of residential location supportive of walking and transit, and the alteration of existing modal patterns for less reliance on the private automobile. The theory applied, The Theory of Planned Behavior, is a widely accepted method for understanding the path from beliefs to attitudes, to intent, and finally to behavior (Bamberg, Aizen & Schmidt, 2004).

A program to lower dependence on the private automobile will require a wide variety of strategies. This paper examines the question of what products and services designed to improve non-auto related mobility might be most effective as part of an over-arching strategy for lowering of Vehicle Kilometers of Travel, and increasing the role of walking and transit.

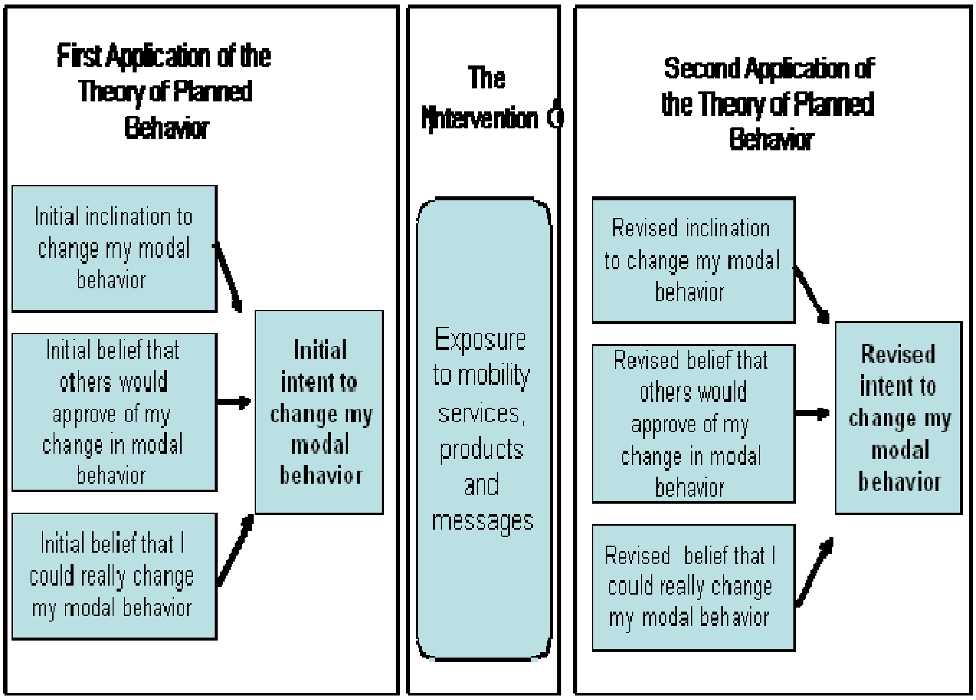
Method: The Theory of Planned Behavior

As part of a larger research plan, the project conducted a survey of 501 participants to explore the factors influencing modal choice once the residential location decision had been made. All of the participants were from metropolitan areas with transit service who either had recently made a residential decision, or were contemplating a residential location decision in the near future. Respondents were drawn from existing samples of individuals who had previously responded to questions about transportation via an Internet-based panel (Adler *et al.*). The sample represents this group of potential transit users, and was not designed to reflect the broader American population.

The Theory of Planned Behavior posits that the immediate antecedent of behavior is intent, as modified by the perceived ability of the subject to undertake the change in behavior. Intent is influenced by three levels of considerations. The Attitude toward the Behavior reflects the subject's inclination to want or not want to undertake the behavior, based on assessments that the new behavior might be desirable, pleasurable or interesting. The Subjective Norm reflects the influence of the subject's immediate personal network of family, friends, and other sources of peer influence. The Perceived Behavioral Control reflects the judgment of the subject about how difficult (or easy) it will be to undertake the new behavior, representing the factor of self efficacy (Stradling, 2004). In the surveying process, the participant is asked a series of questions designed to bring out the latent forces influencing the formation of an intent, in this case the intent to alter one's lifestyle to have greater reliance on transit and walking, and less dependence on the private automobile.

The survey instrument was constructed with three clearly definable phases. First, a "pre-intervention" application of the full Theory of Planned Behavior was undertaken concerning a subject's initial intention to change personal transportation patterns. Second, an "intervention" was undertaken in which the respondents were exposed to different messages and to candidate products and services that might improve the acceptability of the alternative transportation lifestyle. In the third phase of the survey, a second application of the theory was undertaken to allow the documentation of any shifting in scalings which occurred by the end of the survey. That study design is illustrated in Figure 1.

Figure 1: The Study Design



During the intervention phase of the survey, the 501 respondents were exposed to seven separate concepts for making it easier to undertake a lifestyle with more dependence on transit and walking. They were asked to go through several exercises which forced them to make decisions about the mobility options offered. Those seven candidate products and services were:

- Better traditional service to the downtown
- Better traditional service to the rest of the region
- A ‘Smart Card’ to handle all payment requirements
- A community shuttle bus for neighborhood trips
- A community door to door shared taxi system for neighborhood trips
- A car sharing vehicle available within the neighborhood
- A ‘Smart Phone’ which would:
    a. tell the user when the next bus would arrive
    b. tell the user how to make a transit trip home from the actual location if she were lost
    c. have a ‘911’ button that would report her exact location to the police for any reason

The study design allowed for the documentation of the extent of shift in answers to key questions based on the construct of the Theory of Planned Behavior; this was undertaken for the full sample and for four separate market segments created in the project.

Table 1: Characteristics of the Four Market Segments for Mode Change (Highest Values Emphasized)

Four Market Segments	Number of cases	Initial Measure of Intent (Scale from 1- 7)	Revised Measure of Intent (Scale from 1-7)	Transit mode share to work	"For me to reduce pollution by using my car less would be IMPORTANT"	"For me to walk and take public transportation more would be DESIRABLE"
Transit Loyalists	68	**5.19**	**5.45**	**58.8%**	5.50	**5.50**
Environmental Changers	150	4.02	5.44	18.7%	**6.43**	5.09
Happy Drivers	132	3.71	4.30	25.0%	5.17	4.25
Angry Negatives	151	2.26	2.87	14.6%	4.28	2.93
Full Sample	501	3.57	4.36	24.6%	5.32	4.27

Method: Four Market Segments for Modal Change

The research created a market segmentation to help explore the variety of beliefs and attitudes associated with the propensity to change modal behavior toward a greater reliance on walking/transit, and a decreased reliance on the private car, consistent with previous studies (Anable, 2005). Of the four market segments identified, two can be characterized as positive market segments, and two can be characterized as unlikely for change in modal behavior. The four segments for change in modal behavior are summarized here, ranked from highest propensity to change to the lowest.

1. The Transit Loyalists

This group is characterized by their present use of, and understanding of, public transportation services. For them, issues such as the safety of transit services or fear of getting lost are not considered to be determinant, and therefore not important to be solved with new products and services. This group tends to have a very strong idea of what transit is, and how it can improve on doing what it presently does.

This group is characterized by their belief that, if certain conditions are improved, they could become transit users, even though transit does not now live up to their standards. They are further characterized by their belief in environmental causes as a motivation for a change in modal behavior.

2. The Happy Drivers

This group likes to drive, values its automobiles, and has no propensity to like the attributes of a transit oriented life. It should be considered as the moderately negative group.

3. The Angry Negative Group

This group is characterized by its low evaluation of just about every aspect of altering modal behavior, and by the radically low intent of its members to alter their own transportation behavior.

Each variable was structured on a seven point scale, with one as the lowest and seven as the highest scalings possible. The variation in scalings by market segment can be seen in Table 1. The experience of the respondent with actual transit service influences the nature of the attitudes held: Table 1 shows that the Environmental Changers have the highest potential to think that they *should* be driving less to reduce pollution, whereas the Transit Loyalists think that it might be *desirable* to rely more on walking and transit.

Table 2: Increase in Intent and other Direct Measures, by Market Segment. (Change in Scalings from 1-7; Highest Values Emphasized)

Four Market Segments	Increase in 'Intent' to change modal behavior	Increase in 'Attitude' (Personal inclination to want the change)	Increase in 'Perceived Behavioral Control' (Belief that I can change the behavior)	Increase in 'Subjective Norm' (Belief that others will approve and support change)
Transit Loyalists	0.26	0.17	0.11	1.38
Environmental Mode Changers	**1.42**	**0.28**	**0.96**	**2.34**
Happy Drivers	0.58	0.17	0.39	1.49
Angry Negative Group	0.61	0.01	0.42	1.30
Full Sample	0.80	0.16	0.53	1.67

On a scale from one to seven, the “Transit Loyalists” gave a scaling of 5.2 points to the Initial Measure of Intent to Change Modal Behavior, with the “Angry Negatives” registering on only 2.3 on the same scale. There was very little variation by income level over the four groups.

Results: what groups shifted and why?

Table 1 shows that at the commencement of the survey, the Transit Loyalists displayed the highest level of Measure of Intent to change their transportation behavior to become more reliant on transit and walking in the future. After the intervention was completed, and the Theory of Planned Behavior was again applied, the Environmental Mode Changers had shifted their level of Intent to the point where their stated Intent was about as strong as that of the Transit Loyalists.

Table 2 shows the extent of shift in Intent from the base case application of the Theory of Planned Behavior to the post-intervention application of the follow-up survey. This table shows that the Transit Loyalists showed the smallest level of increase in Intent of any of the four segments—smaller even than either of the two negative groups. By contrast, the Environmental Mode Changers showed by far the highest level of shift in Intent; they also showed the greatest increase in Perceived Behavioral Control, *i.e.* the belief that they could succeed in changing their behavior.

Table 2 shows that the most pronounced shift in attitudes concerned the belief that *others* would approve a change in one's transportation behavior (Subjective Norm). Conversely, exposure to the new transit services had the least impact on one's own propensity to believe that the proposed behavior was desirable, pleasurable or interesting (Attitude). Looking both at shift in Attitude and shift in Perceived Behavioral Control, it is clear that the Transit Loyalists were simply not impressed with the strategies offered to them. By comparison, looking at the shift in Attitude and PBC for the Environmental Mode Changers, they were more "moved" by the experience of the intervention, and their measure of Intent shifted accordingly.

The most dramatic shift, however, occurred in the change of ratings assigned to Subjective Norm, which looks at the impact of one's personal social network in one's formation of intent to change behavior. This pattern occurs for all four segments, but is most dominant for the Environmental Mode Changers segment where an increase in the measure of Subjective Norm of about 88% took place

Using a sample of only members of the Environmental Mode Changers segment, correlations were calculated between Revised Subjective Norm and all candidate independent variables. The two variables with the highest correlation were "With the new services available, I would have less concern about being lost or stranded by missing the bus or train" and "If I were to use the new services, I would feel safer from crime and other disturbing behavior." Regression equations were created to better understand the nature of the increase in the belief that others would approve of a personal change in modal behavior. The strongest explanatory power was provided by the same two variables, concerning fear of abandonment, and fear of crime.

Table 3: Ranking of Attributes of New Products and Services

Attributes	Attribute's Rank	Full Sample	*Environmental Changers*	*Transit Loyalists*
I would want to know exactly when the bus or train would arrive	Highest	6.05	6.56	6.01
I would want a transit pass so that I never had to worry about having cash	2	6.02	6.55	6.28
I would want to be able to walk to a nearby store or coffee shop	3	5.97	6.58	6.29
I would want transit service that connects me with the rest of the region	4	5.92	6.48	6.22
I would want to be sure that a taxi would come at any hour	5	5.37	5.95	5.35
I would want a shuttle service to take to activities within the neighborhood	6	5.37	6.07	5.57
I would want frequent transit service (rail or express bus) to the downtown	7	5.28	5.95	5.97
I would want a car on my block that I could rent by the hour (car-sharing)	Lowest	4.42	5.22	4.00

Results: Ranking the Desired Attributes and Products

After the completion of the initial Theory of Planned Behavior survey, the respondents were asked to think about an imaginary neighborhood that already had good sidewalks, and good destinations to walk to. A stated requirement of that imaginary neighborhood was living with fewer cars than at present. The survey question was:

> "Thinking about this imaginary neighborhood, which transportation options would you need to live with fewer cars in your household?'

Table 3 presents the results from this set of questions, presenting the rank order of the attributes offered for the full sample, and the mean scalings given by each of the two most positive segments. In response to this question, the respondents gave the highest rating to: "I would want to know exactly when the bus or train would arrive." The second most desired function was, "I would want a transit pass so that I never had to worry about having cash." Ranking third among the list of desired attributes was "I would want to be able to walk to a nearby store or coffee shop." The least desired attribute was that describing the need for car-sharing.

When a later exercise in the survey forced the respondents to choose between paired options, the full sample gave highest ratings to products providing traditional commuter services, with lower levels of interest in community taxi, community bus or car sharing. Interestingly, the "Smart Phone" product with both bus arrival information and a direct link to the police was ranked as lower priority than the actual improvement of services.

Discussion

The successive application of two iterations of the Theory of Planned Behavior to the question of the possible impact of new services and products revealed both attitudinal shift and lack of attitudinal shift. After the respondents completed the exercises in which they dealt with details of the new products and services, there was little change in the scalings describing their Attitude toward the Behavior, reflecting their assessment of the new behavior being desirable, pleasurable or interesting. However, for all market segments, the greatest shift occurred in the Subjective Norm, reflecting the belief that the transit lifestyle would be more accepted by others in one's social network. If the new products and services could deal with certain underlying concerns about transit, a lifestyle with increased reliance on transit would become more socially acceptable.

The research results might suggest that the increased level of Subjective Norm experienced by the Environmental Mode Changers is associated with latent underlying concerns about crime and abandonment (being lost) on the public transportation system. Looking at the empirical data, we know that the scalings for the perceived beliefs of the *others* (wives, husbands, neighbors, etc.) improved significantly more than did one's own personal rating of the same situation. Perhaps, it is possible that reporting that *others* would feel happier with a higher degree of safety reflects some latent concerns that one has about one's own condition, but is reticent to admit.

The results of the ranking of candidate products and services suggest that further research may be needed. On the one hand, the market research clearly reveals a direct interest in learning about the arrival time of the next bus, as well as a latent underlying concern about being lost on the system, and being exposed to crime or unpleasant behavior. At the same time, when confronted with a specific product designed to deal with these same concerns, the respondents ranked it lower than improvements to conventional services. In terms of priorities, the respondents wanted actual improvements to services first, and information about those services (or gadgets providing information) second.

Future packages of mobility services designed to support a lifestyle with lowered auto dependence will probably include neighborhood-based buses, shared taxis and car sharing (Coogan, 2003). It is clear from this research, however, that concepts which provide services different from the well understood commuter work trip will need to be carefully explained to population. The respondents in the study were given the *opportunity* to reveal a preference for neighborhood-based services, with locally managed shuttle buses augmented by community shared-ride taxis. However, the respondents in our sample gave highest ranking to the products which they knew best (traditional commuter services), and lowest to those with which they were not familiar (car-sharing.)

It can be argued that the community shuttle bus and the community shared ride taxi represent a more radical departure from present modal behavior than the most favored options, and thus were less popular. In the US experience transit gets a high mode share for the work trip, but is simply *not* the mode of choice for getting to the community center, to the doctor, or the neighborhood shopping center. When asked about what they would like to be offered, the respondents showed little interests in options that would alter their basic patterns of modal choice. Evaluations (rankings) were highest for services about which they had actual experience and actual knowledge. Further research in this area could examine the influence of age, specifically old age, on the attitudes towards locally oriented services

Attributes describing the need for new products and services were nearly always valued higher by the environmentally motivated group than by the group with an existing experience base with transit. New products designed to deal with fear of being lost and fear of crime were of higher interest to the group which felt that transit was important, but needed to be improved to their standards before a behavioral change could take place.

This study has revealed an underlying concern for the feeling of being lost, abandoned or not knowing when a bus or train will arrive. We believe that new services and products can be designed to deal with this revealed deficiency, but understand that the potential market will have to be convinced that new services and products can be effective. As mobility planners address this issue, they must be prepared to acknowledge the separate attitudes and beliefs of separate market segments. Products and services designed to deal with these latent concerns will need to be introduced with some care, perhaps through a staged process of implementation.

Acknowledgments

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Defining the issues: What is "Environment and Transport" all about, and how can the fundamental conflict between transport development and environmental protection be resolved?

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Abstract

It is often assumed that in "transport and environment" one has to decide: One can either protect the environment (and reduce transport) or develop transport (damaging the environment). Then, conflicts are manifold.

The paper suggests to change the underlying "objective of transportation" and to separate ends and means, objectives and instruments. The true objective is to allow for "Access" with minimum costs, thus minimizing transport. Then, economic efficiency tools can be used in "transport and environment", too. This allows sustainable development, minimizing not access but total private and social costs for society, including environmental costs. To give proper scarcity signals a continuous process towards less lying prices is unavoidable.

In an example for Saxony, the complete estimation and internalization process is described. Total uncovered external costs are about 1500 € per person per year (2001). The process to reduce these costs includes communication and awareness and acceptance raising.

***Keywords**: Objectives, Definitions, Access, True prices, Efficiency, Sustainable Development*

Introduction

The relation between “transportation” on one hand and “the environment” on the other hand is a difficult one, and has been for quite some decades. Considerable efforts have been made to reduce the environmental impact of transport systems, and although in many cases there have been impressible successes, the overall relation is still stressed. The European Environmental Agency EEA describes the actual situation on their English website as follows (EEA (2006)):

> “Transport
> An efficient and flexible transport system is essential for our economy and our quality of life. But the current transport system poses significant and growing threats to the environment and human health, and even defeats its own objectives (‘too much traffic kills traffic’). The drastic growth in road transport and aviation is the main driver behind this development. The sector is the fastest growing consumer of energy and producer of greenhouse gases in the EU. Technology and fuel improvements have resulted in marked decreases of emissions of certain pollutants. Yet urban air quality in most European cities is still poor. Roads and railways are cutting natural and agricultural areas in ever-smaller pieces, threatening the existence of wild plants and animals. Traffic noise causes human health problems, and over 100 people die on the EU’s roads every day on average.“

This demonstrates the fundamental conflict the field “transport and environment” is facing: On one side, transport is necessary for society, and it is assumed it is in our interest that transport grows. On the other side there are considerable negative environmental effects, and it is assumed it is in our interest that these effects get smaller. Between both sides no general compatible solution seems to exist. This effect divides society into two fractions, and people working in the field “transport and environment”(“transport ecology”) face this conflict as well. However, is it true that we can have only one, that we have to decide on which side we stand? And, moving from individuals to society, is it true that society itself can have only continued transport growth or a better environment? Technical solutions (e.g. catalytical converters) have been very helpful in a number of fields, but the underlying basic trends were not changed. What should society concentrate on? What to do, from an individual point of view, and from society’s point of view? Which recommendations are to be drawn from that for this conference? The following contribution tries to develop a solution out of this conflict and assembles a list for future work.

Society’s Objectives in “Transport” and in “Environment” today

First, the situation within society is to be analyzed more clearly. When discussing the relation between transport and the environment, it is helpful to start with the overall objectives of society in both fields. From that, helpful conclusions can be drawn.

What are the commonly accepted objectives in these two fields? When analyzing the priorities of our societies, e.g. the revealed preferences in official budgets, transport investments hold a dominant position: A lot of money is used to follow "transport objectives". The general opinion within society understands that all transport measures and actions should "improve transport conditions": Engineers should design and build infrastructures and vehicles, both should be "optimized" and made more comfortable, safer, faster and more inexpensive. The actual priority lists of cities, regions and nations often show that it is intended to allow vehicles, persons and goods to move easier, cheaper, safer, faster etc. In the following discussion, this understanding is defined as

Transportation Objective I of society: To improve transport conditions for the users.

Objective I makes the daily (transport) life of people easier, so it is welcomed by almost everybody.

With regard to environmental objectives, citizens, politicians and people often share the perception that the level of environmental damage and levels of pollution, noise, CO_2, aerial consumption, landscape issues, energy use etc. should be reduced. This understanding is defined as

Environmental Objective II of society: To reduce the environmental burden of transport.

Objective II does not change the daily life of citizens too much as it affects more general principles. As long as no detailed measures for persons are feared, general consensus can be expected, too.

Bringing both these objectives of society together is not easy. While it is simple to put it in one single phrase ("Our societies should make transport easier and safer and cheaper and faster and at the same time the environmental costs/damages should be reduced" - in many speeches this is exactly what is said) this statement in market place societies contains a contradiction in itself: Of course, if transport is made faster, safer, easier, more attractive, the population and society will consume more of this good. Thus, the amount of transport itself is increased. Transport growth has many causes, but nobody should be surprised if we get more transport once we make it cheaper or faster. More transport, here, is understood as more passenger-kilometres or vehicle-kilometres or ton-kilometres travelled. Of course, wider and longer travel distances result under ceteris paribus conditions in more energy use, more noise, more pollution and more environmental damage – clearly violating objective I of society. The simple "we want to have both" phrase contradicts itself.

This discrepancy is felt, and it is one reason for the general impression "we can have better transport conditions or more environmental quality but not both at the same time". The conflict spreads into our areas of expertise: "Transportation and environment" is often not perceived as one common area of research or work but instead as a field "where two enemies meet".

Several conclusions can be drawn from that understanding of the two objectives I and II:

- If the situation is as described, then technology is the only help: Only better technology can combine both objectives. Of course, if an engineer develops an engine or a system or some method "to travel better" with the same amount of environmental burden, or to do the same trip with less environmental burden, or to improve both, then everybody feels happy.

- That is why "transport and environment" became a technology-oriented field first. Such technical "win - win" solutions meeting objectives I and II at the same time did and do exist, and looking for them was a main reason to start working in our field decades ago.
- However, the more such solutions are found and implemented, the lesser the chance to continue on that path. The reduction of exhaust emissions from 1 g/km to 0,5 g/km may be easy and cheap, but the reduction from 0,01 g/km to 0,005g/km may be difficult and expensive. From a point those changes become too expensive, and then changing travel behaviour is the only remaining option. This, of course, leads to bitter battles, whether to reduce my travel or not. Those discussions separate people working in our field into experts of one or the other side (but never both). For the field "transport and environment", this is an unhappy development: "Transport experts" against "environmental experts". When it comes to decisions, society has to choose between faster, easier, cheaper transport on one side and a cleaner environment on the other side.
- Benefits of more attractive transport systems are immediately felt by everybody travelling while benefits of a cleaner environment are felt only indirectly and after longer times. It is no surprise that many decisions are taken preferring the "improvement of my travel conditions now" over some "vague environmental improvement for all later". Thus, the environmental objective II is not reachable: Then, the EEA – perception in the introduction can be understood.
- As long as objective I is followed by society and as long as transport conditions become more and more attractive, the environmental situation gets worse under ceteris paribus conditions. Here, a typical ecological feedback system develops: If in conflict between opposing objectives I and II a decision is made to follow (transport) objective I, then increasingly objective II (environment) is failed – creating pressure to reverse the original decision. The more we follow (often short term) transport objective I, the higher the damage in (often long-term) environmental fields, and the higher the pressure to follow objective II. The assumed conflict between transport and environment cannot be resolved by concentrating on objective I alone.

The overriding question remains as to whether the understanding described above is a sound basis for society and for our work. In reality, this perception is widespread, but is it based on a well-founded political decision? Can this understanding be sustained scientifically? Is it helpful at all? In the next paragraph, a change of transport objectives will be suggested.

Changing Society’s Objectives in “Transport” and “Environment”

Is “to improve transport conditions” really a good objective function for society? In practice it is the main objective of planning and transport politics, but can we conclude that the easier, cheaper and attractive transport is, the better for society? The society with the cheapest, fastest and most attractive travel is the society with highest welfare? Unfortunately, this is not the case: The society with the cheapest, fastest and most attractive travel in market place conditions is going to be the society with the most transportation of all: With the most kilometres travelled, with high energy consumption, disperse settlements, loud and polluted streets, high CO_2 – emissions etc. Going to extremes, one recognizes that following objective I only does not ultimately lead to completely satisfying conditions in society, especially when considering dynamic settings and long-term effects.

In addition, objective I is also not completely covering the original purpose of travelling. We all are primarily not travelling because it is easier or less easy but because we want to have access to services: Everybody wants to satisfy certain needs by travelling. Thus, the basic reason to travel is not to travel more or fast or inexpensive but to satisfy human needs which cannot be satisfied staying at home. Persons or goods have to change their location in order to satisfy some needs; e.g. if people are hungry they want to have access to food, or they want to visit friends, to they want to go for a walk or a sunday ride on the motorbike through the mountains. All those individual motives are to be accepted, and they are the original and basic reason for people to move. ACCESS and the needs to change a location are the basic reasons for transportation. As human needs require also goods transports, with regard to needs there is no big difference between passenger and goods transport.

Basic human needs play a fundamental role in human lives. They were even considered to be the basis of the famous Brundtland definition for sustainable development. According to Mrs. Brundtland and the UN (1987), sustainable development is defined as development

- which meets the needs of the present generation
- and allows for future generations to meet their needs.

Needs are the basic reason for our behaviour, and with regard to transportation, they sometimes can be realized only by changing locations: travel becomes necessary. As these needs represent a basis for all life, they should serve as a basis for transportation, too. The transport objective of all measures and actions of society should also concentrate on needs instead of instruments. Consequently, it is suggested to shift objective I away from “to “improve transport conditions” towards “to allow access to services”. For all future work, therefore, I suggest to change the transportation objective:

New Transportation Objective Ia: To allow for ACCESS to services for all people.

Then, societies have to measure and check and allow for access for all groups, areas and people This holds true for all countries of the world, less or more “developed” or “industrialised”: The specific problems may differ but the underlying tasks are the same. Instead of relying on “kilometres” a much more appropriate measure of benefits would be to record needs satisfied. The ratio between ends and means can be quite drastic:

- A society can satisfy quite many needs with only little use of instruments, e.g. in an urban settlement with attractive options for all needs nearby, using walking and biking and public transport etc.: High level of access with little resources and costs.
- Or a society can satisfy location changing needs with a lot of instrument use, requiring people to travel long distances: Low level of access with high levels of pollution and costs.

According to economic wisdom, one always has to separate between the ends and the means, between the needs and the resources necessary to satisfy them. This ratio is called efficiency. Transferring these terms to transport, a new set of conclusions is evolving:

- The overall objective of society’s transport actions and measures is to allow for access; options are to be provided for everybody to satisfy those needs (under some framework conditions). No society can claim to be on a path towards sustainable development if some people do not have a chance to satisfy those needs, see the first part of the Brundtland - definition.
- These objectives can be reached by using different instruments, different amounts of resources and different total cost. An efficient society is a society in which those needs are satisfied at the smallest costs/resources. To satisfy a given level of access societies should install the system with the smallest possible level of transport. Transport should be minimal when a certain level of satisfied needs can be reached with different amounts of travel.
- In addition, the second part of the Brundtland definition asks to take care of future generations´ needs. Their needs are not to be known today, but there is only a chance to satisfy them then if we reduce the environmental burdens of our behaviour today. That is, we have to reduce the risks connected with our behaviour: To satisfy our needs with less consumption and pollution and resources. The conclusion here is the same as above: Transport should be minimal in order to give future generations a chance to satisfy their needs then. Please note: Transport should be minimal (less fuel, noise, pollution, mileage), level of ACCESS should NOT be minimal.

These small changes in the understanding of our transport objective have the potential to significantly reduce the conflict between environmental and transport experts. Now, both sides can agree on serving the location change needs of today’s generation, and transport planers could be happy with such an agreement. The agreement, of course, includes a provision that needs are to be satisfied in an efficient manner with the least level of transport. The least possible level of transport results in return in the smallest burden on the environment, making environmentalists happy. And people working in ”transport and environment” can work with one consistent set of objectives. As efficiency then is a key parameter damage costs in society get smaller. For details of the discussion and the conclusions, see Becker, Gerike and Voellings (1999).

Individual’s Point of View

At that point, the focus of attention has to shift from society’s point of view to aspects of individual people. When making decisions about travel people match supply side options for movement with demand side needs for movement. This personal weighting plays the crucial role in transportation - and it has to be guaranteed that this role stays with the individual because only the individual person knows the acceptability and suitability of this option in the specific situation.

The decision, which travel need is to be satisfied with how much instrument use is an individual one – but does this guarantee efficient decisions? How can society be sure that all citizens choose travel behaviours which allows them to fulfil their needs with the least resources? Does that mean that we have to force our citizens to behave in a certain way?

The answer to that rhetorical question is of course “No”. People have to decide for themselves – but they care mostly about their private costs, private comfort and private benefits. Efficiency, on the other hand, is essential for the entire society, and society has to take care of all costs, those of the users and those of others. As there is a gap between single user and society, there has to be some way to “tell” private people about the real size of costs and benefits to society. With benefits, that is not a problem as practically all benefits of a certain trip rest with the user of transport (traveller or sender/receiver). With costs, however, we share a significant problem because large parts of the total costs of travel are not paid by users. These costs are externalized

- onto other people: Infrastructure costs paid for by other taxpayers, costs of noise abatement for house owners, sicknesses because of vehicle emissions, etc.
- onto other areas: Costs from high ozone concentrations in the troposphere, nitrogenous depositions in other parts of the world, “waste disposal” of European cars in Asia, etc.
- onto other times: Debt repayment for today’s transport costs, flood protection for future generations due e.g. to climate change effects, costs for the recycling of our vehicles, etc.

All these costs should be considered and included in the decision of an individual person. The only possible conclusion is that our societies have to embark on a process to make these costs visible to the user: to establish “true prices”. Economically, it was known for almost a century that no efficient allocation is possible if significant external effects exist. Transportation is possibly the largest sector of society where externalizations are allowed, e.g. in air travel (costs of emissions at high altitudes) or ship travel at high seas (high - sulphur fuel), not to mention road transport worldwide.

The conclusion for the field of “Transport and Environment” is obvious: All our efforts will never provide efficient solutions and reactions if no steps towards less lying prices are implemented. Truer prices are not everything but without truer prices everything else comes down to almost nothing. True prices, see e.g. the ninth principle of the OECD –Vancouver principles for sustainable transportation (OECD (1997)) should cover all costs: Costs for the user as well as costs for other people, other regions and other generations. Prices should “tell the truth as much as possible”.

This task is a huge and long ranging one: We will never know exactly the costs of all travel today for all other generations (because those costs occur in the future). It is obvious that we will never know the “complete and true” costs of a certain trip today. Detailed and “final” internalization prices will never be able to be given. However, this does not mean that we can forget about true prices at all: As “sustainable development” comes down to an endless sequence of steps, the path towards less lying prices also comes down to a process in which societies have to measure and estimate external costs as correctly as possible. This is of course not only a question of pure “scientific work”, but dissemination, political information and acceptance aspects are also important. Cost estimates of “truer prices” will be different from city to city and from nation to nation, and they will change over time – but the process of best possible internalization has to be maintained.

The task here is to design a process: To estimate maybe a lower bound of costs today, to determine appropriate (and cost effective) paths of internalization, and to implement this path. After some time, effects have to be evaluated: Did the damage costs in the considered area go down sufficiently (if no, we have to raise the price; the first guess was too small)? If the price was too high, this also is to be corrected. Such a process will have no end, and it will be different for each society. If we do not embark on this process towards less lying prices, we will have no efficient allocation of resources – that is, we will waste resources. Not embarking on a process towards less lying prices is not economic (because it is not efficient), it is damaging the environment (because we could reach the same results with less environmental damage), it is not social (because the inefficiencies keep us from helping disadvantaged groups), it is unethical (because we use privileges we are not willing to grant to others), and it is not sustainable (because the carrying – capacity of Earth may be damaged).

The difficulties of such a process should not be underestimated, public acceptance of internalization measures is still low but rising, see London or Singapore or Oslo or Stockholm. For the Free State of Saxony, a chain of research at Chair for Transport Ecology went through eight different stages of such an internalization process up to today (see Gerike and Seidel (2005) and Becker, et al. (2002)):

- First a methodology to estimate external costs (and benefits, which are minor) was developed.
- Next, external costs for certain areas were determined. Currently, those areas included are uncovered accident costs, noise, air pollution, climate change, nature and landscape, area consumption, up- and downstream- and segmentation effects, (the latest being almost negligible).

- Then, costs were visualized in a Geographical Information System to show external costs for a certain mode of travel, a certain link of the network, a certain geographical area, a certain vehicle, time or pollutant, etc. In that database it is even possible to estimate the implications of different transport planning measures onto externalities.
- Next, appropriate internalization measures were identified for each environmental field.
- The different measures were combined in a push and pull package, combining measures with usage of the revenue. Revenues can be spent to generate options for the population to avoid paying higher prices. The main role of internalization is NOT to create fiscal revenues but to give incentives to avoid paying those costs. In our package, revenue from polluters was spent for alternative options, public transport, walking and biking, but also measures to promote small neighbourhood shops nearby and dense urban land use were included. All reactions help to avoid paying higher prices, and efficiency increases as pollution gets more expensive.
- Effects of the package were determined with regard to travel behaviour, mode and destination choice, vehicle characteristics and all types of private and environmental costs. A “2020 with internalization” scenario was compared to “2020 business as usual”. The internalization scenario provided the same level of access but with smaller private costs and less environmental damage: Noise –6%, Pollution –11%, climate change –21%, uncovered accident costs –31%.
- Next, acceptance was measured. We polled groups of policy makers and citizens to determine their reaction and attitude. Results can be used to increase public acceptance considerably.
- In parallel steps, information and communication efforts were included. At this time, reports and communication brochures for the general public and policy makers are prepared.
- It should be noted that this process usually starts all over again several times. If policy makers decide to change the environmental priorities or the package itself or the type of measures, previous stages have to be repeated. For results about the external cost estimations and the push and pull package see figures 1 and 2.

Figure 1: Results of the estimation of externaleffects for the Free State f Saxony 2001

[Mio. Ū2001]	road	rail	Air up to 1000 feet	Inland waterway	All modes
Uncovered accident costs	2.349	0	<1	0	2.349
Noise	400	102	2	n.b.	504
Air pollution	1.508	61	<1	3	1.572
Climate change	1.149	42	15	3	1.210
Nature and landscape	198	29	10	n.b.	237
Area consumption	98	n.b.	n.b.	n.b.	98
Separation effects	1	1	n.b.	n.b.	2
Up- and downstream- processes	534	41	2	1	579
Total	6.237	276	29	7	6.550

Figure 2: Package of push and pull measures for internalisation in Saxony (time frame: 2020).

SFP1 pollution charge for cities	City entrance: 1,70Ū per day
SFP2/FSP1: parking management	Shopping malls outside cities: +1Ū / hour
SFP3a: pedestrian zones, car free areas SFP3b environmental standards in public transport SFP3c: biking and walking SFP3d car sharing SFP3f: restrictions for polluting vehicles	zoning EEV for all buses bike system, parking, signalization better organisation noŠdriveŠ zones
FSP2 promotion of transport saving infastructures	development only with public transport
KSP1: CO_2- tax	petrol: +0,32Ū/liter, diesel +0,36 Ū/liter
KSP2c fuelŠ saving driving behaviour	Information
KSP2d renewables and bioŠ fuels	support for filling stations
VSP1 higher standards for driverÕs liability	higher rates for accident drivers
VSP2a speed limits	Autobahn: 120 km/h, urban areas 30 km/h
VSP2b improvement of driversÕ education	several measures
VSP2c higher safety standards	limit for drunk driving reduced to 0,0ä

Conclusions

To resolve the existing fundamental difference between “transport growth” and “environmental protection”, a common set of objectives and efficiency tools is essential. This leads to conclusions

- 1. One has to separate between ends and means in transport: The objective is to provide ACCESS to citizens. To do that, transportation systems should be developed in a way that minimizes the necessary resources and total costs for the desired level of access, hereby minimizing transport.
- 2. The costs of travel are well known and measured, but access levels are not very well researched. This “blind spot” of our societies should be analyzed and measured more carefully.
- 3. Individuals are the ones to decide which needs they want to satisfy at which costs. The transport system and all framework conditions have to indicate true scarcities by a continuous process towards “less lying prices”. Externalization of costs should be restricted. In this long process, scientists, planers, politicians, teachers and the media are to cooperate.
- 4. Once the level of needs satisfied by transport and the total costs used for that access are known, economic efficiency tools can be determined. Those tools are of the following type:
- Transport Efficiency := Access (per group, area, time) / generalized costs for that access.
- 5. These tools can be used to decide upon infrastructure measures, technical innovations, organisational instruments and all other relevant decisions. Without such an efficiency tool, transportation discussions and decisions will follow different and often diverging interests.
- 6. The process towards less lying prices should start with information and education measures; once internalization measures are implemented, they are to be monitored and updated regularly.
- 7. Communication is essential: The acceptance of the population will be of decisive influence. When presenting approaches, people working in “transport and environment” should not start with environmental benefits first: The population will be sceptical about the net benefit for them, and will be afraid of a reduction of their level of Access. Instead, the benefits for the population should be addressed, e.g. as in the following example:
- “We guarantee that your needs will be satisfied, access will be maintained or improved. But total costs of your level of access will be reduced, transport will be shorter, safer, quieter, cleaner or cheaper. Polluting modes of travel may cost more, but options to avoid such modes will be provided. Dynamically, all reactions and adaptations make transport more efficient for everybody. Then, total expenses for the same access will be smaller than today.”

- Two approaches are essential: a change in transport objectives towards focussing on ACCESS, and embarking on a path towards less lying prices in society. Then, solutions in "transportation" will be consistent with those in "environmental fields". "Transport and environment" will then represent a consistent area of work, with fewer fights, more success and even more fun to work in. However, the objective of transport will have to shift from "reducing costs of travel" to "creating access".

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Who breathes in the dirtiest air, and who causes it? Traffic pollution and poverty in Christchurch, New Zealand

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Abstract

This study investigates whether exposure and relative contribution to traffic pollution in Christchurch, New Zealand varies significantly between areas of different socioeconomic status. Geographically-detailed estimates of traffic-related air pollution were compared to area indicators of socioeconomic disadvantage. The results suggest that exposure to pollution is highest among the most disadvantaged groups in society. Furthermore, the groups producing the greatest proportion of vehicular pollution tend to have relatively low levels of pollution exposure. The results of this study suggest that there are clear social injustices in traffic-related air pollution exposure in New Zealand.

Keys-words*: Air pollution, deprivation, environmental justice, New Zealand.*

Introduction

Links between transport and air pollution are well established (Sturm, 2000). More recently research has also begun to consider some of the impacts that transport can have on issues of inequality and social justice, much of which has focused on links between access to transport and the issues relating to social exclusion (Hine and Mitchell, 2001), as well as the impact of transport infrastructure and land use on different social groups (Schweitzer and Valenzuela, 2004). Some researchers have considered issues relating to environmental inequalities, specifically those relating to environmental justice, which has been defined as the "equal access to clean environment and equal protection of issues of environmental harm irrespective of race, income, class or any other differentiating feature of socioeconomic status" (Cutter, 1995). Understanding issues relating to environmental justice is important because if socially disadvantaged communities are exposed to raised levels of air pollution then due to the additional effects of material deprivation and psychosocial stress they are likely to be more susceptible to the health effects of pollution exposure (O'Neill et al., 2003). While environmental justice has received increased attention, most of the research has focused on inequalities associated with industrial pollution, and the vast majority of this research has been carried out in a North American context (Morello-Frosch et al., 2001). Although some research has focused on pollutants such as nitrogen dioxide, where the main source is traffic (Brainard et al., 2002; McLeod et al., 2000; Mitchell and Dorling, 2003), few studies have explicitly focused on inequities in exposure to traffic pollution (Schweitzer and Valenzuela, 2004).

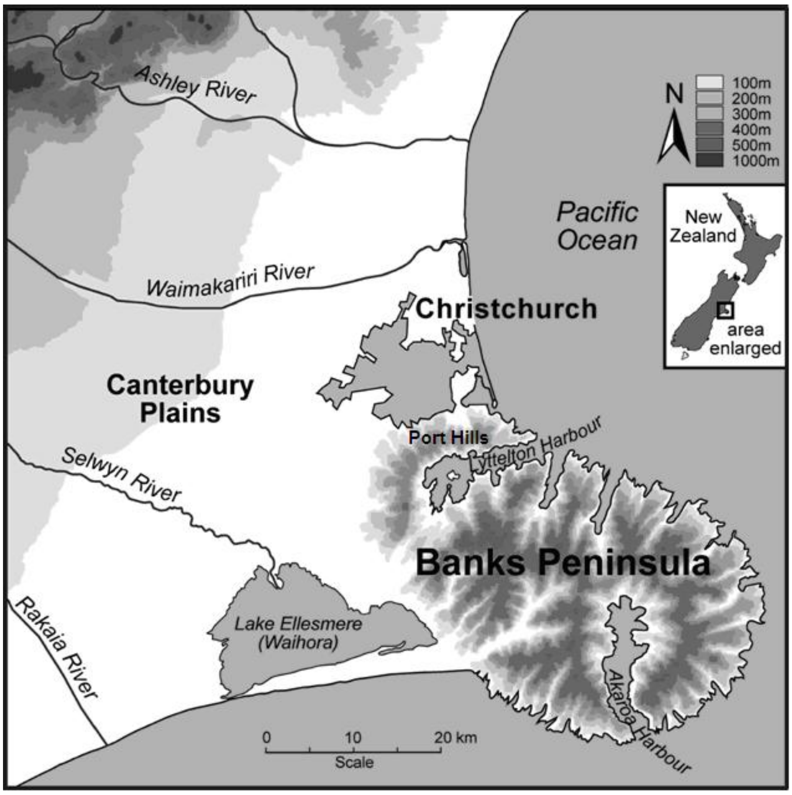
In this study we examine equity issues associated with levels of traffic-related air pollution, specifically PM_{10}, in Christchurch, New Zealand. In New Zealand PM_{10} is the greatest concern in terms of health effects (Kjellstrom, 1999) and a significant wintertime problem pollutant in Christchurch. This paper has two aims; firstly to examine whether disadvantaged groups in Christchurch were more likely to be exposed to higher levels of vehicle pollution; and secondly to estimate to what extent people living in different areas contribute towards the levels of traffic-related air pollution.

Method

1 Study area

The study area for this research is Christchurch, New Zealand, a city of approximately 330,000 situated on the east coast of the South Island (Figure 1). The major source of particulate pollution is from the burning of wood and coal for domestic heating (Scott and Gunatilaka, 2004) although outside of winter months industrial and vehicle emissions dominate. Ground level concentrations of PM_{10} are influenced by the local wind systems at the urban scale. These comprise of up-slope and down-slope winds near the Port Hills, and sea- and land-breezes. These circulation systems are prevalent on days and nights when stagnant synoptic conditions persist over the region that can result in poor air quality. This study is focusing on vehicle-emitted PM_{10}, so the results could be applied to other pollutants for which traffic is the dominant source. New Zealand has introduced National Environmental Standards in 2005(MfE, 2004), which includes a standard for PM10, which is 50ug/m^3 expressed as a 24-hour mean. In addition to the Standard, there is also an annual guideline of 20ug/m^3 (MfE, 2002), which was not adopted as a Standard, "*Guideline levels for pollutants (and averaging periods) not covered by the standards still apply*" (MfE, 2004).

Figure 1: Map of the Christchurch area



2. Estimating vehicle pollution exposure

Accurate estimation of exposure at the right spatial scales is key in studying links between air pollution and environmental justice (Bowen, 2002; Jerrett et al., 2005; Maantay, 2002; Wilson et al., 2005). A number of previous studies have used surrogates of traffic pollution such as proximity to road or vehicle density (Gunier et al., 2003; Weiland, 1994; Zmirou et al., 2004). However these fail to take into account meteorological factors, which is particularly important in cities that are close to mountainous areas, such as Christchurch (Spronken-Smith RA et al., 2002). In this study levels of particulate pollution were simulated across the study area using 'The Air Pollution Model' (TAPM), a PC-based atmospheric dispersion model which combines meteorological and emissions data to estimate pollution levels (Hurley, 2002). The model simulated PM_{10} concentration for one kilometre grid squares across the entire urban area for 1999 from vehicle sources. This dataset was subsequently interpolated to the Census Area Units (CAU) (CAUs are the second smallest unit of dissemination of census data in New Zealand, each representing approximately 2,300 people). A more comprehensive discussion of the methodology used to estimate pollution exposure in Christchurch can be found in (Zawar-Reza et al., 2005)

3. Demographic and socioeconomic data

To examine whether exposure to particulate pollution disproportionately affects the more disadvantaged social groups in Christchurch, the pollution estimates calculated for CAUs across the city were compared data from the 2001. Pollution was compared to; income by calculating the mean levels of predicted annual exposure for different household income groups; social deprivation using the New Zealand Deprivation Index (NZDep 2001) (Salmond and Crampton, 2002); and ethnicity by calculating the mean levels of predicted annual exposure for the four largest ethnic groups in New Zealand (European, Maori, Pacific People and Asian).

4. Analytical methods

In order to examine whether there are exposure inequities with respect to air pollution in Christchurch, the pollution estimates were compared to the demographic and social indicators outlined above by calculating the mean pollution levels in the appropriate data quintiles. Each variable was divided into quintiles in order to consider how pollution varied between areas that were relatively demographically and socially homogenous. This comparison allowed us to examine the relationship between air pollution and ethnicity, income and deprivation in Christchurch. In addition to examining the annual traffic pollution estimates, we also considered whether pollution exposure from traffic was higher in areas where there were greater levels of vehicle ownership, by calculating mean traffic pollution levels for vehicle ownership quintiles.

Results

1. Pollution exposure

The mean modelled traffic pollution value for the CAUs in Christchurch was 1.93μg/m^3, with a minimum of 0.14μg/m^3, a maximum of 4.87μg/m^3, and a standard deviation of 1.07. While the average traffic pollution levels were not particularly high (although the total values including all sources are substantially higher and some exceed the annual guidelines value of 20μg/m^3), there are high levels in some areas and considerable spatial variation between areas.

2. Socio-economic status

The relationship between income and pollution level was examined by calculating the mean vehicle pollution in census areas divided into quintiles according to household income levels (Figures 1 and 2) and deprivation (Figure 4). Figure 2 shows mean vehicle pollution levels by income quintiles ranging from $0 to $30,000 (lower income households). It can be seen that in areas with a greater proportion of households with low income (quintile 5) mean vehicle pollution levels are highest, whereas in areas with a smaller proportion of low income households (quintile 5) mean levels are lower. Conversely Figure 3 shows income quintiles ranging from $50,000 and greater (higher proportion of income households). Here it can be seen that in areas where there are a greater number of high income households (quintile 5) the mean vehicle pollution levels are lower and vice-versa. In summary, in areas with a large proportion of low income households, the mean vehicle pollution levels are higher and in areas where high income households are more common, particulate levels are lower. Examining pollution levels by deprivation (Figure 4) shows a clear gradient from quintile 1 (least deprived) to 5 (most deprived) with those living in the most deprived having areas having the highest levels of pollution.

Figure 2: Mean annual traffic-related particulate pollution in income quintiles (from $0 to $30k) in Christchurch

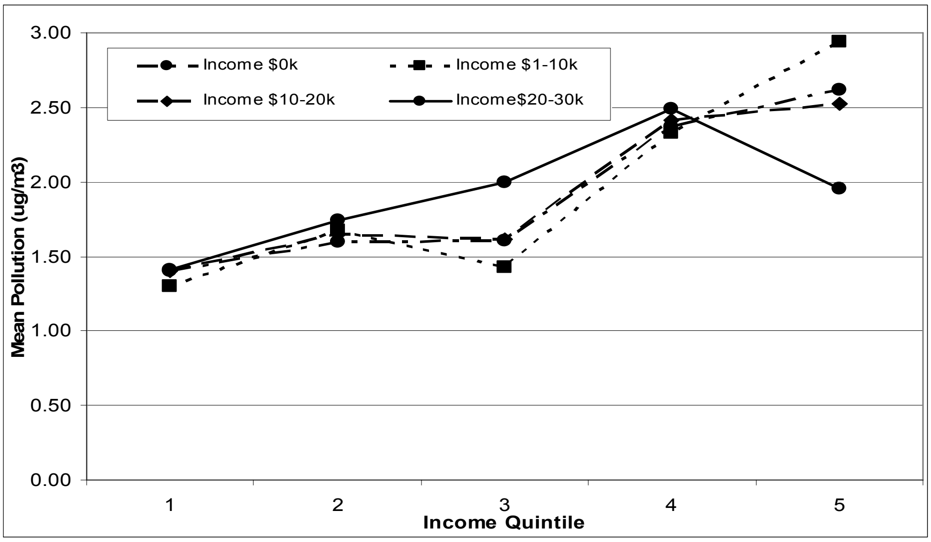
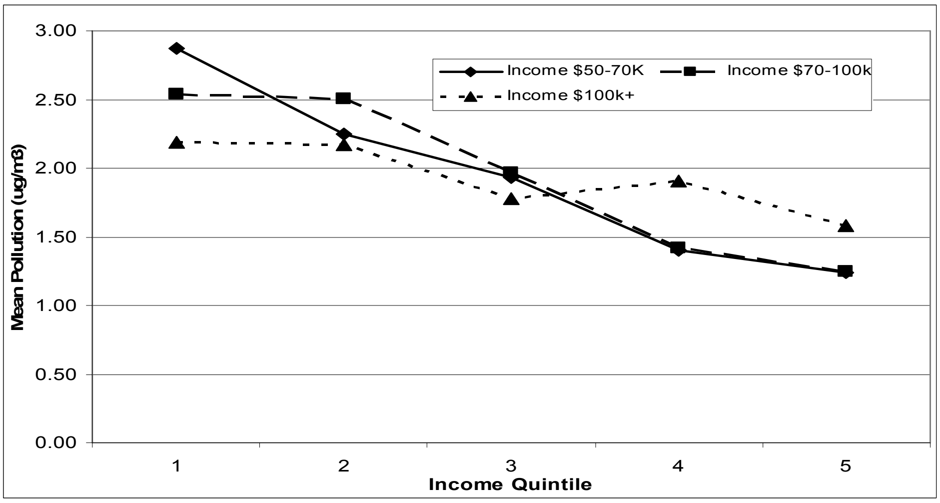


Figure 3: Mean annual traffic-related particulate pollution in income quintiles (from $50k upwards) in Christchurch



3. Ethnicity

The relationship between pollution and ethnicity was examined by calculating the mean vehicle pollution in census areas divided into quintiles according to ethnicity for European, Maori, Pacific Islander and Asian (Figure 5). There is a clear gradient from quintile 1 (least Europeans) to quintile 5 (most Europeans) with increasing proportion of Europeans showing a decrease in level of vehicle pollution. For Maori, Pacific Islanders and Asians the pattern is less clear although pollution levels are generally higher in census areas with increased proportions of those populations.

4. Vehicle ownership

The relationship between vehicle emissions and levels of car ownership was examined by calculating the mean vehicle pollution in census areas divided into quintiles according to the levels of household car ownership (percentage of households with no car, one car, two cars and three or more cars) (Figure 6). For those households without a car, the results demonstrate that mean levels of air pollution from vehicles increased in a linear fashion from quintile one (lowest proportion of households without a car) to quintile five (highest proportion of households without a car). For those households with three or more cars, the reverse pattern is observed with the highest mean pollution from vehicles in quintile one (lowest proportion of households with three or more cars) and the lowest mean pollution in quintile five (highest proportion of households with three or more cars). The pattern for those households with one or two cars is less clear as although mean vehicle pollution levels are lower in quintiles three and four than in quintiles one and two, the mean level of pollution in quintile five is similar to quintiles one and two.

Figure 4: Mean annual traffic-related particulate pollution in deprivation (NZDep) quintiles in Christchurch

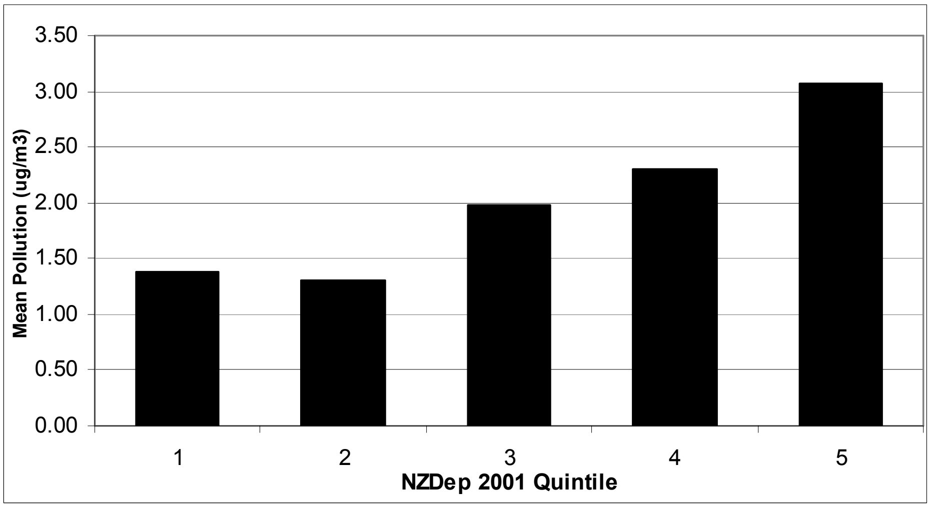


Figure 5: Mean annual traffic-related particulate pollution in ethnicity quintiles in Christchurch

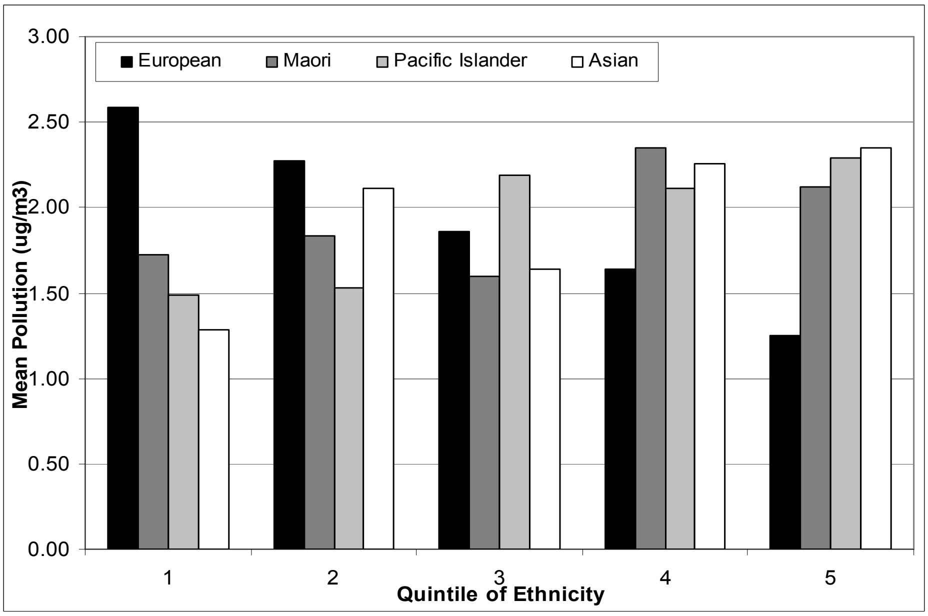
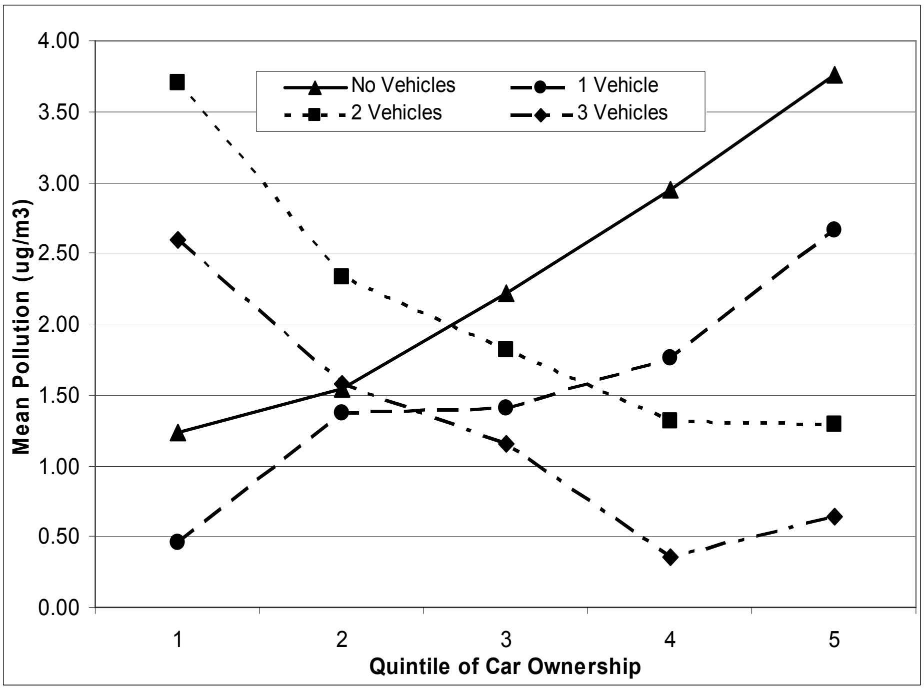


Figure 6: Mean annual traffic-related particulate pollution in vehicle ownership quintiles in Christchurch



Discussion

This research has found that areas of Christchurch with a greater proportion of low income and more deprived households have higher levels of vehicle particulate pollution. In addition, areas with higher levels of traffic-related pollution have greater proportions of Maori, Pacific islanders and Asians. These results demonstrate that there are issues relating to environmental inequity in Christchurch with less affluent and ethnic minority groups being exposed to higher levels of traffic pollution than the more affluent, less deprived people of European ethnicity. These results support previous studies that have found a social gradient in pollution exposure and differences in exposure for different ethnic groups (Brainard et al., 2002; Morello-Frosch et al., 2001; Wheeler, 2004). However few studies have focused specifically and directly on the affects on environmental justice of traffic pollution (Schweitzer and Valenzuela, 2004). Research in California found that areas with higher levels of minority groups and poverty had more than double the level of traffic density compared to the rest of their study area suggesting that this may result in a higher risk of exposure to vehicle-related pollutants (Houston et al., 2004). Other Californian research found that low income households were three times more likely to live in high density traffic areas than those in the highest income households (Gunier et al., 2003). They found similar relationships for ethnicity and based on their analysis concluded that “low income and children of colour have higher potential exposure to vehicle emissions” (Gunier et al., 2003). However our findings in Christchurch add to this, as Christchurch is a relatively small city not renowned for having particularly high levels of traffic pollution yet there are strong geographical disparities in exposure to traffic pollution between deprived and non-deprived areas across the city. Furthermore, these results should be of particular interest because the pollution estimates used were at fine spatial scales using an air pollution dispersion model that includes meteorological inputs.

In addition to examining differences in vehicle pollution exposure, this research has also identified that in areas of higher levels of car ownership, the level of traffic-related air pollution exposure is lower. In other words, those with the highest levels of car ownership are exposed to the lowest levels of vehicle pollution, evidence also noted in two British studies (Mitchell and Dorling, 2003; Stevenson et al., 1999). These results provide some evidence to suggest that those responsible for producing most of the traffic pollution are in fact exposed to the least amount. While this may seem surprising, it is arguably not, based on previous research on environmental justice in Christchurch, albeit not for traffic pollution (Pearce et al., 2006). One explanation would be that people in areas with higher levels of car ownership drive through areas of low car ownership polluting those least responsible for the pollution. However, without information on the nature of emissions from cars in different areas of the city it is not possible to make firm conclusions about the relationship between pollution production and exposure. As Mitchell and Dorling (2003) speculate “could it be that smaller numbers of cars in highly polluted areas are in fact highly polluting cars?” It may be suggested that low income households have fewer cars but that these cars are older and/or their owners are unable to maintain them adequately and they are therefore greater polluters. If this were the case, then improvement in vehicle fleet emissions could have interesting implications. If tighter emission controls applied to new cars only, then environmental inequities between rich and poor may widen as wealthier people get newer cleaner cars, while poorer people retain older dirtier cars. Conversely, if new emissions controls applied to all vehicles, then issues of social injustice may arise as a result of poorer people being priced out of the market as they may be unable to meet the costs of complying with new emissions standards.

Conclusion

This work has focused on Christchurch, a city where accurate fine scale estimates of pollution have been calculated (Zawar-Reza et al., 2005). This is the first study to use pollution estimates for small areas specifically for traffic emissions. The results of this study suggest that exposure to traffic-related pollution is highest among the most disadvantaged groups suggesting that the process of environmental injustice may be operating in Christchurch. In addition it has been shown that not only are those people who live in more deprived areas exposed to higher levels of traffic pollution, but they also own proportionately less cars; the source of that pollution. Furthermore, the groups producing the greatest proportion of vehicular pollution tend to have relatively low levels of exposure. Despite a number of possible caveats, the results of this study suggest that there are clear social injustices in traffic-related air pollution exposure in Christchurch, New Zealand. Future work looking at the whole of New Zealand, including areas of greater traffic volume would be of interest to see if the associations identified exist at a national scale.

Acknowledgments

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NO_2 emissions from passenger cars

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Abstract

Although emission of total oxides of nitrogen (NO_X) have dropped considerably, they remain problematic in relation to road transport, with ambient concentrations of nitrogen dioxide (NO_2) near main roads close to or in exceedance of the limit values set for the European Union for 2010.

At the Transport and Air Pollution conference in 2003 the state of knowledge on the topic was reported. A small number of measurements with non-standardized measurement methods lead to a diffuse picture of the topic. Particular attention has been drawn to the relatively high primary NO_2 emissions associated with oxidation catalysts fitted to diesel passenger cars and certain types of regenerative particulate traps, fitted to HGVs and buses. This paper is based on new insights from dedicated measurement emission programs in the Netherlands and Switzerland, input from the automotive industry (ACEA) and emission tests and ambient air quality measurements in the UK. As such this paper presents a state-of-the-art overview of the current knowledge about the measurement of tail-pipe NO_2 and the causes and impacts of the direct NO_2 emission from modern road vehicles and discusses sensitive technologies.

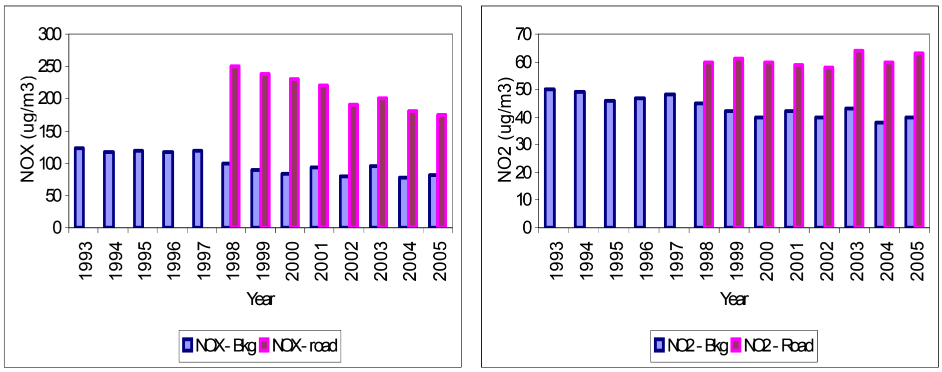
***Keys-words**: Nitrogen dioxide, road traffic emission factors, ambient air quality.*

Introduction

In recent years, the emission of total oxides of nitrogen (NO_X) has come under renewed attention in environmental policy making. Contrary to the last decade, today's focus is not mainly on acidification, but on ground level (tropospheric) ozone formation (smog) and human toxicity of NO_2. NOx emissions mainly stem from anthropogenic combustion processes. Road transport, mobile machinery, shipping, industry and domestic sources all contribute to the emission of NOx. Within the air pollution community, total oxides of nitrogen are routinely referred to as the sum of nitrogen oxide (NO) and nitrogen dioxide (NO_2). At the time concerns were raised about the direct human toxic effects of NOx emission, but NOx emissions from exhaust emissions were also seen as an indicator/tracer for to total reactivity of an exhaust gas stream. This lead to the regulation (on a European level) of a) NOx emissions in the exhaust gas of vehicles and b) the ambient air concentrations of NO_2. The observed discrepancy in regulated components for the exhaust and ambient side stems from the fact that in the exhaust gas a mixture of NO and NO_2 (NOx) is present, while at ambient level mainly NO_2 is found (because of the fast reaction of NO into NO_2 under ambient conditions).

In response to the gradual tightening of the allowable NO_X exhaust limits, under directives of the European Union, an improvement in ambient NO_2 concentrations could be expected. This is indeed reflected in trends in background ambient concentrations, but is not evident at roadside locations, where NO_2 ambient concentrations have appeared to stabilise or even increase, Carslaw and Beevers (2004b). Figure 1, shows this situation, as measured at 8 urban background and 8 roadside sites in the UK. This effect was also measured and analysed for some street canyons in London leading to the conclusion that the proportion of primary NO_2 must be higher than expected and increasings, Carslaw and Beevers (2004/2005a, b).

Figure 1: Annual mean NO_X and NO_2 concentrations from 8 urban background and 8 roadside sites in the UK.



Until recently, it was accepted that the NO_2 fraction in the NOx tail pipe emission was around 5 percent (volume fraction). The figure of 5% was based on relatively old measurements, from vehicles without after-treatment system, but is not widely incorporated into air pollution dispersion modelling tools. This ratio however is crucial when assessing the ambient air levels of NO_2 close to traffic arteries. Under the influence of temperature, sunlight and in the presence of oxidants such as ozone, the equilibrium between the two gases may rapidly change (seconds to minutes). This will lead to the NO_2/NOx ratio in the ambient air, in the area of a few 100 metres downwind of a road, increasing to almost 100% NO_2. The absolute concentration of NOx close the roads is relatively high. In these locations, the proportion of primary NO_2 can be a significant determinant in the potential compliance with air quality standards.

These issues highlight the need for investigating the direct NO_2 emissions from the current vehicle fleet in order to be able to assess the process leading to stagnation in the downward trend in annual mean NO_2 concentrations.

Such an investigation, however, has to deal with analytical measurement problems as well, since the measurement of tail pipe NO_2 emissions is not regulated and for the aforementioned reasons this measuring procedure may be a source of systematical errors, Latham et al (2001). This problem was recognized; the measurement process of tail pipe NO_2 was examined collectively by different research organisations (EMPA, TNO and TRL) and the umbrella organisation for the European auto-industry (ACEA) was consulted for feed-back. As a result a feasible test procedure was selected and used for a large scale measurement programme conducted in 2 laboratories in which in total 63 passenger cars were tested.

This paper focuses on the correct measurement of direct tail-pipe emission of NO_2 and on its fraction in the NOx emission of several vehicle technologies to find out if a substantial shift in the NO_2 fraction and the level of NO_2 emission has occurred.

Measuring tail-pipe NO_2

1. Possible sources for systematical errors

A possible systematic error in the measurement of the relative NO_2 fraction may occur over the time between sampling at the tail pipe and the actual analysis in an analyser. Literature-based evidence suggests a number of potential conversion routes. Some of these, however, stem from reaction mechanisms that occur on a tropospheric level in the atmosphere. Although the ambient conditions are not exactly the same as under laboratory emission test conditions, these conversions have to be taken into account. Regarding the chemical properties of NO and NO_2 some other conversions may be important too when the typical automotive emission sampling environment is considered. Taken from both areas of interest table 1 provides a summary of the potential conversion mechanisms.

The equilibrium between NO and NO_2 is sensitive to ambient conditions such as the presence of oxygen, ozone and other substances, temperature and UV. This sensitivity is suspected to have an effect on the tail-pipe NO_2 emission determined under laboratory conditions with the level of the effect varying from mild to very strong depending on the measuring method, the vehicle, the driving cycle and the actual ambient conditions. Mainly the conversion of NO to NO_2 in the presence of oxygen, influenced by temperature and possibly UV is suspected of influencing the results. The effects of ozone were considered to be negligible compared to these effects, as ozone is only present in very low concentrations in ambient air when compared to the high level of NO_x in diluted or undiluted exhaust gas. A simple test confirmed that NO converts to NO_2 when it is given time to react in a Tedlar bag. The accuracy is further increased if the mixing of exhaust gas with ambient air is prevented or clean (ozone free) air is used for dilution. Sampling emissions from the raw exhaust gas is therefore also recommended.

Table 1: Overview of possible reaction mechanisms. The possible influence per involved component is indicated from very small (<<<) to very large (>>>).

Involves	**Source (exhaust)**	**Exhaust Concentration Level**	**Ambient concentration level**	**Importance for NO_2 measurement**
Ozone	ambient intake air ??	< ?	<	<<<
CO	combustion process	> depending on engine type and conditions	<	?
Radicals	combustion process ??	??	?	?
UV	not present	0	> to >>> depending on weather/location (indoors/outdoors)	> ?
Oxidation	oxygen excess intake air	> to >>>	>>>	>>>
Water	combustion	>>>	>	>
Ammonia	catalysis	> to >>> depending on engine type, applied catalyst and conditions	<	>>>

Another problem, observed for the chemiluminescence principle with its pre-converters, is the sensitivity of this type of analysis for ammonia. If ammonia is present in the exhaust gas, like for example for vehicles with SCR-DeNOx or petrol engines, ammonia may react with NO and NO_2 to produce N_2 and H_2O. This reaction would potentially reduce the measured concentration when the pre-converter is engaged. To what extent this actually influences the final results could not currently be determined and thus requires further investigation.

Hereafter, an outline is given about the latest insights that concern the measurement of NO_2. The insights were obtained from hands-on experience with laboratory experiments.

Table 2: Overview of options for the testing procedure; sampling, conditioning and analysis.

Part of procedure	**Option**	**Remarks**	**Suitable?**
Sampling method	Bag	long delay before analysis->conversion NO->NO_2	No

	Diluted online	effect of dilution air	No
	Raw online		Yes
	Mini-dilution	no experience	??
Sample conditioning	heated lines		Yes
	dehumidifier (wet/dry)	wet; washing NO_2/NH_3	dry Yes wet No
Analysis	Chemiluminescense with NOx to NO converter	SCR reaction with NH_3?	Yes. But possible underestimation in case of NH_3 emission
	Chemical mass ionisation Spectrometry (CM-IS)	kalibration, speed	Yes
	FTIR	Widely used in UK emission measurement programmes	Yes. This technique offers a direct mea-surement of NO_2. Interferences with other compounds needs to be investigated.
	NDUV Non-dispersive Ultra Violet	EPA approved	Yes

2. How to measure NO_2?

Given these insights, a procedure that analyses exhaust gas as *quick* as possible after exiting the exhaust seems the most accurate and reliable. To minimize effects of dilution with ambient air, sampling from the *raw* exhaust gas brings lowest risk of error. FTIR is now widely used in UK measurement programmes, as it remains the only technique that directly measures NO_2. However, *simultaneous* measurement of NO and NOx according the principle of *chemiluminescense* (CLD) seems feasible for the time being as this is a commonly accepted and available method in current emission laboratories. Another argument for this method is the fact that immission measurements also use this principle, so a parallel between both can easily be drawn. Time alignment issues typical for on-line measuring have been investigated by EMPA and did not show any problems in the typical case of simultaneous measurement of NO and NOx for the differential determination of NO_2. Besides, a constant time shift seemed appropriate to obtain good correlation with standard bag measurements. Finally, preconditioning by using *heated sample lines* and *analyses* will prevent the loss of (water-soluble) NO_2 before analysis. However, a few concerns over this measurement method remain:

- Measuring vehicles with significant amount of ammonia in the exhaust gas (like SCR-DeNOx equipped vehicles and petrol cars). Measuring these vehicles with CLD will lead to an underestimation of the actual NO_2 amount.

- The inaccuracy of the differential method at low NOx levels (e.g for modern petrol cars); this inaccuracy results from the fact that the accuracy of a value is very low when it is derived from substraction of two comparitively large numbers which both are already very low and inaccurate by themselves. For the latter it is debatable whether it is a problem or not. The inaccuracy stems from the low levels of NOx and NO_2 and as such may be regarded as of minor importance. On the other hand, the impact of technological developments on petrol cars like for example direct injection might be of interest, but could not yet be monitored very accurately.

Emission measurements 2005.

For an emission measurement programme, performed in 2005, TNO and EMPA tested 63 passenger cars on petrol and diesel, applying the latest insights in the testing methodology for NO_2 and NO. An Eco-Physics CLD chemiluminescence NO-NO_2 analyzer was connected to the main sampling line, close to the vehicle tail pipe. Raw exhaust gas (undiluted), were sampled using heated sampling lines to the instrument. Mass emissions of NO and NO_2 were determined on-line. The vehicle sample involved the technologies and legislative categories as mentioned in table 3.

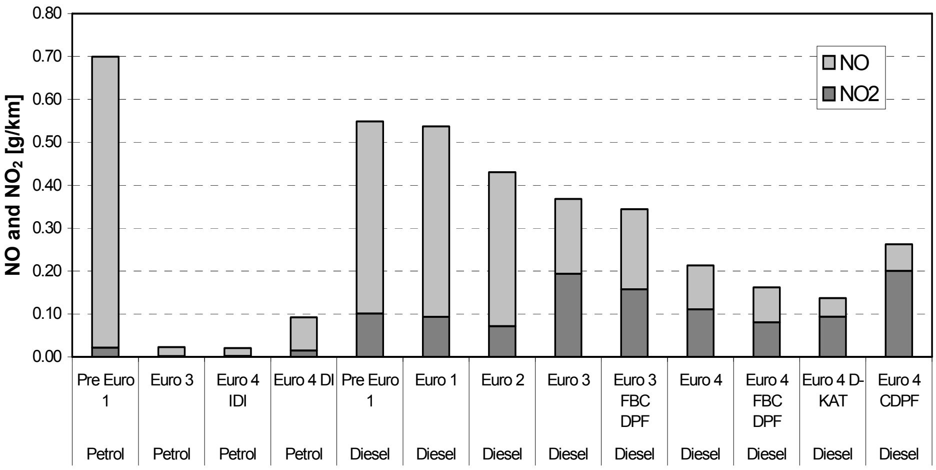
Table 3: Vehicle sample.

Fuel	Legislative category (70/156/EC; M1 ≤2500kg)	Sample size and technology
Petrol	Euro 0	7 various technologies, carburator, multi/monopoint,
	Euro 3	7 Port injection Three Way Catalyst
	Euro 4	10 Port Injection Three Way Catalyst, 2 Gasoline Direct Injection
Diesel	Euro 0	7 diesel
	Euro 1	7 diesel
	Euro 2	7 diesel with oxidation catalyst
	Euro 3	16 diesel with oxidation catalyst, 1 Fuel Borne Catalyst Diesel Particle Filter
	Euro 4	2 diesel with oxidation catalyst, 1 Fuel Borne Catalyst Diesel Particle Filter, 3 Catalysed Diesel Particle Filter, 1D-KAT (combined oxidation /NOx storage catalyst and particle filter) all with oxidation catalyst

The results from the programme show some clear trends and Catalysts coated with platinum are used in aftertreatment systems for Heavy Duty Trucks to supply an excess of NO2 to promote the oxidation of particles. It is likely that this merit is also used for the reduction of particle mass emission of diesel passenger cars. Stimulated by limits for the particulate mass as defined in European legislation (70/220/EC and its amendments) manufacturers sought for cost effective measures to reduce the particle mass emissions and as result applied more active oxidation catalysts to reduce this particle mass. This process probably started already on some Euro 2 cars which can be seen in the results. Some Euro 2 diesel cars already had a somewhat elevated NO2 fraction as shows the almost 35% reached by one car. The average level of the fraction is still as high as the previous legislative categories Euro 1 and 0 as the high figures are remarkably compensated by some rather low fractions as the spread for the Euro 2 diesels show. A Euro 2 diesel car with a low NO2 fraction appeared to be equipped with a replacement catalyst. The low NO2 fraction for this car might therefore be caused by a lack of platinum in this catalyst.

Figure 3). Both the NO_2 emissions as well as the NO_2 fraction of the diesel cars are clearly higher then of the petrol cars. For petrol cars the fraction ranges from a few percent to about 20%, while for diesel cars the fraction ranges from about 5% to almost 80%. For diesel cars a large step from Euro 2 to Euro 3 can be observed for the NO_2 fraction. While the average fraction of Euro 0 to Euro 2 does not vary much and is about 15 to 20%, the Euro 3 diesel cars have a considerably higher NO_2 fraction in the order of 50%. Even though the total NOx emission gradually decreases from Euro 0 to Euro 4, the absolute NO_2 emission also increases sharply from Euro 2 to Euro 3 and continues to vary at a higher level going to Euro 4.

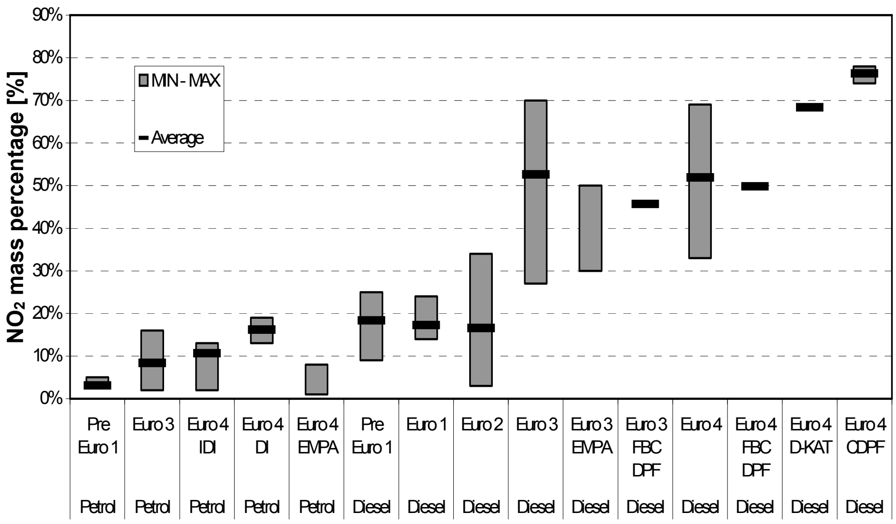
Figure 2: the NO and NO_2 emission of various technologies and legislative categories. The figures are composed of a mix of driving situations, being urban with a cold start, rural and highway driving as measured by TNO-Automotive in 2005.



The differences in NO_2 fractions between petrol and diesel engines can be explained by the difference of the combustion process and in particular the difference in air - fuel ratio at which the engines are operated. For diesel engines their lean operation results in the presence of a lot of oxygen in the exhaust gas, which facilitates the conversion of NO to NO_2. Petrol cars are generally operated around a stoichiometric air - fuel ratio and as a result only little oxygen is present to facilitate this conversion process. Furthermore, for modern diesel cars, a highly active platinum coated oxidation catalyst enhances the NO to NO_2 conversion even more. This latter effect can be observed in the results. Starting on maybe some Euro 2 cars, but clearly present on most Euro 3 cars, platinum coated and highly active oxidation catalysts have a large impact on the NO_2 fraction.

Catalysts coated with platinum are used in aftertreatment systems for Heavy Duty Trucks to supply an excess of NO_2 to promote the oxidation of particles. It is likely that this merit is also used for the reduction of particle mass emission of diesel passenger cars. Stimulated by limits for the particulate mass as defined in European legislation (70/220/EC and its amendments) manufacturers sought for cost effective measures to reduce the particle mass emissions and as result applied more active oxidation catalysts to reduce this particle mass. This process probably started already on some Euro 2 cars which can be seen in the results. Some Euro 2 diesel cars already had a somewhat elevated NO_2 fraction as shows the almost 35% reached by one car. The average level of the fraction is still as high as the previous legislative categories Euro 1 and 0 as the high figures are remarkably compensated by some rather low fractions as the spread for the Euro 2 diesels show. A Euro 2 diesel car with a low NO_2 fraction appeared to be equipped with a replacement catalyst. The low NO_2 fraction for this car might therefore be caused by a lack of platinum in this catalyst.

Figure 3: the NO_2 percentage in the NOx emission of various technologies and legislative categories, including the spread over vehicles (minimum and maximum observed) as measured by TNO-Automotive and EMPA in 2005.



Remarkable is the high NO_2 fraction of diesel cars with a particle filter. Whereas the FBC DPF technology seems to show a lower NO_2 fraction than conventional modern diesels, the DPFs and D-Kat show higher NO_2 fractions. The oxidation catalysts of cars with a particle filter also generate NO_2 to support the combustion of particles and as a result bring about elevated levels of tail-pipe NO_2.

Conclusions

The investigation reported in this paper was started because there were some indications that the fraction of NO_2 in the direct tail-pipe emissions of current on-road vehicles was significantly higher than generally thought and accepted. The primary goal of this exercise was to establish accurate emission values and based on these values draw a possible parallel with current measured ambient air quality situations close to busy traffic arteries in densely populated areas.

Data on direct NO_2 emissions and their fraction in the NO_x emission was gathered from 2 different emission measurement programmes. This data was reviewed with respect to the accuracy, the reliability and the level of the figures presented in these studies. Besides, the causes for the increased NO_2 fraction indicated by the inventories were looked after. This resulted in the following findings:

Available data show a substantial influence of the measurement method on the measured direct NO_2 emissions. The equilibrium of NO and NO_2 is very delicate and can easily be disturbed by several conditions in the measurement process.

Considering the above a measuring procedure was defined which is suitable for the assessment of the current NO_2 situation which mainly focuses on the possibly severe impact of diesel vehicles. This procedure comprises the determination of the NO_2 mass emission by means of *simultaneous* analysis of the NO and NOx concentration from the *raw* (undiluted) exhaust gas, on-line sampled just after the tail-pipe. For the gas analysis an instrument using the *chemoluminescence* principle was proposed.

For this principle of analysis, however, problems regarding ammonia interference should be considered as in the near future SCR-DeNOx systems will probably gain importance in emission inventories. Next to these aftertreatment systems, which are known to have some ammonia slip and thus may cause the interference, petrol engines are also known to emit a substantial level of ammonia compared to their NOx emission. When the suggested procedure is considered for future research into the NO_2 situation and maybe even for adaptation in a type approval system, further research should focus on possible effects of ammonia on the measured level of NO_2.

The oxidation catalysts applied on diesel vehicles nowadays (Euro 3 and Euro 4) are expected to have a strong increasing effect on the direct NO_2 emission. This is due to the strong oxidation capacity of these catalysts.

Active oxidation catalysts coated with platinum are a prominent source for elevated NO_2 emissions, as in this type of catalyst the conversion of NO to NO_2 is promoted by the strong oxidizing environment. An oxidation catalyst is used on most modern diesel cars and on some trucks, for the latter only in combination with a continuous regenerating filter. The quality of an oxidation catalyst to make NO_2 is used as a merit in an exhaust gas aftertreatment system with a diesel particle filter. Here the oxidation catalyst is placed in front of the trap or filter to provide them with an excess of NO_2 to promote instantaneous oxidation of the trapped particles. A major part of this NO_2 exits the tail pipe as only few is used in the process. In oxidation catalysts as applied on diesel passenger cars this process occurs in the oxidation catalyst itself; the produced NO_2 directly regenerates some of the particles passing the catalyst.

The results from the emission measurement programme show some clear trends. The measured level of NO_2 and the fraction of NO_2 in NOx are higher for diesel cars then for petrol cars. For diesel cars the fraction ranges from about 5% to almost 80%. A large step can be observed for diesel cars from Euro 2 to Euro 3. From Euro 0 to Euro 2 the average fraction does not vary much and is about 15 to 20%. For the Euro 3 diesel cars the measured NO_2 fraction is considerable higher and in the order of 50%. The absolute NO_2 emission increases sharply from Euro 2 to Euro 3 and continues at this level to Euro 4. Measurements on 4 cars (3 with a catalysed diesel particle filter and 1 with a D-kat) showed a further increase beyond 50%.

For petrol cars the level of NO_2 emission measured is low compared to diesel cars as both the fraction as well as the absolute level of NOx are much lower than of modern diesel cars. Because of the same reasons no accurate NO_2 fractions could be determined; the figures are too low to determine a reliable figure. Although the figures are low and inaccurate, occasionally they still point out a higher fraction than the generally accepted 5%. This may seem rather unexpected, but may be explained by the fact that a three way catalyst partially acts as an oxidation catalyst.

As petrol engines are dominantly present in the European fleet and as the NO_2 situation regarding these engines is not fully clear and understood, it is advised to include petrol engines in further research, but not before the 'ammonia issue' of chemoluminescense is solved as this principle may cause an underestimation of NO_2 of modern petrol cars.

It is recommended to assess the impact of the fleet penetration of modern diesels on air quality, hereby incorporating the established development of tail pipe NO_2 of modern diesel cars and trucks in different scenarios, distinguishing different penetration ratios of regular diesel vehicles and diesel vehicles with particle filters, especially including the 2010 situation.

Acknowledgements

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Risks of exceeding the hourly EU Limit Value for nitrogen dioxide resulting from road transport emissions of primary nitrogen dioxide

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Abstract

In London, recent analysis of ambient measurements has shown that directly emitted (primary) nitrogen dioxide (NO_2) emissions from road transport sources have increased. These increases appear to be mostly due to certain after-treatment devices, such as oxidation catalysts and particle filters fitted to diesel vehicles. A constrained simple chemical model is used to predict hourly concentrations of NO_2 at a busy roadside site in London. The model performance is shown to be good across the full range of hourly NO_X and NO_2 concentrations over 7 years. A Monte Carlo approach is used to predict future hourly NO_2 concentrations by considering the model errors, uncertainties in future NO_X trends and the inter-annual variability of meteorology. It is shown that if the NO_2/NO_X emission ratio of 22.0 % by vol., as calculated at the end of 2004, is sustained into the future, it is likely that the hourly EU Limit Value will not be met. However, the probability of not meeting the Limit Value in 2010 depends strongly on the meteorological year and varies from 16-88 % depending on the year considered. This work suggests that further increases in the NO_2/NO_X ratio beyond those observed at the end of 2004 would considerably increase the probability of the EU hourly limit for NO_2 being exceeded. It is important that further work is carried out to improve the quantification of NO_2 in vehicle exhausts to determine the likely future risks of exceeding the hourly Limit Value in other European cities.

***Keys-words**: Monte Carlo, nitrogen oxides, diesel particulate filter, London.*

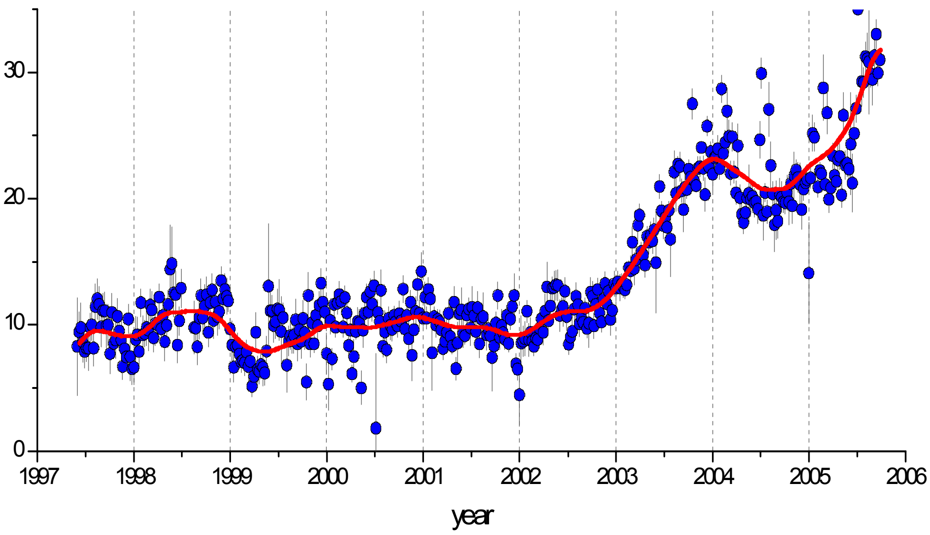
Introduction

In Europe, the EU Daughter Directive (99/30/EC) sets two limits on ambient concentration of nitrogen dioxide (NO_2), which must be achieved by 2010: an annual mean limit of 40 µg m^{-3} and an hourly limit of 200 µg m^{-3} that must not be exceeded on more than 18 occasions each year. In Europe in 2003, the AIRBASE database for monitoring sites in urban areas affected by road traffic shows that 275 out of 456 sites (60 %) with data capture > 80 % exceeded the EU annual mean limit of 40 µg m^{-3} (data available at http://air-climate.eionet.eu.int/databases/airbase). However, only 36 sites (8 %) exceeded the hourly limit value. Of these, Marylebone Road in London recorded the greatest number of hours > 200 µg m^{-3}. These data demonstrate that in urban areas across Europe it will be most difficult to meet the annual mean limit for NO_2. In the UK, dispersion modelling predictions show that at many locations the annual mean limit will not be met in 2010 but it is much more likely that the peak hour limit will (AQEG, 2004).

Recent work has highlighted the growing importance of directly emitted (primary) NO_2 from road vehicles (Carslaw and Beevers, 2005; Carslaw, 2005). This work has shown that in London, the mean proportion of NO_X in the form of NO_2 has increased from 5-7 % (by vol.) in 1997 to around 17 % by vol. by the end of 2003 (Carslaw, 2005). Furthermore, these increases have had an important effect on trends in annual mean NO_2 concentrations. Figure 1 shows the trend in the estimated NO_2/NO_X emission ratio for traffic on Marylebone Road, which is located in a busy (>80,000 vehicles per day) street canyon location in central London. The Figure shows the very clear increase in road vehicle NO_2/NO_X ratio towards the end of 2002. These results also highlight that the primary NO_2 fraction has continued to increase at Marylebone Road, such that by September 2005 the fraction was around 30 % by volume. No work to date has considered the potential effects of these increases on the peak hour limit for NO_2. It is not known if the hourly Limit Value will be met by 2010 due to these recent increases in vehicular NO_2/NO_X ratios, nor what factors (e.g. the increased use of oxidising filters fitted to diesel vehicles) would lead to exceedances. Increases in the ratio of NO_2/NO_X from vehicles is not only of direct concern for NO_2 concentrations; a photochemical modelling study by Jacobson et al. (2004) showed that increased ratios could lead to increased ozone concentrations, albeit under US conditions.

Although increases in the NO_2/NO_X ratio of road traffic have been observed recently, there is little vehicle emissions data available in the open literature that speciates between NO and NO_2. There are however several factors that are likely to be important. First, it is known that diesel vehicles emit more of their NO_X in the form of NO_2 compared with petrol vehicles (Latham et al., 2001). In the UK and the rest of Europe the proportion of passenger cars fuelled by diesel is increasing. Specifically in the UK, the proportion is expected to reach 40 % in 2010 compared with only 12 % in 2000. Diesel cars and light goods vehicles are now fitted with oxidation catalysts. The oxidising environment created by these devices could lead to increased ratios of NO_2/NO_X. Finally, in London, all Transport for London buses are now fitted with catalytic diesel particulate filters (CDPFs). These filters oxidise NO to NO_2 and use the NO_2 to effectively reduce particle mass and number. Ratios of around 40-50 % NO_2/NO_X are reported for CDPF systems (Ayala et al., 2002; Guo et al., 2003).

Figure 1: Estimated weekly NO_2/NO_X emission ratio at Marylebone Road derived using the method of Carslaw and Beevers (2005). Error bars are shown at 2σ.



Objectives

From a simple consideration of the AIRBASE measurements it would seem that exceedances of the EU hourly limit value for NO_2 in 2010 would be very unlikely, even in heavily trafficked locations in large European cities. However, no work has been carried out to assess the risks of such exceedances and the factors that might lead to them. The principal aim of this work is to predict whether the EU hourly Limit Value for NO_2 will be exceeded in 2010 in London and the contribution made by road vehicle primary NO_2 emissions. First, a simple constrained chemical model is used to calculate hourly mean NO, NO_2 and ozone (O_3) concentrations at a busy central London roadside location (Marylebone Road). The model accuracy in predicting hourly concentrations is assessed over 7 years. To project forward to 2010, three major sources of uncertainty are identified: the model uncertainty for hourly predictions of NO_2, that of the projected decline in NO_X concentrations and the inter-annual variability of meteorology. To assess these uncertainties, a Monte Carlo technique is used, which yields information concerning the probability that the hourly Limit Value will be exceeded. Consideration is also given to the source apportionment of NO_2 concentrations to explore the contributory factors that affect high percentile hourly NO_2 concentrations. Finally, these results are discussed in the context of vehicle emissions and emissions technologies as factors affecting current and future concentrations of NO_2.

Method

1. Model description

Carslaw and Beevers (2005b) describe a technique for estimating the NO_2/NO_X ratio from road transport emissions based on an analysis of ambient measurements. Briefly, the increment in NO_X and NO_2 concentration at Marylebone Road above a nearby background site (North Kensington) is partitioned into NO_2 that is chemically derived through the reaction between NO and O_3 and that which is emitted directly by road vehicles. These data are available from the National Air Quality Information Archive, operated by NETCEN (http://www.airquality.co.uk) and further details of the measurement sites can be found in Carslaw and Beevers (2004). The model is constrained by assuming that the difference between roadside and background NO_X concentrations each hour is due to NO_X emissions from road vehicles adjacent to the roadside monitoring site. A simple set of chemical equations is used to describe the time-dependent change in NO, NO_2 and O_3 concentrations as vehicle plumes are mixed with background air. By considering different values of the assumed NO_2/NO_X emissions ratio from road traffic and the time available for the NO–O_3 reaction to take place, the best agreement between modelled hourly NO_2 concentrations and measured roadside concentrations is sought. In practice, several hundred combinations of the NO_2/NO_X emissions ratio and the time available for the NO–O_3 reaction to take place are considered before the best single combination, resulting in the minimum error between modelled and measured NO_2 concentrations, is identified. The technique assumes that the increment in NO_2 concentration above a local background site is controlled by the availability of O_3 and directly emitted NO_2 only. The formation of NO_2 through other routes (e.g., through reactions with VOCs) is assumed to be negligible. The approach yields estimates of hourly NO_2 and O_3, as well as an estimate of the road transport NO_2/NO_X emissions ratio for vehicles using Marylebone Road.

As concentrations of NO_X decrease in London, concentrations of NO_2 will also decrease, whereas O_3 will increase. Therefore, to calculate future concentrations of NO_X, NO_2 and O_3 at Marylebone Road, estimates are required of both the background concentration of these species and the contribution made by the road to NO_X and primary NO_2 on an hourly basis. The latter calculations are straightforward: the road contribution to NO_X is determined by the projected reduction in NO_X and the assumption for the primary NO_2 fraction applied to that concentration. At the background Kensington site, concentrations of NO_X, NO_2 and O_3 for a future year are calculated as follows. Hourly concentrations of NO_X, NO_2 and O_3 are predicted at Kensington by using the constrained model applied to the Kensington site paired with a rural site outside London. The rural site used was Harwell, 50 km west of central London. Concentrations of NO_X were reduced in line with projected trends based on, for example, a decline of 5.0 % yr^{-1}. New hourly concentrations of NO_2 and O_3 were calculated that then provided input to the roadside predictions. Note that it was assumed that the primary NO_2 fraction at calculated at the background site, which varied between 10-15 % by vol. in the period 1998-2004, was assumed to remain constant into the future. In practice, the primary NO_2 fraction at the background site may also increase in future, which would tend to increase predicted NO_2 concentrations beyond those calculated here.

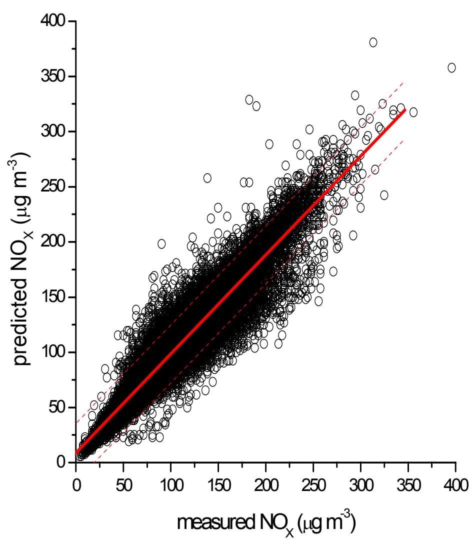
2. Future trend in NO_X concentrations

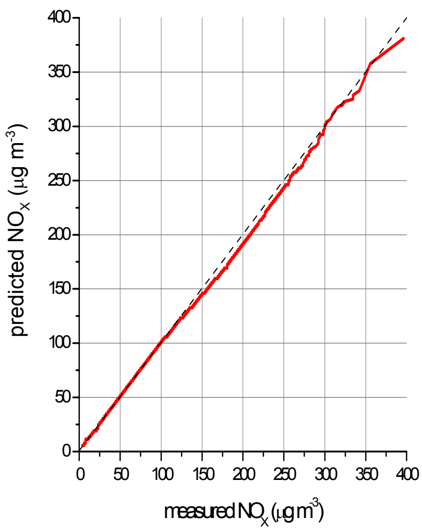
Two approaches have been considered to estimate NO_X concentrations in the period 2005-2015. First, an empirical approach has been used based on the current trend in NO_X at this location. Using annual mean data from 1998-2004, a linear regression showed that there has been a statistically significant downward trend in NO_X over this period, with a mean annual reduction corresponding to 5.0 % yr^{-1}. The first assumption therefore is to assume that this trend continue to 2010 and beyond. An alternative method relies on detailed dispersion modelling of NO_X concentrations using emissions inventories developed for London. Two model predictions at this location have been considered by AQEG (2004). An empirical approach suggests that from 2001 to 2010 the concentration of NO_X at Marylebone Road declines from 335 to 201 $\mu g\ m^{-3}$, corresponding to a mean decrease over this period of 4.5 % yr^{-1}. A more detailed modelling study using ADMS-Urban (Carruthers et al., 1998) gave a decrease from 390 to 220 $\mu g\ m^{-3}$ from 1999-2010; a mean decrease of 4.0 % yr^{-1}. The trend estimates are within $\pm$ 20 % of one another.

3. Quantification of uncertainties and assumptions

The three major sources of uncertainty in the model predictions have been addressed as follows. First, the model uncertainty in predicting hourly concentrations was derived from a comparison of model predictions with measurements from 1998-2004. Figure 2a shows the measured vs. predicted scatter plot together with 95 % confidence intervals. Also shown (Figure 2b) is a Quantile-Quantile (Q-Q) plot of the hourly values, which highlights that the model produces consistent estimates of NO_2 across the entire concentration range of the measurements. The results of the Q-Q plot are important because it demonstrates that the model does not introduce a bias at high (> 200 $\mu g\ m^{-3}$) NO_2 concentrations. Overall, the standard error of the predictions was $\pm$ 14.0 $\mu g\ m^{-3}$ and this is the uncertainty assumed in the model for hourly predictions. The Kolmogorov-Smirnov test showed that the residuals were normally distributed. Second, the base case uncertainty associated with the trend in NO_X was assumed to be -5.0 % yr^{-1} $\pm$ 25 % (1 σ) consistent with observed and predicted NO_X trends. Furthermore, it was assumed that this uncertainty is normally distributed. It was not possible to quantify the uncertainty in future NO_X concentrations based on reported uncertainty information because of a lack of quantitative data e.g. for emissions projections. The uncertainty of $\pm$ 25 % was therefore adopted as a sensitivity test. Finally, future concentrations of NO_X, and in particular high percentile values of NO_X, are also strongly dependent on the assumptions of the meteorological year assumed. The influence that the inter-annual variation of meteorology has on concentrations was considered by projecting forward from each of the 7 years (1998-2004) to years 2005-2015. Monte Carlo simulations for future NO_2 predictions were undertaken by accounting all these uncertainties on the predicted hourly NO_2 concentration. This process resulted in the generation of 1000 sets of hourly NO_2 predictions for a future year, which could then be analysed. The simulations were varied depending on the assumptions for the projected decline in NO_X, the primary NO_2 fraction and the base meteorological year.

Figure 2: a) Hourly modelled vs. measured NO_2 concentrations from 1998-2004 with a linear fit and 95 % confidence intervals, b) Quantile-Quantile plot of measured and predicted hourly NO_2 concentrations from 1998-2004. The dashed line shows the 1:1 relationship.



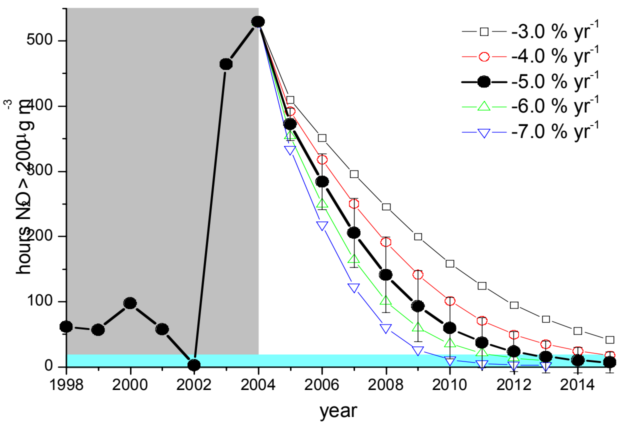


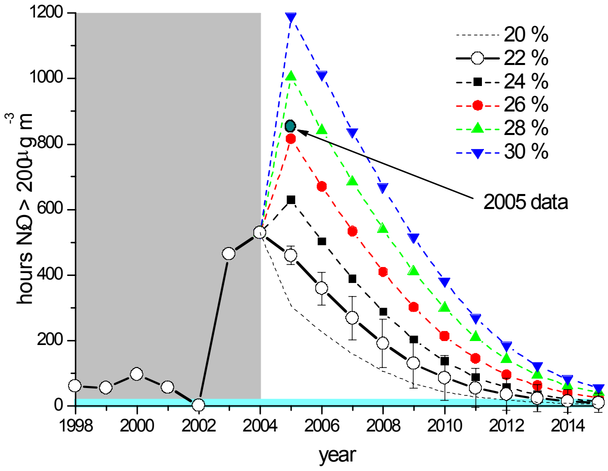
Results

Three principal influences on future peak hourly NO_2 concentrations have been considered in detail. First, the influence of projected decreases in NO_X concentration has been considered. Second, the affect of different assumptions on future primary NO_2 emissions from road traffic on Marylebone Road have been considered. Third, the inter-annual variability of meteorology on peak hour concentrations has assessed. Figure 3a shows the projected number of hours each year where NO_2 is > 200 $\mu g\ m^{-3}$. In this plot, projections have been based on 2004 meteorology (the most recent year in which data were available) and a primary NO_2 level of 22.0 % by vol., corresponding to the value calculated at the end of 2004. The uncertainty in these predictions, shown for the -5.0 % yr^{-1} decline in NO_X, is shown to increase until 2008 and then decrease. The decrease in uncertainty after 2008 is because there is an increasing chance that there are no hours where NO_2 > 200 $\mu g\ m^{-3}$. These results also highlight that the decline in the risk of exceeding the Limit Value is at a slower rate than the increase from 2002-4. Based on 2004 meteorology and projected decreases in NO_X it is likely that the Limit Value will not be met at this location.

Figure 3b shows the effect of the assumptions for primary NO_2 on future hourly NO_2 concentrations. Projections were made from 2004 based on a -5.0 % yr^{-1} decline in total NO_X concentration. Road vehicle primary NO_2 levels from 20 to 30 % by vol. were considered. This plot highlights the high level of sensitivity to future assumptions for the primary NO_2 emission level. In particular, Figure 3b shows that if primary NO_2 levels increase beyond the 22 % observed at the end of 2004, the number of hours where NO_2 > 200 $\mu g\ m^{-3}$ increases considerably. Data for 2005 does indeed reflect this effect: 853 hours were above > 200 $\mu g\ m^{-3}$ and the level of primary NO_2 is estimated to be approximately 25 %. Furthermore, it is clear that if primary NO_2 were to increase beyond 2004 levels, the risks of exceeding the 2010 limit would increase substantially.

Figure 3: a) Values in the shaded grey area show measured number of hours where NO_2 > 200 $\mu g\ m^{-3}$. The effect of different NO_X reduction projections are shown from 2004-2015 assuming a primary NO_2 ratio of 22.0 % by vol. Error bars (1 σ) are only shown on the -5.0 % yr^{-1} projection for clarity. b) Projected hours where NO_2 > 200 $\mu g\ m^{-3}$ for different percentage primary NO_2 assumptions, based on -5.0 % yr^{-1} NO_X reduction. Provisional data for 2005 are also highlighted. Error bars (1 σ) are only shown on the 22.0 % by vol. primary NO_2 for clarity.





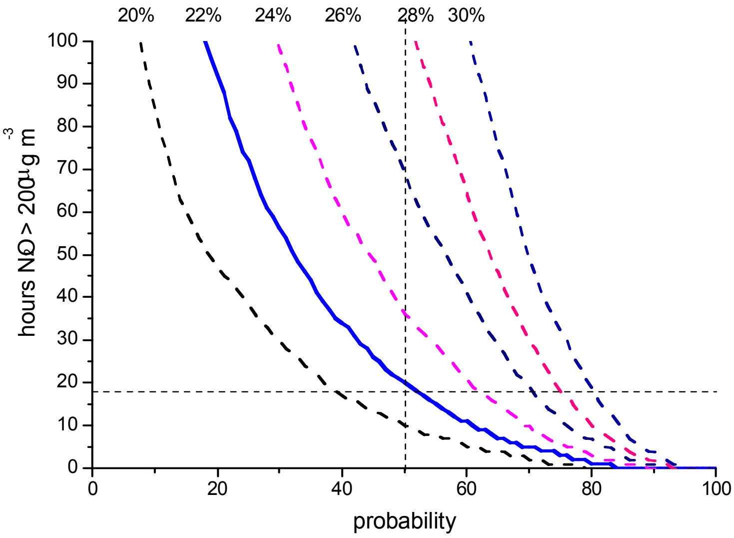
Modelling of 2004 hourly data at Marylebone Road indicates that 66 % of the NO_2 can be identified as being primary in origin for concentrations of NO_2 > 200 □g m^{-3}, which highlights the importance that primary emissions have on high percentile concentrations of NO_2. Interestingly, it is estimated that there would have been 10 ± 4 (2 σ) hours > 200 μg m^{-3} NO_2 in 2004 if primary NO_2 emissions had remained equal to historically typical values of 10.0 % by vol. Therefore, it is very likely that the hourly Limit Value would have been met in 2004 if the NO_2/NO_X emission ratio had remained at 10.0 % by vol. These results also suggest that the potential delay in meeting the Limit Value due to increased ratios of NO_2/NO_X is substantial. For example, if NO_X concentrations continue to decline at historically typical rates and the NO_2/NO_X ratio were to remain at 22.0 % by vol., then it is estimated that the EU Limit Value would not be met until 2014; a delay of 10 years. Any further increase in the ratio beyond 22.0 % would delay compliance still further. The analysis also shows that increases in primary NO_2 have had a strong effect on peak hourly concentrations. Assuming 22 % NO_2/NO_X results in a peak hour concentration of 337 μg m^{-3} in 2004, whereas 9.5 % yields 225 μg m^{-3} NO_2.

Table 1: Probability (%) that the hourly concentration of NO_2 will be greater than 200 μg m^{-3} for at least 18 hours each year in 2010. Numbers in bold text show probabilities > 75 %.

	Primary NO_2 (% by vol.)											
Primary NO_2	20%		22%		24%		26%		28%		30%	
NO_X reduction (% yr^{-1})	-5%	-4%	-5%	-4%	-5%	-4%	-5%	-4%	-5%	-4%	-5%	-4%
1998	11.4	33.2	15.8	41.8	20.6	55.0	32.2	67.8	37.4	74.8	48.8	**79.8**
1999	33.8	70.4	47.0	**78.2**	55.4	**85.4**	64.0	**90.6**	64.0	**92.2**	71.4	**94.8**
2000	43.8	**75.4**	55.8	**85.6**	65.4	**92.2**	72.0	**95.8**	**80.8**	**97.6**	**83.4**	**98.8**
2001	41.0	**75.**	57.0	**87.**	63.0	**92.4**	73.	**97.**	**78.**	**98.4**	**82.**	**99.2**

		4		**0**			0	**2**	**6**		**8**	
2002	8.2	22.0	18.2	47.2	29.6	71.4	50.6	**80.0**	64.2	**92.0**	71.2	**97.8**
2003	68.2	**95.8**	**82.6**	**99.0**	**91.0**	**99.2**	**95.4**	**99.8**	**97.2**	**100.0**	**99.0**	**100.0**
2004	71.8	**96.0**	**87.8**	**99.6**	**95.8**	**100.0**	**98.8**	**99.8**	**99.6**	**100.0**	**99.8**	**100.0**

Figure 4: Probability (%) that a number of hours where NO_2 > 200 μg m^{-3} will be exceeded for different primary NO_2 ratio assumptions in 2010. It is assumed that the NO_X concentration trend is – 5.0 % yr^{-1}. The horizontal dashed line shows the 2010 EU hourly limit value; the vertical dashed line corresponds to the best estimate. Note, that these results consider the uncertainties introduced due to the inter-annual variability in meteorology.



The analysis above was based on projecting forward from 2004. However, the inter-annual variation in meteorology is also an important consideration; particularly for high percentile concentrations of NO_X and NO_2. Furthermore, each year is also influenced by different concentrations of background ozone e.g. in 2003 when background oxidant levels were high. The influence of inter-annual variations in meteorology has been considered by projecting forward from each one of the years 1998-2004 up to 2015. These simulations were used to calculate the probability that there will be an exceedance of the 2010 Limit Value for different assumptions of primary NO_2 and NO_X reduction. Table 1 shows the results of these simulations. It is clear from these results that the inter-annual variation in meteorology has an important affect on the probability that the limit value will be exceeded. For example, for a 20 % primary NO_2 assumption and a 5.0 % yr^{-1} reduction in NO_X, the probability on not meeting the limit varies from 8.2 % (2002 meteorology) to 71.8 % (2004 meteorology). Years that result in a low probability are 1998 and 2002, whereas years that might be considered as “worst-case” are 2003 and 2004. Figure 4 shows the effect of combining uncertainties in the NO_X trend estimate, primary NO_2 and meteorological year. For NO_2/NO_X ratios > 22 %

there is more than a 50 % chance that the hourly 2010 Limit Value will be exceeded.

The effect of increased NO_2/NO_X emissions ratios is very clear at a roadside site such as Marylebone Road, and similar increases have been observed in recent years at other roadside sites (Carslaw, 2005). At background sites it is likely that such increases will be manifest as increases in both NO_2 and O_3, through the photolysis of NO_2. Indeed, any increase to urban O_3 concentrations could be seen as more important than increased NO_2 concentrations because of the health risks associated with O_3. Work is required therefore that considers whether urban O_3 concentrations have increased beyond that expected through reduced titration with NO due to decreased concentrations of NO_X.

Conclusion

This work has shown that at a heavily-trafficked location in central London that the EU hourly Limit Value for NO_2 has been exceeded by a wide margin in 2003 and 2004. Analysis of the measurements shows that these exceedances have been driven by recent increases in the ratio of NO_2/NO_X in vehicle emissions. If the NO_2/NO_X ratio from road traffic remains the same as that at the end of 2004 and NO_X concentrations continue to decrease in line with past trends, then by 2010 the number of hours > 200 μg m^{-3} will be close to 18. However, if the NO_2/NO_X ratio continues to increase, as shown by the analysis of more recent measurements in 2005 and by a consideration of the factors affecting emissions of primary NO_2, then it becomes much more likely that the EU Limit Value will not be met. This paper also considered the effect that the inter-annual variation in meteorology has on peak hourly NO_2 concentrations. These results highlighted that the variation due to meteorology is a major factor controlling the peak hourly NO_2 concentrations. Therefore, the risks of exceeding the EU Limit Value in 2010 will depend strongly on the prevailing meteorological conditions. The re-calculation of NO_2 concentrations in 2004 with an emission ratio of 10.0 % (i.e. a historically typical level), showed that it was likely the Limit Value would have been met. This analysis suggests that the recent increases in the primary NO_2 emissions ratio at this location may delay meeting the Limit Value until 2014 i.e. a period of 10 years; or longer if the primary NO_2 emissions fraction continues to increase. Although only 8 % of European urban locations affected by road traffic exceeded the EU hourly Limit Value in 2003, similar influences are also likely to affect many of these sites, such as increases in diesel vehicles with oxidation catalysts and catalytic diesel particulate filters. It also seems likely that the NO_2/NO_X ratio will continue to increase in the future due to the increased penetration of diesel vehicles in the passenger car fleet. Further emissions testing and modelling work is therefore required to assess the risks of exceeding the EU Limit Value in European cities in 2010.

Acknowledgments

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NO-NO_2 of Diesel Engines with Different Catalysts and Different Measuring Systems

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Abstracts

During the VERT testing of different DPF systems it was remarked, that the oxidation catalyst converts sometimes a big part of NO to NO_2, producing on the one hand a more toxic composition of the exhaust gases and causing on the other hand measuring artefacts, which tend to underestimate of NO_2 and NO_x by the cold NO_x - measurement. The present paper shows some examples of NO_2-production on different engines with different types of DPFs, oxidation catalysts and fuel borne catalyst. Some improvements of NO_2-measuring accuracy are demonstrated.

***Keys-words**: NO_x emissions, toxicity, exhaust gas, oxidation catalyst*

Résumé

Durant les tests de qualité des systèmes FAP (VERT), il a été constaté que les catalyseurs d'oxydation convertissent parfois une grande partie de NO en NO_2 en engendrant ainsi, d'une part, plus de composants toxiques dans les gaz d'échappement et, d'autre part, certains artéfacts qui provoquent une sous-estimation de NO_2 et NO_x pendant la mesure NO_x froide. Le travail présente des exemples de production de NO_2 sur différents moteurs, avec différents types des FAP's catalyseurs d'oxydation et additifs au carburant. Les possibilités d'amélioration de l'exactitude de la mesure NO_2 sont démontrées.

Introduction

The objective of project VERT (Abbreviations: see at the end of paper), [1,2,3], which was promoted by the Swiss, Austrian and German occupational insurances and the Swiss EPA (BUWAL)) since beginning of 90-ties was to consequently improve the air quality at the working places in tunnelling and mining.

The fact, that an oxidation catalyst, which often is used as a key element of the DPF regeneration concept, can increase the NO_2-portion in the exhaust gas, was of big concern to the VERT committee. NO_2 is more toxic than NO and all systems, which increase this component are undesirable in underground as well as in all situations, where people are exposed to higher exhaust gas concentrations, like in congested agglomeration centers.

In the present exhaust gas legislations for on-road vehicles the nitric oxides are measured in summary as volumetric NO_x - concentration and recalculated in the mass-emission by means of the density of NO_2, even if there is usually a relatively low NO_2 content in NO_x at tail pipe.

This consideration does not respond to the objectives of health protection in closed spaces, tunnels and mines, where the development of toxic components, like NO_2 has to be minimized.

In this situation, it was necessary to investigate more carefully the influences of some DPF systems on NO/NO_2 and give more attention to the NO_x measuring procedure.

NO_x in VERT procedure

The first objective of the present work was to measure the NO_x in parallel with cold and with hot measuring installation during the VERT Filter Test (VFT) and VERT Secondary Emission Test (VSET) with different DPF-systems.

In addition to that it should be stated if the accuracy of hot NO_x-measurement can be improved by drying the hot and humid sample gas probe by means of a special Nafion permeability membrane dryer.

The most used engine in the VERT procedure is the 4 cylinder Liebherr D914T, with 6.11 dm^3 and 105 kW.

1. NO_x-measuring systems

The NO_x measuring systems are represented in Fig.1.

The traditional cold system has several coolers and water separators, which allow the contact of the exhaust gas sample with the condensed water and due to that a loss of a part of NO_2, which is particularly water-soluble.

The used cold analyzer was Horiba-CLA-510 chemoluminescence detector with two operating modes to be switched for NO or NO_x.

The hot measuring system keeps the temperature of the sample gas probe – from the heated prefilter, heated line to the heated analyzer – at approx. 190°C, excluding any possibility of water condensation. The used of hot analyzer was also a chemoluminescence detector from Eco Physics CLD 700 EL ht with a simultaneous reading of NO, NO_2 & NO_x.

Nevertheless, a little part of NO_2 can still react with the overheated water vapor and get lost for the NO_x-measurement. To minimize this possibility a Nafion permeability membrane dryer from Perma Pure Inc. was applied.

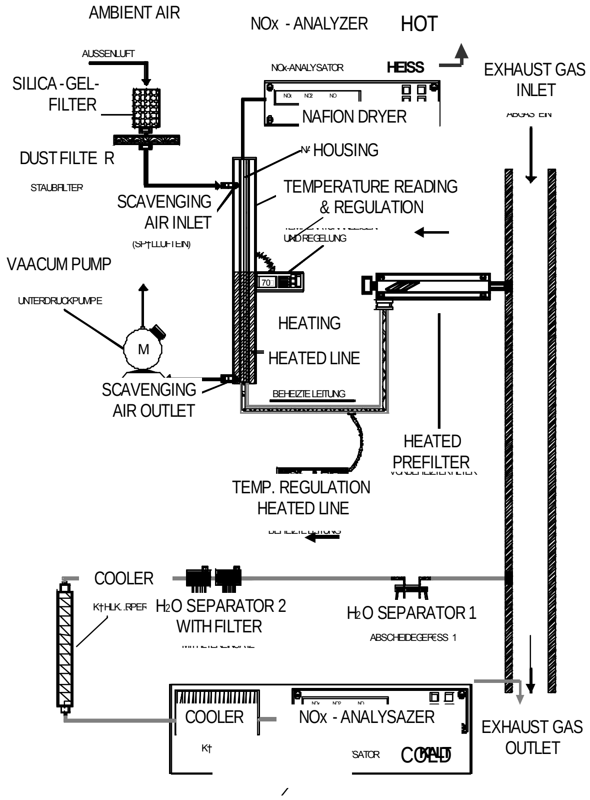


Fig. 1: NO$_X$ measuring systems: hot analyzer with heated line and Nafion Gas Sample Dryer and cold analyzer

2. Principle of Nafion Drying

When gas containing water vapor passes through Nafion tubing, the water is absorbed by and moves through the walls of the tubing, evaporating into the surrounding air in a process called perevaporation. The remaining components in the gas are unaffected.

The reaction is driven by the humidity gradient until equilibrium is reached. If a dry purge gas flows over the exterior surface of the Nafion tubing, water vapor will be continuously extracted from the gas stream inside the tubing until the sample humidity matches that of the purge gas.

Since water is absorbed into the tubing from the vapor phase and evaporates from the exterior of the tubing into the vapor phase, there is no net change of free energy. No external energy source is needed to drive the reaction, other than a supply of dry purge gas. Perma Pure gas dryers enclose one or more strands of Nafion tubing in a shell with fittings to supply a countercurrent purge gas flow. Water vapor from the sample stream is carried away by the purge gas, and the sample is dried without being exposed to condensate or any material that might absorb analyte gases. A purge gas flow rate twice that of the sample flow rate is sufficient to achieve full drying performance.

3. First exemples of DPF's from VFT & VSET

In the first VERT internal quality investigations the NO_x-emissions (as NO and NO_2) were compared with hot and with a cold measuring systems by different DPF's, [4].

The comparisons and a rough averaging of the NO_x-increase with hot measurement are represented in Fig.2.

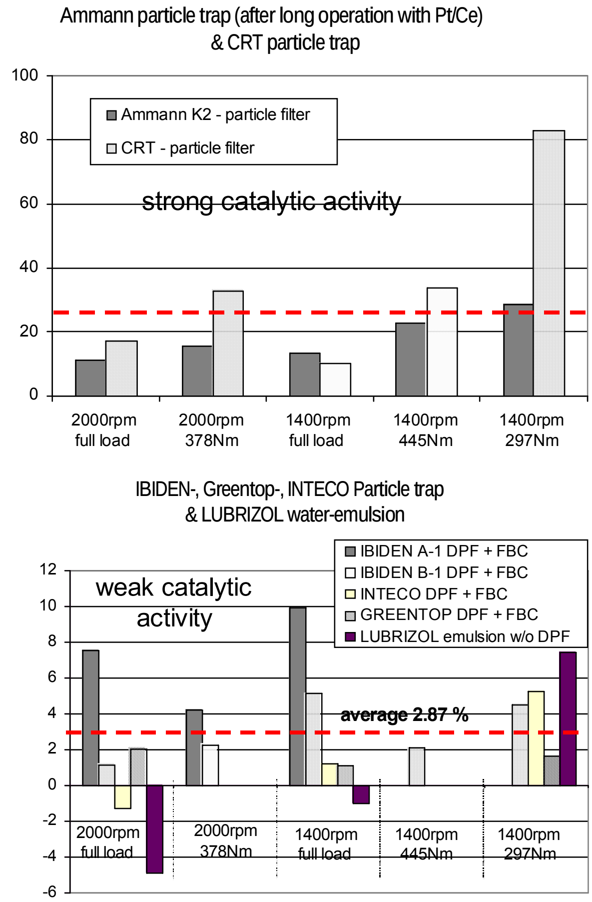


Fig. 2 Increase of the Nox concentration with hot measuring system relatively to the cold measurement.

These comparisons subdivide the DPF systems in two groups according to the catalytic intensity: strong and weak. (distinction: weak i.e. (NO_x hot - NO_x cold) <10%).

The NO_x-values measured with hot system are higher than with cold system –

with strong catalyst in average 27% and with weak catalyst in average 3%.

4. Nafion Dryer

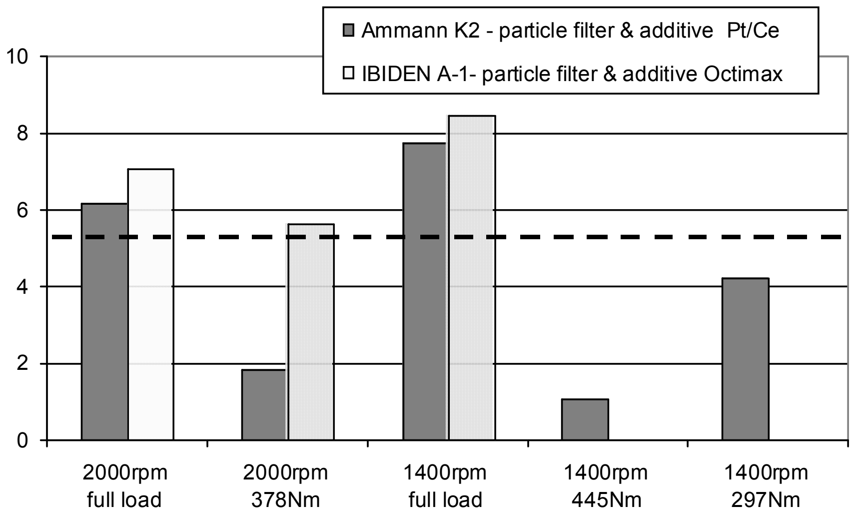
Also the effects of Nafion Dryer with hot measurement were investigated.

With a weak catalytic intensity, where there is a low conversion of NO to NO_2 the use of Nafion Dryer (ND) does not cause any visible improvement of NO_x measuringaccuracy.

In spite of that with strong catalyst and consequently with higher NO_2-rates the ND enables to measure more NO_X, Fig.3.

These results have been confirmed afterwards in several measuring series of didactic projects.

Figure 3: Increase of the NO_X concentration measured hot with Nafion Dryer relatively to the hot measurement without Nafion Dryer



5. Influence of exhaust gas temperature

Fig. 4 shows typical plots of NO_2 /NO_X ratio in function of exhaust gas temperature without and with strong catalytic coating (CRT, or Ox.Cat.).

The used aftertreatment elements were:

- CRT: continuously regenerating trap from HJS (Ox.Cat. + DPF ceramic monolyth, the same which was used in Figure 2 but after some years of storage and sporadic use for didactic projects).
- Airmeex SiC monolyth (without coating).
- Ox.Cat. from Scania (for HD-application, with an intense Pt-coating).

With the lowest t_{exh} with CRT there is no conversion of NO to NO_2, because the catalyst is still to cold. With increasing t_{exh} the catalytic conversion becomes more intense until the maximum of NO_2 /NO_X ratio at approx. 300°C-350°C. With further increase of the temperature the thermal dissociation of NO_2 starts to overcompensate the catalytic conversion causing the decrease of NO_2 share.

In the exhaust gas without CRT (w/o catalyst) there is no increase of the NO_2 /NO_X ratio with the higher temperature. Only at the lowest engine charge and at the lowest t_{exh} there is a remarkable increase of the NO_2/NO_X ratio. Due to the very low NO_X-values in this region of the lowest t_{exh} only very little absolute differences of NO_2 cause relatively big differences of NO_2 /NO_X ratio.

On the other hand it can be assumed for the lowest t_{exh} and lowest gas flows with CRT that due to absorption in the very big and relatively cold internal DPF-surface there are losses of NO_2, which can react with the humidity (HNO_3, HNO_2) and with other substances (nitrates & nitrites).

Figure 4: Effects of different aftertreatment devises on NO_2-portion in exhaust gas engine Liebherr 914 T, 1400rpm, ULSD Diesel S<10ppm FBC Octel Octimax 4810a 1.8x (29ppm Fe/ 7ppm Sr)

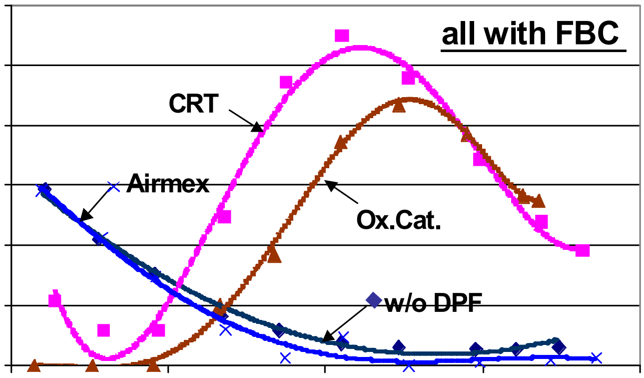


Fig.4 confirms that the catalytic systems (CRT & Ox.Cat.) provoke a strong NO-NO_2 conversion in the temperature window of about 230ºC-430ºC and also losses of NO_2 by the temperatures below 200ºC.

The systems: w/o DPF and Airmeex, as well as the presence of FBC have no influence on NO_2.

Airmex without catalytic coating shows no losses of NO_2 in the low temperature region, which means that not only the large surface, but also the presence of a catalytic washcoat plays here an important role.

6. FBC versus Ox.Cat

Several systematic investigations of the NO-NO2 conversion and the NO_2 /NO_X ratio with hot NO_X-measurement were performed with different FBC's.

The standard procedure on the Liebherr D914T engine was to increase the exhaust gas temperature by means of increasing the torque at constant engine speed of 1400 rpm.

All measuring series were performed with the same lube oil and with the same fuel ULSD S<10ppm.

Fig. 5 visualizes clearly that there is no influence of Fe/Sr-additive on the NO_2-values, this with and without CRT.

Also the overdosing of those additives has no significant impact on the engine-out NO_2 emissions and on the NO_2 /NO_X-ratio.

These results with additive represent short term effects, because the working periods with FBC were not very long. Nevertheless there are experiences especially with Pt-additives, which show a strong increase of a catalytic activity of a trap due to the collected deposits of FBC (see Ammann K2 in Fig. 2 & 3).

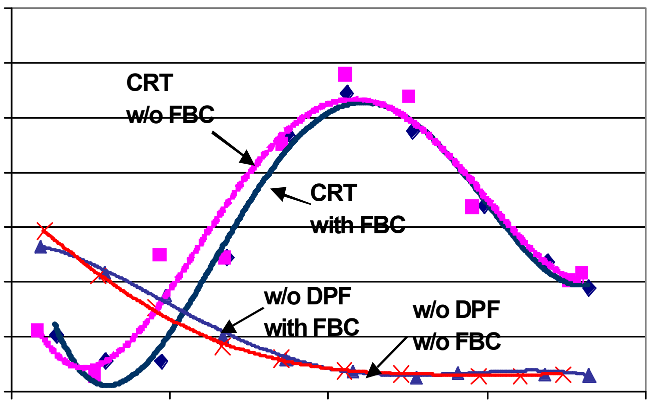
Platinum is generally known as a strong catalyst and also the Pt-coatings (as well as Pt-deposits) show the strongest NO-NO_2 conversion.

The increased NO_2/NO_X-ratios were confirmed in several VERT-tests with different DPF-systems.

There are some base-metal-coatings, which have no influence on NO_2 at all.

The differences of results with different Pt-coating are due to the coating itself, but are also caused by different constructions of DPF, i.e. different space velocity and different local thermal situation.

Figure 5: Influence of FBC on NO_2-ratio with/without CRT engine Liebherr 914 T, 1400rpm, ULSD Diesel S<10ppm FBC Octel Octimax 4810a 1.8x (29ppm Fe/ 7ppm Sr)



Experiences on other engines

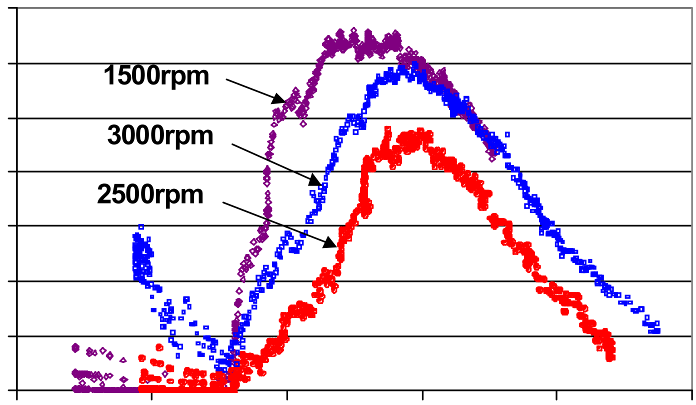
1. VW-TDI Test-Engine, 4 cyl., 1896 cm^3, 85 kW

Fig. 6 shows the examples of NO_2/NO_x ratio and NOx-concentrations with the original catalyst at other constant engine speeds. The NO_2 ratio i.e. also the NO_2-conversion depends on the conditions: pressure, temperature and residence time in the catalyst; therefore different maximal values of NO_2/NO_x at different temperatures result.

Other important experiences on this engine are :

- By the lower NOx-emission values, below t_{exh} 380ºC there are almost no differences of NOx (cold / hot), but the little differences of NO_2 cause the very big differences of NO_2/NOx -ratio.
- The reduction of NOx in the catalyst is only an artefact of the cold measurement, where the higher NO_2-share after the catalyst is more absorbed in the condensation water provoking more NOx-losses, [5].

Figure 6: NO_2/ NO_X on the VW-TDI engine at 1500-2500-3000 rpm, w/o add.



2. Peugeot 406 with DPF, 4 cyl., 1997 cm^3, 79kW

This is a representant of the first DPF-system, which was introduced in series production of passenger cars.

The DPF regeneration is promoted by the FBC, by the Ox.Cat. upstream of the DPF, but first of all by the potential of the Common Rail diesel injection system to increase the exhaust gas temperature very quickly according to the necessity. The control of the DPF system is perfectly integrated in the vehicle OBD.

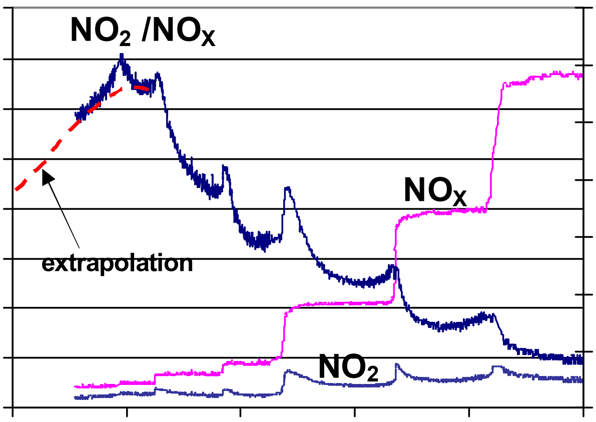
Fig. 7 shows the NO_x-results during driving the vehicle at 80km/h and varied torque on the chassis dynamometer.

The DPF is placed nearer to the engine outlet, than on the Liebherr engine, therefore at the lowest engine charge with low absolute NO_x-values the attained temperatures before DPF are high (approx. 350°C). With increase of temperature and torque the NO_2/NO_x-ratio decreases, like in usual case “without catalyst”. This is due to the upper limit of the temperature window for NO_2-conversion (until approx. 430°C), where the thermal dissociation predominates.

In spite of the presence of a catalyst in the DPF regeneration system of Peugeot there is no major problem about NO_2-emission, because the temperature window of the maximum

NO_2-production coincides with the lowest absolute engine-out emissions.

Figure 7: NO_2/NO_X after catalyst and particule trap on PEUGEOT 406 with FAP engine at 1715 rpm, 80km/h, on roller test bench



3. VW-Transporter, TDI, 4 cyl., 2461 cm^3, 75 kW

Similarly to Peugeot this vehicle (LDV) was driven on the chassis dynamometer at constant speed with increasing torque.

Figure 8: VW Transporter 2.5 TDI at const. speed 74 km/h, engine at 1800rpm NOx, NO_2 and temperature before cat. (after engine) and NO_2 after cat. Force at wheel : 0 - 1500 N

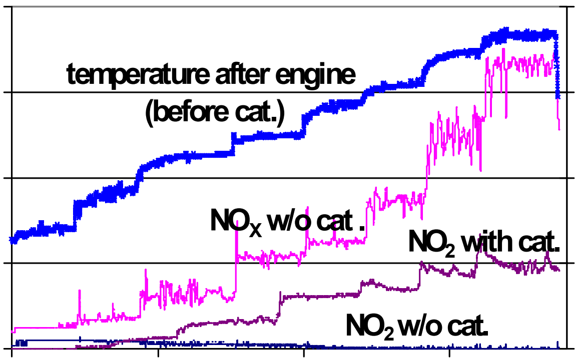


Fig. 8 shows the time plots of temperature, NO_x & NO_2 before catalyst at 74 km/h during the stepwise increasing engine load. A simple tube replaced the catalyst in this test. The trace of NO_2 after catalyst is result of an identical measuring procedure with catalyst installed.

A considerable NO_2-production in the ox.cat. takes place until approx. 450°C. Fortunately a big part of this temperature window (below 300°C) responds to very low absolute NO_2-values. The NO-NO_2-conversion in the ox.cat. of this LDV is similar to the conversion on the Liebherr engine.

Summarizing it can be said, that the oxidation catalysts on passenger cars, or LDV's convert also NO to NO_2, but the problem of absolute NO_2-emissions is less acute than on the HD-engines due to more frequent low load engine operation and sometimes due to a nearer position of the catalyst to the engine, which moves the window of maximum NO_2-conversion to the very low absolute NO_2-values. Obviously, the composition of catalytic coating influences the conversion rate.

Conclusions

- With a strong oxidation catalyst in the exhaust system the hot NO_x-measurement allows to recognize the NO_x-values, which are in average of the investigated cases 27% higher, than with cold NO_x-measurement,
- A supplementary use of a Nafion-permeability sample dryer (still in case of a strong catalyst) enables to measure in average 5% higher NO_x-values,
- With a weak catalytic influence in the exhaust system the hot NO_x-measurement yields higher values until 3% and the use of Nafion dryer is not necessary,
- The present configuration of Nafion dryer is not appropriate for transient measurements,
- With a coated catalyst (Ox.Cat), or with catalytic surface filter (CSF) there is a maximum of NO_2 / NO_x-ratio typically in the exhaust gas temperature range of 300°C – 350°C,
- With fuel borne catalyst (FBC), or with noncatalyzed DPF + FBC there is no NO - NO_2 conversion, nevertheless FBC can affect (increase) the conversion activity of an Ox.Cat, or DPF in particulary due to the long term deposits,
- The above tendencies were confirmed on different engines.

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Abbreviations

AEEDA... Association Européenne d'Experts en Dépollution des Automobiles
AFHB … Abgasprüfstelle der Fachhochschule, Biel CH, www.hti.bfh.ch - Auto - AFHB
(Lab. For Exhaust Gas Control,
Univ. of Appl. Sciences, Biel-Bienne, Switzerland)
BUWAL.. Bundesamt für Umwelt, Wald und Landschaft (Swiss EPA, SAEFL)
CARB … Californian Air Resources Board
CRT … continuously regenerating trap
CSF … catalytic surface filter
DPF … Diesel Particle Filter
EMPA… Swiss Federal Laboratories for Materials Testing and Research
EPA … Environmental Protection Agency
FBC … fuel borne catalyst
LDV … light duty vehicle
LSD … low sulfur diesel
ND … Nafion Dryer
NO … nitrogen monxide, nitric oxyde
NO_2 … nitrogen dioxide, nitric dioxyde
NOx … nitric oxides = NO + NO2
OBD … on board diagnostics
Ox. Cat… oxidation catalyst
PAH … policyclic aromatic hydrocarbons
PM … Particulate Matter, Particle Mass
SAEFL… Swiss Agency for Environment, Forest and Landscape
SUVA ... Schweizerische Unfall Versicherung Anstalt (Swiss Occupational Insurance)
texh ... exhaust gas temperature
TTM … Technik Thermische Maschinen, Switzerland
ULSD ... ultra low sulfur diesel
VERT … Verminderung der Emissionen von Realmaschinen im Tunnelbau
(Swiss – Austrian – German project, DPF retrofitting in underground)
VFT … VERT Filter Test
VSET ... VERT Secondary Emissions Test

Modelling Road Transport Emissions on European Scale

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Abstract

A conceptual framework is presented that links a transport emission tool, a spatial resolution tool and an optimisation model for the assessment of abatement strategies. The objective is to model road transport emissions on European scale with a high spatial resolution and to use this information for a detailed evaluation of cost-effective abatement measures. Taking Italy and Sweden as examples, different road types (rural, urban and highway) as well as different types of emissions (hot and cold start emissions with emission factors from HBEFA) are considered to model air pollutants such as NO_x and PM_{10}. Additionally, non-exhaust PM_{10} emissions from tyre and brake wear as well as from road dust suspension are included and an assessment for the abrasion from studded tyres in urban Sweden is given. Vehicle fleet projections for 2020 are used to assess effects on emissions. It can be shown that in 2000, Italian two-wheelers and pre-Euro passenger cars had a significant impact on urban air quality and that in 2000 as well as in 2020, the majority of urban PM_{10} emissions in Sweden derives from non-exhaust sources.

Keywords: *road transport, spatial resolution, particulate matter, exhaust emissions, non-exhaust emissions.*

Introduction

Emissions caused by road transport are a main contributor to air pollution, especially in urban areas. Primary particulate matter (PM) such as PM_{10} and secondary aerosols formed by gaseous precursors in the atmosphere have significant impacts on health and increase the risk for heart diseases, according to WHO (2003). Although stringent EURO-emission standards have led to a decrease in ambient air concentrations of PM, NO_x and other gaseous pollutants, human exposure is still unacceptable in highly polluted urban areas.

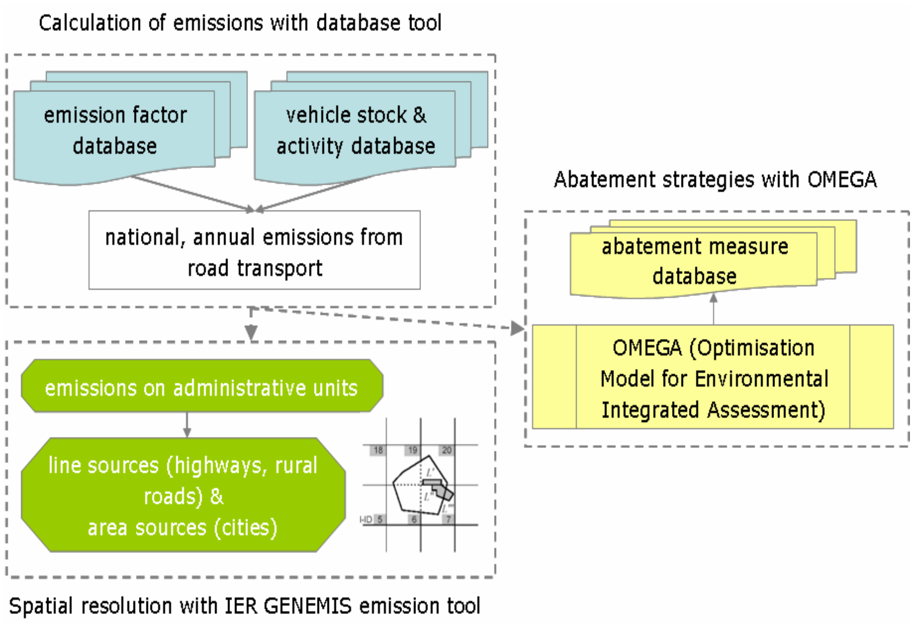
Road transport emission models are used for the quantification of air pollutants such as PM_{10} and NO_x. Together with chemistry transport models their results are taken to map and assess ambient air quality. For this, national total emissions have to be disaggregated on different scales with a high spatial resolution. With the help of integrated assessment models such as OMEGA, Reis et al. (2005), modelled emissions can finally be used to identify the most effective abatement strategy.

A conceptual framework is presented to link the above mentioned aspects to a toolbox of transport emission modelling and assessment of abatement measures on different spatial scales. In a whole, the model framework will be able to answer questions in the field of transport policy, emission legislation and regional/local effects of abatement measures. One main objective is a transparent and consistent European-wide quantification of transport induced air pollutants, including state-of-the-art methodology on exhaust and non-exhaust emission sources. A further objective is to distinguish between vehicle types, fuels and exhaust technologies, including a differentiation of Euro and pre-Euro emission standards which could be interesting for retrofit purposes. Another objective is to develop further methodologies for the spatial resolution of national emissions.

The concept and first results are presented using Italy and Sweden as an example to account for country specific climatic conditions and fleet composition. In Italy, especially in urban areas, two-wheelers are one of the major means of transportation. As motorcycles and mopeds have been included only recently in EU-wide emission standards, they are still a source for relatively high emissions. Vasic and Weilenmann (2006) show that in comparison to passenger cars, two-wheelers emit 2.7 and 16 times more CO and HC, respectively (Swiss fleet emissions in tons per year). In Sweden, tyre and road surface wear caused by studded tyres has a significant influence on the ambient PM emission level as described in Omstedt et al. (2005). In Nordic countries (Sweden, Finland, Norway), studded tyres are used during winter and spring by the majority of light duty vehicles and passenger cars.

Methodology

Figure 1: Model framework for the assessment of road transport emissions



A tool for emission calculation, a tool for spatial resolution of emissions and an optimisation model will be combined to a model framework (Figure 1) that enables the user to evaluate impacts of abatement strategies on regional or local emissions. One example could be the impact of an urban ban on driving for heavy duty vehicles on urban and non- urban emissions. Thus, transport and air pollution topics beyond the scope of one single model can be covered. In the following, methodological aspects of transport emission calculation and spatial resolution are described.

1. Emission factor datasets

Automotive emissions are usually modelled based on vehicle stock, the corresponding activity data and emission factors. For the model framework, this would comprise emission factors for different vehicle categories, technologies, fuels and road classes. Exhaust cold start and evaporation emissions are subject to temporal variations according to ambient temperature and should be calculated including regional or local climatic conditions. Non-exhaust emissions are varying depending on fleet composition and automotive equipment (as for studded tyres) as well as climatic conditions (as for suspension of road dust). A detailed calculation guideline can be found in European Environment Agency (2001).

Following the objective of consistent emission modelling, an emission factor dataset is needed that is applicable throughout Europe and refers to the same level of detail as the vehicle stock and activity data. It should also allow for adaptation to varying climatic conditions.

In HBEFA (2004), country specific information from Germany, Switzerland and Austria is used to model so called traffic situations, e.g. inner-city stop-and-go. Those traffic situations are the basis for a highly detailed emission factor data set with factors for hot, excess cold start and evaporation emissions. With the high number of traffic situations given in HBEFA, with the differentiation in vehicle technologies (Euro and pre-Euro emission standards) and with the envisaged high spatial resolution of transport emissions, the Handbook is suitable for the transport emission model calculations. As driving patterns are varying from country to country (e.g. different speed limits on highways), appropriate driving situations are chosen from HBEFA according to average speed per EU-15 country (given in COPERT III, Ntziachristos and Samaras (2000)). Cold start and evaporative emissions are temperature-dependant, therefore suitable seasonal temperature profiles are chosen from HBEFA according to national average seasonal temperatures. With these profiles, cold start and evaporation emission factors can be defined per country and season to account for climatic conditions.

NO_x and particulate matter from diesel exhaust emissions are modelled with emission factors from HBEFA. The PM_{10} fraction and exhaust PM_{10} emissions from gasoline engines are calculated using information from Pregger (2006). Non-exhaust emissions comprising tyre and brake wear, road dust suspension and the influence of studded tyres in Nordic countries during winter months are assessed with emission factors data sets listed in Table 1.

Table 1: Emission factors used for the calculations

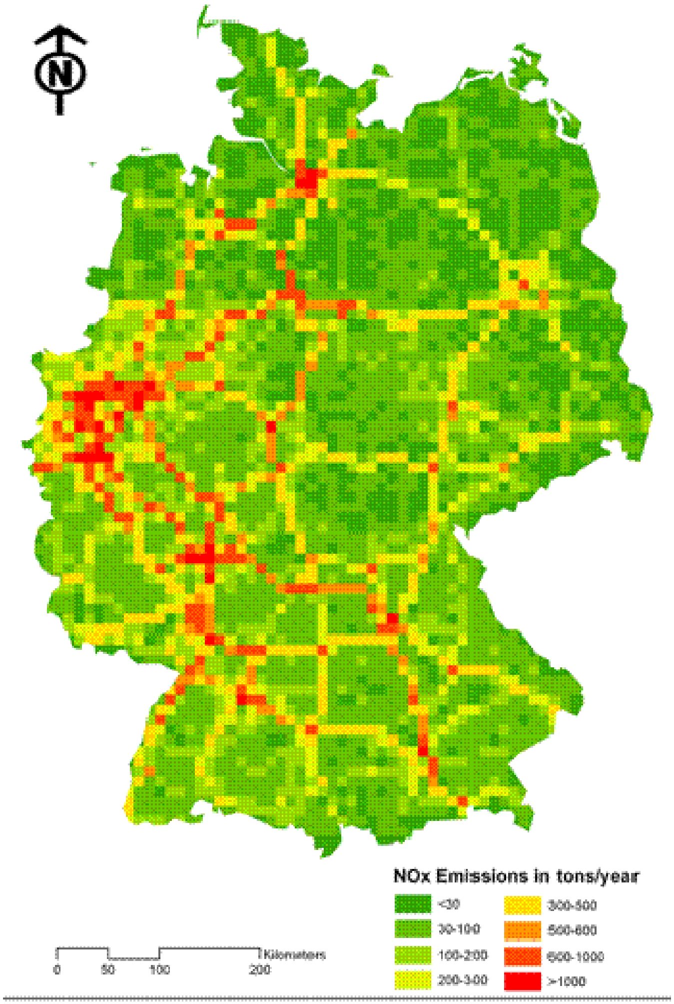
Type of emission factor	Vehicle category	Source
PM_{10} exhaust	diesel engines (passenger cars, light and heavy duty vehicles), gasoline engines (PC, LDV, two-wheelers)	Pregger (2006)
Brake wear	PC, LDV, HDV, two-wheelers	Garg et al. (2000)
Tyre wear	PC, LDV, HDV, two-wheelers	Gebbe et al. (1998), Rauterberg-Wulff (1998)
Road dust suspension in Italy and Sweden	PC including LDV, HDV	Düring et al. (2005)
Road dust suspension in Sweden (urban only)	Fleet average	Omstedt et al. (2005)

2. Spatial allocation of national transport emissions

In accordance to the calculation methodology, a top-down approach is chosen for the spatial allocation of road transport emissions. A split into urban, rural and highway driving is available for EU-15 within COPERT III. With the help of statistical data (e. g. from EUROSTAT) such as vehicle stock, population or road length per road category, country total emissions are allocated to administrative units (NUTS 1, 2 or 3) as a first stage of spatial allocation. These geographically resolved emissions are finally intersected using GIS (geographic information system) tools and digitised road maps covering Europe. The spatial resolution of line sources leads to a significantly better allocation instead of modelling area sources based on land use data. The emissions can be resolved on different spatial scales and grid resolutions and be provided as input data for atmospheric dispersion modelling. In this context, the IER GENEMIS emission model based on the work of Wickert (2001) will be updated and further developed.

An example for highly resolved road transport emissions is given for Germany in Figure 2. NO_x country total emissions reported to EMEP (2005) for the year 2000 were spatially distributed using the spatial resolution tool (the IER GENEMIS emission model). The spatial differentiation of traffic flow in mesoscale (10 x 10 km grid) enables to visualise traffic emissions and effects of emission abatement strategies identified at national scale. If annual traffic census data for single road segments is available, this information can be used in addition to information on road length to represent a spatial variation of traffic flow. This bottom-up approach was incorporated in the generation of Figure 2. The methodology for allocation and grid intersection is shown for line and area sources in Figure 3.

Figure 2: Example for top-down-allocation of emissions and detailed spatial resolution: NO_x emissions from road transport in Germany 2000 (data taken from EMEP(2005)



Results

Because of the focus on spatial resolution, the transport emission model framework has a European and a regional perspective. National and regional traffic properties concerning fleet composition, technological standards and climate conditions are an important input to spatially resolved emission calculation. For Sweden and Italy, stock and activity data and their projections to 2020 were taken from TRENDS, a road transport database for EU-15, see LAT et al. (2002).

Fleet composition is very different for both countries (Figure 4), and the high amount of two-wheelers in Italy is of special interest for urban air quality as motorcycles and mopeds have only been included recently in EU-wide emission standards. Figure 5 demonstrates that two-wheelers are of equal importance for PM_{10} exhaust emissions as diesel and gasoline passenger cars.

Vehicle technology and spatial distribution of emissions are further important aspects of transport induced air pollution. For Italy it could be demonstrated that in 2000, rural traffic was responsible for a large part of NO_x emissions and that pre-Euro passenger cars were the dominating NO_x emitter (Figure 6). Their disappearance in 2020 will lead to a remarkable drop in the overall emission level. For rapid emission reduction, however, technical reduction measures could be a reasonable and cost-effective alternative.

Figure 3: Methodology for the spatial resolution of transport emissions with highway and rural emissions as line sources and urban emissions as area sources, Wickert (2001)

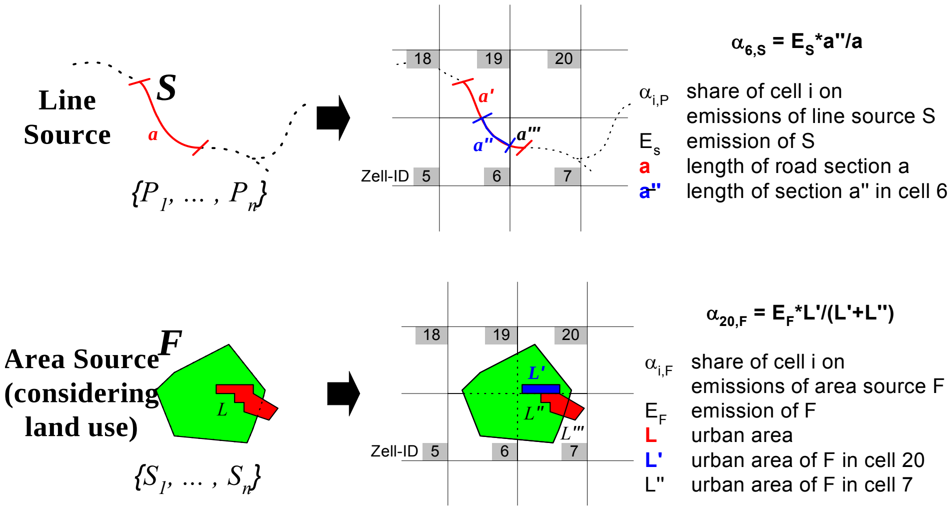


Figure 4: Fleet composition in Italy (left) and Sweden (right) in 2000 (HDV=heavy duty vehicles; PC=passenger cars; LDV=light duty vehicles; LPG=liquified petroleum gas)

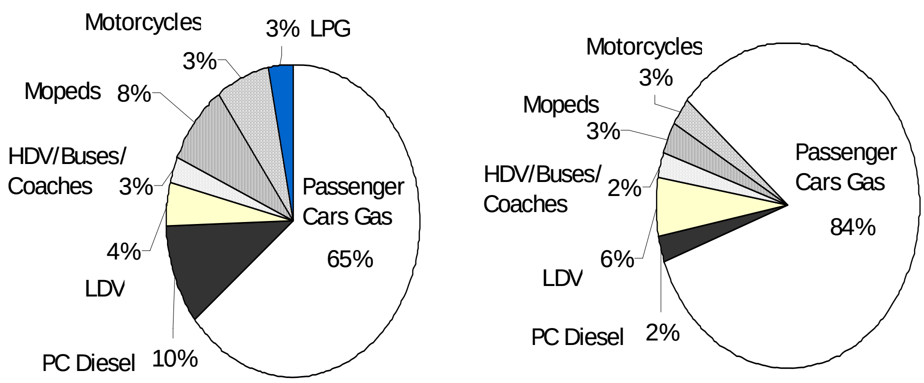
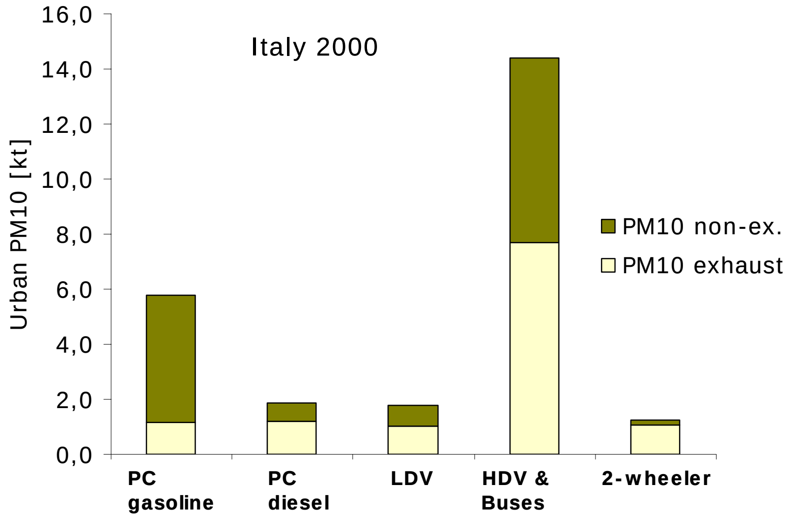


Figure 5: PM_{10} fleet emissions from exhaust and non-exhaust sources in urban Italy in 2000



At the case of urban Sweden it could be shown that non-exhaust PM_{10} emissions from suspension, brake and (studded) tyre wear dominated the PM_{10} emission level already in 2000 (Figure 7). In 2020, this domination will be even more distinct due to decreasing exhaust emissions and due to increasing non-exhaust emissions because of increasing stock and activity.

Figure 6: NO_x-emissions from passenger cars in Italy in 2000 and 2020

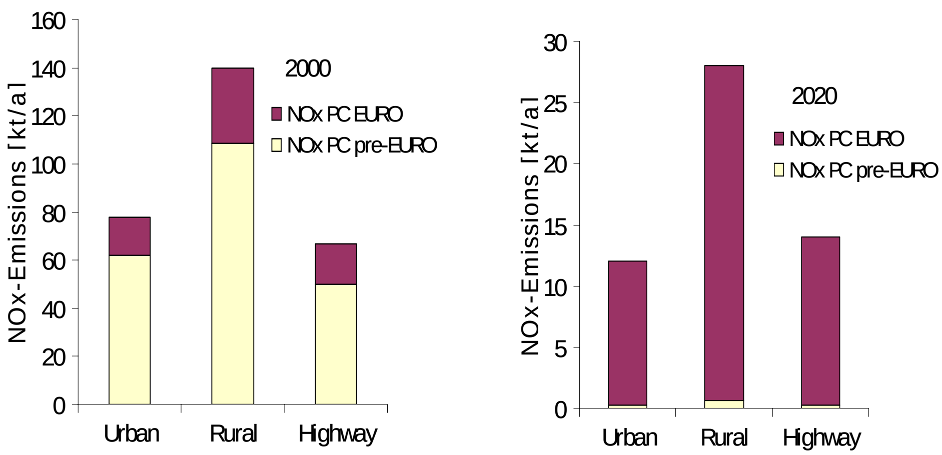
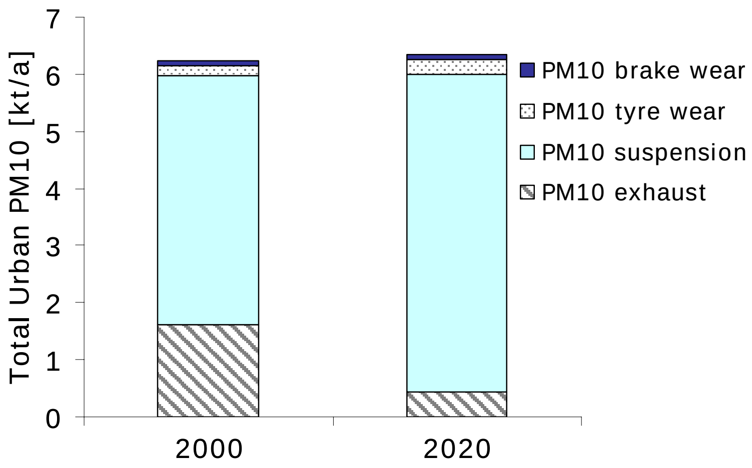


Figure 7: Total urban PM_{10} for Sweden in 2000 and 2020



Discussion

For road transport emissions in high spatial resolution, a bottom-up methodology based on traffic counts per road category decreases the uncertainty of emission modelling. At EU-15 or EU-25 scale, however, data in such a level of detail is not accessible at the moment. Instead, modelled stock and activity data and other information on the spatial distribution of traffic have to be used. Traffic census data on European scale would significantly decrease the uncertainty in modelling emissions from road transport.

Another source of uncertainty is the emission factor dataset, as shown by Kuehlwein (2004). HBEFA is an emission factor database with an Austrian, German and Swiss scope, whereas COPERT III offers a European-wide harmonised data set. HBEFA emission factors are, however, based on different driving cycles rather then on average speed. Because of different methodologies, the emission factor data sets lead to different results, as described in Niederle (1999). On the basis of Swedish diesel and gasoline passenger cars in 2000, Figure 8 shows the percentage deviation of NO_x emissions calculated with COPERT III from NO_x emissions calculated with HBEFA. NO_x emissions modelled with COPERT III emission factors are generally higher, especially in cities (with the exception of diesel passenger cars in rural areas).

Further uncertainties exist in the field of non-exhaust emissions. More measurements are needed to verify emission factors, size distribution, processes, quantification and calculation methodologies. Non-exhaust emissions are a main source for airborne PM_{10}. Their relevance for air quality will increase with decreasing exhaust emissions due to stringent EURO emission standards, especially in urban areas. Compared to exhaust emissions, however, only a small share of non-exhaust emissions have an aerodynamic diameter < 2.5 μm (Figure 9). Due to increasing health effects with smaller diameters, non-exhaust and exhaust emissions have thus to be differently evaluated.

Figure 8: Deviation of NO_x emissions calculated with COPERT III from NO_x emissions calculated with HBEFA (passenger cars, Sweden 2000)

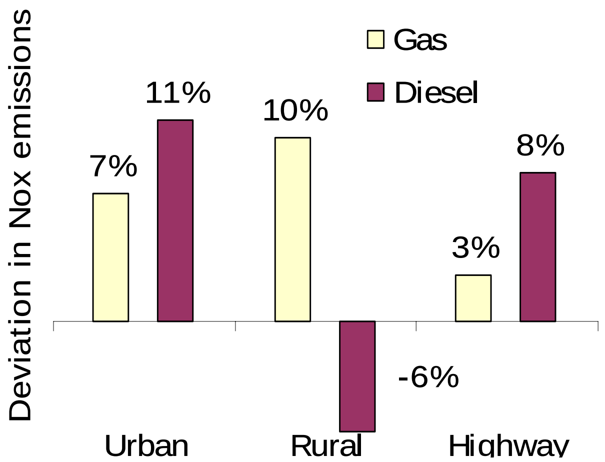
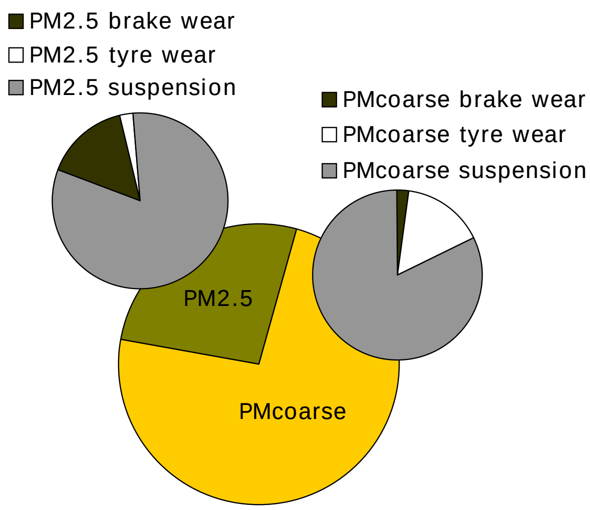


Figure 9: Composition of non-exhaust PM_{coarse} for urban passenger cars in Italy ($PM_{coarse} = PM_{10} – PM_{2.5}$)



Conclusion

The concept to link a tool for road transport emission modelling to a model for spatial resolution and to a model for the assessment of abatement measures is useful for comprehensive research in the field of air quality policy. To identify the most cost-effective abatement strategy for road transport emission levels on European scale, it is not only crucial to know which vehicle types, technologies and processes are the dominating sources, but also where these air pollutants are emitted.

A top-down approach for the calculation and spatial resolution of road transport emissions is suitable for a EU-15 and a EU-25 scope as consistent base data in sufficient level of detail is not available for a bottom-up approach. Indicators such as vehicle stock per administrative unit, trade and industry statistics or population densities can be taken for the disaggregation of road transport emissions. An equivalent level of detail as demonstrated in the case of Germany leads to detailed information on emission patterns and should be considered for future model development.

With a differentiation of vehicle categories, fuels and technologies, with the spatial split provided in COPERT III and with projections from TRENDS it is possible to assess emissions caused by road transport in detail for 2000 and for 2020. With this information, adequate national and regional abatement strategies can be designed.

Non-exhaust PM_{10} emissions from suspension, brake and (studded) tyre wear dominate the PM_{10} emission level already today. In 2020, this domination will be even more distinct due to decreasing exhaust emissions and increasing transport activities. However, high uncertainties exist on non-exhaust emission factors, processes and the effects of PM_{10} on air pollution and health, and further research is necessary.

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Influence of passenger car auxiliaries on pollutant emission factors within the Artemis model

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Abstract

The impact of the auxiliaries and particularly Air Conditioning on emissions (CO_2, CO, HC, NOx, and particles) is investigated. To this aim, various data from European laboratories are used and analysed. Parameters linked to technology and to climatic conditions are investigated. The main distinction is made between gasoline and diesel vehicles. A physical model is proposed to extrapolate the excess emissions at low temperature (below 28°C) and with solar radiation, together with a statistical model.

***Keys-words**: Air Conditioning, auxiliary, emission, atmospheric pollutants, passenger car, climatic condition.*

Résumé

Les effets des auxiliaires et plus particulièrement de la climatisation sur les émissions (CO_2, CO, HC, NOx et particules) sont étudiés. Pour cela, des données expérimentales de plusieurs laboratoires européens ont été rassemblées et analysées. Les paramètres liés à la technologie et aux conditions météorologiques sont évalués. La principale distinction est opérée par le type de carburant : essence ou diesel. Un modèle physique a été développé pour déterminer les émissions pour des faibles températures (inférieures à 28°C) et selon le rayonnement solaire, ainsi qu'un modèle statistique.

***Mots-clef :** climatisation, auxiliaire, émission, polluants atmosphériques, véhicule particulier, condition climatique.*

Introduction

The Artemis (Assessment and Reliability of Transport Emission Models and Inventory Systems) study is aiming at developing a harmonised emission model for road, rail, air and ship transport to provide consistent emission estimates at the national, international and regional level. A workpackage is aiming at improving the exhaust emission factors for the passenger cars and light duty vehicles, by enlarging the emission factor database, especially for effects of auxiliaries.

A European Climate Change Programme working group estimated that the usage of air conditioning (AC) systems under average European conditions causes an increase of fuel consumption between 4 and 8 % in 2020 (ECCP, 2003). A recent study valuated an increase of fuel consumption in 2025 below 1 % (Hugrel & Joumard, 2004). That is why it is proposed to undertake a state-of-the-art review of this area, to include fleet characteristics and a collection of data on auxiliaries (Roujol, 2005). Studies about air conditioning have been done in Europe focussed on the evaluation of individual passenger car emission due to AC (Barbusse et al., 1998; Gense, 2000; Pelkmans et al., 2003; Weilenmann et al., 2004), or on the improvement of AC (Benouali et al., 2003). A major study about AC impact has been carried out in the framework of Mobile 6 by the USEPA, focussed on the real use of AC in real conditions (Koupal, 2001) and on the effect of air conditioning running at full load on regulated pollutants (Koupal & Kremer, 2001).

Excess fuel consumption and CO_2 emission data analysis

Air conditioning database is made up of experimental data from 3 European laboratories (Utac and Cenerg in France, Vito in Belgium), i.e. 27 vehicles and 146 tests. Driving cycle, number of vehicle tests, type of vehicle, experimental objectives vary with experimentation. The choice of vehicles covers the main types of vehicle (small and large vehicles), different propulsion systems (gasoline and diesel) and the emission standards (mainly Euro 1, but also Euro 3 and 4). The climatic conditions are specific to each laboratory, but have been chosen in order to represent severe climatic conditions. The small size of the database allows us to perform a simple statistical analysis. According to Mobile 6, emitter classes, vehicle type, driving cycle, emission AC off and mean speed have to be distinguished to estimate effect of AC. At this short list, we can add, as proposed by Benouali et al. (2003), the regulation type and the compressor technology type.

The excess emission of pollutants due to air conditioning is the difference of emission with and without air conditioning running in the same condition. We have first to decide the type of unit to express the excess fuel consumption due to AC: in volume per distance unit or in volume per time unit. For physical reason (no strong relation between cooling demand and vehicle speed), it seems that volume per time (l/h for instance) is better.

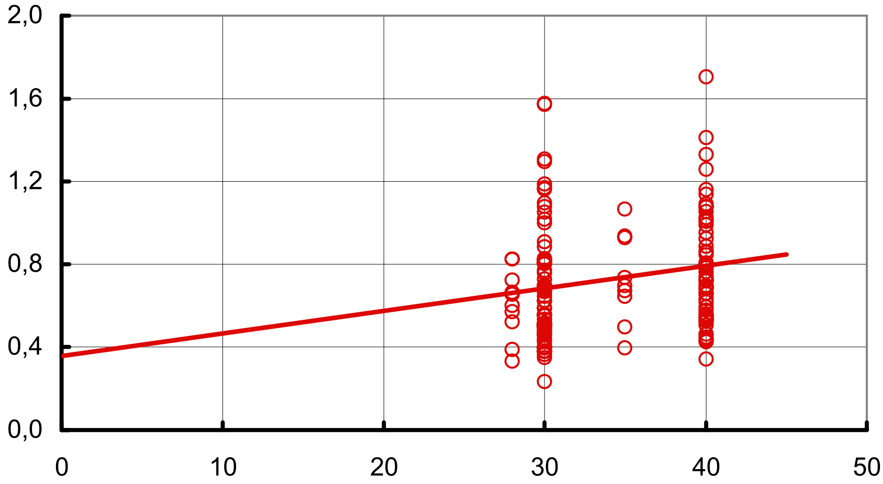
The mean speed has little impact on excess fuel consumption, but variance test indicates that the relation is statistically significant. The relationship is mainly influenced by the data at 90 and 120 km/h constant speed. It seems due to the engine efficiency, which varies with load and engine speed. The effect of AC on fuel consumption is partially hidden by the improvement of engine efficiency, but not at high speed or load. A similar conclusion is given in a recent experimental study on two vehicles in real driving conditions (Roumégoux et al., 2004). The effect of speed is explained by the fact that the engine load for EUDC cycle is particularly low. For real driving cycle, engine load is slightly higher, and fuel consumption due to AC should be quite independent of the speed or type of driving cycle.

Technological parameters analysed are parameters connected to the vehicle engine, to the AC system and to the body shape of the vehicle. The data are displayed according to the engine size, the fuel type, the vehicle size, the type of compressor and the type of regulation. In order to get enough data per class, only 4 types of vehicles are distinguished (see Table 1). The results show that the fuel consumptions are quite close with large standard deviations. Therefore we assume that the fuel consumption of AC does not depend on technical parameters.

Table 1: Average fuel consumption due to air conditioning (l/h) for the 4 vehicle types.

vehicle type				fuel consumption	
	fuel	AC regulation	nber veh.-tests	average	st. dev.
Small, Medium 1	Gasoline	manual	38	0.7	0.2
	Diesel		55	0.68	0.22
Medium 2, Large, SUV	Gasoline	automatic	25	0.75	0.34
	Diesel		28	0.85	0.35

Figure 1: Excess fuel consumption (l/h) due to AC versus outside temperature (°C), with linear regression.



The climatic conditions and set temperature have certainly a huge influence on AC running, and then on pollutants emissions. No experimentation is performed according to the solar radiation, although, according to Barbusse et al. (1998), solar load represents 45 % of the total load of the air conditioning. According to Figure 1, the variation of excess fuel consumption with the outside temperature is lower than expected: although the uncertainty of the measurements, the outside temperature at which there is no cooling or heating, obtained by linear extrapolation, seems to be below 0°C. Theoretically, the relation between fuel consumption and outside temperature is quite linear because of convective heat gains linearly linked with the difference between outside and inside temperatures. That seems to demonstrate that AC is running quite close to full load for outside temperature higher than 28°C.

An extrapolation of these data is therefore non applicable. As the experiments do not allow us to take into account temperature below 28°C and solar heat radiation, a physical model is therefore developed.

Air conditioning physical modelling

The physical phenomena taken into account are the heat exchanges of the cabin with outdoor, the heat exchange on evaporator of air conditioner, the air conditioner and the engine running.

The passenger compartment modelling is based on a description of heat exchange as it is usually done in mono-zone thermal building modelling (Bolher et al., 2000). Air temperature and humidity in the cabin is assumed to be uniform. Heat exchanges governing temperature of cabin are due to the global heat exchange coefficient, UA ($W.m^{-2}.K^{-1}$), the untreated air flow rate due to permeability, m_p ($kg.s^{-1}$), the internal heat gains due to occupants and electrical equipments, A_{int} (W), the solar gains, A_{sol} (W), and the treated air flow, m_t ($kg.s^{-1}$).

The modelling of solar gains (Fraisse & Virgone, 2001) depends on the direct and diffuse solar radiation, the position of the sun in sky and the geometric and physical properties of the vehicle window. Temperature and flow rate of treated air flow are regulated in order to maintain cabin air temperature to set temperature.

The thermal mass of the vehicle's interior has an effect in dynamic behaviour, increasing cooling demands during cool down for instance, but has no effect during steady state cooling and is therefore neglected. Weilenmann et al. (2004) have studied initial cool down, by combining the effect of initial cool down of the overheated passenger compartment and the effect of cold start. Two counteracting effects occur: Because of thermal mass, AC running involves more power than at steady state, and AC running involves that engine compartment is heated much faster than without AC running. These two effects compensate each other, and excess emission due to initial cool down in comparison to steady state emission is in the same order of magnitude than the cold start excess emission in the same temperature conditions.

With the internal temperature T_{int}, the temperature of treated air T_t, and the outside temperature T_{ext}, the conservative equation of energy is:

$$(m_t + m_p)\cdot T_{\text{int}} - (m_t \cdot T_t + m_p \cdot T_{ext}) = A_{\text{int}} + UA \cdot (T_{ext} - T_{\text{int}}) + A_{sol}$$

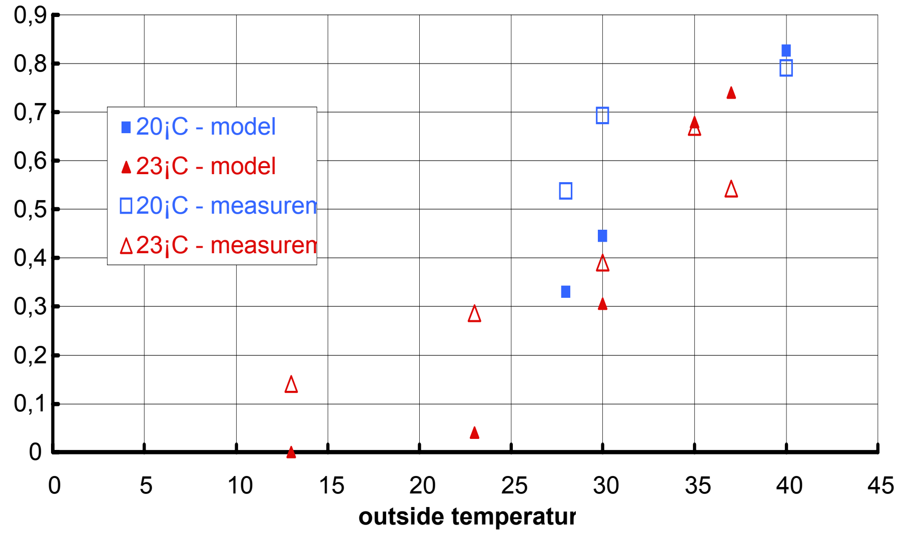
The internal temperature is chosen according to the thermal comfort theory (Fanger, 1972). The conditions of thermal comfort are a combination of skin temperature and body's core temperature providing a sensation of thermal neutrality and the fulfilment of body's energy balance. From ASHRAE standard 55 (1992) and Charles (2003), 23°C is chosen as default value. The sensible heat exchange P_{sens} at evaporator to maintain internal temperature at the comfort temperature can be deduced, and, if air treated rate m_t is known, air treated temperature T_t can be calculated:

$$P_{sens} = m_t \cdot (T_{ext} - T_t) = (m_t + m_p + UA)\cdot (T_{ext} - T_{\text{int}}) + A_{sol} + A_{\text{int}}$$

Heat exchange at the evaporator can cause dehumidification of air treated. The average surface temperature humidity of air treated across AC evaporator depends on the heat transfer coefficients of evaporator and the temperature of coolant. With the air side heat exchange efficiency, it allows us to calculate the average surface temperature and humidity of outlet air. With assumption on a minimum air flow rate of 300 m^3/h and a minimum average surface temperature of 0°C, the air treated temperature can be calculated. We assumed that the efficiencies of AC and engine are constant. For energy efficiency of the engine, experimental data show that running conditions of the engine have a small effect on CO_2 emissions due to air conditioning. According to Park et al. (1999), the main parameters on AC efficiency are the temperature conditions, but the effects of temperature on energy efficiency are lower than on cooling demands.

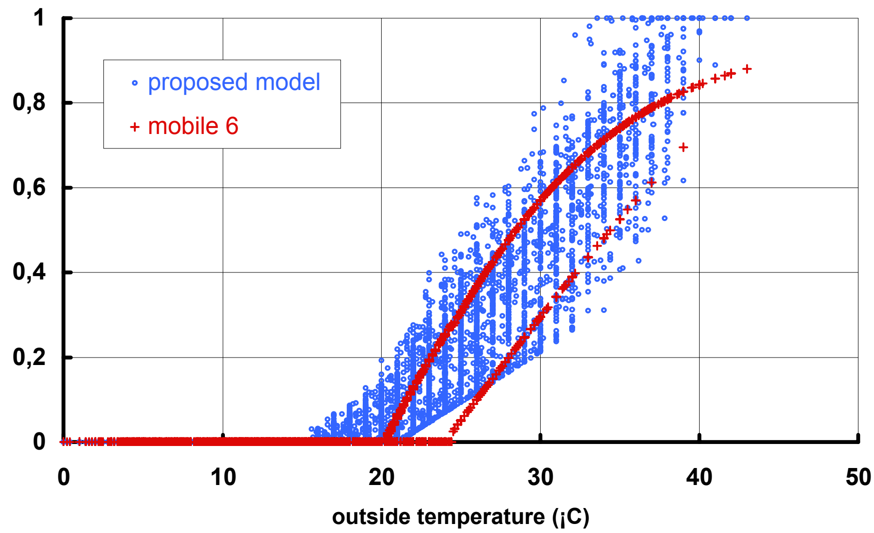
The model is applied to all experimental conditions either presented in section 1, or by Weilenmann et al. (2004), with temperature range resp. of 28-40°C and 13-37°C. The results of the model are compared to the experimental results (see Figure 2). They are quite close for temperature higher than 30°C. From 20°C to 30°C, the model underestimates the fuel consumption; And below 20°C, hourly fuel consumption from model are null, but experimental excess fuel consumption can be linked to the electrical consumption of ventilation to avoid windscreen fogging.

Figure 2: Comparison of the results from model and from experiments as a function of outside temperature for two internal temperatures (20 and 23°C).



A second comparison is done with the Mobile 6 model of demand factor based on experimental measurements. Demand factor is defined by Mobile 6 as the fraction of running time of AC, but can be also defined as the ratio of part load power consumption to the full load power consumption, estimated at 0.85 l/h. The Mobile 6 model and the proposed model are applied with hourly weather data of Seville in Spain, which has the closest climate in Europe to the climate of Denver where vehicle were followed in order to determine demand factor in Mobile 6. In order to take into account the solar loads, Mobile 6 distinguishes daytime and night, and our model calculates the solar loads for each climatic condition described in the weather data. As shown Figure 3, demand factors obtained by Mobile 6 and our model are quite close for temperature higher than 20°C. Below 20°C, demand factor from Mobile 6 model is null but slightly above 0 for our model because of solar loads heating.

Figure 3: Comparison of the Mobile 6 model (upper curve for daytime and lower curve for night) with the proposed model (set temperature at 23°C).



We consider that the model satisfied our objective, which is to determine hourly fuel consumption in non-tested weather conditions. The differences between results from model and data from EMPA (Figure 2) at temperature below 20°C are not well understood and required additional experiments at these particular conditions.

1. Simplified model and weather data

A physical model of excess fuel consumption due to AC seems to be too complex to be implemented in an inventory software as Artemis. Therefore we computed the physical model with weather data of 91 regions all over Europe, and looked for a relationship by statistical regressions between hourly fuel consumption and the following explicative variables: ambient temperature, humidity, position of sun in the sky, and solar radiation, replaced by the hour in the day. The general form of the simplified model is:

$$hfc = a_{1,wf} + a_{2,wf} \cdot T_{ext,wf} + a_{3,wf} \cdot T_{\text{int}} + a_{4,wf} \cdot h + a_{5,wf} \cdot h^2 \qquad \text{with } hfc \geq 0$$

with:

hfc: hourly excess fuel consumption (l/h)

$T_{ext,wf}$: external temperature provided by hourly, daily or monthly weather data (°C), which contain resp. 8760, 365 and 12 values

T_{int}: set temperature in the cabin; default value is 23°C

h: the hour (between 1 and 24)

$a_{1,...5}$: coefficients depending on the location

The coefficients a_1 to a_5 are available for each location. But in addition, two other sets of coefficient a are provided: The first set is given according to 6 modified Köppen climate classification, based on the annual and monthly averages of temperature and precipitation (DOE, 2004), and the second set corresponds to an average.

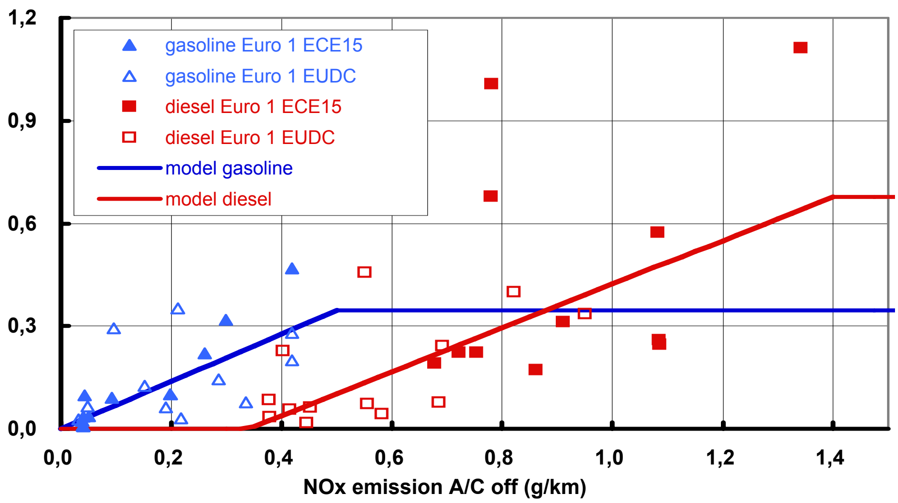
The excess fuel consumption and CO_2 emission for a fleet is calculated by summing hfc according to the number of vehicles with AC running for a given road segment, expressed in number of vehicles per hour.

Excess pollutants emissions analysis

Data available for pollutant emissions (CO, HC, NOx, PM) due to AC are rare in comparison with data available for CO_2 emission, mainly because only 13 gasoline and diesel vehicles are tested.

As it was shown in section 1, AC system is running quite close to the full load at the test conditions (outside temperature > 28°C), where pollutants emissions are assumed to be full load ones. An example of data is shown in Figure 4: NOx emission and effect of AC are larger during the urban driving cycle ECE15 than during the extra-urban cycle EUDC. For each pollutant a relationship is proposed between excess emission and hot emission without AC (Figure 4). Results of gasoline vehicles are in accordance with the theoretical explanation proposed by Soltic & Weilenmann (2002): as long as the increased torque does not cause a air fuel mixture enrichment, an increase in the exhaust temperature, a slight reductions of HC and CO emissions, and an increase of NOx emission are expected. If an increased torque level causes an increase of enrichment, CO and HC emissions will also increase.

Figure 4: NOx excess emission versus NOx emission AC off according to the fuel and driving cycle for Euro 1 vehicles, for urban ECE15 and extra-urban EUDC driving cycles, with the corresponding modelling.



For the pollutants emissions modelling, we assume that pollutants emissions at part load are a fraction of pollutant emissions at full load, with a fraction equal to the demand factor.

Because of the lack of data, only a distinction between the gasoline and diesel vehicles is proposed. The model do not explicitly distinguish the age of vehicle, because we consider it has no influence on excess CO_2 emission. The effect of emission standard on pollutant emission is taken into account through the hot emission, which depends on standard emission.

For the future vehicles, some counteracting effects occur: Firstly, technological improvements of efficiency of AC system are expected:

- \- By reducing the thermal load of the vehicle (Türler et al., 2003; Farrington et al., 1998, 1999) through the use of advanced glazing which reduces the transmission of infrared solar radiation. The improvement of air cleaning allows reducing the amount of outside air, reducing by the way thermal load and power consumption of fan. Advanced regulation of ventilation allows ventilating parked vehicles reducing the peak cooling load.
- \- By increasing energy efficiency ratio of AC system (Benouali et al., 2002; Barbusse & Gagnepain, 2003). The first improvement will be due to the improvement of AC components as the external control of compressor, the electrical compressor, a high efficiency heat exchanger. At long term, alternative technologies are investigated as magnetic cooling, desiccant cooling, and absorption.

Secondly, the evolution in the vehicle design and in the leakage refrigerant standard will certainly increase the CO_2 emission due to the use of AC. The constraint against refrigerant leakage drives to use alternative refrigerant with a lower Global Warming Potential as HFC 152a and CO_2. These alternative refrigerants have the drawback to reduce the efficiency of AC system because their lower thermodynamic properties. The use of alternative refrigerant as the CO_2 allows using AC system as a heat pump in order to warm passenger compartment, made more and more difficult by the development of high efficiency engine which could reduce the possibility to use the engine heat to warm the passenger compartment and which justify the development of reversible system.

At short time, we assume that these two effects compensate each other. No correction is proposed for future vehicles.

Other auxiliaries

The excess fuel consumption due to other auxiliaries can be express in l/h as for AC. According to Soltic & Weilenmann (2002), we evaluated an average excess fuel consumption of 0.075 l/h for an electrical load of 160 W corresponding to dip headlight. We assume that excess fuel consumption is proportional to electrical load of each auxiliary. In order to be in accordance with excess pollutant emission due to AC, we proposed to use a similar way for excess emission due to auxiliaries. Excess pollutant emission due to AC at a given conditions is a fraction to excess pollutant emission at full load. This fraction is calculated as a ratio of excess fuel consumption at given condition to excess fuel consumption at full load. We proposed to use the same model by replacing the excess fuel consumption of AC by the excess fuel consumption of auxiliaries. For instance, in the case when headlights are use, the value of fraction is 0.075/0.85.

Conclusion

The different analyses show that the excess fuel consumption expressed in l/h is quite independent to the speed or to the traffic situation. No significant technological parameters are found. That does not mean that no relation exists between excess fuel consumption and technological parameters, but that the number of data is not sufficient to extract this type of relation or that the technological solutions are too close each other.

The excess fuel consumption due to air conditioning is well know in warm conditions because of the large number of experiments. It is quite different in usual climatic conditions with solar radiation, because of the reduced number of experiments. To approach the behaviour of AC system at these conditions, a physical model is proposed and compared to experimental data. According to the objective of the model, the results show a good agreement in warm conditions. At usual conditions, the model underestimates the excess fuel consumption without understanding the reason. Effect of AC in usual conditions is an important way of investigation because of the occurrence of these conditions in comparison to warm conditions. In the model, based on the usual comfort theory, we assume that the set temperature is 23°C for all the vehicles equipped with AC, but experiments on real world vehicles with air conditioning could improve the knowledge of user's behaviour.

The model proposed is a part of the new European emission inventorying model Artemis.

Acknowledgments

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Development and evaluation of hybrid vehicle emission and fuel consumption factors based on measurements over real world cycles

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Abstract

The reduction of transport generated CO_2 emissions is currently a problem of global interest. Hybrid electric vehicles (HEVs) are considered as one of the most promising technological solutions for the abatement of transport generated gaseous emissions. Currently the number of HEVs in the market remains limited, but this picture is expected to change in the years to come. In this context, the development of HEV emission factors will provide the necessary information for predicting future emissions and evaluating various scenarios and policies. This paper presents the measurements conducted on a Prius II HEV using both legislated and real world (ARTEMIS) driving cycles. Two sets of measurements were conducted using the ARTEMIS measuring protocol. The results of the measurements revealed the improved energy efficiency of the vehicle compared to conventional ones and verified that HEVs are indeed a fuel efficient and relatively "clean" technology that bares great potential. These results were used for the development of HEV emission factors as presented.

Key-words*: Hybrid electric vehicles, emission factors, fuel consumption, CO_2 emission*

Introduction

Main goal of this paper is to produce representative emission factors for HEVs, based on the evaluation of the operation and performance of one HEV as regards gaseous emissions and fuel consumption, with an emphasis on CO_2 emissions, both under type approval and real world driving conditions. Historically, HEVs first appeared at the Paris Salon exhibition of 1899 (Wakefield 1994). Since then numerous efforts were made by various manufacturers to produce a mass production HEV. Not until 1997 did this situation change, when the Toyota Prius I and the Honda Insight models first appeared in the Japanese market (Ehsani et al. 2005). Today HEVS are an established solution in the U.S. market, which is closed to the diesel engine, for drivers that want to purchase vehicles of low fuel consumption. In the European market which is more diesel oriented only 3 hybrid models are currently available but this is expected to change in the future.

The following evaluation is based on experimental measurements conducted on a Toyota Prius II HEV. It should be noted that the specific model is the only HEV introduced so far in the European market in a broad scale and represents the majority of the HEV sales in Europe. This means that the performance of this vehicle especially with respect to fuel consumption should not be taken as a future average for all HEVs, since the solutions adopted by each manufacturer may vary substantially. On the other hand, these measurements are of special scientific interest as HEVs are expected to increase significantly in number in the European fleet, as projected by TREMOVE model (De Ceuster 2005). In this case it will be important to have the necessary tools for evaluating their contribution in the pollutant emissions and run realistic scenarios concerning the future trends in European road transportation. In the following first information is provided regarding the measurement protocols that were adopted. Then the results of the measurements are presented and commented. Finally the conclusions drawn from these measurements are summarised and follow-up activities are proposed.

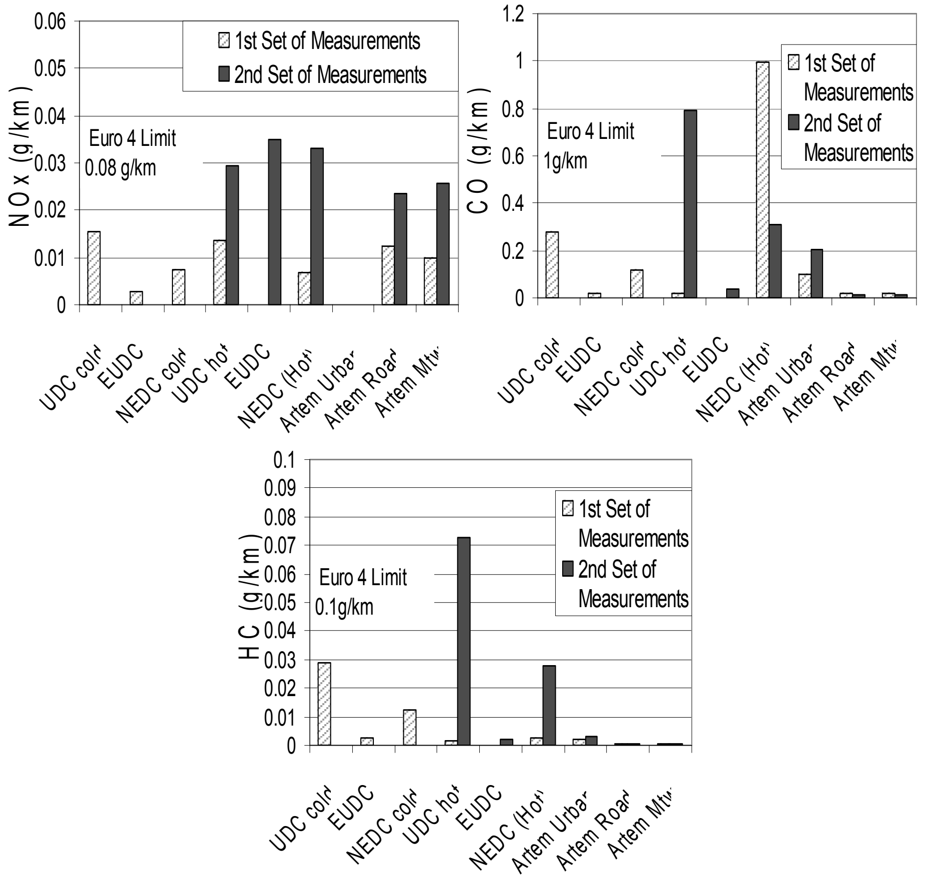
Methodology

Main objective of the measurements that were conducted was the quantification of the emissions of gaseous pollutants and fuel consumption, not only under type approval driving conditions (legislated cycles) but also under real life driving conditions. Therefore the Artemis protocol was used, which includes one cold New European Driving Cycle - NEDC (the combined legislated driving cycle), one hot Urban Driving Cycle -UDC (urban sub-cycle of NEDC) and the sequence of Artemis driving cycles. The Artemis cycles, developed to simulate real conditions of vehicle operation, are distinguished into 3 parts: Artemis urban cycle (URBAN) for urban driving conditions, a semi-urban cycle (ROAD) simulating the operation of the vehicle in a regular medium speed road, while the extra urban cycle (MOTORWAY) simulates the operation in high speed freeway.

For Prius II, Toyota specifies that different reference mass should be used for fuel consumption (1360kg) and gaseous pollutants measurement (1470kg) during the type approval test (Toyota 2005). Since the emphasis of our measurements was on fuel economy, it was decided to use in all cases the 1360kg inertia. During the measurements all regulated gaseous emissions were measured using the legislated method. Two sets of measurements were conducted on two different vehicles, one supplied by TOYOTA and one by a private owner. The first vehicle was relatively new (8000km mileage), while the second was somewhat older (34000km mileage).

Figure 1: Legislative gaseous pollutant emissions for all test cycles (2 repetitions)

Figure 1: Emissions de polluants législatifs de toutes les cycles du protocole



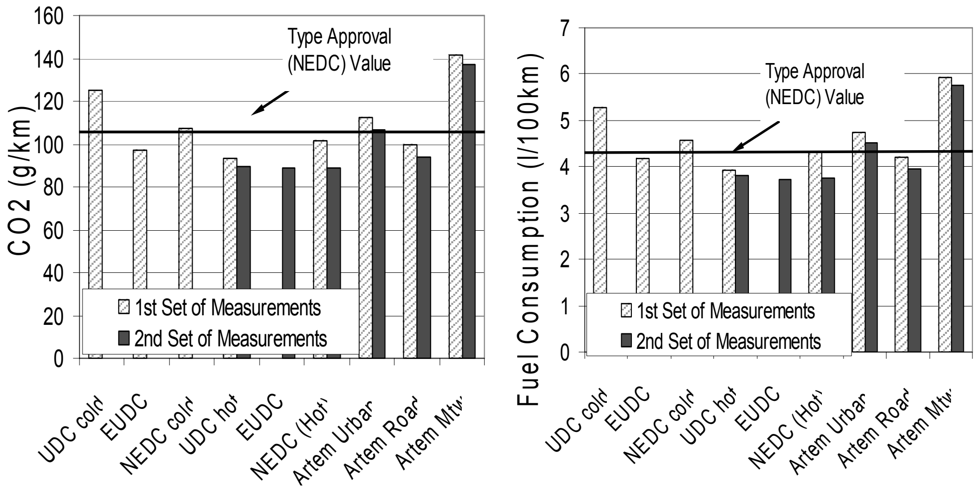
Results

Figures 1 and 2 summarize the average measurement results, for each driving cycle of the 2 measurement sets. The values in the box indicate the EURO IV emission limit. It should be reminded that, as in the legislated procedure, the NEDC measured in the Artemis is a cold-start NEDC driving cycle and that is why it is being referenced as “NEDC cold”. In the second set of measurements the cold cycle is omitted due to technical problems that occurred during the measurement. As shown in Figure 1, the gaseous emissions of the vehicle are lower than the existing emission standard for all driving cycles. The measurements showed that the majority of the vehicle's emissions occur from "emissions events" that take place within the cycles non-systematically (except in cold UDC). When comparing the results of gaseous pollutants (Figure 1) with those of the type approval it appears that the measured values are different than those provided by the manufacturer.

This can be attributed to an extent to the aforementioned difference in the equivalent inertia used for the measurements and for the type approval. In spite of these deviations from the manufacturer values, the scatter of the two repetitions is relatively small, a fact that underpins the reliability of the measurements. More specifically, during the cold start urban driving cycle (UDC cold) which is the only driving cycle presenting a relatively high emission level, the gaseous emissions measured are close to the type approval values and the scatter is within the range accepted by legislation. The overall picture of the measured NEDC is satisfactory.

Figure 2: CO_2 Emissions and Fuel Consumption (FC) for all test cycles

Figure 2 : CO_2 Emissions et consommation de carburant (FC) de toutes les cycles du protocole



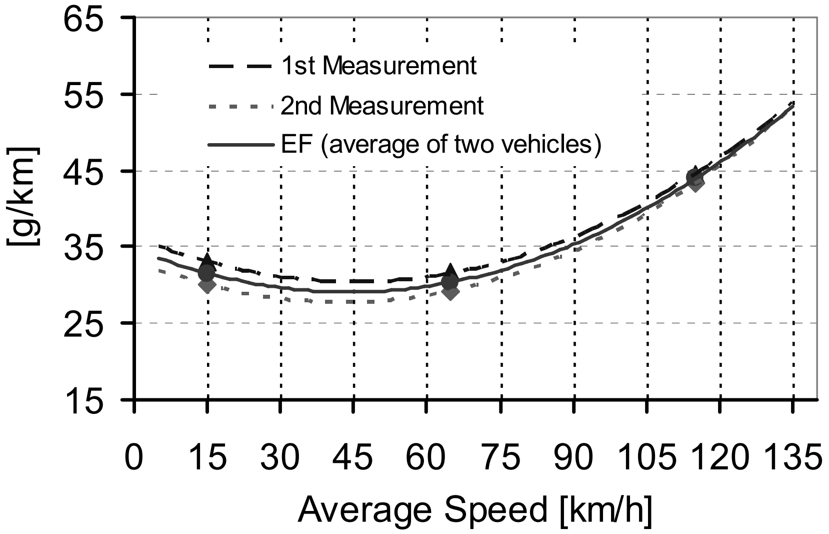
CO_2 emissions and fuel consumption are close to the officially reported values and less than the 5% deviation accepted by legislation. Contrary to gaseous emissions, in the case of fuel consumption the reference mass used matched the official one. At this point it is important to say that there was a differentiation of 3-10% of the fuel consumption between the two sets of measurements as presented in Figure 3. This differentiation is considered to be quite high even for 2 different vehicles of the same model. As it will be presented onwards the vehicle fuel consumption is highly affected by the operating temperature, the rise of which increases the efficiency of the vehicle. In the particular measurements the ambient temperature was 20°C for the 1st measurement and 24°C for the second, a factor that may possibly explain the reduced fuel consumption in the second case. Similar results are presented by El Khoury and Clodic (2004) for measurements at 28°C. In order to understand the cause of this effect in fuel economy, it is necessary to examine the electrical system of the vehicle.

The accurate measurement of HEV's performance requires monitoring of the vehicle battery's state of charge (SOC). A differentiation of SOC during the tests can cause important changes in the operating scheme of the vehicle. Figure 4 shows how the operation of the vehicle over a UDC can be affected by the battery SOC. The Figure presents the instantaneous CO_2 emissions for 3 different cases: battery charging, battery discharging and battery close to average SOC over cycle.

It is important to note that after some time the operation of the engine becomes almost the same in all cases signalizing that the vehicle reaches a balance point. Therefore, gaseous emissions are considered those that occur over a driving cycle that presents zero differentiation between the initial and the final SOC – i.e. zero ΔSOC. In order to ensure the validity of the results additional measurements of fuel consumption were conducted over the same driving cycles, which showed that the effect of ΔSOC on the measurements was minimal; especially for the Artemis cycles where it was almost zero. Hence, it is concluded that the battery ΔSOC was not responsible for the observed deviations in fuel consumption.

Figure 3: Fuel consumption measurements and proposed consumption factor function

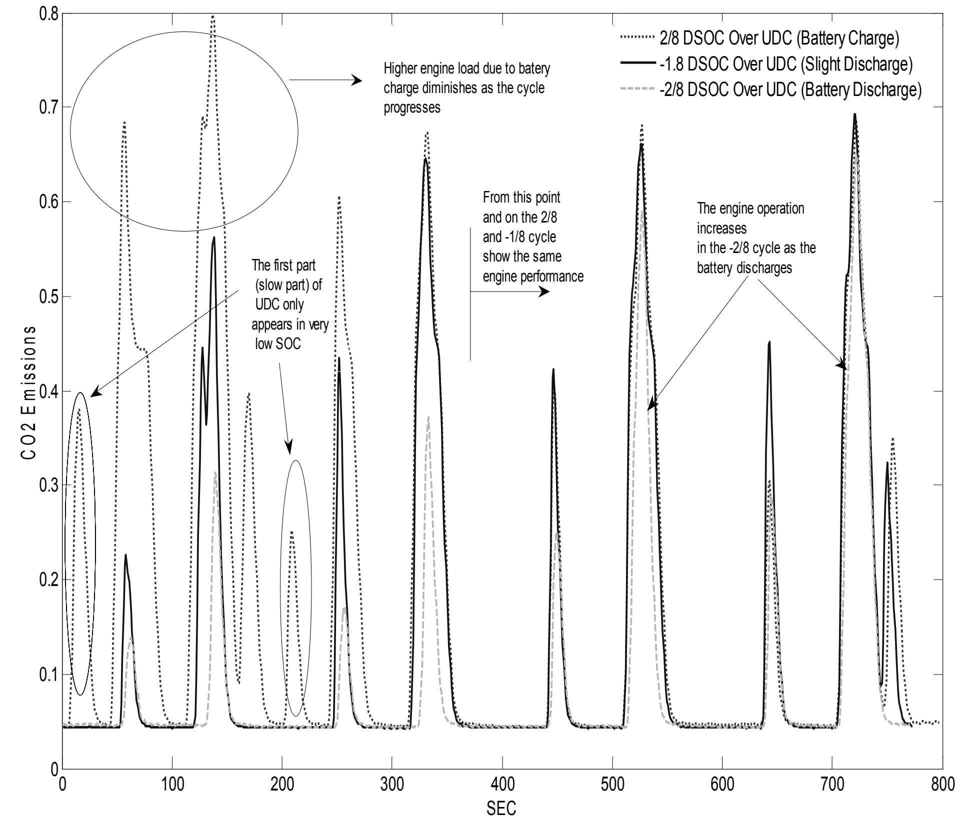
Figure 3: Measures de consommation et facteur de consommation



Another factor that can cause differentiation in fuel consumption is ambient temperature. The temperature at which the measurements were conducted was 27ºC, higher than the 20ºC at which type approval tests are usually conducted (although within the legislated range of 20 to 30ºC). These 7 degrees temperature difference would have a negligible effect on the fuel consumption of any conventional vehicle, but for HEVs it must be taken into account. The vehicle performance is significantly affected not only by the battery charge / discharge, but also from battery’s overall capacity. Generally, a higher battery capacity is experienced at higher temperatures, resulting in reduced fuel consumption, as in the case of these measurements. This is due to the powertrain management of the vehicle. As Figure 5 shows, at steady state conditions the powertrain operation presents periodical differentiations. It is clear from the graph that the vehicle is powered for a short time only by its electrical system discharging the battery. When a critical SOC level is reached (for Prius estimated at about 70% of the battery's capacity) the thermal engine is started, powering the vehicle and charging the battery. This periodicity occurs in all cases. What is differentiated between various vehicle speeds is the duration of this cycle and eventually the ratio between electrical and thermal operation. It must be mentioned that for velocities higher than 65km/h the thermal engine never stopped, but its operation was shifting between two different operating points. As a result of this control strategy, the vehicle economy is highly affected by the battery's capacity. According to Yaegashi (2005), ambient temperature affects the battery of the Prius with colder weather reducing its capacity and therefore affecting the fuel economy of the vehicle.

Figure 4: Vehicle operation over the UDC for different states of charge (SOC)

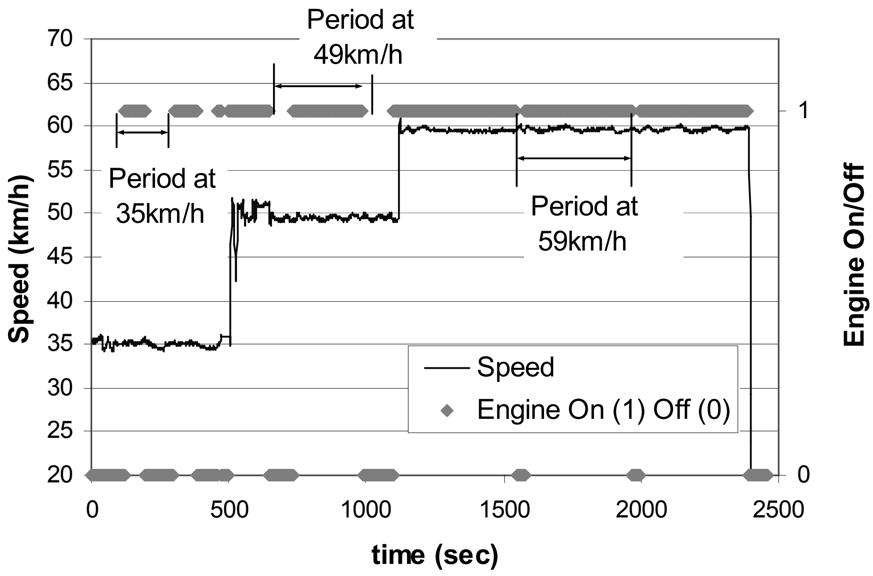
Figure 4: Opération du Prius II sur UDC pour trois différents conditions du charge



El Khoury and Clodic (2004) presented similar results, measuring a fuel consumption of 3.6 l/100km. The results of El Khoury and Clodic refer to a hot start NEDC conducted at 28^{o}C (prior to the measurement cycle a conditioning cycle was run and then the vehicle was left to soak for 1h at 28^{o}C). The difference in fuel consumption with the measurements of this study (3.76l/100km for hot NEDC at 24^{o}C) may be attributed to the slight difference in vehicle coast down curve applied by El Khoury and Clodic and the starting temperature conditions.

Figure 5: Prius II Hybrid system operation at steady state conditions

Figure 5: Opération du system hybride Prius II en conditions continues



Emission factors development

The information from the measurements was used for the development of average speed dependent emission factors (EF) for HEVs, proposed for inclusion in the ARTEMIS database (Samaras and Geivanidis 2005). The usual approach of a second degree polynomial function was followed, the parameters of which are presented in Table 1.

Table 1: Equations for hybrid vehicle emission factor functions

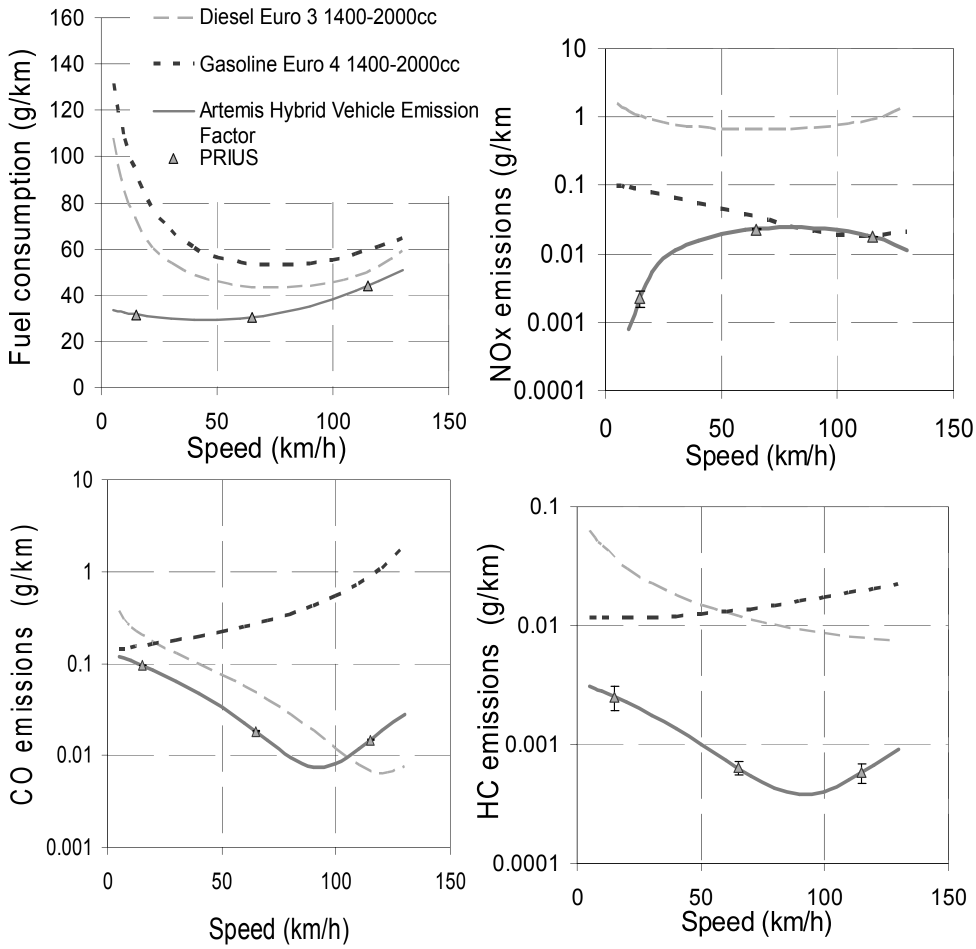
Tableau 1 : Equations pour les facteurs d'émission des véhicules hybrides

Pollutant	Equation	a	c	e
CO (g/km)	y=a+cx+ex^2	0.1322586	-0.0027016	1.46E-05
HC (g/km)	y=a+cx+ex^2	0.0034430	-6.69332E-05	3.65E-07
NO_x (g/km)	y=a+cx+ex^2	-0.00895196	0.0008237	-5.1E-06
Fuel Consumption (g/km)	y=a+cx+ex^2	34.749	-0.2572765	0.002928

In comparison to the conventional Euro 3 & 4 vehicles, the emission and fuel consumption factors of HEVs are lower. Therefore hybrid technology can be considered cleaner in comparison to the conventional one. A comparison of the difference between HEV and conventional diesel/gasoline emissions and fuel consumption is presented in Figure 6. Note that the conventional vehicle functions refer to Euro 3 compliant diesels and Euro 4 compliant gasoline vehicles, all of the1400c-2000cc capacity class. Note also that the y-axis of Figure 6 is logarithmic.

Figure 6: Comparison of hybrid vehicle emissions with conventional Diesel (Euro 3) and Gasoline (Euro 4) vehicles (Source: ARTEMIS database)

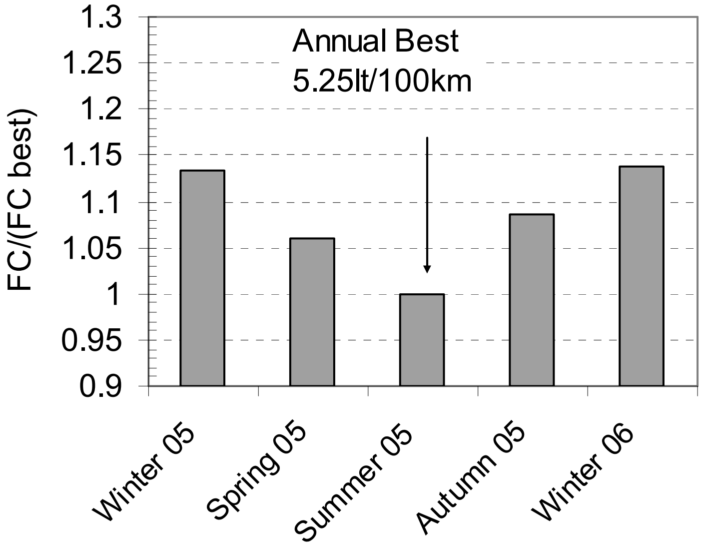
Figure 6: Comparaison des émissions des véhicules hybrides avec ces des véhicules Diesels (Euro 3) et Essence (Euro 4) (Source: ARTEMIS database)



When examining the above curves one could argue that the low emissions of HEVs should not be considered as representative for all HEVs, since they are derived from measurements of a single model. This is only partially true, because the specific hybrid model represents today the majority of the global HEV sales. It is expected that hybrid models that will appear in the near future will follow a similar trend in being not only very fuel efficient, but also very clean. In fact, advertising campaigns of vehicles such as Lexus RX400h and Ford Escape, stress not only the low fuel consumption but also their especially low gaseous emissions.

Figure 7: Annual fluctuation of a Prius II fuel consumption

Figure 7: Fluctuation annuelle de la consumation d'un Prius II



With respect to fuel consumption factors an important issue that should be considered as described in section 3 is the effect of ambient temperature fluctuation. In parallel to the measurements described above, the authors closely monitor the actual on-road operation of another Prius II vehicle, recording its fuel consumption and other data since November 2004. In Figure 7 the average seasonal variation of fuel consumption of the vehicle is presented. It must be noted that the operation of the specific vehicle does not vary significantly throughout the year; therefore the differentiations observed should be mainly associated with ambient temperature fluctuation. As shown, the optimal fuel consumption occurs during summer when temperatures are higher. In winter on the other hand there is a 13% increase, while spring and autumn lay in between. These numbers are fully in line with measurements presented by Toyota (Yaegashi 2005) concerning older Prius models. It can therefore be concluded that a temperature correction is necessary when calculating CO_2 emissions and fuel consumption of HEVs. The fuel consumption factors presented in this paper correspond to an ambient temperature of 22^{o}C, which is close to the annual average temperature in Greece and the type approval temperature. It is expected that the fuel consumption values would be 6-7% higher for temperatures around 10^{o}C, and approximately 5% lower for temperatures around 30^{o}C. These differences must be accounted for in cases where more detailed analysis is necessary.

Conclusions

Based on the findings of this study the following conclusions can be drawn:

- The measurement results are in line with the specifications of the manufacturer and with similar measurements conducted by other researchers.
- The gaseous emissions of the vehicle were found lower than the Euro 4 emission standards. It is important to note that the levels of emissions are maintained at the same low levels over cycles that simulate real driving conditions (Artemis).
- Fuel consumption is quite low compared to that of other vehicles of the same inertia class and CO_2 emissions remain below 140 g/km in most cases. The contribution of the electric system in the vehicle fuel economy is very important, especially at low speeds (<40km/h) and decreases with increasing vehicle speed.
- Fuel consumption over the Artemis driving cycles remained within the range of 3-6 l/100km. Under urban driving conditions, fuel consumption was found to be more than 50% lower than the average equivalent conventional gasoline vehicle. This benefit becomes even higher for low average speed driving (stop-and-go).
- Fuel consumption is affected by ambient temperature, due to the effect the latter on battery capacity. Higher temperatures tend to increase the capacity and thus improve the penetration potential of the electrical system which leads to better fuel economy.
- Emission factors were successfully produced from the results of the measurements. When used for calculating CO_2 emissions and fuel consumption in the future it is advised to apply the correction factors presented for compensating for ambient temperature effects.
- The emission factors developed are based on a single vehicle model which currently accounts for the majority of HEV sales. Nevertheless the emission factors should be revised in the future when more models enter the market.

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Modelling particle formation: An helpful tool to interpret measurement results

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Abstract

Vehicle manufacturers and oil companies need precise information concerning particle formation and the modeling of the concerned processes. Certain assumptions were made concerning their formation process, especially nucleation (conversion gas-liquid) of the sulfuric acid - water mixture which would play an important role. However, the participation of some hydrocarbons in the formation process cannot be excluded.

A numerical model has been developed, taking into account the various phenomena induced in particles formation: nucleation, growth by condensation, coagulation etc. as well as the variations in temperature and concentrations along the exhaust line and the sample line (dilution, catalyst, filter particles). This model was successfully validated by comparison with experimental results obtained on a chassis dynamometer. The sensitivity studies allow to evaluate the influence of the parameters "fuel" (sulphur content), "motor" (speed, fuel consumption,…) and "measurement conditions" (temperature, dilution ratio, humidity, residence time,…). These sensitivity studies increase our knowledge on the processes involved and provide an useful tool to better interpret the results of the particle size measurements.

Keys-words: *Particle formation, size distribution, particle modelling, measurements, exhaust gas vehicle*

Résumé

Les constructeurs d'automobiles et les pétroliers sont à la recherche d'informations précises concernant la formation des particules et la modélisation des processus mis en jeu. Certaines hypothèses ont été avancées concernant leur mode de formation, notamment la nucléation (conversion gaz – liquide) du mélange *eau – acide sulfurique qui aurait un rôle important, sans que l'on puisse exclure la participation de certains hydrocarbures.*

Nous avons développé un modèle numérique prenant en compte les différents phénomènes intervenant dans la formation des particules: nucléation, croissance, coagulation etc. ainsi que les variations de températures et de concentrations le long de la ligne d'échappement et de la ligne de prélèvement. Le modèle a été validé, avec succès, par comparaison avec les résultats d'une campagne de mesures. Des études de sensibilité ont été menées et ont permis d'évaluer l'influence des paramètres "carburant" (teneur en soufre), "moteur" (vitesse, consommation de carburant, …), et "conditions de mesures" (température, taux de dilution, humidité, temps de résidence,…). Ces études de sensibilité ont permis d'accroître notre compréhension des phénomènes mis en jeux pour mieux interpréter les résultats des mesures de particules.

***Mots clé**: Formation des particules, granulométrie, modélisation, mesures de particules, gaz d'échappement des véhicules.*

Introduction

Air pollution is an important problem, which affects industrialised OECD countries as well as emerging countries. The pollutants emitted by vehicles are directly concerned because they contribute to the deterioration of the air quality in large cities. Among these pollutants, particles have been shown to be harmful to human health, Afsse (2004). The finest particles of size lower than 10µm (PM10) or even 2,5µm (PM2,5) are likely to penetrate deeply in the lungs were they can be deposited and were they can release some toxic products such as PAHs, Tissort (1999); Mauderly (2001), Kagawa (2002). The diesel particles are often pointed out because they can induce respiratory and cardiovascular diseases. A better understanding of their formation modes in the engines and in the exhaust line is mandatory to be able to reduce them by new engine technologies or new formulation axes for diesel fuels. Therefore, car manufacturers and oil companies need more precise information concerning these particle formation and evolution.

In these last years, an increasing number of studies has been devoted to the experimental determination of particle size distribution in exhaust systems, Baumgard (1996), Shi and Harrison (1999, 2000), Abdul-Khalek et al. (1998, 1999, 2000). However, the methodologies used, the vehicles tested and the operating conditions were strongly different from one study to another and therefore it is very difficult to compare the results.

In this context, we have combined two complementary approaches: a modelling study and an experimental measurement of particle formation and evolution. The measurement techniques include the particle-measurement instrument of SMPS type (Scanning Mobility Particles Sizing) or the ELPI (Electric Low Presssure Impactor) which make it possible to count particles in the size range 7 - 300 nm. Thanks to these apparatuses, ultra-fine particles (diameter of about 10 nm to 20 nm) were measured, as well as larger particles, for gasoline or diesel vehicles. If the formation and the nature of the solid particles (diameter of about 100 nm) have been often studied, Auphan-de- Tessan (1999), Beaulieu (2001), it is not the case for these ultra-fine particles which can probably be described as liquid "droplets" or aerosols.

Objectives

The objective of the present work was to develop a numerical code able to model the formation and evolution of the ultra-fine aerosol and to study its interactions with the larger particles. Certain assumptions have already been made concerning the formation processes Kittelson, (1999), Vouitsis et al (2005). These ultra-fine particles could be formed from the nucleation (gas – liquid conversion) of the acid sulphuric - water mixture, Jaecker – Voirol et al., (1987, 1988), Kulmala and Laaksonen (1990) present in exhaust gas, but the participation of certain hydrocarbons cannot be excluded. Belot et al. (2004) has confirmed this assumption showing that the sulphur content of the fuel is a very important parameter for the resulting particle size distribution. However, the observed bimodal distribution of the aerosol size distribution suggest that two or more coupled phenomena are occuring:

- the formation of liquid aerosols by homogeneous nucleation, followed by growth and coagulation of the nuclei leading to the ultra-fine part of the aerosol,
- heterogeneous phenomena (nucleation, growth...) for which the soot particles formed during the combustion process act as condensation nuclei, leading to the larger fraction of the particulates.

The "Particulates" European program, Ntziachristos et al (2004), has demonstrated that the nucleation phenomenon is highly influenced by the "external" conditions, such as the exhaust line temperature and sampling conditions (dilution ratio, dilution air temperature, residence time in the sampling lines, ...)

To better understand which physical processes are involved in the formation of diesel particles, it was decided to develop a numerical model which aim was to simulate the particle emission properties of a vehicle and more specially the size distribution of the particles which, on another hand, were measured on a chassis dynamometer, with a SMPS. This model would be an essential tool to explore the impact of after–treatment systems and sampling conditions.

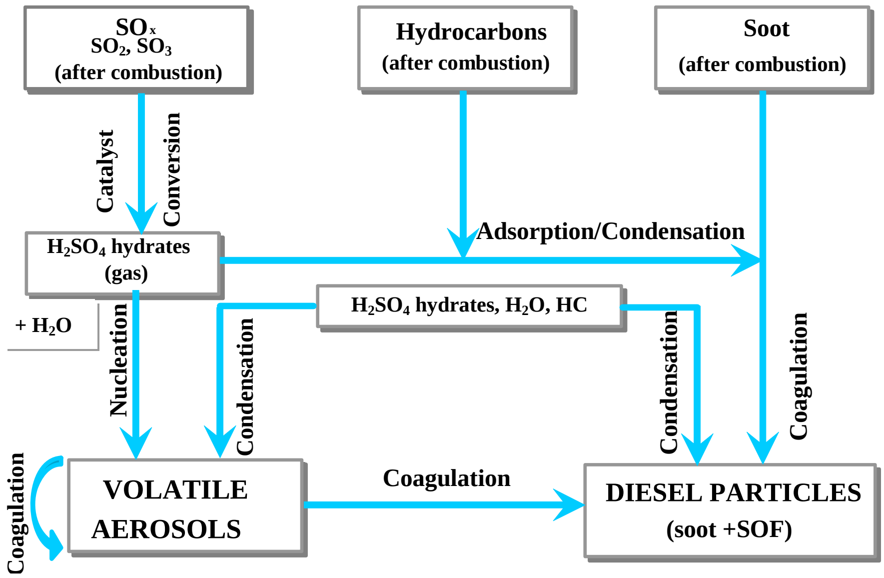
Model development

Initially, we started by adapting existing thermodynamic models that were developed to simulate the formation of particles in the atmosphere. The various theories presented in the literature: nucleation (homogeneous or heterogeneous), growth and coagulation were transcribed in the form of dedicated data-processing models.

The model can be divided into different parts (see Fig 1) taking into account SO_2, soot and hydrocarbons. The evolution of these constituents is governed by the nucleation, the condensation and the coagulation processes.

Figure 1: Scheme A.Si.Di.Si Model for particle formation

Figure 1 : Schéma du modèle A.Si.Di.Si de formation des particules



1. Nucleation

Nucleation is a general word used to describe, from a thermodynamic point of view, the transitions from a metastable phase to another more stable phase. This word tends to replace the French term of "germination". In this article, "nucleation " will concern only gas to liquid transition of the water - sulphuric acid mixture. Nucleation of this mixture has been studied earlier in the eighty's, but it was under atmospheric conditions. It was then necessary to update the thermodynamic functions for a temperature range 400-600 K. The rate of homogeneous nucleation J (the number of nuclei formed in $1 cm^3$ and in 1 s) is governed by the following equation :

$$J = C \exp(-\Delta G^*/kT)$$

Where ΔG^* is the free energy required to form a critical nucleus adapted by Jaecker – Voirol (1987), Noppel (1998), T the temperature, k the Boltzmann constant and C is a slowly varying frequency factor firstly determined by Reiss (1950) and adjusted by Stauffer (1976).

Note that only the H_2SO_4 – H_2O mixture is concerned by nucleation. There is no possibility of hydrocarbon nucleation because their concentrations are much too low.

2. Condensation

The model developed to describe the condensation process is base on the Fukuta and Walter (1970) theory. Condensation is mainly controlled by the rate of incorporation of acid molecules N_a according to:

$$dN_a/dt = 4 \pi R_p D_g (P_a - P_a^0) f(Kn, \alpha)$$

where R_p is the particle radius, D_g the diffusion coefficient for gas, P_a the vapour pressure of the acid in the gas phase, P_a^0 is the acid equilibrium vapour pressure at the surface of the droplet and f(Kn,α) is a correction due to non continuum effects and imperfect surface accommodation, depending on the Knudsen number Kn and the molecular accommodation coefficient α. H_2SO_4, H_2O and hydrocarbons are also able to condense on particle surface. For H_2O condensation, we have applied the assumption made by Mirabel and Katz (1974) i.e. to limit H_2O incorporation, it is supposed that there is a thermodynamic pseudo- equilibrium between two acid molecules incorporation.

3. Coagulation

Particles may come into contact because of Brownian motion or other forces and they can form a larger particle which volume is equal to the sum of the initial volumes. If N_k is the particle concentration per cm^3, the number of new particles k formed by coagulation is:

$$\frac{dN_k(t)}{dt} = \frac{1}{2}\sum_{j=1}^{k-1} K_{j,k-j} N_j(t) N_{k-j}(t)$$

The ½ factor avoids to count twice the same collision and $K_{i,j}$ is the coagulation coefficient between the two particles i and j. The number of k particles, which disappear by coagulation, is given by:

$$\frac{dN_k(t)}{dt} = N_k(t)\sum_{j=1}^{\infty} K_{k,j} N_j(t)$$

The global variation of the concentration k particles is :

$$\frac{dN_k(t)}{dt} = \frac{1}{2}\sum_{j=1}^{k-1} K_{j,k-j} N_j(t) N_{k-j}(t) - N_k(t)\sum_{j=1}^{\infty} K_{k,j} N_j(t)$$

The coagulation is a very important process for the particle size distribution because it modifies both the size and the concentration of particles.

4. The initial conditions

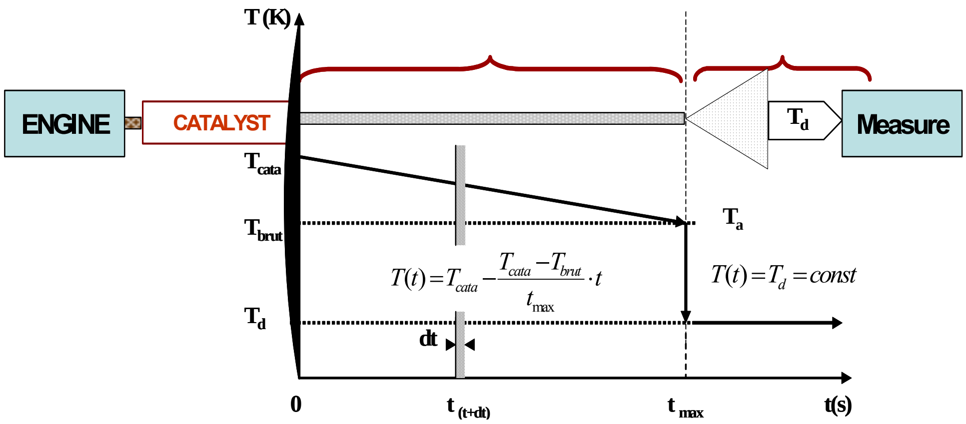
For the simulation, the most important data to introduce in the model are :

- The sulphuric acid concentration which is evaluated using the sulphur content of the fuel, the conversion efficiency of the catalyst which itself depends on the catalyst temperature.
- The water concentration which is calculated using the combustion equation and the relative humidity of the air of dilution coming in the CVS (Constant Volume Sampling).
- The hydrocarbons: as it was impossible to take into account all the hydrocarbons. We have used an hypothesis similar to the one of Abdul-Khalek (2000) and simulated the hydrocarbons with only a mixture of three species C16 (30%), C19 (20%) and C25 (50%) Nguyen Thi, (2005).

- Soot distribution: To simulate the interactions between the ultrafine and the larger part (soot) of the aerosols, the soot size distribution is needed. The model offers two possibilities : to introduce a soot size distribution measured after a thermo denuder or to generate this distribution using an initial two classes distribution of soot precursors, Nguyen Thi (2005).
- The evolution of the temperature in the exhaust line which is a key point of the model (see Fig 2). As the simulations begin just after the catalyst, the knowledge of its temperature is very important because it has a strong impact on H_2SO_4 concentration. We have assumed a linear decrease between the catalyst temperature and the CVS where the temperature of the dilution gas is taken into account.

Figure 2: Modelisation of the temperature variations

Figure 2 : Modélisation des variations de température



Results and discussion

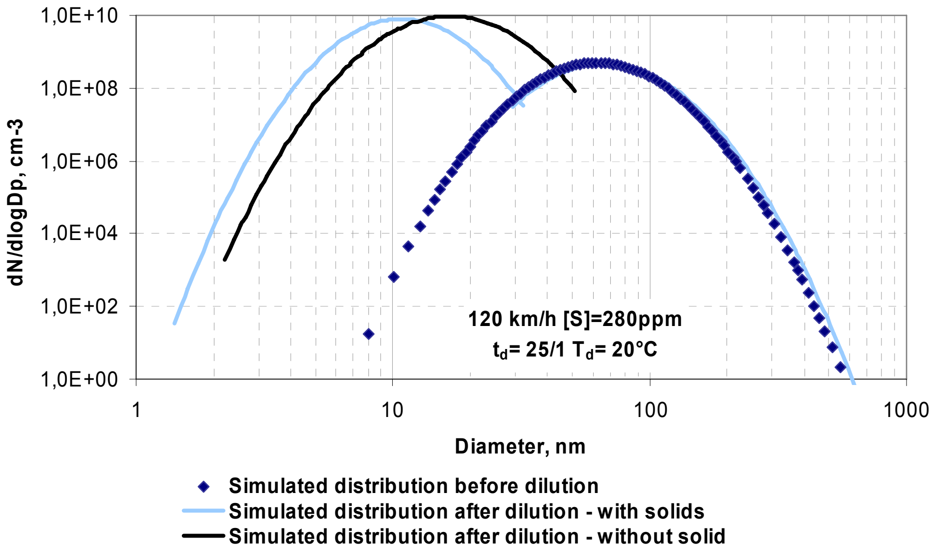
The model was used to test the sensitivity of the processes to different variations of the physical parameters: temperature, relative humidity, acid sulphuric content etc. As we have seen that particle formation is the result of a complex mixing of various processes such as homogeneous and heterogeneous nucleation, growth by condensation and coagulation, the variation of the physical parameters will help to determine what are the key processes.

1. Evolution of liquid aerosols

The homogeneous nucleation of the H_2SO_4 – H_2O mixture is responsible of the formation of the finest particles (∅ < 20 nm) that are mainly liquid aerosols. The H_2SO_4 – H_2O homogeneous nucleation could not occur upstream of the CVS, because of the too high temperature. It is only when exhaust gases are diluted in the CVS that nucleation can be efficient (see Fig. 3). Even with a fuel sulphur content as low as 5 ppm, homogeneous nucleation still produce very fine liquid aerosols (∅ ≈ 10 nm). In presence of soot, liquid aerosols condense on the carbon surface, decreasing the number of smallest particles. The concentration of hydrocarbons is too low to allow nucleation. They can only condense on soot particles.

Figure 3: Impact of dilution on homogeneous nucleation and liquid aerosol formation

Figure 3 : Impact de la dilution sur la nucléation homogène et la formation des aérosols liquides



2. Comparison with measurements

The model A.Si.Di.Si (Aerosol Size Distribution Simulation) was validated, successfully, by comparison with the results of measurements done during the European "Particulates" program The "Particulates" goal was to propose a reliable, repeatable and reproducible method to measure the size repartition and emission level of exhaust particles. During this program different vehicles where tested on chassis dynamometer, with a special design for the particle measurements allowing variations of temperature, relative humidity, dilution ratio and duration in the taking line. The sensitivity studies allow to evaluate the influence of the parameters "fuel" (sulphur content), "vehicle" (speed, fuel consumption,…), and "conditions of measurements" (temperature, dilution ratio, humidity, residence time,…).

The model gives the same answers in term of influence of the different parameters (see table 1). Increasing fuel sulphur content facilitates the nucleation and the growth by condensation, so both the number and the size of particles increase. Increasing the relative humidity leads to the same effects. Increasing the temperature of the diluted gases decreases the nucleation rate and then decreases the number and the size of particles. For the dilution ratio, the results of the measurements are not so clear, but in most of the measurements we do not know the temperature of the diluting gas. We have observed with the model that a variation of only 5°C of the diluted gases temperature can change the impact of the dilution ratio. Increasing the residence time of the particles in the sampling line induces a decrease of their number and an increase of their size. Coagulation is directly responsible of this impact.

Table 1: Impact of the different parameters on the nucleation mode particles

Tableau 1 : Impact des différents paramètres sur les particules formées par nucléation

Nucleation mode				
	Measurements		Model A.Si.Di.Si	
Parameters	Particle number	Particle diameter	Particle number	Particle diameter
Fuel sulphur content↗	↗	↗	↗	↗
Relative humidity ↗	↗	↗	↗	↗
Temperature ↗	↘	↘	↘	↘
Dilution ratio ↗	↗↘	↗↘	↗↘	↗↘
Residence time ↗	↗↘	↗↘	↘	↗

These results allow to conclude that this model represents well the different processes involved in the formation of the particles and their measurements.

We have successfully simulated the measurements obtained with two different Euro III vehicles. The first one was equipped with a silicium carbide particulate filter (DPF), and the second had no filtration device. The tests were done on chassis dynanometer, for stabilised speeds (120 km/h and 50 km/h). The particle size distribution was measured with an SMPS. As one can see on figures 4 and 5, the model reproduces quite well the experimental data in both cases.

Figure 4: Comparison model/ measurement: Vehicle equipped with particulate filter

Figure 4 : Comparaison mesures / modèle : Véhicule équipé d'un filtre à particules

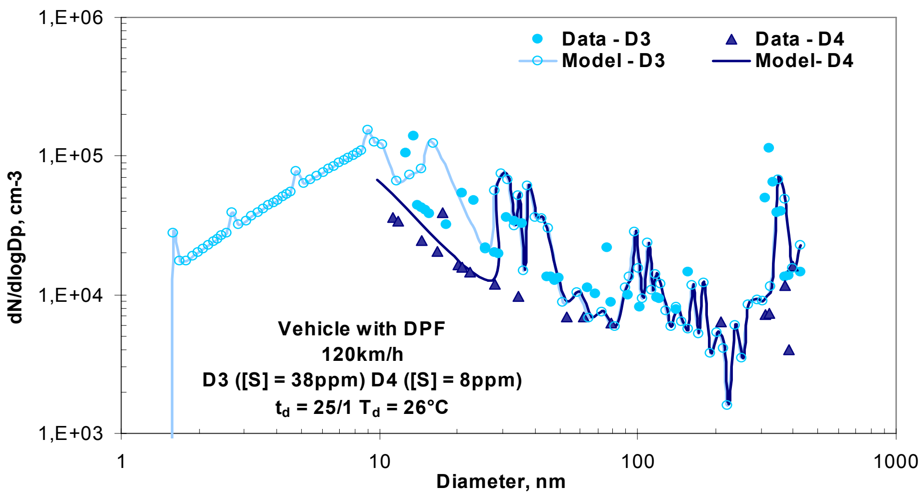
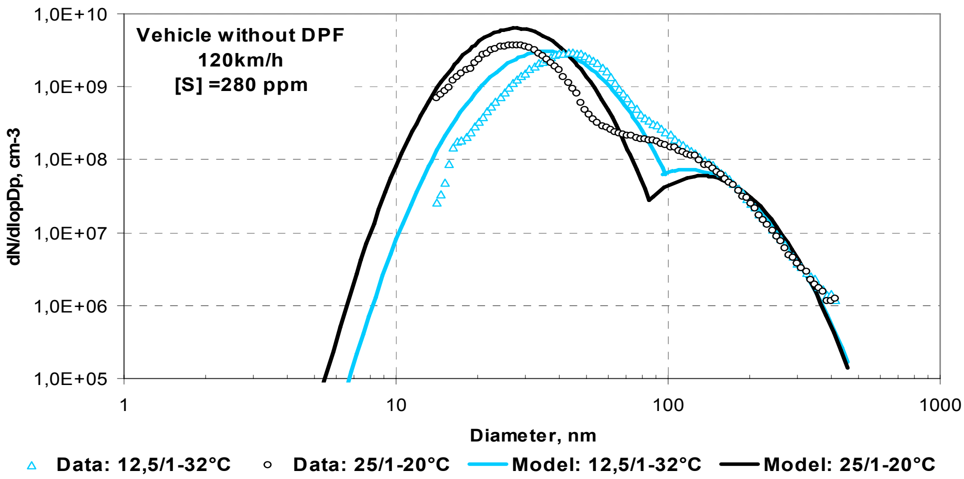


Figure 5: Comparison model/ measurement: Vehicle without particulate filter – Variation of dilution ratio and temperature of diluted gases

Figure 5 : Comparaison mesures / modèle : Véhicule non équipé d'un filtre à particules – variation du taux de dilution et de la température des gaz dilués.



Conclusion

The A.Si.Di.Si model was developed to simulate the formation and the size repartition of particles in the vehicle exhaust gas. This is a powerful tool, which was validated by comparison with the data base "Particulates". This model is able to represent the size repartition of the particles whatever the analysis system (SMPS, ELPI, with or without thermodenuder), the vehicle equipment (particle filter or not). It is a reliable tool for better controlling the influence of the measurement conditions (Temperature, humidity, duration, dilution ratio).

This model provides an improved comprehension of the phenomena involved in the formation and the evolution of the particles in the vehicle exhausts. We have proved that nucleation could occur even with fuel sulphur contents of 5 ppm. The formation of liquid aerosols is possible only after the dilution process, since before dilution the temperature is too high. The influence of the catalyst, which converts SO_2 into H_2SO_4 is fundamental for the formation of these liquid aerosols.

The measurement conditions have a great influence on the size distribution specially for the liquid aerosols. This model could be very helpful for the definition of standardized measurement conditions.

Acknowledgments

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Emission factors of unregulated atmospheric pollutants for passenger cars - Task 322 of the EU ARTEMIS Project

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Abstract

EU ARTEMIS project focused on the assessment and reliability of transport emission models and inventory system, and the sub-task 322 entailed measurements of unregulated pollutants from passenger cars. As the result of this work a database of unregulated exhaust pollutants was developed, providing a unique and extensive set of data to study and define emission factors for several non-regulated pollutants. Based on the current knowledge on the atmospheric pollutants from mobile sources, and taking into account limitations of Artemis data on unregulated pollutants, the following "Priority Toxics" were selected for determination of emission factors: benzene, 1,3-butadiene, ethylbenzene, toluene, xylenes, n-hexane, formaldehyde, acetaldehyde, acrolein, benzo[a]pyrene and sum of six most carcinogenic PAHs from the combined "semivolatile+ particulate" phase. For major part of unregulated pollutants, the emissions level decreased step by step, when advancing from pre-Euro-1 cars towards Euro-4 emission class cars. In some cases, like 1,3-butadiene emission, emissions were close to zero with cars from Euro-1 to Euro-4 emission class. Generally, emissions of individual hydrocarbons from diesel cars were at lower level than those from petrol fuelled cars, whereas aldehyde emissions on the other way around.

***Keys-words**: light-duty, passenger cars, unregulated, emissions, atmospheric, toxics, benzene, 1,3-butadiene, formaldehyde, acetaldehyde, PAH*

Introduction

There are a number of atmospheric pollutants from mobile sources, which have been found to be toxic or risk to human health or environment, but not regulated by legislation. Unregulated emissions have been studied widely for a long time. However, measurement protocols and analysis methods are not harmonized due to lacking control of laws and standards. Despite of various studies on unregulated emissions, representative emission factors (EF) of unregulated emissions for different emission categories of cars do not exist.

There are several lists of "Priority Air Toxics" that define the most harmful compounds that should be taken into account for mobile exhaust gases. These lists have been defined *inter alia* by U.S.EPA and EU from various starting points, and thus there are also some differences in the outcome. However, a common conception is that at least volatile organic compounds (VOC) and the polyaromatic hydrocarbons (PAHs), are necessary to assess more accurately.

Objectives

Sub-task 322 "Exhaust emissions of unregulated pollutants" of EU ARTEMIS project entailed measurements of non-regulated pollutants from passenger cars of current technology (Euro 3 and Euro 4), including hot and cold start tests. The objective was to define emission factors for unregulated emissions from passenger cars. In principle this task provided an extensive test matrix, which enables to analyse EFs for various toxic pollutants, even though the database of unregulated emissions is limited when compared to ARTEMIS database of regulated emissions. Furthermore, due to variations in analytical methods, there were significant limitations regarding comparability of the unregulated data between different laboratories. This paper describes the emission factors of the unregulated emissions, and discusses how these factors meet the reliability criteria.

Experimental

1. General description of the task, test matrix

The emission tests were carried out in five laboratories: VTT, EMPA, INRETS, Istituto Motori (IM) and KTI. In addition, Université du Littoral Côte d'Opale (ULCO), Université de Savoie (US), and Université du Droit et de la Santé de Lille (ULCO) conducted chemical analyses of the samples taken in tests conducted at INRETS.

The main emphasis was given on vehicles representing current and new technologies. However, there is still quite substantial amount of old vehicles in the vehicle park in almost all countries. Therefore, the test fleet comprised also vehicles of Euro 1 and even pre-Euro 1 cars without catalyst. The unregulated emissions were tested in parallel with regulated emissions. Thus the complete detailed description of the cars, test cycles, the testing conditions etc. is found from the reports of regulated emissions (Joumard et al., 2005).

Normal test temperature was deemed to be over 18 °C in the reporting of the unregulated emissions. EMPA and VTT carried out tests also at low ambient temperatures, -7 and -20 ºC, in addition to normal ambient conditions, but those results are not shown in this paper. IM has found a substantial influence of the fuel temperature on emissions, particular care was taken that this kind of effects shall not affect to the results. Therefore, adequate cooling was provided.

Local, commercial grade fuels were used. Thus the fuel composition varied from one laboratory to another to some extent, which is discussed in the other ARTEMIS reports (Renault & Altran, 2002).

Table 1: General description of the test matrix.

	EU emission standard	Laboratory	Temperature[a]	Unregulated emissions[b]			Number of cars	
				VOC	Carbonyl	PAH[c]	per laboratory	per emission group
gasoline	pre-EURO-1	INRETS	>18	x	x	x	5	
		EMPA	>18, low	limited[d]	no	no	6	
		ISTITUTO MOTORI	>18	x	x	x	1	**12**
	EURO-1	INRETS	>18	x	x	x	5	
		ISTITUTO MOTORI	>18	x	x	x	1	**6**
	EURO-2	INRETS	>18	x	x	x	5	
		ISTITUTO MOTORI	>18	x	x	x	2	
		KTI	>18	x	no	x	1	
		VTT	>18, low	x	no	no	1	**9**
	EURO-3	INRETS	>18	x	x	x	3	
		EMPA	>18, low	x	x	no	6	
		ISTITUTO MOTORI	>18	x	x	x	4	
		VTT	>18, low	x	no	no	4	**17**
	EURO-4	VTT	>18, low	x	no	no	2	2
diesel	pre-EURO-1	INRETS	>18	x	x	x	2	2
	EURO-1	INRETS	>18	x	x	x	3	3
	EURO-2	INRETS	>18	x	x	x	10	
		EMPA	>18, low	limited[d]	no	no	6	
		ISTITUTO MOTORI	>18	x	x	x	1	
		KTI	>18	x	no	x	1	**18**
	EURO-3	INRETS	>18	x	x	x	2	
		ISTITUTO MOTORI	>18	x	x	x	2	**4**

[a] hot-start and cold-start tests; "low" means -7 and -20 °C [b] not analysed with all car/cycle combinations

[c] INRETS analyzed PAHs separately from particulate and semivolatile phase, whereas IM and KTI from combined samples

[d] "limited" means that only some light aromatics were analyzed

The test matrices for the unregulated emissions at different laboratories are shown in a Table 1. From six to nine petrol-fuelled cars were tested in each emission category from pre-Euro-1 to Euro-3, but only two Euro-4 cars. Euro 4 cars were not widely available at the time of tests, and the spread in EFs between individual Euro 4 cars is expected to be narrow, when compared to older technology. When diesel cars are considered, Euro-2 emission class cars were well-represented with 18 cars tested. Four diesel cars is deemed to be fairly representative set to produce emission factors for Euro-3 emission class, especially as these cars were tested at two laboratories, whereas only two pre-Euro-1 diesel cars and three Euro-1 diesel cars were measured, and thus not assumed to be representative enough. In addition, Euro 4 diesel cars were not tested at all.

As a summary, matrix of petrol fuelled cars covered fairly well all emission categories, whereas diesel cars did not cover all emission categories. Emission factors for pre-Euro-1 and Euro-1 emission class diesel cars are not representative due to low sample size.

Table 2: Test cycles used at different laboratories. All cars were not tested using all cycles mentioned in the Table (characteristics of cycles: André 2004, Joumard et al., 2005).

INRETS	EMPA	ISTITUTO MOTORI	KTI	VTT
"Artemis" test cycles[a]				
Artemis.HighMot_motorway	Artemis.motorway_150	Artemis.motorway_130		
Artemis.HighMot_urban	Artemis.rural	Artemis.rural		Artemis.rural
Artemis.LowMot_motorway	Artemis.urban	Artemis.urban		Artemis.urban
"Inrets", "European" and "FTP" test cycles[a]				
Inrets.urbainfluidecourt_INRETS	Inrets.urbainfluidecourt_I	Inrets.urbainfluidecourt_I	Inrets.urbainfluidecourt_III	Inrets.urbainfluidecourt_I
	Inrets.urbainfluidecourt_II	Inrets.urbainfluidecourt_II		Inrets.urbainfluidecourt_II
	Inrets.urbainfluidecourt_III	Inrets.urbainfluidecourt_III		Inrets.urbainfluidecourt_III
	Inrets.routecourt_I (&II&III)	Inrets.routecourt_I (&II&III)	Inrets.routecourt_I (&II&III)	
	Legislative.ECE_2000	Legislative.ECE	Legislative.ECE	Legislative.ECE
	Legislative.EUDC	Legislative.EUDC	Legislative.NEDC	Legislative.EUDC
	Legislative.US_FTP2 (&3)	Legislative.ECE_2000		Legislative.ECE_2000
Other cycles[a]				
modernHyzem motorway_part_1	EMPA BAB			
modernHyzem road	Handbook.R1_I (&II&III)			
modernHyzem urban1	Handbook.R2_I (&II&III)			
modernHyzem urbanx2	Handbook.R3_I (&II&III)			
	Handbook.R4_I (&II&III)			

[a] all cycles not tested with all cars

The test cycles used in the measurements are shown in Table 2. As this was not a strictly coordinated effort, there were not many common cycles at different laboratories. The test cycles at EMPA, IM and VTT showed the highest congruity. General procedures used in the exhaust emission measurements are reported in other Artemis reports (Joumard et al., 2005). Overall, all tests were carried out on chassis dynamometer, fitted with a constant volume sampling system using purified air for the dilution air of exhaust gases.

2. Unregulated emissions measured

A number of unregulated species (URP) were measured, *inter alia,* individual hydrocarbons, carbonyl compounds, PAHs from particulate and/or semivolatile phase. Furthermore, some other emissions, e.g. particle size distributions, methane, ammonia, nitrous oxide and others measured with FTIR, were also studied in some laboratories, but the results are not shown in this paper. Individual compounds analyzed varied from laboratory to another. In addition there were variations in the analysed compounds also within laboratories, because different set of unregulated compounds were analysed depending on car/cycle combinations.

ARTEMIS URP database was created in Task 322. The data is based on true analyses results, e.g zero in the database should mean that the result is zero, whereas blank cell should indicate that no analysis was carried out. Groups, like "VOCs" and "alkanes", are not included in the database. The data obtained at normal temperature for "Priority Toxics" was analysed for outliers, these were discussed with responsible laboratories, and discarded or corrected.

Artemis Task 322 produced data on a huge number of different unregulated compounds. However, when considering the inconsistency of data, and other reliability factors, it was decided to focus on the priority toxic species, which were defined in this study as follows:

- benzene - 1,3-butadiene
- ethylbenzene - toluene
- xylenes - n-hexane
- formaldehyde - acetaldehyde
- acrolein - benzo[a]pyrene from combined "semivolatile+particulate" portion
- sum of 6 most carcinogenic PAHs from combined "semivolatile+particulate" portion: benz(a)antracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, indeno (1,2,3-cd)pyrene, dibenz(a,h)anthracene

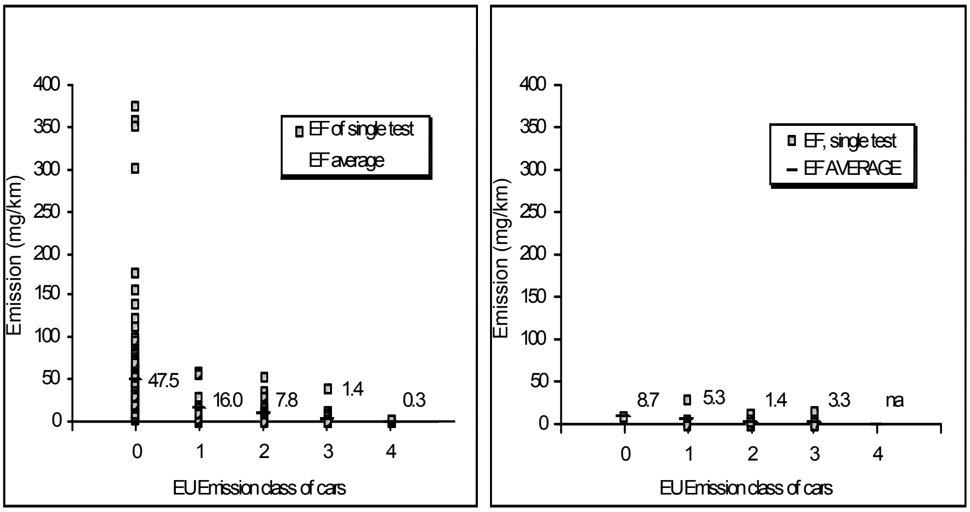
3. Analysis methods

Sampling procedures, measurements and other analysis methods are described in Annex 1 per laboratory, as they vary from one lab to another.

Results

Round-robin of unregulated components was not included in the Artemis project, but comparison of laboratories was carried out using the results from the same test cycle with the same emissions group of cars. Comparability of the laboratories was fairly good as concerns benzene and formaldehyde emission, even though benzene level was somewhat higher at INRETS, and formaldehyde level lower at EMPA, than respective emissions at other laboratories. The widest range in emissions reported by the laboratories was seen in PAH results. The lowest benzo[a]pyrene level was obtained at INRETS, intermediate level at KTI and the highest level at IM. This is specifically important due to the fact that the test matrix on PAH emissions was not as extensive as e.g. matrix on benzene and formaldehyde.

Figure 1: Benzene results from individual tests, hot-start tests at normal temperature.



An example of benzene results from individual tests is shown in Figure 1 and average results for various emissions in Figure 2. The average results, deviation and the number of measurements are summarized in Table 3.

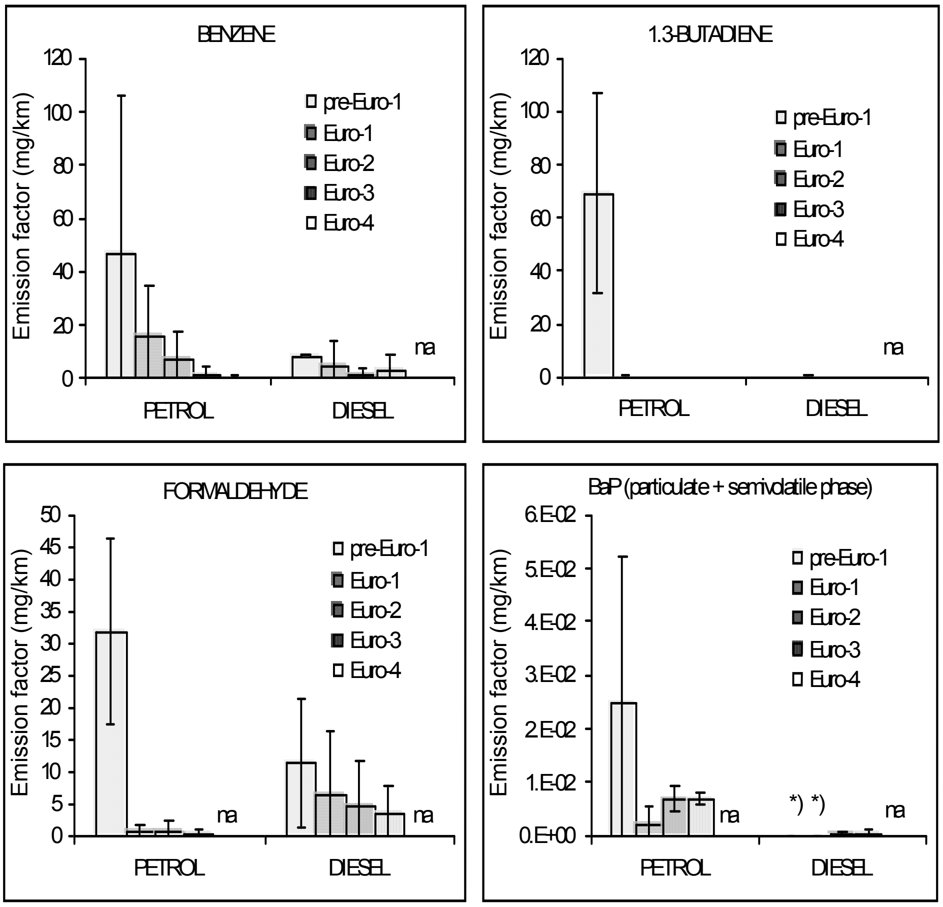
Benzene emissions from pre-Euro-1 emission level petrol-fuelled cars varied a lot, from very low level of emissions up to almost 400 mg/km. This applied to other “Priority VOCs”, as well. Technology variations of the cars belonging to pre-Euro-1 emission class are huge. Some cars are equipped with carburettors, some with fuel injection systems, either less or more developed ones. Some cars are equipped with three-way-catalyst. This explains the wide variation in the emission factors.

Table 3:Summary of the emission factors (EF) on “Priority toxics” from different emission categories of cars, hot-start tests at normal temperature. Note: Sample size of Euro-4 petrol cars (2 cars tested), Pre-Euro-1 diesel cars (2 cars tested) and Euro-1 diesel cars (3 cars tested) was too low for final conclusions.

			GASOLINE			DIESEL		
EU emis. standard	hot/ cold	Temperature	Average mg/km	Stdevp mg/km	Number of tests	Average mg/km	Stdevp mg/km	Number of tests
benzene								
pre-EURO-1	hot	>18	47.5	58.9	147	*8.7*	*0.0*	*1*
EURO-1	hot	>18	16.0	18.5	14	*5.3*	*9.0*	*8*
EURO-2	hot	>18	7.8	10.0	55	1.4	2.1	174
EURO-3	hot	>18	1.4	3.3	190	3.3	5.3	10
EURO-4	hot	>18	0.3	0.7	10	na	na	na
1.3-butadiene								
pre-EURO-1	hot	>18	*69.3*	*37.4*	*8*	*na*	*na*	*na*
EURO-1	hot	>18	0.38	0.53	10	*0.21*	*0.13*	*4*
EURO-2	hot	>18	0.00	0.00	29	0.00	0.00	8
EURO-3	hot	>18	0.03	0.10	61	0.00	0.00	7
EURO-4	hot	>18	0.0	0.0	10	na	na	na
ethylbenzene								
pre-EURO-1	hot	>18	na	na	na	*11.2*	*6.9*	*5*
EURO-1	hot	>18	*4.0*	*4.5*	*8*	*0.9*	*1.3*	*10*
EURO-2	hot	>18	12.1	23.6	36	6.3	8.7	38
EURO-3	hot	>18	4.4	13.1	34	*22.6*	*15.3*	*3*
EURO-4	hot	>18	0.0	0.0	10	na	na	na
toluene								
pre-EURO-1	hot	>18	208.1	204.4	147	*31.7*	*27.9*	*5*
EURO-1	hot	>18	15.6	12.5	14	*12.7*	*18.3*	*11*
EURO-2	hot	>18	16.0	32.9	60	3.0	10.0	187
EURO-3	hot	>18	2.5	9.8	191	*6.2*	*7.1*	*9*
EURO-4	hot	>18	0.2	0.5	10	na	na	na
hexane								
pre-EURO-1	hot	>18	*67.5*	*44.8*	*8*	*na*	*na*	*na*
EURO-1	hot	>18	3.7	3.6	10	*na*	*na*	*na*
EURO-2	hot	>18	1.0	1.1	25	*0.3*	*0.7*	*8*
EURO-3	hot	>18	0.1	0.3	49	*0.7*	*1.6*	*7*
formaldehyde								
pre-EURO-1	hot	>18	32.0	14.4	18	*11.4*	*10.0*	*13*
EURO-1	hot	>18	0.8	1.0	31	*6.4*	*10.0*	*20*
EURO-2	hot	>18	1.0	1.4	51	4.8	6.8	52
EURO-3	hot	>18	0.4	0.5	65	3.6	4.1	20
acetaldehyde								
pre-EURO-1	hot	>18	11.5	11.4	18	*7.7*	*6.1*	*13*
EURO-1	hot	>18	0.7	0.6	31	*6.6*	*8.3*	*20*
EURO-2	hot	>18	0.7	0.9	50	4.1	5.5	52
EURO-3	hot	>18	0.2	0.2	65	1.7	2.4	20
acrolein								
pre-EURO-1	hot	>18	*2.6*	*0.1*	*3*	*1.5*	*2.0*	*12*
EURO-1	hot	>18	0.0	0.1	9	*0.8*	*1.2*	*6*
EURO-2	hot	>18	0.4	1.2	13	0.3	0.7	29
EURO-3	hot	>18	0.0	0.0	12	na	na	na
Sum of priority VOCs (from data above)								
Pre-Euro 1	hot	>18	438	371	varies*	*72*	*53*	varies*
EURO-1	hot	>18	41	41	varies*	*33*	*48*	varies*
EURO-2	hot	>18	39	71	varies*	20	35	varies*
EURO-3	hot	>18	8.9	27	varies*	38	36	varies*
EURO-4	hot	>18	0.6	1.3	varies*	na	na	varies*
benzo(a)pyrene								
Pre-Euro 1	hot	>18	0.025	0.027	8	**	**	*3*
EURO-1	hot	>18	0.002	0.003	11	**	**	*8*
EURO-2	hot	>18	0.007	0.002	39	0.000	0.001	53
EURO-3	hot	>18	0.007	0.001	47	0.001	0.001	24
Sum of 6 PAHs								
Pre-Euro 1	hot	>18	*0.112*	*0.104*	*8*	**	**	*3*
EURO-1	hot	>18	0.008	0.007	11	**	**	*8*
EURO-2	hot	>18	0.004	0.010	23	0.002	0.006	37
EURO-3	hot	>18	0.005	0.007	47	0.003	0.003	24
Euro 4	hot	>18	na	na	na	na	na	na

*) varies with compound, e.g. benzene from pre-Euro-1 gasoline cars: 147 tests, but 0 on ethylbenzene
**) average EF is not representative (see Chapter 6)
Font in italic means low sample size, average EF may not be representative
na = no data available

Figure 2: Average results of benzene, 1,3-butadiene, formaldehyde and Banz(a)pyrene, hot- start tests at normal temperature. [markers: na=not analysed; *) too low sample size]



Petrol fuelled cars from Euro-1 to Euro-4 emission level showed drastically lower benzene emissions on average than pre-Euro-1 cars. The major reduction in mass emissions was seen when Pre-Euro-1 cars were compared to Euro-1 emission level cars. Average benzene emissions were 48 mg/km for pre-Euro-1, 16 mg/km for Euro-1, 8 mg/km for Euro-2, 1.4 mg/km for Euro-3 and only 0.3 mg/km for Euro-4 emission level petrol cars. The development of the vehicle and aftertreatment technology after Euro-1 emission level is seen clearly regarding benzene emissions.

Benzene emissions from diesel fuelled cars were on average below 10 mg/km, and for all individual diesel cars below 30 mg/km. Benzene emission from pre-Euro-1 diesel cars was on average about one fifth of that from Pre-Euro-1 petrol cars. Benzene emissions from diesel cars were not highly dependent on the emission class of cars. However, from Euro-2 onwards benzene emissions from diesel cars were at very low level, close to Euro-3 petrol cars. Furthermore, it is noticeable that benzene content of petrol was below 1% at different laboratories.

1,3-butadiene emissions from pre-Euro-1 emission level petrol fuelled cars varied a lot, similarly to benzene emissions. The highest values observed were even above 100 mg/km, which is a huge emission level for 1,3-butadiene. Cars from Euro-1 to Euro-4 emission class showed 1,3-butadiene emissions close to zero. With diesel cars, only a limited number of 1,3-butadiene analyses were carried out. However, a few results with Euro-2 and Euro-3 emission level cars showed close to zero emission level for 1,3-butadiene.

Formaldehyde emissions from pre-Euro-1 petrol fuelled cars were high with an average around 30 mg/km, but variation of the results was also high. Petrol cars from Euro-1 and onwards showed low formaldehyde emissions around 1 mg/km. This is similar trend as was seen for individual hydrocarbons showing influence of the development of engine/aftertreatment technologies on emission.

Formaldehyde emissions from diesel fuelled cars acted differently than those from petrol cars. The emissions were more or less at the same level, or moderately improving when moving from pre-Euro-1 to Euro-3 emission class cars. Euro-1 petrol cars showed only around 1 mg/km formaldehyde emission level, whereas formaldehyde emission from Euro-1 diesel cars was 6.4 mg/km on average. Formaldehyde emission level from pre-Euro-1 diesel cars was around 10 mg/km, which seems to be significantly lower level than that from respective petrol cars (around 30 mg/km), but one must keep in mind that sample size of Pre-Euro-1 and Euro-1 diesel cars was too low to draw definitive conclusions.

Acrolein emissions were generally at low level, but the trends similar, when compared to formaldehyde and acetaldehyde emissions. Acrolein emission level was below 3 mg/km from pre-Euro-1 cars and below 1 mg/km from cars representing Euro-1 emission class onwards. This applies to both petrol and diesel fuelled cars.

Sum of priority VOCs were calculated from average emission factors of benzene, ethylbenzene, toluene, xylenes, n-hexane, 1,3-butadiene, formaldehyde, acetaldehyde and acrolein. Generally the sum of “Priority VOCs” followed the general trends seen for individual VOC species. When average results are concerned, reduction in “Sum of Priority VOCs” is impressive when moving from Pre-Euro-1 to Euro-4 cars: 438 mg/km -> 41 mg/km -> 39 mg/km -> 9 mg/km -> 0.6 mg/km.

Polyaromatic hydrocarbons (PAHs) are shown as combined results from particulate and semivolatile phase. INRETS analysed PAHs from particulate and semivolatile phase separately, which is reported by Joumard et al. (2004). Number of data records on PAHs was lower than e.g. on benzene and formaldehyde - some emission categories of cars were covered only at one laboratory. A clear difference in PAH levels obtained at INRETS and IM was seen, and thus simple averages of emission factors may give misleading results. However, the emission categories of cars can be compared at least within laboratory.

BaP and sum of 6 PAHs were higher for petrol fuelled cars in pre-Euro-1 emission class than cars from Euro-1 to Euro-3 emission class. With diesel cars, based on INRETS data, both BaP and sum of 6 PAHs decreased step by step when moving from pre-Euro-1 to Euro-3 emission level cars. IM tested only Euro-2 and Euro-3 group cars, also in this case an improvement in PAH emissions was seen when moving from Euro-2 to Euro-3 emission class cars. So, evolution of diesel cars was seen as benefit in PAH emissions when data from INRETS and IM was studied separately. However, due to different PAH emission levels from these laboratories, and non-harmonic set of cars, average emission factors does not show this trend. Vice versa, average emission factors show even higher emission factors for e.g. Euro-1 petrol cars than for respective diesel cars. Only 2-3 diesel cars were tested from pre-Euro-1 and Euro-1 emission groups, and thus average emission factors are not shown in summary Table 3 for these emission categories. The PAH results were generally in-line with literature, e.g. two recent reports on automotive PAH emissions (Doel et al. 2005, Aakko 2006).

Conclusions

The sub-task 322 in ARTEMIS entailed measurements of unregulated pollutants from passenger cars of new technology, including hot and cold start tests. Based on the current knowledge on the atmospheric pollutants from mobile sources, and taking into account limitations of Artemis data on unregulated pollutant, the following “Priority Toxics” were selected for determination of emission factors: benzene, 1,3-butadiene, ethylbenzene, toluene, xylenes, n-hexane, formaldehyde, acetaldehyde, acrolein, benzo[a]pyrene from combined “semivolatile+particulate” phase, sum of 6 Priority PAHs from combined “semivolatile+particulate” phase.

Pre-Euro-1 emission class petrol cars showed the highest emission level on average regarding almost all unregulated pollutants. For major part of unregulated pollutants, the emissions decreased step by step when moving from Euro-1 to Euro-4 emission class. In some cases, like 1,3-butadiene emission, emissions were close to zero with cars from Euro-1 to Euro-4 emission class. Emissions from pre-Euro-1 petrol cars vary a lot between individual cars due to wide scale of different technologies in this group (e.g. carburettors or fuel injection systems, some cars not equipped with catalyst).

Diesel cars tend to be more insensitive than petrol cars as regards differences in hydrocarbon emissions from cars representing different emission classes. Generally, emissions of individual hydrocarbons from diesel cars were at lower level than those from petrol-fuelled cars, whereas vice versa as regards aldehyde emissions.

PAH results varied significantly between laboratories. However, within laboratory results showed that both benzo(a)pyrene and sum of 6 PAHs generally decreased step by step when moving from pre-Euro-1 to Euro-4 emission level cars.

The was some uncertainties in the ARTEMIS data on unregulated emissions, e.g. test methods and analysis capabilities for unregulated emissions were not identical between all participants and very few replicate tests were conducted. This limited the amount of data that can be considered comparable, and support statistically solid conclusions, although a considerable amount of data and individual EF’s were collected. However, it is concluded that reasonable emission factors were determined for “Priority toxics” from selected test conditions (normal test temperature, hot-start tests). This applies for the emission categories with representative set of cars. In this respect, the most problematic emission categories were Euro-4 petrol cars, Pre-Euro-1 diesel cars and Euro-1 diesel cars, as the sample size in these emission categories was so too low for the final conclusions.

Acknowledgments

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The rising importance of two-wheelers emissions – a comparison to cars

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Abstract

Passenger cars are the primary means of transportation in Europe. Over the past decade, a great deal of attention has therefore been paid to reducing their emissions. This has resulted in notable technical progress, leading to unprecedentedly low exhaust emissions. In the meantime, emissions from motorcycles have been ignored due to their subordinate role in traffic. Even though the motorcycle fleet is small in comparison with the car fleet, and logs lower yearly mileage per vehicle, their contribution to traffic emissions has become disproportionately high.

Based on exhaust emission measurements of a fleet of modern two-wheelers and of a fleet of actual passenger cars different comparisons are shown. These emission measurements are based on real world cycles (CADC) to get a representative emission values for urban, rural and highway driving separately.

Keys-words*: two-wheelers, motorcycle, car, tailpipe, emission, pollutant, fleet.*

Introduction

Since tailpipe emissions were first recorded in the eighties, a great deal of attention has been given to passenger cars because their number and fleet mileage outweigh all other vehicles. In the past 20 years, statutory limits have been successively reduced, and the technologies implemented have led to notable reductions in emissions of such pollutants as CO, HC and NO_x.

As in other European countries, two-wheelers are not the primary means of transportation in Switzerland. Their number and mileage per vehicle, as well as fuel consumption are lower than in the case of passenger cars. As a consequence, the importance of their emissions has been underestimated in legislation for a long period, giving manufacturers little motivation to improve after-treatment systems. Only slight progress has therefore been made in reducing emissions from powered two-wheelers.

The real significance of their emissions in comparison with modern passenger vehicles is the key topic of this study. The emissions of CO, HC, NO_x and CO_2 from two-wheelers are contrasted with those from gasoline-powered passenger cars. The investigations are based on measurements from vehicles of both types which were on the market in 2001.

A similar study was performed by Chan et ali. (1995) using gasoline cars available at that time. Compared with cars with catalytic converters, cars without catalytic converters and two-cycle motorcycles were found to emit 12 times more HC and CO. However, these emissions ratios have since increased again due to technical progress in reducing emissions from cars. In contrast to earlier measurements from Chan et ali. (1995) and Tsai et ali, (2000), which used statutory driving cycles such as the ECE, , the measurements presented here were obtained using real-world cycles – i.e. cycles derived from driving behaviour studies, which thus give a more realistic representation of the situation on the road. For the sake of comparison, the same real-world driving cycle was used for both vehicle types.

The measurement of tailpipe emissions of CO, HC, NO_x and CO_2 from eight two-wheelers is described in the Methods section. Different comparisons with 17 Euro-3 gasoline-powered passenger cars are given in the section on Emission Comparison. The two groups of vehicles are compared first on their emissions per kilometre, second on the yearly emission per vehicle and third on the fleet emissions per year.

Methods

1. Vehicles

The eight two-wheelers tested were chosen because they were considered to be representative of the Swiss fleet in 2002 with regard to their chassis type, engine capacity, operating principle, after-treatment system and mileage (Vasic et ali., 2006). Vehicle specifications are shown in Table 1. The vehicles were in use and made available by private owners on request. To ensure the results were as close to reality as possible, all vehicles were tested without prior maintenance or adjustments.

The vehicles were from statutory periods FAV 3 (Swiss emission regulations for two wheelers: Verordnung über die Abgasemissionen von Motorrädern, (1988), similar to Euro 1 regulations) and Euro 1. Since the changeover from one to the other involved no tightening of the emission limit values, it was assumed that the emission behaviour of the vehicles had not changed.

Table 1: Specifications of two-wheelers.

Tableau 1: Données techniques des deux-roues.

Vehicle no	1	2	3	4	5	6	7	8
Make	Yamaha	Piaggio	Piaggio	Yamaha	Honda	Suzuki	Honda	BMW
Model	YN 50	Skipper	Vespa	YP 250	Shadow	VS 800 GLP	VFR 800 FI	R1150GS
Type	scooter	scooter	scooter	scooter	motorcycle	motorcycle	motorcycle	motorcycle
Emission class	Euro-1	FAV 3	Euro-1	Euro-1	FAV 3	FAV 3	Euro-1	Euro-1
Capacity [cc]	49	124	124	250	583	805	782	1130
Engine type	2 str.	2 str.	4 str.	4 str.	4 str.	4 str.	4 str.	4 str.
Fuel system	carb.	carb.	carb.	carb.	carb.	carb.	injection	injection
Aftertreatment	oxi-cat	oxi-cat	no	no	no	sec. air valve	3-way cat	3-way cat
Mileage [km]	11'222	15'472	13'951	22'724	5'364	29'466	32'223	31'474

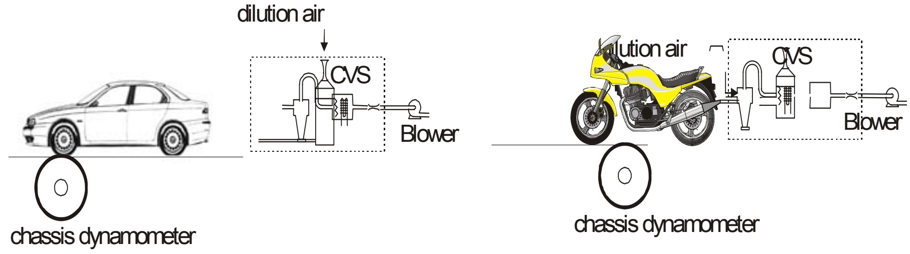
2. Test setup

Emissions measurements were performed at EMPA on a chassis dynamometer test bench in a climate chamber. During all the tests presented, the temperature in the chamber was kept at 23 °C with a relative humidity of 60%. A variable-speed fan was placed in front of the vehicles to simulate the cooling air stream from real driving.

The usual emission testing setup for cars or trucks involves a closed connection between the tailpipe and the CVS dilution system. However, the small engines of two-wheelers produce a comparatively low exhaust gas flow. With a closed connection to the dilution system, the pressure at the tailpipe would be significantly below ambient pressure. The CVS ventilation would support the engine in ejecting the exhaust gas. This aid would clearly falsify the emissions. A more appropriate test setup involving open dilution was therefore chosen. The tailpipe was connected to the CVS plant in an open way, such that the exhaust gas was drawn off diluted with room air. The test setups for cars and two-wheelers are shown in Figure 1.

Figure 1: Test setup for emissions testing of cars and two-wheelers.

Figure 1: Configurations pour mesures d'émissions sur voitures et deux-roues.



3. Emissions and signals

A sample of diluted exhaust gas was fed into sampling bags and analysed offline for its CO, HC, NO_x and CO_2 content. The standard equipment used for this purpose (HORIBA Mexa 7400 series) fulfils certification requirements. In addition to the bag measurements, the diluted exhaust gas and several other signals such as tailpipe temperature, lambda and engine speed were recorded continuously at a rate of 10 Hz for detailed analysis. All the equipment, i.e. dynamometers, CVS and analysers, is approved for homologation.

4. Cycles

Emission measurements were performed on the two-wheelers using the following three real-world driving cycles. All tests started with warm engines.

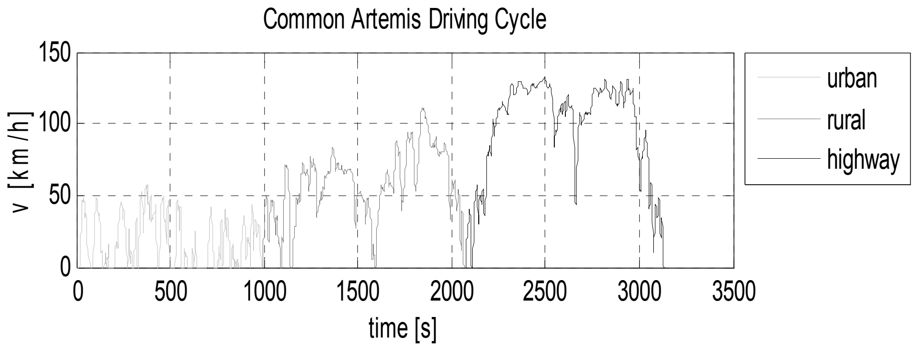
CADC The Common ARTEMIS Driving Cycle developed within the European research project ARTEMIS (2001) (Assessment and Reliability of Transport Emission Models and Inventory Systems) represents mean European driving behaviour for passenger cars (De Haan & Keller, 2001). This real-world driving cycle is divided into patterns for urban, rural and highway driving. Every vehicle was tested in those patterns which were suitable in view of its maximum power (Figure 2).

WMTC (Worldwide Harmonized Motorcycle Emissions Certification Procedure) This cycle was developed to replace various existing legislative cycles for two-wheelers (such as ECE for Europe) with standard worldwide real-world driving cycle (WMTC, 2005). As for the CADC, it is divided into urban, rural and highway driving.

FHB (Fachhochschule Biel cycles) These cycles were developed by the Biel University of Applied Science of to reflect driving in and around this Swiss town. The 'center' pattern represents inner-city driving and, combined with the 'periphery' pattern, models urban driving behaviour.

Figure 2: Speed patterns of driving cycles.

Figure 2: Courbes de vitesse des cycles de conduite.



The characteristics of the test cycles are given in Vasic et ali. (2006). Only the CADC data will be used for the subsequent comparison to cars.

5. Results

The obtained emission factors for CADC are shown in Table 2 and in Figure 3 (for emission values of individuals see Vasic et ali. 2006). The plots highlight the large differences between individual vehicles employing the various technical concepts. Because of its low engine power, vehicle 1 was only tested in the urban part of the cycle, and vehicles 2 and 3 only in the urban and rural parts.

Table 2: mean emissions and standard deviations from two-wheelers in CADC driving cycles.

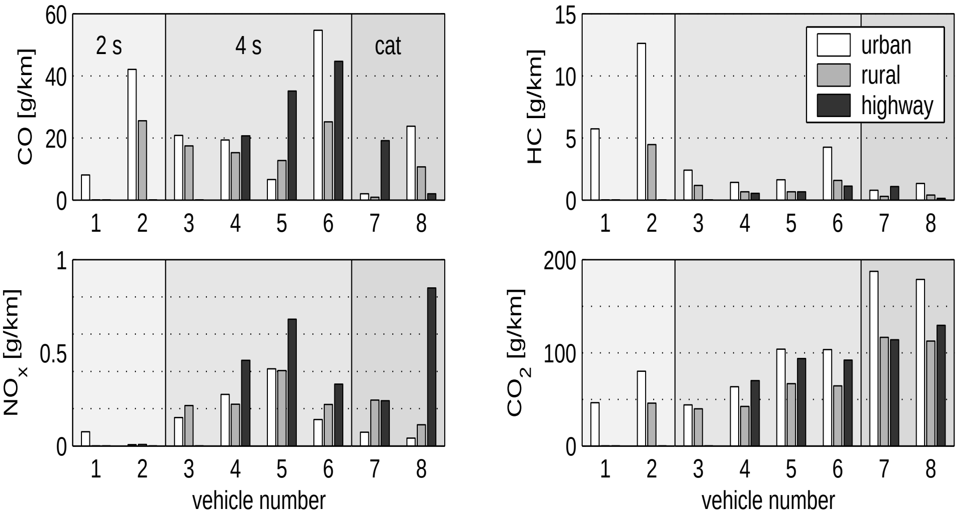
Tableau 2: Emissions moyennes et écarts-type des deux-roues durant cycle de conduite CADC.

	CO [g/km]		HC [g/km]		NO_x [g/km]		CO_2 [g/km]	
	avg.	std.	avg.	std.	avg.	std.	avg.	std.
urban	22.2	18.2	3.77	3.95	0.148	0.136	100.9	55.6
rural	15.4	8.6	1.32	1.46	0.205	0.122	69.7	32.4
highway	24.3	16.4	0.70	0.41	0.512	0.250	99.9	22.7

The oil in the two-cycle fuel and incomplete combustion resulted in high HC emissions for vehicles 1 and 2. In addition, vehicle 2 also produced high CO and almost no NO_x emissions, a result that indicates a very rich mixture. The lambda signals for vehicles 3, 4 and 6 indicated a consistently rich mixture with values around 0.8 to 0.9, while vehicle 5 was lean in the urban part of the cycle, roughly stoichiometric in the rural part and rich in the highway part. The secondary air valve in vehicle 6 seemed to be the main reason for its low HC emissions.

Figure 3: Emissions from two-wheelers in CADC driving cycle; shaded background groups vehicles into two-cycles, four-cycles and vehicles with a three-way catalytic converter.

Figure 3: Emissions des deux-roues durant cycle de conduite CADC ; les différentes couleurs de fond regroupent les véhicules en catégories 2-temps, 4-temps et véhicules équipés de catalyseurs 3 voies.



Vehicles 7 and 8 both had a controlled 3-way catalytic converter and fuel injection systems. Their technology was thus similar to that of modern passenger vehicles. Nevertheless, their lambda values were not the same as for cars and differed from one motorcycle to another. The mixture of vehicle 7 stayed lean, with lambda around 1.1 in urban and rural parts, which resulted in very low CO and HC levels. The additional power required for the highway part was produced using a rich mixture with lambda values decreasing to 0.8. This considerably increased CO emissions. In contrast, vehicle 8's catalytic converter performed poorly in the urban and rural parts, where lambda values fluctuated with large amplitudes around one. Only on the highway part was the mixture mainly stoichiometric, enabling the catalytic converter to efficiently reduce CO and HC.

6. Car measurements used for comparison

The emissions of 17 gasoline-powered passenger cars from the Euro 3 statutory period were measured at EMPA in 2001/02, see Stettler et ali (2004). These in-use vehicles were loaned from private owners in order to create a realistic maintenance situation. They were chosen to reflect the real-life composition of Switzerland's vehicle fleet. Like the two-wheelers, the cars were driven on a chassis dynamometer test bench with a cooling fan in a climate chamber at temperature of 23 °C. In contrast to the measurements on the two-wheelers, a closed connection was used to hitch the tailpipe to the CVS dilution system in accordance with common practice (see Figure 1). Emissions of CO, HC, NO_x and CO_2 were sampled in several driving cycles, one being the CADC, the emissions from which are compared here. Table 3 lists the mean values of the cars' emissions.

Table 3: Mean values and standard deviations of 17 Euro 3 cars' emissions in driving cycle CADC used for comparison.

Tableau 3 : Emissions moyennes et écarts-type de 17 voitures Euro 3 durant cycle de conduite CADC utilisés à titre de comparaison.

	CO [g/km]		HC [g/km]		NO_x [g/km]		CO_2 [g/km]	
	avg.	std.	avg.	std.	avg.	std.	avg.	std.
urban	0.57	0.91	0.017	0.015	0.089	0.069	278.4	53.4
rural	0.85	0.97	0.018	0.016	0.052	0.030	160.4	24.5
highway	3.04	3.00	0.031	0.017	0.065	0.048	192.4	19.4

Emission comparison

The values from Table 2 and 3 were used to compare the powered two-wheelers with the gasoline passenger cars. The two-wheelers met FAV 3 and Euro 1 standards, while the cars complied with Euro 3. Thus vehicles' emission behaviour was the same as that of vehicles on sale in 2001.

1. Comparison of mean unit emissions

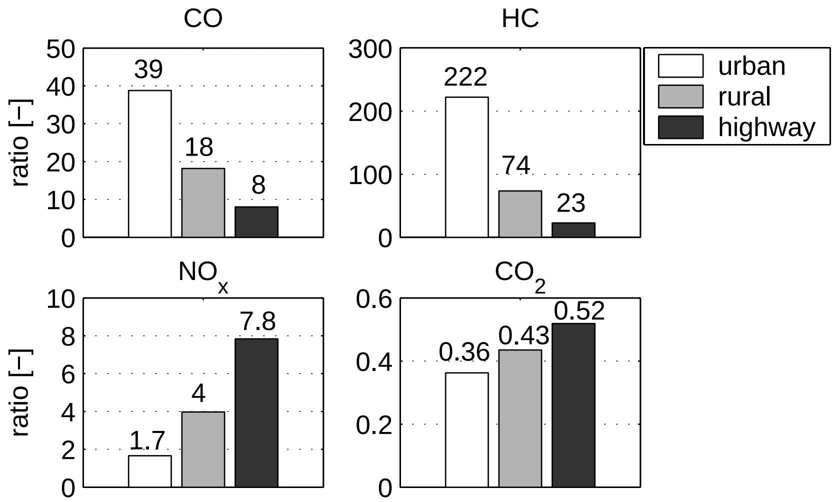
First two-wheelers' mean unit emissions in g/km are compared with those of passenger cars. The ratio of the mean unit emission of all two-wheelers and all cars is calculated for every driving pattern. Because of their low power it was possible to test only seven out of eight two-wheelers in the 'rural' part of the cycle and only 5 in the 'highway' part. However, the two-wheelers were selected so as to be representative of the Swiss fleet, which means that the results should yield realistic proportions.

Figure 4 shows the ratios of mean emissions in g/km from two-wheelers and cars. For CO and HC this ratio is largest at slower speeds, while the opposite is true for NO_x. Frequent acceleration and load changes in urban driving result in enrichment and incomplete combustion. This is assumed to be the main reason for this observation. The emissions ratio is particularly high for HC (factor of 222!) in urban driving, and this can be attributed to two-cycle vehicles and four-cycle vehicles with a very rich mixture. The NO_x emission ratio may appear surprising, as the two-wheelers' generally rich mixture should create very little NO_x. Obviously, there is more thermal NO_x in the two-wheelers' emissions greater than in the catalysed exhaust gas of the passenger cars.

Since two-wheelers are lighter than passenger cars, their fuel consumption and CO_2 emissions are also lower. Nevertheless, the weight-and payload-specific fuel consumption of the two-wheelers is still quite high.

Figure 4: Ratios of mean unit emissions [g/km] of two-wheelers and passenger cars.

Figure4: Rapport entre les émissions moyennes [g/km] des deux-roues et des voitures.



2. Comparison of fleet emissions

In Switzerland, as in other European countries, motorcycles and cars are used on different occasions. Two-wheelers are used primarily are local urban transport and leisure. Their use obviously depends on weather conditions, which is why the average two-wheeler has a lower yearly mileage than a passenger car. In this section, we therefore present two different comparisons: First, the yearly emissions from the average motorcycle are compared to the average car, and the unit emissions from above are therefore multiplied by the average yearly mileages. Second, the emissions from both fleets are compared, thus accounting for the total numbers of vehicles in both groups.

The number of vehicles sold in 2001 is given in INFRAS (2004), along with yearly mileages on urban, rural and highway routes. With regard to the test bench measurements, it was assumed that two-wheelers with engine capacities of less than 50 cc are to be found only in urban regions, while vehicles with engine capacities up to 125 cc are used in urban and rural situations and the rest in all driving patterns. Fleet mileages and average mileage per vehicle are presented in Table 4.

Figure 5 shows the ratio of yearly emissions from the average two-wheeler to yearly emissions from the average car. The ratios are lower in comparison with Figure 4 because of two-wheelers' lower mileages, but they are still quite high. The yearly HC emission of the average two-wheeler in urban traffic is up to 49 times that of the average car.

Table 4: Fleet and vehicle mileage from INFRAS (2004).

Tableau 4: Flotte et kilométrages provenant d' INFRAS (2004).

	two-wheelers			passenger cars		
	FAV3 and Euro-1			Euro-3		
		yearly mileage			yearly mileage	
	sales	fleet [10^6 km]	vehicle [km]	sales	fleet [10^6 km]	vehicle [km]
urban	48'077	57	**1'194**	151'867	821	**5'406**
rural	27'001	44	**1'640**	151'867	856	**5'635**
highway	25'717	31	**1'200**	151'867	839	**5'524**

Finally, to demonstrate the overall impact of emissions at national level, the yearly output of both fleets is compared. Using the mileages and stocks from Table 4, emissions from both fleets are compared in Figure 6. Despite the lower mileage and smaller number of vehicles, the motorcycle fleet produces CO emissions that are higher by a factor of up to 2.7, and HC emissions that are higher by a factor of 16 in urban conditions. It may be supposed that two-wheelers clock up most of their mileage in good weather conditions. As these conditions are also conducive to the formation of tropospheric ozone, emissions of HC and NO_x gain additional significance, see Tsai et ali (2003). It must therefore be borne in mind that the ratios presented here refer to an average value over the year and are probably higher on ozone-critical days.

Figure 5: Ratios of mean yearly emissions [kg/vehicle/year] from two-wheelers and passenger cars.

Figure 5: Rapport entre les émissions moyennes annuelles [kg/véhicule/an] des deux-roues et des voitures.

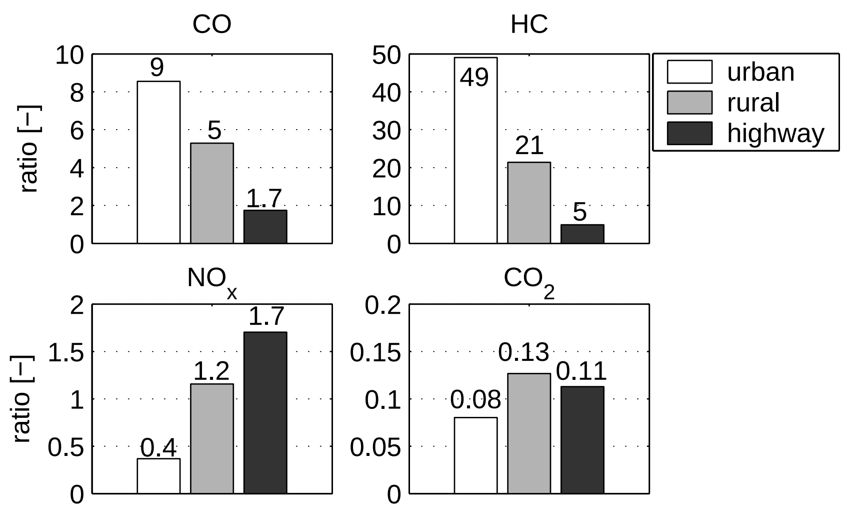
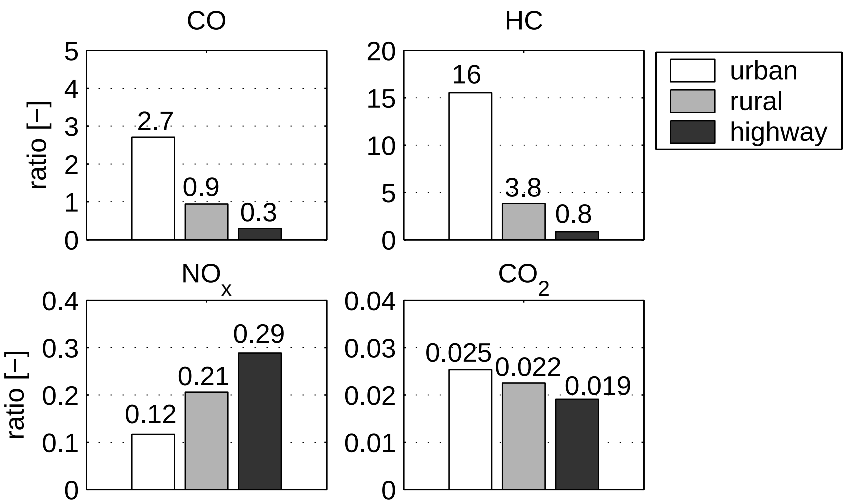


Figure 6: Ratios of fleet emissions [tons/year] from two-wheelers and passenger cars for the Swiss fleet.

Figure 6: Rapport entre les émissions de flotte [tonne/an] des 2-roues et des voitures pour la flotte suisse.



Discussion

Several comparisons show that the powered two-wheelers sold in 2001 produced significantly higher emissions of all pollutants except CO_2 than gasoline-powered passenger cars from the same sales period. Whether in a direct comparison of mean unit emissions (in g/km), mean yearly emissions (in kg/vehicle/year) or fleet emissions (in tons/year), the two-wheelers' HC and CO emissions were all, and often significantly, higher.

The substantially higher HC emissions are mainly caused by two-cycle machines, which emit more HC than motorcycles with four-cycle engines as confirmed in Chan et ali (1995), Tsai et ali (2000) and Chen et ali (2003). However, the use of technologies similar to those employed in cars (three-way catalytic converters in vehicles 7 and 8) does not yield similar results either. It must be assumed that work on implementing the lambda control loop has not been performed with the same care as for cars.

Overall, emissions from motorcycles have become relevant compared to those from modern passenger cars. Even if they account for a comparatively small number of vehicles, motorcycles' impact on traffic emissions cannot be overlooked. Directive 2002/51/EC of the European Parliament and Council is a step in the right direction. With the introduction in 2006 of new emissions limits which are intended to correspond to Euro 3 gasoline cars, and with checking procedures for the correct operation of emission control systems, motorcycle emissions are expected to decrease.

Acknowledgments

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Validation of a hierarchical emission model: statistical analysis of diesel and gasoline EURO 3 automobile emissions from the ARTEMIS data base

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Abstract
A statistical hierarchical emission model was developed to analyse and determine average vehicle emission factors. It was based on the emission data base collected within the EU Fifth Framework Project "Assessment and reliability of transport emission models and inventory systems" (ARTEMIS). This paper illustrates the preliminary results of statistical analysis and model validation of emissions for a representative group of diesel and gasoline EURO 3 passenger cars. The model was validated on independent data sets using several statistical criteria. The emissions predicted by the emission model for the independent data sample were statistically compared with observed emissions and with values calculated by a polynomial. Our results indicate that the proposed approach is effective in the exploratory analysis of emission data and is a potential tool in predicting emissions of new trips.

Key words*: ARTEMIS, average hot emission factors, hierarchical statistical emission model, PLS, validation.*

Introduction

Vehicle emission assessment has assumed institutional importance worldwide due to the major environmental impact of transportation, especially in metropolitan areas. Besides pollutants which have a negative effect on human health, attention has focused on CO2 due to its effect on climate, as stated by the Kyoto protocol and enshrined in subsequent legislation. The models currently used to evaluate emissions include that of Copert, Ntziachristos and Samaras (2000), and the German Handbook, INFRAS (1999), developed in Europe, as well as Mobile, EPA (2003), and EMFAC, EMFAC (2000), developed in the USA. They predict emissions with different approaches, but are all based solely on mean trip speed as the only kinematic input parameter, and are mostly intended for inventory use of emissions produced in large areas over a long time span. Except the German Handbook model which qualitatively characterizes driving behaviour by considering different traffic situations together with mean speed, they cannot differentiate emissions of two trips with same mean speed but with different distribution of modal events like acceleration, deceleration and cruise, although the effect of acceleration on emissions is well known (Joumard et al., 1995).

The EU Fifth Framework Project ARTEMIS (Hickman et al., 2003) has collected a comprehensive emission data base for a broad range of automobiles with different technology, powering fuel, homologation and displacement class, tested on a consistent scenario of real driving cycles representing vehicle operating conditions on urban and extra-urban roads and in different (from congested to rush) traffic situations.

Within this context, a statistical modelling approach has been proposed (Rapone 2005) for the exploratory analysis of emissions in the data base and for the prediction of emissions produced by vehicles in micro trips. The proposed emission model seeks to improve the sensitivity of mean speed emission models, including terms for acceleration in the regression equation, and allows for the main findings of micro emission models (Barth et al., 2002; Lee et al., 2001) as well as the kinematics characterization utilized to analyze and classify real driving cycles in the ARTEMIS project (André, 2004). A minimal set of explicative variables was determined from the vehicle dynamic equation, and from the frequency distribution of acceleration events at different speeds. Variables were divided into two conceptually meaningful blocks, and then a hierarchical multi-block PLS method was applied to calculate the hierarchical emission model (HEM).

In developing the HEM, data from measurements performed with ARTEMIS CADC (Common Artemis Driving Cycles) under testing conditions which can be assumed as "reference hot conditions" were analysed. Ten case studies were conducted, dividing vehicles into classes considering powering fuel (diesel oil and gasoline), vehicle homologation (EURO 1, 2, 3 and 4) and engine displacement. The results have been widely reported (Rapone et al., 2005). The models calculated proved effective at explaining the effect of driving cycles and individual vehicles on emission factors. Prediction of emission factors for the test cases was, however, affected by the fact that CADC driving cycles did not fully represent the range of real vehicle operating conditions.

On this basis, to improve the HEM a validation effort is in progress to analyse the whole ARTEMIS data base, considering emissions for a wider scenario of driving cycles, including tests performed in other European national projects. The rest of the paper presents the model structure and validation results for a diesel and a gasoline case study.

Validation Methodology

1. Emission Model

The response Y is the unit emission (expressed in g/km) of CO, HC, NOx and CO2 measured in a driving cycle (DC). Because emissions are correlated, a multivariate regression method was adopted, based on a multivariate response **Y** (whose components are CO, CO2, HC and NOx). A log transform of Y is used to account for non-normality. Two sets of explicative variables were assumed as Y-predictors. The first block is defined from the dynamic vehicle equation, given that part of the emission variation can be explained by the variation in exhaust mass, which in turn is a function of: mean running speed (MV), mean square speed (MV2), mean cube speed (MV3), running time (T_RUNNING), mean product (a(t)•v(t)) when v(t)>0 and a(t)>0 (M_VA_POS). Moreover, idling time (T_IDLE) considers emission production during idling time, and the reciprocal of driving cycle length (INVDIST) takes into account that unit emission mass (calculated as emission mass in a test divided by DC distance and expressed in g/km) is to be predicted. The second block of X-variables is determined by the joint empirical distribution of second-by-second instantaneous speed/acceleration {v(t),a(t)} of the driving cycle. X-variables are obtained by the (cumulated) frequency of {v(t),a(t)} in each of 42 cells obtained by the intersection of six speed classes and seven acceleration classes. Since explicative variables are numerous and mutually correlated, the Partial Least Squares (PLS) regression method based on principal components was assumed to calculate models (Wold et al., 2001). Finally, a Multivariate Hierarchical Multi-block PLS method is adopted to consider both the contributions of the two blocks of explicative variables in one model (Westerhuis et al., 1998). Following this approach, a set (t_1, t_2,…, t_k) of principal components (X-scores) is estimated separately for each block of variables, fitting a PLS base model to each block. The super-block regression model (called top-model) is then constructed by applying the PLS regression of Y-variables on super scores made by summing the scores of the two base models. This approach enables one to analyse and predict emissions utilizing regression models based on the two blocks of X-variables (either separately or combined) according to the best fit for different data prediction sets. Details of the model structure and statistical methods applied are reported in Rapone (2005). In this paper we present top model results, performing the regression of emissions on the combined two blocks of explicative variables.

To account for the non-normality of the emission data a logarithmic transformation was used. However, when log predictions are back transformed to untransformed linear scale by simply making the exponential of predicted values, the expected value of Y tends to the median and the model may tend to underestimate fleet emissions. To obtain correct values a retransformation formula correcting the bias was adopted both for mean (Duan, 1983) and confidence intervals (Land, 1974).

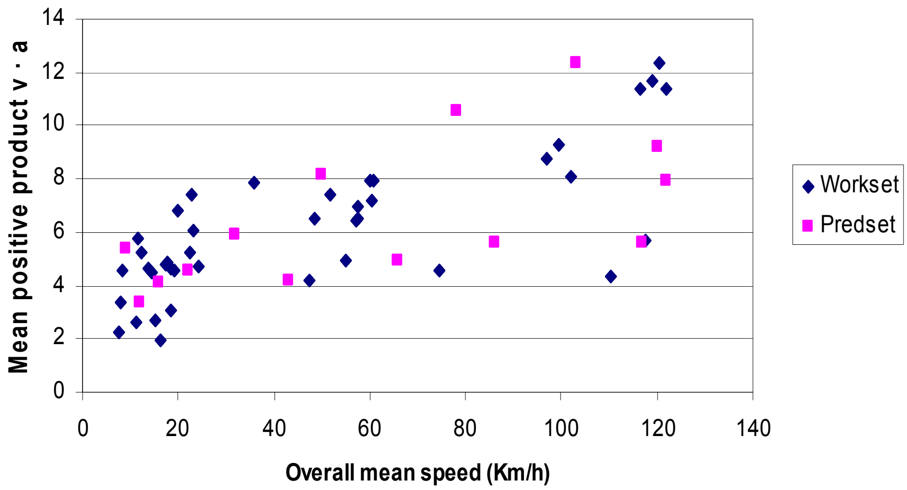
2. Vehicle and driving cycles

Making reference to ARTEMIS data base version 20050114, the vehicle fleet was divided into ten sets according to homologation (EURO1 to EURO 4) and displacement. Three displacement classes were assumed (1200-1400 cc, 1400-2000, over 2000) when data were available; otherwise data were grouped to obtain a consistent sample. Seven consistent samples were identified for gasoline passenger cars, three for diesel passenger cars.

For each case the model was validated by dividing the whole set of measurements into two data sets: a working set used to calculate the regression models (WORKSET), and an independent prediction set (PREDSET) used to predict emissions by means of the regression model coefficients calculated by the model fitted to WORKSET. Data of the working set are relative to bag values of complete ARTEMIS reference driving cycles (CADC) and to emission measurements performed with different driving cycles within other projects by TUG, TRL, OSCAR, CNR IM and MODEM (Andrè, 2004). Data of PREDSET are measurements relative to ARTEMIS sub-cycles, which are parts of urban, rural and motorway CADC. This choice was motivated by the fact that driving cycles of the working set cover a considerable multidimensional variation of driving behaviour, while ARTEMIS sub-cycles represent specific traffic/road situations detectable in a micro trip of a passenger car in real use. In figure 1 overall mean speed and m_va_pos values are reported for the driving cycles of WORKSET, as well as for ARTEMIS sub-cycles of PREDSET.

Figure 1: M_va_pos vs. Mean overall speed - WORKSET and PREDSET driving cycles.

Figure 1: M_va_pos vs. vitesse globale moyenne – cycles de conduite WORKSET et PREDSET



3. Validation statistics

Different statistical measures of effectiveness may be used in model validation (Fomunung, 2000; Schulz, 2000). As a measure of the model's closeness to observed data, the mean prediction error (MPE), or "model bias", was adopted. It is given by the mean of the difference between modelled values and observed values, and assesses over/underprediction of the model by the sign of difference. The mean absolute prediction error is estimated by the root mean square error of prediction (RMSEP), as follows: $RMSEP = \sqrt{\frac{1}{N}\sum_{i=1}^{N}(\hat{y}_i - y_i)^2}$ where y_i is the i-th observation, $\hat{y}_i$ is the corresponding i-th value predicted by the model and N is the number of observations. To take account of sample-to-sample variation in predicted emissions, 95% confidence intervals of emissions predicted by the model for each driving cycle were calculated on the prediction set.

Results

This section describes some results of the HEM validation carried out with two case studies concerning EURO 3 1400- 2000 cc diesel and gasoline passenger cars. Prediction ability of the HEM PLS top model is compared with a second order polynomial emission model calculated on the WORKSET. For each case, preliminary statistical analysis was carried out to identify the presence of outliers. First, any cars to be considered as abnormally high emitters over the entire set of driving cycles are identified by estimating the percentage car effect on emissions (Rapone et al., 2005). Outlier analysis is then performed on individual emission measurements for each driving cycle. Statistical methods based on data distribution and robust scale measures were applied. Finally, after the model fit, observations with a residual (Y predicted – Y observed) in the normal probability plot greater than four standard deviation, were considered as outliers. Abnormal vehicle emissions and identified outliers were dropped from the data set.

The matrix of observed data is sparse for both cases. The number of vehicles is unbalanced with respect to driving cycles, while the number of measurement repetitions is highly unbalanced for different combinations of vehicles and driving cycles.

Table 1: Diesel working set - HEM Model goodness of fit

Tableau 1: Ensemble de réalisation pour gazole: qualité de l'ajustement du Modèle HEM

	WORKSET			WORKSET+PREDSET		
	NObs	R2	RMSEE	NObs	R2	RMSEE
CO	270	0.410	1.1163	484	0.408	1.0485
CO2	299	0.707	0.1415	548	0.796	0.1420
HC	294	0.389	0.8320	542	0.454	0.7551
NOx	299	0.3818	0.2762	548	0.498	0.3121

Table 2: Diesel prediction set - Measures of model predictability

Tableau 2: Ensemble de prévision pour gazole – mesures de prévisibilité du modèle

PREDSET	NObs	MEAN OBS	HEM_MPE	HEM_RMSEP	POL_MPE	POL_RMSEP
CO	214	0.0657	-0.031495	0.152586	-0.0033	0.1443
CO2	249	188.86	-9.451825	53.04814	-0.11	36.79
HC	248	0.0225	0.075984	0.077852	0.0044	0.0160
NOx_	249	0.8996	-0.112766	0.382354	-0.0600	0.3429

1. Diesel passenger cars

Data for 21 cars, from about 300 observations for WORKSET and 200 observations for PREDSET, were collected from the data base for EURO 3 1400-2000 cc diesel cars.

In table 1 the goodness of fit of the HEM top model to WORKSET data is shown for different pollutants. Number of observations (NObs), the explained fraction of variance estimated by the determination coefficient R^2, and the standard prediction error RMSEE are reported. Model fit is good only for CO2, while for other emissions R^2 is less than sufficient (R^2<.5). This is due to the critical nature of CO, HC and NOx emission data more than to model effectiveness: the component of variance to be attributed to driving cycles is less than the summation of components due to experimental error and vehicle effect (for example for NOx it results in 38% of total variance contributed by the driving cycle, 56% by vehicle effect and 6% by error). Recalculating the model on pooled PREDSET and WORKSET data almost gives a sufficient fit both for HC and NOx.

The effectiveness of the HEM model in predicting emissions of PREDSET driving cycles, compared with predictions of a second order polynomial model calculated on WORKSET data, is shown in table 2. CO2 is underpredicted (negative MPE values) by the HEM, confirmed by high values of CO2 RMSEP. The HEM was less effective for CO2 and NOx than the polynomial. Observed and predicted emissions are shown in figures 2 and 3, where point and .95 confidence intervals of predictions are reported for CO2 and NOx, respectively, versus overall mean speed (VMOA). Kinematic characteristics of several Artemis sub-cycles are very different from WORKSET driving cycles and corresponding HEM predictions are affected by a considerable error.

Figure 2: Observed and predicted CO2 versus overall mean speed for Diesel PREDSET

Figure 2: CO_2 observé et prévu contre vitesse globale moyenne pour PREDSET gazole

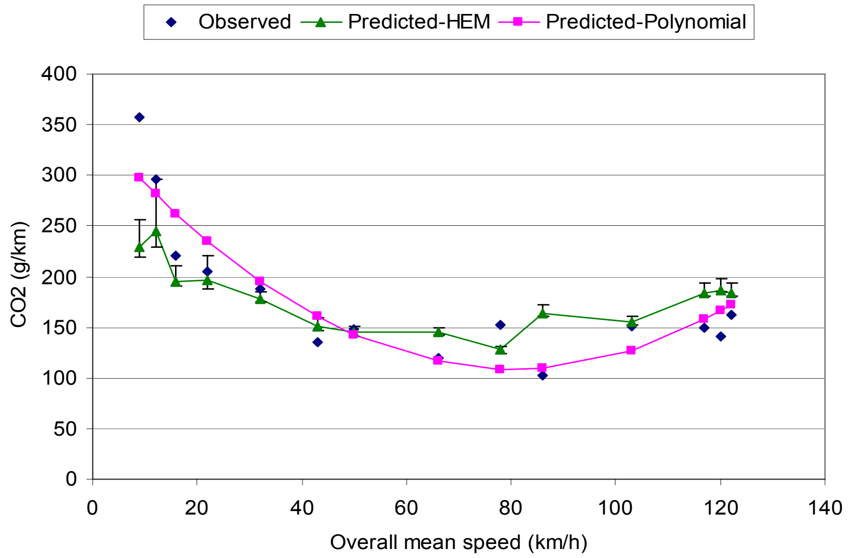
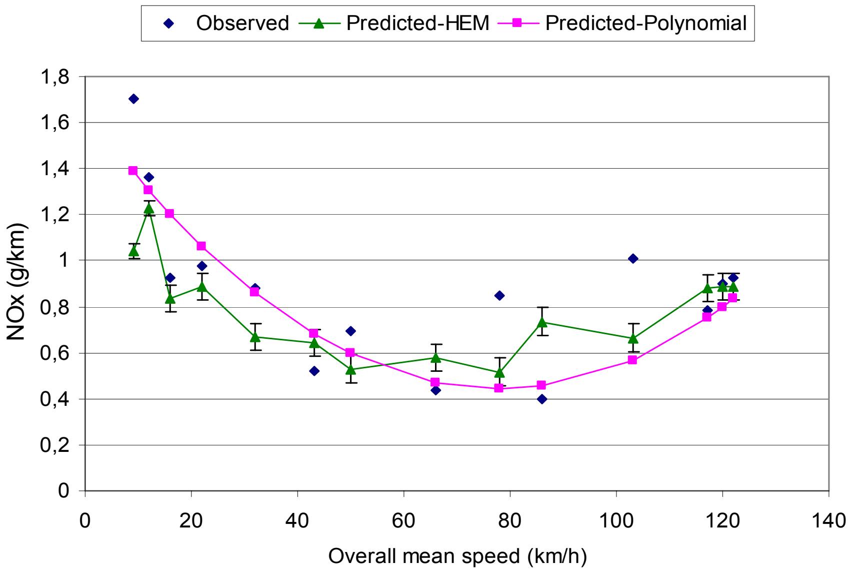


Figure 3: Observed and predicted NOx versus overall mean speed for Diesel PREDSET

Figure 3: NO_x observé et prévu contre vitesse globale moyenne pour PREDSET gazole



2. Gasoline passenger cars

Data from 16 cars, amounting to about 90 observations for WORKSET and 240-270 observations for PREDSET, were collected from the data base for EURO 3 1400- 2000 cc gasoline cars.

The goodness of fit of the HEM top model to the working set of data was good for CO2, sufficient for CO, poor for HC and for NOx (R^2 values in table 3). As a reference, the variance component contributed by driving cycles, expressed in percentage of total observed data variance (computed by analysis of variance) is 85.8% for CO2, 64.7% for CO, 23.4 % for HC and 45.7 % for NOx.

Table 3: Gasoline working set - HEM Model goodness of fit

Tableau 3: Ensemble de réalisation pour essence: qualité de l'ajustement du Modèle HEM

	WORKSET			WORKSET+PREDSET		
	NObs	R^2	RMSEE	NObs	R^2	RMSEEP
CO	84	0.5868	1.3469	332	0.3263	1.4111
CO2	89	0.8118	0.1167	355	0.8427	0.1391
HC	85	0.2239	0.94205	322	0.1711	1.0764
NOx	89	0.0953	0.7254	355	0.2560	0.9371

Table 4: Gasoline prediction set - Measures of model predictability.

Tableau 4: Ensemble de prévision pour essence – mesures de prévisibilité du modèle

PREDSET	NObs	MEAN OBS	HEM_MPE	HEM_RMSEP	POL_MPE	POL_RMSEP
CO	248	0.1452	-0.0906	0.1849	0.0289	0.3707
CO2	266	231.822	13.62	55.11	0.072	38.82
HC	237	0.0139	-0.0072	0.01759	0.0002	0.0148
NOx	266	0.0775	0.0013	0.0937	0.0131	0.1012

The ability of the HEM model to predict emissions of driving cycles of PREDSET, compared with results from a second order polynomial model calculated on WORKSET data, is shown in table 4. An overprediction results for CO2 (positive MPE values), confirmed by high RMSEP values. For other pollutants HEM predictions were poor: mean and RMSEP are the same order of magnitude, polynomial performance is comparable (worse for CO). Observed and predicted emissions are shown in figures 4 and 5, where point and .95 confidence interval predictions are reported for CO and HC respectively. Large confidence intervals for the predicted emissions are caused by the high values of standard error of Y prediction. HEM explains multidimensional emission variability not explained by the polynomial. However, because X-variables of some PREDSET driving cycles are outside the multidimensional range of the working set of data, corresponding observed emissions of CO and HC lie outside the trend predicted by the HEM (and by Polynomial), based on WORKSET.

Figure 4: Observed and predicted CO versus overall mean speed for Gasoline PREDSET

Figure 4: CO_2 observé et prévu contre vitesse globale moyenne pour PREDSET essence

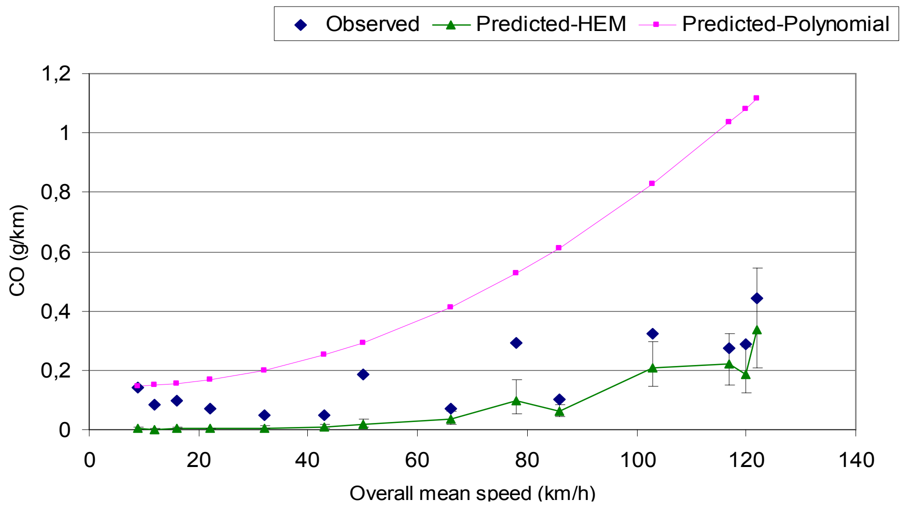
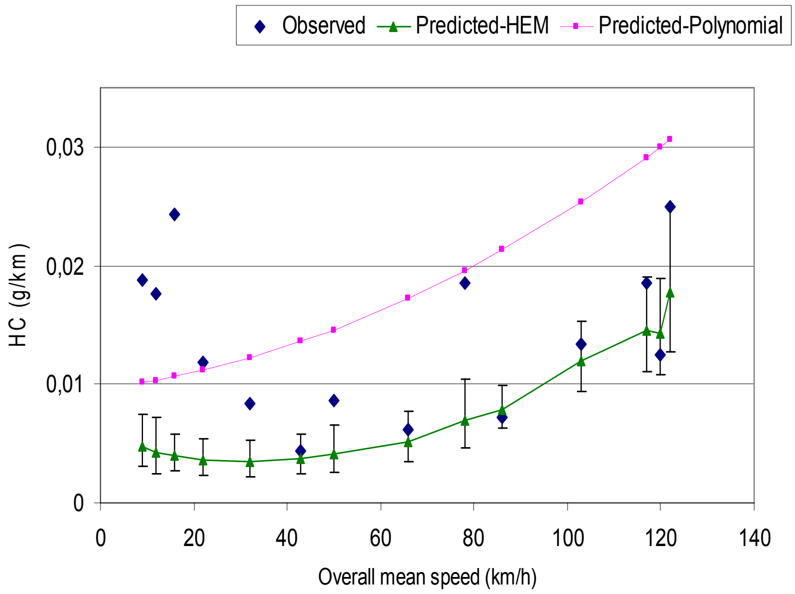


Figure 5: Observed and predicted HC for versus overall mean speed for Gasoline PREDSET

Figure 5: HC observé et prévu contre vitesse globale moyenne pour PREDSET essence



Conclusion

Preliminary results summarized in this paper showed the limits and difficulty of statistical methodology using existing data to predict emissions. The nature of emission data, generally characterized by a greater effect of vehicles (and of vehicle-driving cycle interaction) than driving cycles upon CO, HC and NOx data variance, makes it difficult to fit any model to data. Some ARTEMIS sub-cycles, being parts of a driving cycle, showed a significantly different kinematics from WORKSET driving cycles, thereby making prediction critical. As a consequence, for these data sets the HEM model proved to be effective in exploratory data analysis but only a potential tool for prediction: it was unable to explain trends not significantly revealed in the analysed data. These preliminary results suggest improvements be made to the statistical approach, giving greater consideration to information regarding vehicles and critical driving cycles. A possible way forward would be to consider also an interval estimation of emission factors, and to give for each sample a secondary emission factor relative to higher emitters or give an emission factor for a central fraction of vehicles determined by their percentage effect on the prediction.

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Interpretation of multi-metric particulate matter data monitored near a busy London highway

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Abstract

Continuous measurements of several particulate matter metrics at the Marylebone Road supersite are investigated in this paper and compared to those measured at urban background sites in order to define local contributions due to the traffic. Diurnal and weekly variations are also examined and compared to traffic data. Particle mass (both TEOM and gravimetric), particle number size distributions, particulate elemental and organic carbon, particulate sulphate, nitrate and chloride were measured at Marylebone Road between 2002 and 2004, as well as a number of relevant gases. Local vehicle emissions significantly contributed to most particulate components. Only the inorganic ions showed a much larger contribution from outside the street canyon. Local vehicle emissions contributed considerably to measured elemental carbon and particle numbers (dominated by small particles). Results show that local emissions of most particulate components at Marylebone Road could be mostly attributed to the heavy-duty vehicles that represented on average 10% of the traffic. Only the particulate organic carbon was more closely associated with the light duty traffic in agreement with previous studies.

Keywords*: on-road emissions, heavy-duty traffic, particle mass, particle numbers*

Introduction

Many studies have linked adverse health effects with exposure to airborne particulate matter but it is still unclear which particle metric or component has the most significant health effects. Particle number or particle surface may be a more relevant metric for human health effects than the particle mass but on the other hand, many particulate matter chemical constituents have been highlighted as toxic (e.g. Donaldson et al., 1998 ; Maynard and Maynard, 2002). Particulate matter concentrations and especially particle number concentrations are very high in the vicinity of roads and on-road vehicles are known to be a large contributor to particulate matter in the urban environment. In particular, in London, traffic contributed 80% of the emissions of particles in 2001 (Dore et al., 2003).

Because the behaviour of the most volatile compounds emitted by vehicles depends on dilution conditions (Kittelson, 1998; Shi and Harrison, 1999; Shi et al., 2000) and the contribution of non-exhaust emissions is large (Abu-Allaban et al., 2003; Charron and Harrison, 2005), roadside measurements complement dynanometer studies for the assessment of particulate matter from vehicle emissions.

This paper investigates particulate emissions from on-road vehicles using multiple continuous measurements and traffic data at the Marylebone Road supersite located in Central London.

Methods

The Marylebone Road supersite is located on the kerbside of a major arterial route within the City of Westminster in London and a part of the inner London ring road. The surrounding buildings form an asymmetric street canyon. Traffic flows of over 80,000 vehicles per day pass the site on 6 lanes with frequent congestion. Lanes 1 and 6 are bus and taxi lanes. The instruments are in a cabin and sampling inlets are less than 5 metres from the road.

Two Tapered Element Oscillating Microbalances (TEOM Model 1400AB Rupprecht and Patashnick R&P) are used to monitor PM_{10} and $PM_{2.5}$ particle mass concentrations. Air at 16.67 L min^{-1} is sampled through a R&P PM_{10} impactor inlet and for $PM_{2.5}$ measurements, the TEOM is fitted with a Sharp-Cut Cyclone inlet. The inlet of the TEOMs is heated to 50°C in order to eliminate the effect of condensation or evaporation of particle water. This heating of the aerosol stream induces losses of semi-volatile species and consequently the TEOM instrument gives lower readings than filter-based methods (e.g Allen et al., 1997, Ayers et al., 1999; Charron et al., 2004).

Three Partisol instruments Model 2025 from R&P are also installed at Marylebone Road in order to measure simultaneously on a daily basis PM_{10} and $PM_{2.5}$ mass and inorganic ions in PM_{10}. These instruments use the same size-selective inlet designs as the ones used on the TEOM with the same flow rate of 16.67 L min^{-1}. Temperature and pressure sensors are provided in the instrument with internal regulators to maintain the temperature within $\pm$ 5°C of ambient.

Quartz fibre filters (Whatman QMA 47 mm diameter filters, 0.6 µm pore size) have been used for the collection of particulate matter for mass measurements. Blank and dust-loaded filters are handled according to the same protocols in accordance with the requirements of prEN 12341. Filters are equilibrated for 48 hours within an air-conditioned weighing room at a temperature of 20°C and a relative humidity of 50% before weighing on a balance with a resolution of 10 µg.

1 µm size PTFE filters have been used for inorganic ion measurements. Filters are stored at 4°C before water extraction (pre-wetting using propan-2-ol) and analyses for chloride, nitrate, sulphate using ion chromatography. Extraction and measurement methods are fully described in Abdalmogith and Harrison (2005).

Particulate elemental (EC) and organic (OC) carbon are measured on a semicontinuous basis using R&P Series 5400 Ambient Carbon Particulate Monitors. Particulate matter is collected on an impactor. The amount of OC is determined by oxidising the particulate matter at a temperature of 340°C (CO_2 is measured); and the EC by oxidising the resultant particulate matter at 750°C. Sampling is carried out over 3-h periods. Jones and Harrison (2005) reviewed different studies comparing R&P 5400 monitors with other methods; the EC may be underestimated because of the low collection efficiency of particles with aerodynamic diameters below 140 nm and the OC may be underestimated due to the heating of the impactor at 50°C.

Particle counts and size distributions are determined using a Scanning Mobility Particle Sizer (SMPS). The SMPS system comprises a Model 3071A Electrostatic Classifier (EC) which separates the particles into known size fractions, and a model 3022A Condensation Particle Counter (CPC) which measures their concentration. The configuration of the EC allows the measurement of particles in the range 11-450 nm diameter. Particles smaller than 50 nm are underestimated by the SMPS because of the efficiency of the 3022A CPC and losses by Brownian diffusion and diffusional broadening of small particle trajectories by the 3071A EC.

The site is also equipped with continuous monitors recording gases including CO, SO_2, NO_x and VOCs and with traffic counters. All six lanes provide vehicle counts for 6 vehicle classes using a loop monitoring system (detailed in Charron and Harrison, 2005). From the 6 classes available, we have formed two classes: the light duty class (cars and motorcycles) and the heavy-duty class (rigid and articulated lorries, buses, coaches). Traffic data are available from January 2002 to May 2004 (after this date, problems with the loop of lane 6). From 2002 to 2004 about 78% of the light vehicles on urban UK roads were petrol vehicles (estimated from the National Atmospheric Emissions Inventory, www.naei.org.uk). Virtually all heavy-duty vehicles were diesel-powered. Marylebone Road had an average traffic flow of 3600 vehicles per hour (ranging from 310 to 4700 vehicles per hour) with a mix of light duty vehicles (on average 90%) and heavy-duty vehicles (on average 10%).

Data included in the present study were measured from January 2002 to December 2004 except SMPS data (available until March 2003 because of instrument malfunctions). Data measured at Marylebone Road are compared with data measured at two London urban background sites where the same instruments are deployed. Bloomsbury is located in Russell Square Gardens in the centre of London at about 2 km to the east of Marylebone Road (SMPS and TEOM $PM_{2.5}$ and PM_{10} data) and London North Kensington is located in a residential area about 4 km to the west of Marylebone Road (gravimetric $PM_{2.5}$ and PM_{10}, particulate sulphate, nitrate, chloride, EC and OC).

Results

Average concentrations measured at Marylebone Road between 2002 and 2004 are presented in Table 1; as well as the local increments calculated by difference between Marylebone Road and urban background data. As expected, average PM_{10} and $PM_{2.5}$ TEOM concentrations are smaller than gravimetric concentrations. Average differences are small; but differences could be larger for individual episodes. Perhaps more surprising, absolute local increments of PM_{10} and PM_{coarse}

(defined as $PM_{2.5-10}$) concentrations are similar for gravimetric and TEOM concentrations and the difference between the local increment of gravimetric $PM_{2.5}$ and the local increment of TEOM $PM_{2.5}$ is small (Table 1). This suggests that volatile material at 50°C contributed little to particle mass emitted by vehicles and the aerosol volatile at 50°C would mainly be in the fine fraction. Accumulation mode and coarse mode particles emitted by vehicles (that represent most of the particle mass) would mostly be constituted by organic materials involatile at 50°C, soot aggregates and metals.

The nitrate and sulphate are major chemical species at Marylebone Road. Sulphate and nitrate are assumed to exist as ammonium sulphate and ammonium nitrate with their bound water estimated according to the method of Harrison et al. (2003). Note that nitrate and bound water would not be measured by the TEOM. On average, ammonium sulphate and ammonium nitrate comprised 29% of the gravimetric PM_{10} mass measured at Marylebone Road; with possible larger contributions during episodes of high PM_{10} concentrations (Charron and Harrison, 2006). However, the increments in sulphate and nitrate due to the local traffic (calculated by difference between concentrations measured at Marylebone Road and concentrations measured at North Kensington) are small. Local increments of 0.34 µg m^{-3} is found for the sulphate (0.61 µg m^{-3} converted into ammonium sulphate with bound water) and of 0.39 µg m^{-3} for the nitrate (0.66 µg m^{-3} for the ammonium nitrate with bound water). This means that local increments in sulphate and nitrate represented on average 14% and 15% of the total sulphate and nitrate concentrations respectively. Not surprisingly, the variability of sulphate and nitrate at Marylebone Road is mostly explained by the advection of polluted air masses from regions of high precursor emissions. Ammonium sulphate and ammonium nitrate represented on average 4% and 5% of the local increment in gravimetric PM_{10} (2.3% and 3% for sulphate and nitrate). The result for the sulphate is in good agreement with estimates from tunnel studies (Allen et al; 2001; Gillies et al., 2001; Geller et al., 2005). Allen et al; (2001) and Geller et al. (2005) have shown that the heavy-duty vehicles dominated primary sulphate vehicle emissions, although the current use of ultra-low sulphur diesel should minimise emissions of primary sulphate. On the contrary, the same studies found contributions for the nitrate smaller than ours and Kuhn et al. (2005) found for the nitrate little difference between freeway and background sites. The local increment in chloride is very small, on average 0.15 µg m^{-3} or 1% of the local increment in gravimetric $PM_{10.}$ This small increment might possibly be associated with resuspended particulate matter. Gillies et al. (2001) reported an emission factor for the chloride in PM_{10} that was a factor 4 smaller than the emission factor for the sulphate. The average ratio SO_4^{2-} to Cl^- increments at Marylebone Road is smaller (around 2).

Average particulate EC and OC concentrations were larger than sulphate and nitrate concentrations (Table 1). Particulate EC and OC showed larger local contributions than those of the inorganic ions (Table 1). The local increments in EC and OC were respectively 74% and 45% of the total EC and OC measured at Marylebone Road. In agreement with Jones and Harrison (2005), a larger contribution of sources outside the canyon is found for the OC. The secondary organic aerosol contribution to the total OC and may be estimated using the linear relationship between OC and EC at Marylebone Rd and assuming that the EC only comes from primary sources. On average 45% of the OC measured at Marylebone

Road is estimated to be secondary organic carbon, suggesting a small contribution of primary OC to background OC.

On the other hand, the EC is predominantly locally emitted in agreement with the significant contribution of traffic, mostly diesel vehicles. However the relative contributions of EC and OC to local emissions of PM_{10} were on average 18% and 15% (21% if the OC is converted into organic mass using the 1.4 factor, see Harrison et al., 2003). These estimations are smaller than those from tunnel studies (Gillies et al., 2001; Geller et al., 2005). The low value for the EC is a possible result of most of EC from vehicle emissions being in the fraction below 180 nm (Geller et al., 2005; Kuhn et al., 2005) which is poorly measured by the R&P instrument. Particulate ammonium nitrate, ammonium sulphate, organic mass and elemental carbon on average accounted for 54% of the PM_{10} increment at Marylebone Road.

High particle numbers were measured at Marylebone Road and the modes in the particle number size distributions mostly ranged from 20 nm to 35 nm, which is in good agreement with other roadside/kerbside studies (e.g. Kuhn et al., 2005; Wåhlin et al., 2001; Wehner et al., 2002). For the calculation of local increments, the SMPS size spectra are integrated over the following size intervals : 11-450 nm, 11-30 nm, 30-100 nm and 100-450 nm. The local increment in numbers of the whole SMPS size spectrum is estimated to be on average 29,748 cm^{-3} or 71% of particle numbers. However, large differences are found for the different particle size ranges: the relative local increments of 11-30 nm 30-100 nm and 100-450 nm were respectively on average 84%, 62% and 51%. The smaller the particles, more local they are. Traffic is a strong emitter of very small particles that have a short lifetime (e.g. Zhang et al., 2004) and results indicate that the public exposure to small particles is much greater near roadways. The significant contribution of background sources to accumulation mode particles (100-450 nm) is consistent with the that of $PM_{2.5}$.

Table 1: Average concentrations measured at Marylebone Road between 2002 and 2004 and local increments due to traffic calculated by difference with background data. P0.25 and P0.75 are the 25 and 75 percentiles.

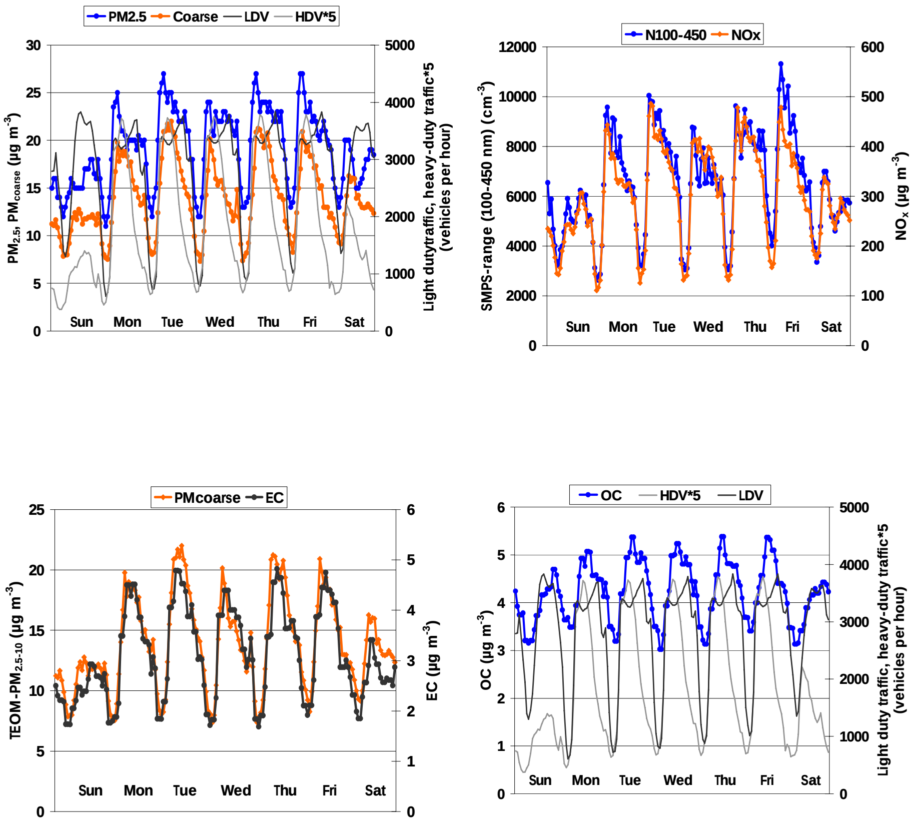
Tableau 1 : Concentrations moyennes mesurées à Marylebone Road entre 2002 et 2004 et contributions locales générées par le trafic calculées par soustraction des concentrations urbaines de fond. P0.25 and P0.75 sont les percentiles 0,25 et 0,75.

		PM_{10} grav.	$PM_{2.5}$ grav.	PM_{coarse} grav.	PM_{10} TEOM	$PM_{2.5}$ TEOM	PM_{coarse} TEOM	OC	EC	SO_4^{2-}	NO_3^-	Cl^-	N 11-450 nm	N 11-30 nm	N 30-100 nm	N 100-450 nm	NO_x
	Units	µg m^{-3}	µg m^{-3}	µg m^{-3}	µg m^{-3}	µg m^{-3}	µg m^{-3}	µg m^{-3}	µg m^{-3}	µg m^{-3}	µg m^{-3}	µg m^{-3}	cm^{-3}	cm^{-3}	cm^{-3}	cm^{-3}	µg m^{-3}
Conc. measured	Median	35.0	22.0	12.0	34.4	19.3	14.3	4.3	3.2	2.6	2.7	0.81	46,430	20,476	19,087	5,796	300
	P0.25	27.4	14.4	7.5	27.0	14.2	10.0	3.4	2.1	1.7	1.4	0.21	27,058	11,540	11,164	3,422	198-407
	P0.75	44.6	30.0	18.5	41.6	24.6	18.2	5.3	4.5	4.1	4.7	1.91	70,089	31,769	29,745	8,852	
Local increments	median	13.1	7.9	6.4	13.1	7.0	6.5	2.0	2.4	0.34	0.39	0.15	29,748	15,541	10,584	2,682	207
	P0.25	7.3	4.0	3.0	6.2	3.0	3.0	1.2	1.4	0.17	0.19	0.05	10,354	6,614	2,828	587	93
	P0.75	18.7	12.5	13.1	22.3	12.0	11.2	2.8	4.0	0.52	0.81	0.32	53,678	7,049	20,881	5,529	359
	% to Total	36%	33%	50%	40%	38%	49%	45%	74%	14%	15%	18%	71%	84%	62%	51%	

Median diurnal and weekly variations have also been examined (Figure 1).

Figure 1: Median weekly cycles for $PM_{2.5}$, PM_{coarse}, particle numbers from 100 to 450 nm, NO_x, EC, OC measured at Marylebone Road and for the vehicle counts.

Figure 1 : Cycles hebdomadaires médians des concentrations massiques des fractions de $PM_{2.5}$, PM_{coarse}, du nombre de particules de tailles 100 à 450 nm, des concentrations en NO_x, carbone élémentaire et carbone organique particulaires et des comptages de véhicules.



Median weekly cycles are used to smooth the variability of concentrations due to meteorology and dispersion processes. $PM_{2.5}$, PM_{coarse}, particulate elemental and organic carbon and particle numbers displayed strong diurnal and weekly variations. The weekly median cycles of gases known to be from vehicle emissions and the weekly averaged heavy-duty and light duty traffic cycles have also been examined. Low values during the night corresponded to the low traffic densities. A clear workday to weekend difference is also seen for all particulate concentrations. The light duty traffic (motorcycles, cars and light good vehicles) reach maximum density on Sunday afternoon and on workdays from 18:00 to 20:00 during the evening commuting time. On the contrary, the heavy-duty traffic shows a strong workday to weekend difference with maximum density during the midday period (12:00-13:00) of the workdays. The workday to weekend difference for all particulate components is clearly related to the heavy-duty traffic. The different particulate components could be sub-divided into sets with similar patterns. Associations are confirmed by a hierarchical cluster analysis done on median weekly cycles (Figure 2). The gases NO_x, CO, benzene and SO_2 were also included in the cluster analysis.

Particle numbers from 30 to 450 nm, $PM_{2.5}$ and NO_x concentrations showed similar weekly cycles. Patterns are influenced by heavy-duty and light duty traffic volumes while our previous study (Charron and Harrison, 2005) showed that the smaller volume of heavy-duty traffic (<10% from 1998 to 2000) contributed on average 62% of PM_{10} exhaust emissions (that are mostly in the fine fraction). The increment in $PM_{2.5}$ was mostly associated with exhaust and brake wear emissions; while, a large part of the local increment in PM_{coarse} concentrations was inferred to arise from resuspended road dust emissions. The SO_2 is clustered with this group; while its median weekly cycle is closer to those of PM_{coarse} and EC (not shown).

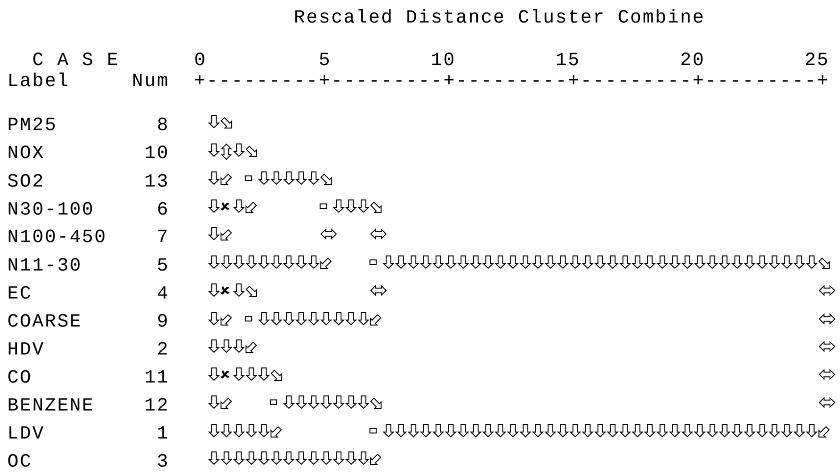
A second set is clearly composed of coarse particulate matter and elemental carbon that closely associate with the heavy-duty traffic. This excellent agreement between PM_{coarse}, elemental carbon and the heavy-duty traffic suggests that the contribution of the heavy-duty traffic is much stronger for these particulate components than for $PM_{2.5}$ and numbers of particles larger than 30 nm. The particulate elemental carbon is a known tracer of diesel vehicles. Heavy-duty vehicles at Marylebone Road are virtually all diesel vehicles and have larger emission factors than light vehicles. Because a large part of local PM_{coarse} emissions are associated with non-exhaust emissions, the results indicate that the heavy-duty traffic is a much larger contributor to non-exhaust emissions than the light duty traffic.

The CO and benzene are clustered with the light duty traffic in agreement with the large proportion of petrol cars in the light vehicle fleet. Similarly, Beauchamp et al. (2004) have not found any relationship between aromatic hydrocarbons, CO concentrations and the heavy-duty traffic; while, they found that NO_x and PM_{10} concentrations could be largely attributed to the heavy-duty traffic. The number of small particles (11-30 nm) and the particulate organic carbon have diurnal variations which are different from the other pollutants. Charron and Harrison (2003) discussed the case of small particles for Marylebone Road. Particles in the 11-30 nm range measured between 1998 and 2000 did not show a clear relationships to traffic volumes and peaked in the early morning showing an inverse association with air temperature. It was concluded that this size range contains freshly nucleated particles formed during the rapid cooling of the exhaust gases diluted with ambient air. More recently, studies have shown that these nucleation mode particles are mainly made of volatile materials (e.g. Wehner et al., 2004) and the role of ambient temperature on particle number concentrations from vehicle emissions have been observed elsewhere (Janhäll et al., 2004; Kuhn et al., 2005). However nucleation mode particles measured from January 2002 to March 2003 showed variations closer to 30-450 nm particles than than those measured in the 1998-2001 dataset and as a consequence, nucleation mode particles are clustered with the $PM_{2.5}$ group

(Figure 2).

Figure 2: Dendrogram representing the result of a hierarchical cluster analysis (Wards method, square Euclidean distances of standardized data (Z score) is used as a similarities measure). The distances between joined clusters are the lengths of the horizontal lines.

Figure 2: Dendrogramme représentant les résultats d'une classification hiérarchique



The particulate organic carbon showed strong diurnal variations with smaller workday to weekend difference than the other particulate components. On workdays, the OC showed maximum concentrations from 14:00 to 15:00 and on weekends in the evenings. The different patterns observed on Saturday and Sunday give strong evidence that the median weekly cycle of the OC is related to the traffic. The hierarchical cluster analysis more closely associates the OC with the light duty traffic. Geller at al. (2005) found a greater percentage of OC in light duty emissions than in heavy-duty emissions. The pattern observed at Marylebone Road suggests a more important contribution of the light duty traffic than to the other particle components.

Conclusion

The London supersites offer an unique opportunity to characterize vehicle emissions under real-world conditions thanks to known traffic flows and parallel multiple continuous measurements at roadside and urban sites. Knowledge from these databases could be used to assess scenarios of emission reductions or the exposure of people living or commuting in the vicinity of roadways. The present study highlights the fact that particle metrics and components that have been associated with adverse health effects (e.g. EC, particle numbers) have much larger concentrations near traffic sources and mostly arise from the heavy-duty vehicles which are on average about 10% of the total traffic.

Acknowledgments

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The Mobile Source Contribution to Observed PM_{10}, $PM_{2.5}$, and VOCs in the Greater Cairo Area

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Abstract

Cairo, Egypt is generally classified as one of the world's "megacities", with an estimated population in excess of 20 million people in the greater Cairo/Giza area. It also suffers from high ambient concentrations of atmospheric pollutants. As part of a major program to improve air quality, a source attribution study (SAS) to determine contributions from various sources to the observed pollutant levels was performed. Intensive monitoring studies were carried out during the periods of February/March and October/November 1999 and June 2002. PM_{10}, $PM_{2.5}$, polycyclic aromatic hydrocarbons (PAHs), and volatile organic compounds (VOCs) were measured on a 24-hour basis at sites representing background levels, mobile source impacts, industrial impacts, and residential exposure. Mobile sources were found, in general, to be the major source of $PM_{2.5}$ and VOCs; with open burning and secondary species also being important contributors to the observed $PM_{2.5}$. Major contributors to PM_{10} included geological material, mobile source emissions, and open burning. Since mobile source NO_x and SO_2 emissions contribute to the observed secondary species and part of the geological material can be attributed to resuspended road dust, the mobile source contribution to PM_{10} and $PM_{2.5}$ throughout the region is likely greater than 50%. Mobile sources are also a major source of the observed VOCs.

Key-words: Source apportionment, mobile source emissions, particulate matter, megacities.

Introduction

Cairo, Egypt suffers from high ambient concentrations of atmospheric pollutants (Nasralla, 1994, Sturchio et al., 1997), including particulates (PM), carbon monoxide (CO), oxides of nitrogen (NO_x), ozone (O_3), and sulfur dioxide (SO_2). Nasralla (1994) reported particulate lead concentrations that ranged from 0.5 $\mu g/m^3$ in a residential area to 3.0 $\mu g/m^3$ at the city center. Sturchio et al. (1997) measured total suspended particulate (TSP) and lead concentrations using stable isotopic ratios ($^{207}Pb/^{204}Pb$ and $^{208}Pb/^{204}Pb$) at eleven sites in Cairo. Lead and TSP concentrations ranged from 0.08 $\mu g/m^3$ and 25 $\mu g/m^3$ respectively at Helwan to over 3 $\mu g/m^3$ and 1100 $\mu g/m^3$ respectively at the city center.

Rodes et al. (1996) measured fine ($PM_{2.5}$) and coarse (PM_{10}-$PM_{2.5}$) concentrations as a part of a source apportionment study in Cairo from December 1994 through November 1995. The annual average PM_{10} concentrations exceeded the 24-hour average U.S. standard of 150 $\mu g/m^3$ at all sites except Ma'adi and the background site. An attempt was made to attribute the high PM levels to specific sources using the Chemical Mass Balance (CMB) source apportionment model. PM_{10} mass was dominated by the coarse fraction, suggesting a strong influence of fugitive dust sources. Emissions from mobile sources, oil combustion, and open/trash burning dominated the $PM_{2.5}$ apportionments.

In order to develop and implement a pollution-control strategy and to reduce the health impact of air pollution in Cairo the Cairo Air Improvement Project (CAIP) was established. As part of the CAIP, source attribution studies were performed to assess the impact of various sources. In this paper we report PM and VOC source attribution results of ambient monitoring studies performed during the periods of February 21 to March 3, 1999 and October 27 to November 27, 1999 (Abu-Allaban et al., 2002), and an additional PM study performed June 8 to June 26, 2002. Particular emphasis is placed on evaluating the mobile source contribution to the observed elevated pollutant levels.

Experimental Methods

For the 1999 studies, six sites were selected from CAIP network (Abu-Allaban et al., 2002), while two additional sites were monitored in 2002. In order to evaluate inter-site changes over this period, we will only discuss the results from the six sites where measurements were performed in both 1999 and 2002. The six sites included: Background/Upwind (Kaha), Industrial/Residential (Shobra El-Khaima and El Massara), Traffic (El Qualaly Square), and Residential (Helwan and El-Zamalek). Sites were distributed in a north-south orientation (see Abu-Allaban et al., 2002) as follows: Kaha, Shobra, El Qualaly, El-Zamalek, El Massara, and Helwan. During the sampling periods, the prevailing winds tended to come from the north. Sampling periods were chosen to represent high pollution events (e.g., the Cairo “black cloud” that appears during the fall) and different seasons. For the winter 1999 intensive, samples were collected on a daily basis during the period 21 February to 3 March, 1999. The fall 1999 period took place during 29 October to 27 November 1999, while the summer 2002 intensive was performed during the period of 2 June to 28 June 2002. For the latter two measurement periods a non-sequential PM sampler was used and so samples were collected every other day.

Ambient $PM_{2.5}$ and PM_{10} samples were collected using the sampling protocol described by Watson et al. (1994). All samples were of 24-hour duration. During the February/March, 1999 study, samples were collected daily, while in the October/November, 1999 and June, 2002 studies samples were collected every other day. Samplers designed to collect samples for chemical analyses were utilized. In addition, we sampled for polycyclic aromatic hydrocarbons (PAHs). The PAHs were critical to apportion the carbon components of the PM based on the uniqueness of PAH compounds associated with motor vehicles and other combustion sources (Fujita et al., 1998). PAH samples were collected on Teflon-impregnated glass fiber filters followed by an adsorbent cartridge of polyurethane foam and XAD-4 resin (TIGF/PUF/XAD-4). For the VOC phase of the study, whole air samples, analyzed for C_2-C_{12} volatile hydrocarbons (VOC's), were collected with 6-liter stainless steel canisters following the protocol described by Zielinska et al. (1997).

Source emissions samples were collected using methods similar to those used in the ambient sampling program. In addition, bulk soil and road dust samples were collected at each of the ambient-sampling sites. Emissions from various sources including brick manufacturing, cast iron foundry, copper foundry, lead smelting, refuse burning, Mazot oil combustion, refuse burning, and restaurants were sampled. Individual motor vehicle emissions were sampled from heavy- and light-duty diesel vehicles, spark ignition automobiles, and motorcycles.

Chemical analyses were performed on Teflon-membrane and quartz-fiber filters following the methodology described by Watson and Chow (1993). The PAHs samples were analyzed following the protocol described by Fujita et al. (1998). For the VOC samples, C_2-C_{12} hydrocarbons were determined using gas chromatography with a flame ionization detector (GC/FID).

The Chemical Mass Balance (CMB) receptor model was used to apportion PM and its chemical constituents to their sources (Watson et al., 1990). This model has been approved for receptor modeling studies by the USEPA and has been successfully applied in urban and rural studies worldwide. The CMB procedure requires several steps. First, the contributing sources must be identified and their chemical profiles must be entered. Then the chemical species to be included in the model must be selected. The next step is the estimation of the fractions of each chemical species contained in each source type and the estimation of the uncertainties in both the ambient concentrations and source contributions. The final step is the solution of the set of chemical mass balance equations. These procedures are described in detail in an application and validation protocol (Pace and Watson, 1997).

Results

1. Mass and Inorganic Chemical Species

Mass concentrations in both size fractions (Table 1, measured mass column) were higher at all sites (except for $PM_{2.5}$ at Shobra) during the fall 1999 sampling period (Abu-Allaban et al., 2002). Shobra exhibited the highest average PM_{10} and $PM_{2.5}$ mass concentrations. The lowest values in both size fractions were observed at Helwan, a residential location. The background/upwind site, Kaha, generally had the lowest or second lowest mass levels, indicating most of the pollutants were from local sources. The only chemical species with a significant long-range transport component is NH_4Cl, which is formed from bleaching operation in the Nile delta.

The correlations between measured and reconstructed (sum of species) were very high, which indicates that the data quality for the particulate measurements was quite good. The sum of the species consistently accounted for 70-80% of the measured mass. The difference is accounted for by the fact that the sum of species does not contain oxygen associated with geological species (e.g., Al, Fe, Si) or hydrogen, oxygen, nitrogen, and sulfur associated with organic carbon.

The $PM_{2.5}/PM_{10}$ ratio varied from 0.3 at El Massara to 0.8 at Shobra. El Massara is an industrial/residential location impacted by emissions from nearby cement plants. The ratio of 0.3 at El Massara is consistent with coarse particle emissions from those activities. Shobra is a highly industrialized site with a number of lead smelters in the vicinity. The ratio of 0.8 is likely due to the impact of fresh combustion emissions, although it is still unusually high. One might have also expected very high ratios at El Qualaly, the mobile source site; however, the observed ratio was 0.4. This site also had high levels of crustal species in the PM_{10} fraction, indicating resuspended road dust, which reduced ratio.

Crustal components (Si, Ca, Fe, and Al) were significant at all sites. The majority of crustal material was in the coarse (PM_{10}-$PM_{2.5}$) fraction. The highest concentrations of PM_{10} crustal species were found at Shobra and El Massara, probably as a result of fugitive dust emissions from industrial operations at these sites. Organic carbon (OC), and elemental carbon (EC) were major components of PM at all sites. Potential sources include mobile emissions, open burning, and fossil fuel combustion. The highest average PM_{10} OC levels were observed at El Qualaly and Shobra.

2. Organic Chemical Species

Table 2: Summary of VOC source attribution results at the six intensive sites during the winter and fall, 1999 (average ± standard deviation, ppb).

Site	Date	Predicted	Measured	Evaporative Emissions	Motor Vehicles	Industrial	Compressed Natural Gas
El Massara	Winter	395 ± 24	407 ± 15	27 ± 6	96 ± 10	215 ± 18	57 ± 11
El Massara	Fall	797 ± 63	763 ± 21	80 ± 16	183 ± 26	389 ± 46	145 ± 30
El Qualaly	Winter	1931 ± 103	1849 ± 22	220 ± 45	1008 ± 67	578 ± 58	124 ± 28
El Qualaly	Fall	2242 ± 147	2037 ± 50	371 ± 61	1047 ± 88	639 ± 89	186 ± 46
Al Zamalik	Winter	981 ± 53	879 ± 32	140 ± 23	449 ± 32	304 ± 31	88 ± 17
Al Zamalik	Fall	1399 ± 95	1281 ± 33	267 ± 39	544 ± 52	437 ± 61	150 ± 33
Helwan	Winter	334 ± 19	365 ± 13	19 ± 7	175 ± 12	112 ± 11	28 ± 6
Helwan	Fall	570 ± 36	628 ± 15	88 ± 15	210 ± 20	195 ± 22	77 ± 15
Kaha	Winter	368 ± 22	375 ± 13	16 ± 5	94 ± 9	203 ± 17	56 ± 9
Kaha	Fall	452 ± 33	462 ± 11	32 ± 7	93 ± 12	226 ± 24	100 ± 18
Shobra	Winter	833 ± 47	914 ± 14	62 ± 15	304 ± 25	375 ± 33	93 ± 17
Shobra	Fall	1258 ± 121	1149 ± 29	162 ± 28	361 ± 44	534 ± 89	201 ± 64

Fall VOC concentrations were generally higher than those observed in winter (Table 2). The highest average VOC concentrations were found at El Qualaly during both fall (2037±1369 ppb) and winter (1849±298 ppb). This is consistent with the high volume of mobile source emissions expected at this site. The next highest VOC concentrations were found at Al Zamalek (winter VOC = 1282±965 ppb; fall VOC =

879±213 ppb) and at Shobra (fall VOC = 1149±822 ppb; winter VOC = 914±171 ppb). The lowest VOC concentrations were found at El Massara, Helwan, and Kaha.

The temporal variations of VOC were consistent and largely invariant among the six sites during winter. All sites seemed to experience a minimum in VOC concentrations at the end of February, 1999. During fall, all sites experienced VOC maxima on November 22, 1999 and elevated concentrations on the two days leading up to it. Again, the lowest concentrations in fall were found at El Massara, Helwan, and Kaha.

The most abundant VOCs were isopentane and n-pentane, which are associated with evaporative emissions from motor vehicles, C_2 compounds (e.g., ethane, ethene), propane, isobutane, and n-butane, which come from compressed natural gas (condensed natural gas) and LPG (liquefied petroleum gas). MTBE (methyl tertiary-butyl ether), a gasoline additive, toluene, and benzene were also abundant compounds.

3. PM Source Attribution Results

The PM source attribution results are presented in Table 1. A summary, by site, of the CMB results is presented below:

Al Zamalek: The major sources of PM_{10} were geological material, Mazut oil, mobile sources, and open burning. Most of the secondary sulfate, ammonium chloride, and combustion (motor vehicle, open burning, and oil) were in the fine fraction. Lead and copper smelter contributions were small at this site.

El Qualaly : This site was dominated by mobile source emissions. PM_{10} was dominated by geological, mobile source, and open burning. About half of the mobile and open burning emissions were in the $PM_{2.5}$ fraction. The lead and copper smelter contributions to PM_{10} were significant at this site

Helwan: PM_{10} at this site was dominated by geological material. While the marine contribution was almost entirely in the coarse (PM_{10}-$PM_{2.5}$) fraction, mobile, Mazut, and open burning contributions at this site were almost entirely in the fine fraction. Copper smelter contributions were detected at low levels (1-2%) in both size fractions.

Kaha: The largest contributors to PM_{10} were open burning and geological dust. Mobile source and Mazut oil contributions were small compared with other sites. $PM_{2.5}$ was dominated by open burning.

El Massara: The PM_{10} fraction was dominated by geological material, cement, mobile sources, and open burning. The fine fraction was dominated by open burning and mobile emissions. The lead and copper smelter contributions were found almost entirely in the fine fraction.

Shobra: The most unusual aspect of this location is the high PM Pb levels. Eighty percent of the lead contribution was in the $PM_{2.5}$ fraction. Most of this Pb is in the form of fresh emissions from secondary Pb smelters in the vicinity. Contributors to PM_{10} included geological material and mobile source emissions. The $PM_{2.5}$ apportionment shows a similar distribution of source contributions, which is consistent with dominance of fine particles at this site.

Table 1: Summary of PM2.5 and PM10 source attribution results for the six intensive sites (average ± standard deviation, μg/m3).

Location	Year	Size	Measured Mass	Predicted Mass	Geological Material	Lead Smelters	Copper Smelters	Steel Industry	Heavy Oil	Motor Vehicles	Open Burning	Marine	Ammonium Sulfate	Ammonium Nitrate	Ammonium Chloride	Cement Plants
El Massara	Winter 1999	PM10	186 ± 9	187 ± 18	65 ± 6	1 ± 0	1 ± 0	1 ± 1	2 ± 0	12 ± 3	44 ± 4	11 ± 1	3 ± 7	0 ± 0	3 ± 0	44 ± 14
El Massara	Winter 1999	PM2.5	61 ± 3	67 ± 3	7 ± 1	1 ± 0	0 ± 0	1 ± 0	3 ± 0	13 ± 2	19 ± 2	2 ± 0	4 ± 1	3 ± 0	5 ± 1	7 ± 1
El Massara	Fall 1999	PM10	317 ± 17	348 ± 33	145 ± 15	0 ± 0	1 ± 0	4 ± 1	1 ± 0	11 ± 3	103 ± 9	5 ± 1	13 ± 5	13 ± 2	16 ± 13	36 ± 23
El Massara	Fall 1999	PM2.5	107 ± 6	116 ± 6	12 ± 1	0 ± 0	1 ± 0	2 ± 0	4 ± 1	12 ± 2	48 ± 5	0 ± 0	9 ± 1	5 ± 1	17 ± 1	5 ± 1
El Massara	Summer 2002	PM10	175 ± 9	202 ± 17	86 ± 8	0 ± 0	0 ± 0	1 ± 1	1 ± 0	8 ± 2	54 ± 5	6 ± 1	8 ± 2	8 ± 1	0 ± 0	29 ± 15
El Massara	Summer 2002	PM2.5	48 ± 3	61 ± 3	6 ± 1	0 ± 0	0 ± 0	0 ± 0	1 ± 0	14 ± 2	25 ± 2	1 ± 0	8 ± 1	2 ± 0	0 ± 0	3 ± 0
El Qualaly	Winter 1999	PM10	220 ± 11	191 ± 11	77 ± 9	7 ± 1	3 ± 0	5 ± 1	3 ± 0	33 ± 4	34 ± 4	3 ± 1	1 ± 3	6 ± 1	18 ± 2	0 ± 0
El Qualaly	Winter 1999	PM2.5	85 ± 4	72 ± 3	3 ± 0	2 ± 0	2 ± 0	1 ± 0	6 ± 1	21 ± 2	14 ± 1	0 ± 0	5 ± 1	4 ± 0	13 ± 1	0 ± 0
El Qualaly	Fall 1999	PM10	252 ± 13	269 ± 17	70 ± 10	3 ± 0	3 ± 0	4 ± 1	2 ± 0	43 ± 6	104 ± 10	2 ± 1	8 ± 8	11 ± 1	20 ± 2	0 ± 0
El Qualaly	Fall 1999	PM2.5	135 ± 7	149 ± 7	3 ± 1	2 ± 0	3 ± 0	1 ± 0	5 ± 2	46 ± 4	56 ± 5	0 ± 0	9 ± 2	6 ± 1	19 ± 1	0 ± 0
El Qualaly	Summer 2002	PM10	136 ± 7	137 ± 8	68 ± 6	1 ± 0	0 ± 0	2 ± 0	1 ± 0	24 ± 2	25 ± 3	3 ± 1	6 ± 2	6 ± 1	0 ± 0	0 ± 0
El Qualaly	Summer 2002	PM2.5	59 ± 3	56 ± 3	1 ± 0	0 ± 0	0 ± 0	0 ± 0	2 ± 1	29 ± 2	11 ± 1	1 ± 0	9 ± 1	1 ± 0	0 ± 0	0 ± 0
Al Zamalek	Winter 1999	PM10	127 ± 6	112 ± 7	29 ± 4	2 ± 0	1 ± 0	3 ± 0	3 ± 0	16 ± 2	22 ± 2	4 ± 1	1 ± 2	6 ± 1	23 ± 2	3 ± 4
Al Zamalek	Winter 1999	PM2.5	62 ± 3	58 ± 3	1 ± 0	1 ± 0	1 ± 0	1 ± 0	6 ± 0	9 ± 1	12 ± 1	0 ± 0	5 ± 1	3 ± 0	18 ± 1	0 ± 0
Al Zamalek	Fall 1999	PM10	249 ± 13	262 ± 17	80 ± 9	1 ± 0	2 ± 0	5 ± 1	2 ± 0	24 ± 4	108 ± 9	6 ± 1	7 ± 9	11 ± 1	15 ± 4	0 ± 0
Al Zamalek	Fall 1999	PM2.5	132 ± 7	143 ± 7	5 ± 1	1 ± 0	2 ± 0	1 ± 0	5 ± 2	29 ± 4	60 ± 5	1 ± 0	11 ± 2	7 ± 1	20 ± 1	1 ± 0
Al Zamalek	Summer 2002	PM10	99 ± 5	98 ± 7	41 ± 5	0 ± 0	0 ± 0	2 ± 0	1 ± 0	19 ± 3	19 ± 2	4 ± 1	8 ± 2	5 ± 1	0 ± 0	0 ± 0
Al Zamalek	Summer 2002	PM2.5	40 ± 2	39 ± 2	2 ± 0	0 ± 0	0 ± 0	0 ± 0	2 ± 0	14 ± 2	10 ± 1	1 ± 0	9 ± 1	1 ± 0	0 ± 0	0 ± 0
Helwan	Winter 1999	PM10	88 ± 5	85 ± 8	27 ± 4	0 ± 0	1 ± 0	2 ± 0	1 ± 0	13 ± 3	18 ± 2	5 ± 1	2 ± 3	4 ± 0	2 ± 1	11 ± 5
Helwan	Winter 1999	PM2.5	29 ± 2	30 ± 2	1 ± 0	0 ± 0	0 ± 0	0 ± 0	1 ± 0	8 ± 1	10 ± 1	0 ± 0	4 ± 1	2 ± 0	3 ± 0	0 ± 0
Helwan	Fall 1999	PM10	146 ± 9	190 ± 15	78 ± 11	0 ± 0	2 ± 0	3 ± 1	1 ± 0	13 ± 3	63 ± 7	6 ± 5	8 ± 4	7 ± 1	5 ± 1	4 ± 5
Helwan	Fall 1999	PM2.5	100 ± 6	84 ± 5	5 ± 1	0 ± 0	2 ± 0	2 ± 0	2 ± 1	9 ± 2	39 ± 4	1 ± 0	8 ± 1	4 ± 1	10 ± 1	1 ± 0
Helwan	Summer 2002	PM10	142 ± 7	142 ± 9	73 ± 7	0 ± 0	0 ± 0	3 ± 1	0 ± 0	22 ± 3	25 ± 4	5 ± 1	5 ± 2	8 ± 1	0 ± 0	0 ± 0
Helwan	Summer 2002	PM2.5	48 ± 3	54 ± 3	10 ± 1	0 ± 0	0 ± 0	1 ± 0	1 ± 0	15 ± 2	16 ± 2	1 ± 0	6 ± 1	2 ± 0	0 ± 0	0 ± 0
Kaha	Winter 1999	PM10	93 ± 5	97 ± 6	12 ± 3	0 ± 0	0 ± 0	1 ± 0	2 ± 0	13 ± 3	35 ± 3	3 ± 1	2 ± 2	6 ± 1	23 ± 2	0 ± 0
Kaha	Winter 1999	PM2.5	50 ± 3	51 ± 2	0 ± 0	0 ± 0	0 ± 0	0 ± 0	3 ± 0	4 ± 1	19 ± 1	0 ± 0	5 ± 1	4 ± 1	15 ± 1	0 ± 0
Kaha	Fall 1999	PM10	205 ± 11	237 ± 14	52 ± 6	0 ± 0	0 ± 0	1 ± 1	1 ± 0	14 ± 3	123 ± 10	9 ± 4	6 ± 6	11 ± 1	18 ± 2	0 ± 0
Kaha	Fall 1999	PM2.5	111 ± 6	130 ± 8	5 ± 1	0 ± 0	0 ± 0	0 ± 0	3 ± 1	11 ± 3	82 ± 8	0 ± 0	8 ± 1	7 ± 1	14 ± 1	0 ± 0
Kaha	Summer 2002	PM10	100 ± 5	97 ± 5	39 ± 3	0 ± 0	0 ± 0	1 ± 0	1 ± 0	9 ± 1	34 ± 3	4 ± 1	5 ± 2	5 ± 1	0 ± 0	0 ± 0
Kaha	Summer 2002	PM2.5	35 ± 2	33 ± 2	3 ± 0	0 ± 0	0 ± 0	0 ± 0	1 ± 0	9 ± 1	12 ± 1	0 ± 0	7 ± 1	0 ± 0	0 ± 0	0 ± 0
Shobra	Winter 1999	PM10	265 ± 14	219 ± 10	51 ± 5	53 ± 5	10 ± 1	11 ± 1	4 ± 0	21 ± 4	40 ± 5	6 ± 1	0 ± 0	6 ± 1	18 ± 2	0 ± 0
Shobra	Winter 1999	PM2.5	216 ± 11	171 ± 7	36 ± 3	43 ± 4	9 ± 1	14 ± 1	9 ± 3	26 ± 4	5 ± 1	3 ± 1	2 ± 2	5 ± 1	16 ± 1	2 ± 1
Shobra	Fall 1999	PM10	360 ± 19	349 ± 25	90 ± 10	17 ± 3	7 ± 1	14 ± 2	4 ± 0	23 ± 5	147 ± 13	4 ± 7	5 ± 11	12 ± 2	14 ± 2	11 ± 11
Shobra	Fall 1999	PM2.5	174 ± 9	203 ± 11	20 ± 2	12 ± 2	6 ± 1	2 ± 0	9 ± 3	26 ± 5	89 ± 9	0 ± 0	12 ± 2	7 ± 1	20 ± 2	0 ± 0
Shobra	Summer 2002	PM10	154 ± 8	143 ± 7	52 ± 5	11 ± 2	3 ± 0	9 ± 1	3 ± 0	15 ± 2	34 ± 4	5 ± 1	5 ± 2	6 ± 1	0 ± 0	0 ± 0
Shobra	Summer 2002	PM2.5	61 ± 3	60 ± 3	2 ± 0	7 ± 1	2 ± 0	4 ± 0	6 ± 1	13 ± 2	15 ± 2	1 ± 0	10 ± 1	1 ± 0	0 ± 0	0 ± 0

4. VOC Source Attribution Results

The CMB results are presented for each winter and fall sample in Table 2. The VOC apportionments were also consistent spatially and temporally. On-road mobile and industrial (e.g., lead smelter/LPG) emissions dominated contributions to VOC at all sites during both seasons. Further, evaporative emission contributions were relatively higher during the fall season. This may be related to the observation that average temperatures were about 5 ^{o}C higher during the fall as opposed to the winter season. Considering the results in table 2, total mobile source emissions (defined as evaporative plus mobile source tailpipe emissions) made up the majority of the VOC emissions at Al Zamalek, El Qualaly, and Helwan.

Assessment of the Mobile Source Impact

The motivating factor for this study was to quantitatively assess the contribution of various sources to the observed pollutant levels. This is a critical step if the regulatory authorities are to implement effective pollutant control strategies. Further, it is surprising how commonly held beliefs as to the nature of the pollutants are often at odds with the results of a source attribution study. For example, most of the problem in Cairo was blamed on industrial emissions. While this is true in the Shobra area, open burning and mobile source emissions are clearly important. This raises the issue of the magnitude of the mobile source contribution. In order to assess the total mobile source contribution to the observed pollutant levels, we need to look at both the direct emissions (i.e., tailpipe) and the contributions from other mechanisms (e.g., resuspended road dust, secondary PM, evaporative emissions, etc.). Direct tailpipe emissions are <25% of the PM_{10} (~60 μ/m^3) and <50% of the $PM_{2.5}$ in the most extreme case (El Qualaly). However, gas phase mobile source emissions (i.e., NO_x and SO_2 from diesel fuel) can react in the atmosphere to form secondary PM. In addition, a significant fraction of geological component is due to resuspended road dust. When these contributions are considered, the mobile source contribution to both PM_{10} and $PM_{2.5}$ is >50% for all locations.

Similarly, for VOC emissions, we can attribute evaporative emissions and mobile source tailpipe emissions to mobile sources. For three of the locations (Zamalek, El Qualaly, and Helwan) the mobile source contribution to the observed VOCs was >50%, while it was less than this at the remaining three sites.

Recommendations to Improve Air Quality

Clearly, mobile source emissions are the major contributor to the elevated PM and VOC levels in the Cairo area; however, this work has shown that industrial emissions and open burning are also significant contributors. In order to improve air quality we recommend the following:

General recommendations: (1) Institute a program to reduce traffic congestion. This will reduce mobile source emissions and resuspended road dust; (2) Implement a comprehensive enforcement program to ensure industrial compliance with air quality regulations; (3) Develop policies to encourage retrofitting existing industrial sources with lower emitting technologies; (4) Involve the public and other stakeholders in the planning and implementation of any improvement strategies; and (5) Perform routine measurements of air pollutants (e.g., CO, SO_2, NO_x, O_3, in addition to PM) in the greater Cairo area.

Specific recommendations: (1) Initiate a program to collect garbage and eliminate open burning in both residential and agricultural areas. This will reduce the open burning component of PM; although cooking will still contribute to this component; (2) Expand and enforce the vehicle emissions and testing program to reduce mobile source emissions. This will also reduce NO_x emissions, which would lead to a reduction in the observed ammonium nitrate component of PM; and (3) Reduce geological emissions by paving the roads to reduce resuspended road dust and implement control measures at industrial facilities to reduce fugitive emissions from piles of raw materials.

Summary and Conclusions

An intensive $PM_{10,}$ $PM_{2.5}$, and VOC sampling program was carried out at six sites in the greater Cairo area during a winter period from February 18 to March 4, 1999 and during a fall period from October 29 to November 27, 1999. Additional PM measurements were performed during a summer period from June 8 to June 26, 2002. Medium volume samplers were used to collect $PM_{2.5}$, PM_{10}, and PAH samples for subsequent chemical analysis and source apportionment modeling. Canister samplers were used to collect VOCs.

The CMB receptor model coupled with source profiles measured during the CAIP and from previous studies was used to estimate source contributions to $PM_{2.5}$ and PM_{10} mass. Depending on the sites, major contributors to PM_{10} included geological material, mobile source emissions, and open burning. $PM_{2.5}$ tended to be dominated by mobile source emissions, open burning, and secondary species. VOC concentrations were generally higher in fall than in winter. El Qualaly, the site chosen to represent mobile emissions, displayed the highest average VOC concentrations of any site, by factors of 2 or more, in both winter (1849 ppb) and fall (2037 ppb). The major contributors to VOCs at all sites in both seasons were mobile source emissions and industrial emissions. We interpret the latter to represent industrial processes that may be fueled by LPG. When we consider the various mechanisms by which mobile sources can lead to elevated PM, we find that >50% of the observed PM_{10} and $PM_{2.5}$ is due to emissions from this source. In addition, mobile sources are a major contributor to ambient VOCs.

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Air pollution at street level in selected European cities

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Abstract

Air pollution at street level is studied using a sequence of air quality models covering the regional, urban and local scales. The urban scale model OFIS is driven by the results of the regional scale EMEP model and the local scale OSPM model is applied using OFIS urban background concentrations, in order to estimate air pollution at traffic hotspots. The TRENDS and COPERT 3 models are used to estimate vehicle emissions for the reference year (2000) and two scenarios focusing on the introduction of Euro 5 and Euro VI compliant vehicles. Air quality model calculations are presented for NO_2 and PM_{10} and for both scenarios, improved air quality at urban and street scale is predicted for the year 2030, in line with the NO_x and PM emission reductions assumed, though for PM_{10} the model results show that additional measures may be needed in order to meet air quality standards in all cities considered.

***Key-words**: air quality modelling, street canyons, vehicle emissions.*

Introduction

Until recently, the design of European air pollution abatement strategies has focused on the analysis of regional scale concentrations. However, ambient concentrations of certain air pollutants show strong variability at a much finer scale (urban and local scale) and it appears necessary to also include these scales in an Integrated Assessment (IA) modelling framework. During the Clean Air for Europe programme (CAFE), the City-Delta project used a number of tools to study the increased concentrations observed at urban scale across Europe and sought ways to account for these elevated values in the overall IA framework, as described in Cuvelier et al. (2004). Measurements at street level stations show that concentrations are systematically higher in areas with increased traffic and recent studies show that most of traffic related PM emissions are in the fine particulates range (e.g. Zhu et al. (2002)). The evidence of adverse health effects related to exposure to fine particulate matter is continuously emerging (e.g. Pope et al. (2002)) and therefore it appears necessary to also include the local scale in the design of integrated air pollution abatement strategies.

Methodology

1. Urban emissions and air quality

In line with the nature of air quality assessment and air pollution abatement strategies implemented by the European Commission (2001), a multi-pollutant, multi scale approach was adopted to study the increased air pollution levels at traffic hotspots of twenty European cities. The current situation (reference year 2000) and two scenarios projecting emissions in 2030 (Current Legislation-CLE and Maximum Feasible Reductions-MFR scenarios) were studied and are described in detail in Cofala et al. (2005). The analysis was performed for NO_2, NO_x, PM_{10} and $PM_{2.5}$ using a complete regional-urban-local scale modelling sequence. The urban scale OFIS model by Arvanitis and Moussiopoulos (2003) was applied driven by results of the regional scale model EMEP as described in Simpson et al. (2003). In turn, the local scale OSPM model developed by Berkowicz (2000) was applied using OFIS results to derive the necessary urban background conditions. Air quality model results for the reference year (2000) were evaluated against background and street level monitoring station data from Airbase (URL1) and were found to be in good agreement as shown in EEA (2006).

The urban emission inventories required by OFIS were developed in the frame of the MERLIN project for twenty cities through the application of the European Emission model, according to Friedrich and Reis (2004), Schwarz (2002) and Wickert (2001). The urban emission projections for the year 2030 were predicted according to the emission control scenarios CLE and MFR, as described in Cofala et al. (2005). Since information of this type was only available at country level and not at city level, the emission reductions according to the two scenarios for the year 2030 were calculated for each country. The emission reductions at urban level were then considered equal to those at country level, in order to obtain the urban emissions for the year 2030.

2. Local emissions and air quality

For the local air quality analysis, specific street canyon characteristics were required in order to define particular case studies (types of streets) in each city. Three hypothetical street canyon configurations were selected (wide, square and narrow) defined in van den Hout and Teeuwisse (2004). In the present paper, the NO_2 and PM_{10} annual mean concentrations in the narrow canyon configuration, assuming an average daily traffic of 20,000 vehicles per day, are presented. The traffic emissions considered in the local air quality modelling differ from city to city according to the specific fleet composition and the contribution of each vehicle category to the total street emissions. The heavy duty vehicle (HDV) percentage and the average vehicle speed (26 km/h) used in the calculations were selected in accordance with the typology methodology by van den Hout and Teeuwisse (2004), which foresees one of two discrete values (7 % or 15 %). Based on results obtained from the TRENDS model, as described in Giannouli et al. (2006) and the TREMOVE model as presented in De Ceuster et al. (2005) for the country scale, the larger value (15 %) was used only for the city of Lisbon. As the precise HDV percentage in the fleet can significantly influence the modelled concentrations, for a limited number of cities the effect of an increased HDV percentage was investigated using two different fleet configurations. The first is based on the typology methodology and assumes 7 % HDVs, while the second configuration assumes 15 % HDV.

Concerning the calculation of PM_{10} concentrations, since neither the regional scale air quality model (EMEP), nor the urban scale model (OFIS) account for natural primary PM sources such as windblown dust (African dust or local soil resuspension), sea salt or organic aerosols, a constant value of 17 $\mu g/m^3$ was assumed in all cities to account for these PM sources, as described in EEA (2006). This assumption may overestimate or underestimate natural sources, e.g. it should perhaps be larger in the case of cities located in dry costal areas of Southern Europe where PM sources such as African dust, local soil resuspension and sea salt would make a larger contribution. However, it must be noted that primary PM_{10} emission data are not as robust as those for other air pollutants and this fact combined with the complex formation, deposition and resuspension processes leads to uncertainties in the modelled PM_{10} ambient concentrations.

Local scale emissions for the reference year (2000) were calculated by means of the COPERT 3 emission model, which is described in Ntziachristos and Samaras (2000), using as input vehicle activity and fleet data originating from the TREMOVE and TRENDS models. Moreover, in the case of two-wheelers, updated NO_x and PM emission factors were used, which were produced by LAT (2004).

Emissions produced at street level for the year 2030 were estimated through generalised attenuation factors, which were calculated according to the two scenarios, CLE and MFR. In order to obtain the aforementioned attenuation factors for NO_x and PM emissions, vehicle activity data (1995-2020) from the TREMOVE model were inserted in the TRENDS model. Then, emission results were calculated using the COPERT 3 model at country level. Emission estimates for the two scenarios considered were also produced by applying suitable emission reductions, based on the introduction of Euro 5 and Euro VI vehicles (for passenger cars and heavy-duty vehicles respectively) to the emissions calculated by COPERT. The emission estimates produced according to each scenario were then extrapolated up to the year 2030 and attenuation factors were calculated for each scenario. Finally, new street emissions up to the year 2030 for the street canyons located in the 20 urban areas considered were calculated by applying the above attenuation factors to the reference year (2000) emissions.

For the purpose of this study it was considered that the base case scenario of TREMOVE approximates a “business as usual” scenario corresponding to the CLE scenario. The MFR scenario was considered to represent the maximum reductions that can be achieved through emission control measures. For that reason, the MFR scenario was considered to simulate the emissions produced from future vehicle technologies, namely from the Euro 5 technology for passenger cars (PCs) and light-duty vehicles (LDVs), and the Euro VI technology for HDVs. More specifically, focus of the MFR scenario is on the probable limit values of NOx and PM emissions, which are suggested on the basis of the discussions on Euro 5 and Euro VI held at EU level, as described in European Commission (2004).

Table 1 shows the emission standards corresponding to the future vehicle technologies, which were adopted in the framework of this study for NOx and PM emissions. With regard to PM, the cases suggested with Diesel Particulate filter (DPF) as the technical measure, the actual reduction used was 90 %. This can be justified by the fact that if a DPF is used to satisfy a legal limit, its reduction effect in real life might go far beyond the legal limit. It should also be mentioned that, according to the COPERT methodology, PM emissions from gasoline passenger cars and light-duty vehicles are considered negligible. For that reason, emission reductions for these vehicle categories are not considered, as shown in . The aforementioned assumptions on the future emissions limits of road vehicles were conducted prior to the recent release, by the European Parliament and Council (2005), of the proposed limit values for Euro 5 vehicles. These values for NO_x and PM emissions are also displayed in , for reasons of comparison with the corresponding values assumed in the framework of this study according to the MFR scenario. From , it can be observed that the values considered for the MFR scenario exceed the proposed Euro 5 limits for all vehicle categories. For that reason, the air emission levels predicted according to the MFR scenario will be lower than the expected emission levels resulting from the application of the proposed Euro 5 limits.

Table 1: Reduction percentage of NO_x and PM emissions according to the MFR scenario and corresponding values proposed by the European Parliament and Council for Euro 5 vehicles.

	PC - LDV Gasoline	PC - LDV Diesel	HDV
PM	N/A	DPF	DPF
PM (Proposed Euro 5 values)	0.005 g/km	-80%	N/A
NO_x	-40%	-40%	-85%
NO_x (Proposed Euro 5 values)	-25%	-20%	N/A

Figure 1: Attenuation factors (%) of NOx emissions in Germany, according to the CLE and the MFR scenario, with respect to the reference year (2000) emissions

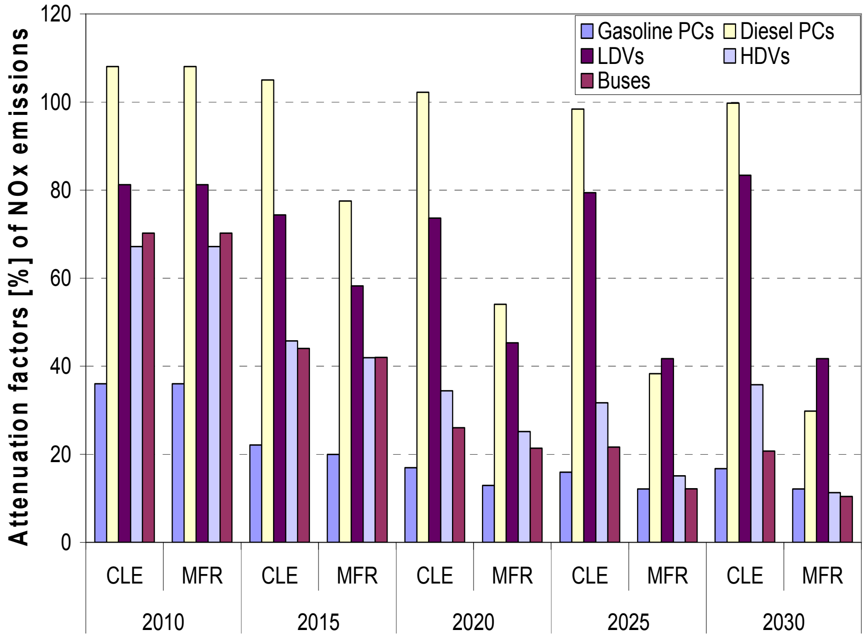
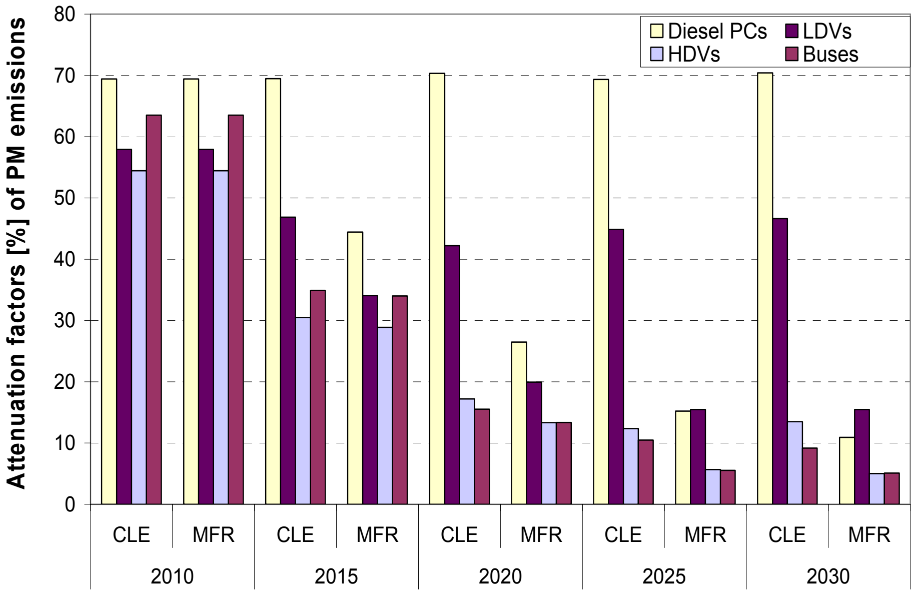


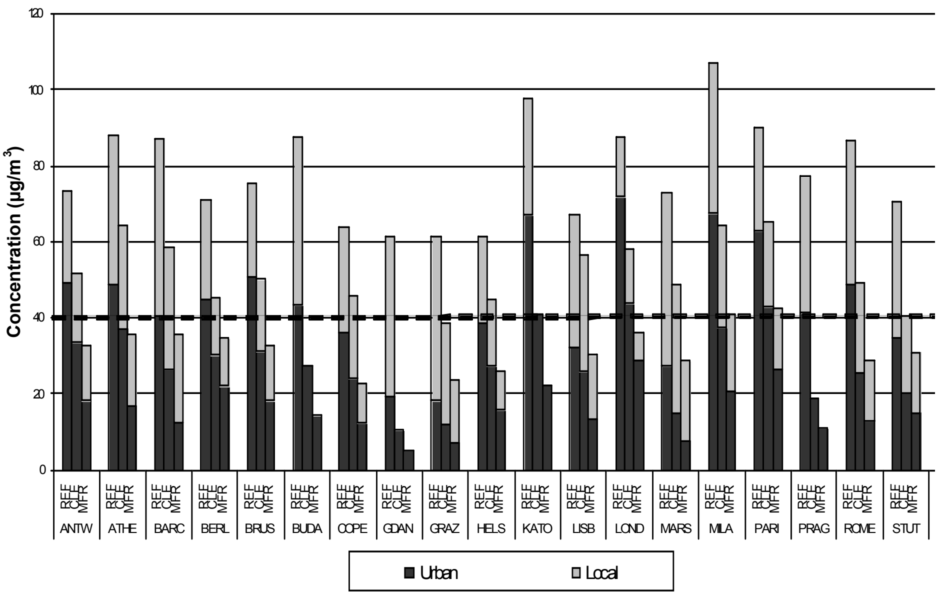
Figure 2: Attenuation factors (%) of PM emissions in Germany, according to the CLE and the MFR scenario, with respect to the reference year (2000) emissions



Results and Discussion

Examples of the attenuation factors for Germany, which were calculated according to the two scenarios for NO_x and PM emissions, are presented in Figures 1 and 2 respectively. The attenuation factors displayed in these figures cover the time period 2010 – 2030 and reflect the expected reduction in the emissions for each year and vehicle category, compared to the respective emissions values produced for the reference year (2000). From Figures 1 and 2, it is evident, that according to the MFR scenario, the most significant reductions can be observed for the emissions of diesel passenger cars and light-duty vehicles, due to the high reduction factors considered for these vehicle categories (see), as well as to high replacement rates considered for these vehicles in the TRENDS model.

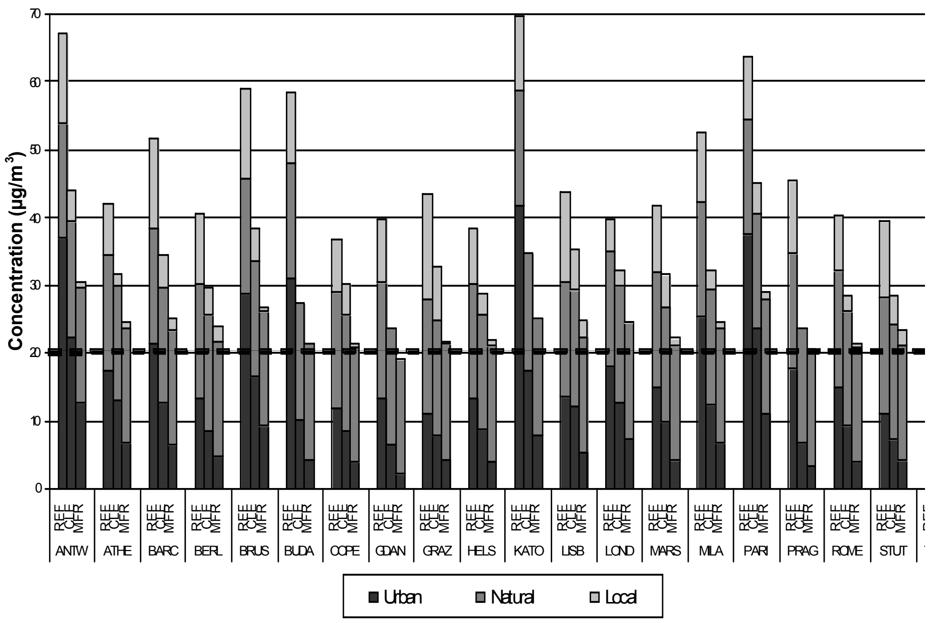
Figure 3: Annual mean NO_2 calculated at urban scale (OFIS) and the additional street increment calculated at local scale (OSPM) for cities across Europe in the reference year (2000) and the CLE and MFR scenarios (2030). The dashed line shows the limit value for 2010 according to Directive 1999/30/EC.



In Figures 3 and 4, the annual mean concentrations for NO_2 and PM_{10} at urban and local scale are presented for the reference year (2000) and the CLE and MFR scenarios. The limit value for NO_2 and the indicative limit value for PM_{10}, both set to apply from 2010 onwards, are also shown for comparison. In both figures, the urban concentrations correspond to urban background air quality levels in the city centre (calculated using the OFIS model), whereas the local concentrations correspond to the urban traffic concentrations expected to be observed at street level in a narrow street canyon configuration. It should be noted that due to the lack of reliable vehicle fleet data for non EU-15 countries for the year 2000, unrealistic emission attenuation factors were calculated for the projection year 2030 leading to much too high reductions in the street level concentrations; hence these cities were not considered in the scenario analysis.

In the case of NO_2, the measures considered by the CLE scenario lead to a significant reduction of urban emissions and hence the air quality at urban scale is predicted to be within air quality standards in most cases. However, at street level the CLE scenario is not enough to meet the air quality standards, since in most cities the concentrations in the narrow canyon configuration are predicted to be above the 2010 limit value. In the MFR scenario, the urban background concentrations are predicted to drop significantly, in line with the stricter measures assumed and the street level concentrations in the narrow canyon configuration are also expected to fall below the 2010 limit value in almost all cities. In terms of street increments (street level minus background concentrations) the modelled street increment in the reference year (2000) ranged from 16-53 µg/m^3 depending on the city, whereas in 2030 the range drops to 14-36 µg/m^3 in the CLE and 7-24 µg/m^3 in the MFR scenario, in line with the stricter vehicle emission standards, which are expected to apply in the following years.

Figure 4: Annual mean PM_{10} calculated at urban scale (OFIS) and the additional street increment calculated at local scale (OSPM) for cities across Europe in the reference year (2000) and the CLE and MFR scenarios (2030). The dashed line shows the indicative limit value for 2010 according to Directive 1999/30/EC.



The situation for PM_{10} is rather different, since although the urban emissions are significantly reduced in the CLE scenario, the urban background concentrations in 2030 (including the natural contribution, see methodology section) are still predicted to be above the indicative limit value of 20 µg/m^3 which may apply from 2010 onwards, but below the limit value of 40 µg/m^3 which applies since 2005. In accordance with the emission reductions of the MFR scenario, urban background PM_{10} concentrations are predicted to drop in 2030, though they still remain above the indicative limit value. At street level, the increased traffic emissions lead to elevated concentrations, therefore the indicative limit value is expected to be exceeded even more at the local scale. In terms of street increments, the contribution in 2000 is estimated to range between 5-15 μg/m^3 depending on the city, but the introduction of Euro 5 and Euro VI technology vehicles leads to a reduced contribution in 2030, estimated to range between 2-8 μg/m^3 and 0.2-2.4 μg/m^3 in the CLE and MFR respectively. It should be noted that in Figure 4, for both the CLE and MFR scenarios, it is assumed that the natural contribution to PM_{10} concentrations (see methodology section) remains constant at 17 μg/m^3, i.e. the same as in 2000 (reference year). It is uncertain how this estimate may change in 2030, depending on the climatic conditions.

Finally, the number of HDVs in the fleet plays a significant role in the concentrations observed at street level. In order to study the sensitivity of local air quality on the percentage of HDVs assumed, for seven cities (Athens, Berlin, Milan, Rome, Stuttgart and Thessaloniki) air pollution in the narrow canyon configuration was studied assuming instead of 7 %, 15 % HDVs. This assumption leads to an increased estimate of all pollutant concentrations and the increase depends on the specific composition of the HDV fleet in each city. In countries such as Greece (Athens and Thessaloniki) where old technology and more polluting vehicles are still used, the increase is larger than in German or Italian cities. An assumption of 15 % HDV in the fleet increases street level NO_2 concetration by 5-7 μg/m^3 and PM_{10} by 4-6 μg/m^3, depending on the city, as described in EEA (2006).

Conclusions

The application of the multi-pollutant, multi-scale model approach using the EMEP, OFIS and OSPM models, combined with the generic values assumed by the typology methodology, lead to reasonable results. The two future emission scenarios that were studied, predict reduced urban background and street level concentrations for the year 2030, though the largest improvement is achieved with the MFR scenario. In terms of meeting the established annual limit value for NO_2, urban background concentrations are predicted to be below the limit in almost all cities in the CLE scenario, but above the limit in narrow street canyons. Only the MFR scenario projects concentrations below the limit value at street level across all cities. The situation is worst for PM_{10}, since both scenarios predict urban background and street level PM_{10} concentrations to be above the indicative 2010 limit value, though at a policy level it remains uncertain whether this limit will indeed be agreed to apply. In most cities, there seems to be no problem in meeting the 2005 limit value, even according to the less optimistic CLE scenario.

In terms of the specific street configurations (aspect ratio, HDV percentage in the fleet, etc.) the parameters and the corresponding ranges of values selected in accordance with the typology methodology, also lead to satisfactory results. However, the methodology needs to be studied further to evaluate the sensitivity of the street emissions and consequent air quality calculations to the specific parameter values, with particular focus on worst case situations, which pose the greatest threat to human health. Overall, the successful application of the regional-urban-local scale modelling sequence demonstrates the feasibility of such an approach using well documented modelling tools and provides a first step towards a methodology that can be included in the already established Integrated Assessment framework used for the design of European air pollution abatement strategies.

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SIMAIR – Evaluation tool for meeting the EU directive on air pollution limits

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Abstract

Almost all Swedish cities need to determine air pollution levels – especially PM10 – close to major streets. SIMAIR is an Internet tool that can be used of all Swedish municipalities to assess PM10, NO_2, CO and benzene levels and how they compare to the EU directive. SIMAIR is delivered to the municipalities with all data pre-loaded and is meant to be used prior to decisions if, and where, monitoring campaigns are required. The system includes a road and vehicle database with emission factors and a model to calculate non-tailpipe PM10 emissions. Regional and urban background contributions are pre-calculated and stored as hourly values on a 1x1 km grid. The local contribution is calculated by the user, selecting either an open road or a street canyon environment. Preliminary comparisons with measurements show that SIMAIR is able to identify critical roads where PM10 and NO_2 levels reach the standard limits.

***Keys-words**: air pollution, EU directive, traffic model, dispersion models*

Introduction

The EU directive on air pollution levels for PM10, NO_2, CO and benzene, mirrored in Swedish legislation, has had far reaching consequences for Swedish city administrations. Of special importance is the PM10 legislation, as the Swedish EPA estimates that around 80% of 350 population centres – some with less than 10 000 inhabitants – will have to assess PM10 concentrations. The critical levels are generally expected to be caused by local traffic emissions (for PM10 this includes non-exhaust emissions) together with long range transport. Also NO_2 levels are close or above the legislation limits in a number of cities, while CO and benzene is not expected to exceed the standards in any Swedish city.

In Sweden the long range transport is especially important for the urban background PM10 levels, typically contributing, as a yearly average, to 70% in the south parts and 50% in the northern parts of the country (Forsberg et al., 2005). Also NO2 has a significant long range component and the local NO/NO_2 ratio has a strong dependence on the ozone concentrations in the incoming air.

An important characteristic of the PM10 levels in Scandinavian cities is that domination of non-exhaust PM emissions as compared to combustion particles. In Stockholm the emission of mechanically generated particles, mostly due to road and tyre wear, is as an yearly average 8 times higher than the emission of vehicle exhaust particles (Forsberg et al., 2005). This exceptional high non-exhaust emission, as compared to other parts of Europe, is explained by the wintertime use of studded tyres and sanding on the roads.

The fact that the EU directive includes upper concentration limits for percentiles of daily averages, prevents the use of simple superposition of the regional, the urban and the local contributions. Episodes of high regional contributions may be correlated or anti-correlated with high local contributions (low mixing). The SIMAIR system is therefore based on hourly time series calculations of the different contributions. This allows a straightforward statistical treatment of the simulated concentrations before comparing with EU standards.

The SIMAIR system, except being an advanced coupled model system, also includes a traffic emission database covering the entire road network of Sweden. The end user of SIMAIR, found at municipality level, can directly perform model calculations close to streets in his own area, as all input data are preloaded. A highly efficient user interface, accessed from an ordinary Internet browser, allows for a GIS-like operation of the system.

Methods

The characteristics of the SIMAIR system is to use the best available emission and dispersion models on different scales, but at the same time present a very simplified functionality. Small city administrations do not have personal nor economical resources to spend much time on following up air quality standards. All information needed should be available on delivery of the evaluation tool and the time to get an answer for a particular road link should be as close as possible to immediate. A SIMAIR customer logs into the system through an Internet browser, without any downloads, and will be able to launch a one year simulation by zooming in over a city map and clicking on the road link of interest. In 10 seconds there will be a report with all information concerning a comparison of the EU standards. Moreover, the contribution of long range (outside the city), urban (all sources within the city, excluding the road link itself) and local (traffic on the street itself) sources are quantified.

In order to achieve this performance, the SIMAIR system is based on a combination of pre-calculated concentrations from models of different scales and output from local street models that need a few seconds for a one year calculation. The system is based on the handling of true hourly time series, both for meteorology as well as the individually calculated long range, urban and local contributions. This is principally to allow the best possible determination of extreme values, but hourly time series also allow a simpler evaluation against measured time series concentrations.

In what follows, a description will be given of the different parts of the system.

1. Road and traffic information

The Swedish government has issued directives to create a nationwide road database, containing up-to-date information that fulfils particular quality standards. The Swedish National Road Database, NVDB, is unique in its scope and content even on an international scale. The aim of NVDB is to meet the immediate and long-term need for fundamental road information, and to be accessible both to the public and private sectors. SIMAIR uses extracts from the NVDB system to achieve, for all road links in Sweden, individual information such as functional road class, speed limit, number of lanes, road width etc.

The NVDB database contains information concerning traffic volume only on state-owned roads. In order to provide information also on municipality roads, simulations with the traffic demand model SAMPERS and the traffic model EMME/2 are performed. SAMPERS is calibrated for both passenger and goods transport using traffic volumes on state-owned roads and important municipality roads. With the EMME/2 model flows on state-owned roads and major municipality roads are simulated. Traffic flows on other roads are put on the areas of the demand model. These have not been used at the moment, but in a later version of SIMAIR they will be used for calculations of background concentrations. The SIMAIR database will thus contain, for all road links with a major traffic flow, information also of the annual average traffic volume and the heavy duty vehicle percentage.

Emission factors for light and heavy duty vehicles, running under different road conditions concerning speed limits, visibility and accessibility, were taken from the Swedish emission model EVA (Hammarström and Karlsson, 1994). During 2006 the EVA model will be replaced by ARTEMIS (http://www.trl.co.uk/artemis/). Replacing the emission model with ARTEMIS will give large advantages. ARTEMIS represents the most recent knowledge in Europe of emissions from vehicles and will from 2006 also be used for national inventories of emissions from road transport in Sweden. ARTEMIS, and consequently also SIMAIR, will be updated yearly with description of the Swedish vehicle fleet. SIMAIR will thus provide a bottom-up traffic emission inventory for both the regulated air pollutants as well as for certain greenhouse gases.

Critical data for the simulation of the traffic impact on air pollution concentrations in city centre streets, so called street canyons, are the height of the surrounding buildings and the street canyon width, i.e. the distance between the façades of the two surrounding buildings. Although the latter can be crudely estimated from the road width as given in the NVDB database, there is no national register for building heights in Swedish cities. Default values has been implemented in order to yield “conservative” (in this case high) estimates of building heights, however this kind of data should be revised and corrected by the municipality user.

2. Regional background concentrations

Modelling of air pollution and deposition on the regional scale was performed using the multi-scale atmospheric transport and chemistry (MATCH) model (Robertson et al., 1999). The photochemical model includes approximately 130 thermal and photochemical reactions between 59 chemical components. The MATCH model is driven by the weather forecast model HIRLAM (High Resolution

Limited Area Model) and uses a 44x44 km grid over Europe. Emissions are taken from the EMEP 50x50 km inventory (http://www.emep.int). The MATCH model has participated in various model evaluation exercises, latest within the European Tracer Experiment (Warner et al., 2004) where it was ranked as the best model.

The regional background concentrations are evaluated for each city or population area included in SIMAIR. The contributions from sources outside Sweden and inside Sweden are taken together and evaluated directly from the simulated MATCH Europe results. However, measurement data from three rural stations in southern, middle and north Sweden are used to support the PM10 model calculations. This is necessary as the model does not include secondary organic aerosol formation and would otherwise sub estimate PM10 levels.

The regional NO_2 contribution is further separated into the long range contribution from sources outside Sweden and that from sources inside Sweden but outside the city itself. This has been achieved by running another MATCH calculation on a 11x11 km grid (named MATCH Sweden), only covering Scandinavia and with only Swedish emissions taken from SMED (Database for Swedish Emissions to the Environment). The SMED consortium elaborates the emission inventories needed for the report of GHG to the European Commission, obligation under the Climate Convention. Although the geographical resolution required for the EU reports is only 40x40 km, SMED has elaborated emissions on a 1x1 km grid. The calculation is further supported by monthly data from a Swedish precipitation chemistry network with about 25 stations and daily (for ozone hourly) data from some 10 EMEP stations in Norway and Sweden.

The difference between the MATCH Sweden results and the MATCH Europe results, supported by the measurements, is interpreted as the contribution from sources outside Sweden and this part is then distributed over the entire 11x11 km grid through Optimal Interpolation (OI). The basic idea behind this procedure is that the regional contribution from sources outside Sweden varies spatially in a much smoother way than what the calculated differences do. The OI calculations are based on a method used for ordinary meteorological mesoscale analysis (Häggmark et al., 2000).

The regional contribution of NO_2 from Swedish sources will also reflect the influence of the emissions inside the city itself. As the coupled model system includes a separate urban model calculation, a simple addition of regional and urban model output would thus include the urban impact twice. The strict solution is to run the MATCH Sweden model excluding the emissions of the city of interest. However this would require 350 MATCH Sweden calculations extra in order to cover all Swedish population centres. In order to economize this need, a procedure with city masks have been developed. Only MATCH Sweden cells inside specified city areas plus a surrounding 17 km wide buffer zone are “active”. Those masks are applied over the Swedish emissions and used as input to a second run with the MATCH Sweden model. In that run, concentrations are forced to zero as soon as they are advected outside the masked areas. This means that no city can give impact on another, the only impact there are is over the city itself. If the result of the masked MATCH Sweden simulation is subtracted from the first and original MATCH Sweden simulation, the remaining part will express the desired regional contribution. Strictly the subtraction of NO_2 fields should not be done, as NOx chemistry is non-linear.

However, the two model runs uses the same analyzed ozone fields and comparisons with the correct method (only one simulation excluding the emissions from the city itself) show that the subtraction method is sufficiently accurate. A number of 8 masks, each with 10-42 cities included, is sufficient to cover all 350 population centres in Sweden.

3. Urban background concentrations

Urban background contributions are simulated on a 1x1 km grid using two different model approaches. For ground-level sources, e.g. traffic exhausts, an adjoint approach suggested firstly by Berkowicz (2000a) is used. The model is based on the determination of an influence area upwind a receptor point, within which all emissions are aggregated to a final concentration. Each of the cells within the urban 1x1 km grid will then constitute a receptor point. For the calculation of the final concentration, the impact of the emissions within the influence area – that forms a cone - are weighted as a function of distance between the source and the receptor point. The weighting function has a transversal sine-like shape giving more weight to the sources close to the centreline of the influence cone. The other function is a vertical dispersion scale that increases with distance, giving less weight to distant sources. The parameterization of the scaling functions are taken from meteorological parameters like the friction velocity and the convective velocity scale. There is also a parameterization of the height of the atmospheric surface layer, putting an upper limit on the vertical dispersion.

The dispersion of stack emissions are treated in a separate Gaussian point source model, forming part of the DISPERSION model (Omstedt, 1988).

The two urban models are fed with meteorological data taken from the routinely operated MESAN system (Häggmark et al., 2000). All available measurements from both manual and automatic stations, including radar and satellite information, are analyzed on a 11x11 km grid and with 3 hour time resolution. Variables used in the SIMAIR model system include wind speed and direction at 10 m height, temperature at 2 m, cloudiness, global radiation, sensible heat flux, friction velocity, humidity and precipitation.

Emissions as input to the urban model calculations are mainly taken from the SMED database, the same as used for the MATCH Sweden simulation. Primary PM is not included in the SMED databases, so vehicle tailpipe emissions are scaled from NOx emissions. Among PM point sources the most important sector is small-scale wood burning, for which SMHI has compiled a database on a 1x1 grid.

As earlier stated the traffic is contributing to PM emissions both through engine exhaust and through mechanically generated particles mainly due to road wear, the last one being the dominating part in Sweden. For that purpose SIMAIR includes an emission model for non-tailpipe particles (Omstedt et al., 2005). One important parameter in the model that controls the suspension of road dust in the air is road surface moisture. This is calculated every hour from a budget equation taking into account precipitation, evaporation and runoff. During wet conditions a road dust layer is built up from road wear which strongly depends on the use of studded tyres and road sanding. The dust layer is reduced during dry road conditions by suspension of particles due to vehicle-induced turbulence. The dust layer is also reduced by wash-off due to precipitation. Direct non-tailpipe vehicle emissions due to the wear and tear of the road surface, brakes and tyres are accounted for in the traditional way as constant emission factors.

Urban background calculations of PM10 in SIMAIR includes emissions both from the 1x1 km databases as well as non-tailpipe emissions calculated from information of traffic work and meteorological conditions.

4. Local models for roadside concentrations

While regional and urban contributions are pre-calculated, there are two local street models available for direct execution by the SIMAIR users. If the road of interest is not surrounded by buildings or obstacles that reduces the dispersion of traffic emissions (open road conditions), then a Gaussian line source model is used (Gidhagen et al., 2004). However, if there are houses at one or both sides, a street canyon model OSPM (Berkowicz, 2000b) is used. Both models, and in particular the OSPM model, have been evaluated in many different environments. The striking characteristic is that these kind of models are very fast, a one year simulation of hourly values in two receptor points – one at each side of the road – takes a few seconds to complete.

For the open road model geometrical properties of interest is the width of the street and the number of lanes, possibly separated by a median strip. Before the calculation the user specifies the distance of the receptor points from the roadside. Receptor height is pre-set to 2 m. The vertical dispersion coefficient in the Gaussian dispersion calculation takes into account initial values influenced by traffic generated turbulence and then both convective and mechanical contributions. One-hour average concentrations are calculated by averaging concentrations in a wind direction interval, parameterized from measured standard deviations of wind direction.

The OSPM model, as implemented in SIMAIR, needs some geometrical data. Except for road width and number of lanes, also the width between the surrounding houses and house heights on the two sides form part of the road link information. OSPM has one part which is a direct plume model, following the estimated wind direction at the bottom of the street canyon. The other part, taking care of the contribution from the re-circulation, is calculated by a simple box model. OSPM assumes stability conditions inside the street canyon to be neutral. The two receptor points are located at 2 m height and 2 m from the building façades.

Both the open road model and the OSPM model use the earlier mentioned vehicle emission model for non-tailpipe emissions and they also include a calculation method for the transformation of NO to NO_2 developed by Hertel and Berkowicz (1989).

Results

Currently (January 2006) the SIMAIR system is available for approximately 60 Swedish cities, among them the Stockholm Metropolitan area. Although a systematic evaluation of the SIMAIR system against measured PM10 and NO2 levels has still not been initiated, some preliminary comparisons have been made.

Figure 1 shows a comparison from Kungsgatan in Uppsala some 70 km north of Stockholm. This street has, as a yearly average, 18 000 vehicles per day of which 5% are heavy duty vehicles. The measurement site is within a street canyon, although there is small parking lot just where the monitor is located. On the opposite side of the road the house height is 20 m. The width of the street canyon is assumed to be 20 m. The parking lot, normally full, is assumed to constitute a "house" of 2 m height. Of the personal cars 76% are assumed to use studded tyres during the winter.

Figure 1: Measured and simulated total PM10 levels at Kungsgatan, Uppsala, during 2001. Also the simulated urban and regional contributions to total PM10 levels are shown (total concentration is the sum of regional, urban and local (street) contributions. The TEOM measurements have been corrected with a factor 1.2 to compensate for evaporative losses.

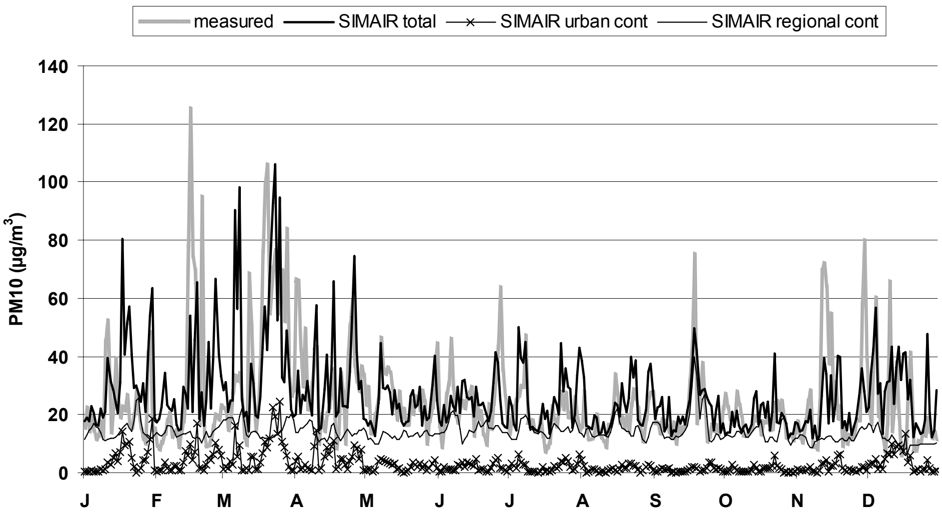

Figure 1 shows that SIMAIR simulates a total PM10 concentration that reasonable well follows the measured time series (output from SIMAIR is hourly values, here the data have been averaged to daily values in order to make the comparison easier to visualize). Note especially the stronger peaks during springtime, this is when the street starts to dry up and large amounts of PM is lifted to the atmosphere due to vehicle induced turbulence. The contribution from the rest of the city (urban contribution) is mostly small, except during the springtime peak of non-tailpipe emissions. Note also the high PM contribution from regional sources, during summer being the dominating part.

In Table 1 the statistical measures of the EU directive (and the Swedish legislation) are given as determined by measurements and by SIMAIR. As can be seen this particular street does not exceed the limit values (40 µg/m^3 for a annual averages and 50 µg/m^3 for 90-percentiles), but the 90-percentile is not far from the latter limit. The goal with the SIMAIR system is to provide a rapid and cheap identification of all roads that may be close to or above the standards, and for which measurement should be performed. Strictly it is not necessary that SIMAIR provides simulated hourly time series that follows exactly measured concentrations, it is enough if the model system outputs annual averages and percentiles (as in Table 1) that gives the same conclusion as if a measurement would have been performed: Does the simulated levels motivate a measurement campaign to be initiated? In the case of Kungsgatan in Uppsala, this is clearly the case.

Table 1: Measured and simulated average and percentile values for PM10 at Kungsgatan, Uppsala, during 2001. The TEOM measurements have been corrected with a factor 1.2 to compensate for evaporative losses.

	measured	SIMAIR
Regional contribution (annual average)		14.0
Urban contribution (annual average)		3.0
Street concentration (annual average)	25.1	27.0
90-percentile (daily averages)	46.1	42.5
98-percentile (daily averages)	76.0	66.2

Table 2 lists the traditional statistics used for model evaluation. A lower RMSE and a higher correlation coefficient can best be achieved by introducing measured time variations of local traffic intensity (in SIMAIR city streets all over Sweden have an identical time variation pattern) and also local data of street maintenance and road surface conditions, the latter determining the emission of road dust. For the Uppsala street, located just north of Stockholm, the non-tailpipe emissions constitute, as an annual average, 75% of total vehicle PM10 emissions. However, during springtime when then streets dry up and the vehicles still use studded tyres, the non-tailpipe emission of PM10 particles raises to 10-15 times the mass of the engine exhaust particles. During summer conditions vehicle emissions of PM10 are more similar to those of central Europe, with about similar sizes of the road and tyre wear part as compared to tailpipe combustion particles.

Table 2: Comparative statistics of the measured and simulated daily average values of PM10 at Kungsgatan, Uppsala, during 2001.

	measured	SIMAIR
Number of data points	364	364
Average	25.1	27.0
Standard deviation	17.4	14.2
RMSE	16.4	
Correlation coefficient	0.47	

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Mapping of air quality and human exposure along motorways

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Abstract

The Danish regulatory air pollution model OML, which originally was developed for use with stationary pollution sources, has now been adapted for modelling pollution from motorway traffic. The main model extension is in implementation of the traffic-produced turbulence into description of the Gaussian plume dispersion parameters. The traffic-produced turbulence has a significant influence on the dispersion conditions close to the traffic sources. The model parameterisation was validated against a three month measurement campaign of NOx and NO_2 at different distances from a busy section of a motorway. Subsequently the model was applied for mapping of air quality and human exposure along the entire motorway network of the County of Roskilde in Denmark. Modelling was carried out for all residential addresses up to a distance of 1,000 m from the motorway, and results were compared to European Union limit values for NO_2.

Keywords: *Air quality, model, measurements, human exposure, motorways, GIS.*

Introduction

The County of Roskilde in Denmark has some of the busiest motorways leading to Copenhagen and plans exist for enlargement of these motorways. Against this background the county wanted to through light on the environmental conditions along the motorways and the impacts for the citizens living along the motorways. Therefore, the objectives of the project were to develop an air quality model for motorways, validate the model against a measurement campaign and apply the model for mapping of the air quality and human exposure along the entire motorway network in the County of Roskilde. Such a study has not previously been performed in Denmark and therefore provides a new insight into pollution levels from motorway traffic.

Methodology

1. Measurement campaign

Continuous measurements of nitrogen oxides (NOx) encompassing hourly data on nitrogen monoxide (NO) and nitrogen dioxide (NO_2) were carried out during a three month campaign in 2003, (from September 17 to December 18) along the most busy motorway section in Denmark - the Køge Bugt Motorway that carries about 100,000 vehicles per day. The motorway section is part of the large motorway network in the Greater Copenhagen Area. Three monitor stations were located south-east from the motorway in a line perpendicular to the motorway at distances of 1m, 50 m and 100 m from the edge of the outermost emergency lane. Additionally, a fourth monitoring station was located at a distance of 259 m, north-west of the motorway. The measurements from this station were used for determination of the background pollution in the case of winds blowing towards the three other stations. A meteorological mast equipped with a sonic anemometer was also located at this background station and the data from this mast were used as the meteorological input for the model (Jensen et al 2004).

2. Model development

The focus of this paper is on application of the model for mapping of air quality and human exposure along motorways and only a summary description of the model is provided here. More details are given in Jensen et al. (2004).

OML is a modern Gaussian plume model intended to be used for distances up to about 20 km from the source (Olesen et al. 1992). The source is typically one or more stacks, and possibly also area sources. Typically, the OML model is applied for regulatory purposes. In particular, it is the recommended model to be used for environmental impact assessments when new industrial sources are planned in Denmark. The model can be used for both high and low sources. It is a characteristic of the OML model that it does not use traditional discrete stability categories, but instead describes dispersion processes in terms of basic boundary-layer scaling parameters, such as friction velocity, Monin-Obukhov length, and the convective velocity scale. Thus, before being used by the model, meteorological measurements must be processed by a pre-processor. In the OML model, the Gaussian dispersion parameters σ_y and σ_z are not - as in conventional operational models - functions only of stability category and distance from the source. Instead, they are continuous functions of several boundary layer parameters. The dispersion parameters are regarded as the result of contributions from several mechanisms: convective turbulence, mechanical turbulence, plume buoyancy and building downwash. Their dependence on source height is taken explicitly into account.

In order to adapt the model for description of pollution from motorway traffic an additional turbulence production mechanism was introduced – the traffic-produced turbulence. The traffic-produced turbulence has a significant influence on the dispersion conditions close to the traffic sources and neglecting this term leads to a significant overestimation of the resulting pollution levels. The largest overestimation occurs under low wind speed conditions when the traffic-produced turbulence is the dominant turbulence production mechanism. The concept used to model the traffic-produced turbulence is similar to the method applied in the Operational Street Pollution model (OSPM) (Berkowicz 2000a). However, in order to take into account the diminishing effect of the traffic-produced turbulence with the distance from the motorway, an exponential decay term was introduced. The parameterisation of this decay term was deduced from analyses of the campaign measurements and comparison with model results.

An important feature of the model is that the motorway emissions are treated as area sources and not as a single line source. The along-wind extension of the traffic source has a significant influence on the concentrations calculated at distances close to the traffic. Due to the method by which the calculations for area sources are made in OML, each of the traffic lanes is divided in a number of small rectangular area sources.

3. Traffic and emissions

Detailed traffic data from each lane (3 lanes in each direction) was collected based on automatic counting equipment that the Danish Road Directorate permanently has running on the chosen motorway section. The time resolution on vehicle counts was 15 minutes that was aggregated to hourly data. The counting equipment identifies the numbers of vehicles with a high certainty but the grouping into vehicle categories is too uncertain. Therefore, the vehicle composition (passenger cars, vans, small trucks, heavy trucks, buses) was based on a few manual counts carried out by the Danish Road Directorate close to the chosen motorway section. The traffic data was used to generate hourly diurnal profiles for each vehicle category for working days, Saturdays and Sundays. Similar emission profiles were generated based on emission factors from the emission module of the OSPM model. Validation of emission factors under urban conditions are given in Berkowicz et. al. (2006) and Ketzel et al. (2003).

For the model evaluation campaign a total length of 1,483 m of the motorway section was considered as emission source around the monitor stations (742.5 m in each direction). Each lane of the motorway was described with rectangular area source of 4 m x 37 m.

The NO_2 concentrations are calculated in OML using a simple chemistry model, which describes formation of NO_2 due to oxidation of the directly emitted NO by ozone. However, the fraction of NO_2 which is also directly emitted by traffic is not negligible. For this study this fraction was assumed to be 15% but a large uncertainty connected to this value must be expected. Very little is known about direct NO_2 emissions under highway driving conditions and with a large proportion of heavy-duty diesel vehicles. However, recent studies for urban conditions indicate that primary NO_2 exhaust emission fraction has increased in recent years due to increased penetration of diesel vehicles and particle filters, see Carslaw & Beevers (2005).

4. Mapping of air quality and human exposures along the motorway network

The model was applied for mapping of air quality and human exposure along the motorway network in the county of Roskilde (about 70 km motorway) within a distance up to 1,000 m from the motorway. Calculations were carried out for 2003 (Jensen et al. 2005a).

A new GIS application in Avenue (ArcView GIS 3.3) was developed to semi-automatically generate input data for the air quality model for the entire motorway network (emission sources, receptor points and exposure areas).

In the air quality model the motorway was represented as rectangular area sources e.g. one direction with three lanes was represented by 11 m x 110 m rectangular area sources. This was done to simplify calculations.

To be able to calculate human exposure, exposure areas were defined with a length up to 1,000 m and up to a distance of 1,000 m from the motorway with categories of: [0..25m[, [25..75m],]75..150m],]150..300m],]300..600m],]600..1,000m]. Receptor points were generated in the centre of the exposure areas at distances of 50 m, 112 m, 225 m, 450 m and 800 m. All addresses located within an exposure area are assigned the concentration level modelled at the receptor point.

Background levels were modelled with the Urban Background Model (Berkowicz 2000b) for a receptor grid net of 1x1 km^2 covering the motorway network. A GIS based traffic and emission database for the Greater Copenhagen Area that includes all roads was used to provide emission data (withdrawing the emission from the motorway network to avoid double counting). Traffic was based on modelled traffic data from the 'Ørestadstrafikmodel'. However, for the motorway network traffic levels, vehicle composition and travel speed was based on data from the Danish Road Directorate. The regional air pollution contribution was based on measurements from a representative regional background station (Keldsnor) and meteorological data was from another representative regional background station (Lille Valby). The variation in modelled NO_2 background concentrations was from 12 $\mu g/m^3$ to 15 $\mu g/m^3$. Since the variation was modest, four classes of background conditions were defined and each section of the motorway network was allocated to one of the four classes.

Calculations were carried out for NO_2 and PM_{10} and compared to European Union ambient air quality limit values (Council Directive 1999).

In the model calculations it is assumed that the dispersion is in flat terrain. The dispersion will be affected in locations where the motorway is below the terrain or noise barriers/earth banks are located along the motorway. In these cases, the initial dispersion will probably be larger leading to lower concentrations further away from the motorway. The model is likely to overestimate concentrations under these conditions. The model does not take into account the influence of building topography and its restriction on dispersion. Hence, the model would largely underestimate concentrations under street canyon conditions. However. such conditions does not exist at the Danish motorway network.

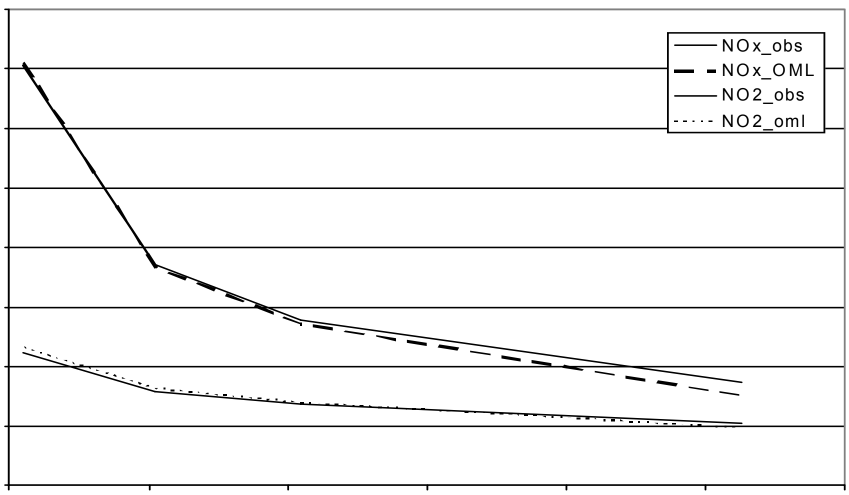
The model only includes traffic as source. Emissions from space heating, district heating plants and industrial sources were not considered. The contributions from these sources were not quantified. However, a qualitative assessment indicates that NOx emissions from these sources are of less importance compared to traffic. Though, particle emissions from wood burning used for residential space heating might influence PM_{10} concentrations along motorways provided a high density of wood burning stoves in residential areas next to the motorways.

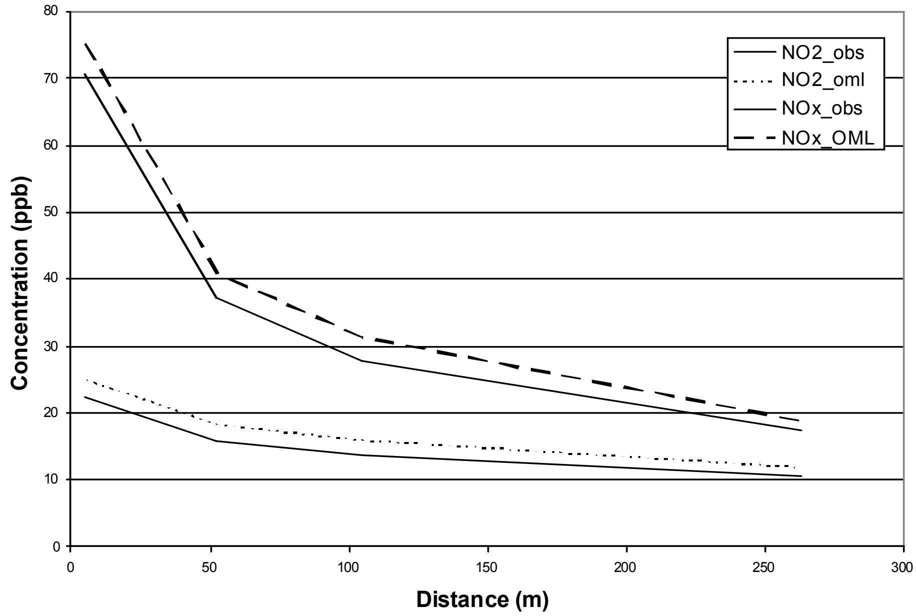
Results and discussion

1. Application of the model for mapping of air quality and human exposure

To be able to apply the model for mapping purposes it was necessary to simplify the description of the motorway network. For model development and validation the motorway was represented by rectangular area sources of 4 m x 37 m representing each lane individually. However, detailed traffic data does not exist on a lane by lane basis. Therefore, the motorway was described as rectangular area sources of 8 m x 80 m for 2 lane motorways and 11 m x 110 m for 3 lane motorways for each direction. This means that the emission is evenly distributed over one direction and not on individual lanes. The emissions are not evenly distributed since traffic levels are highest in the centre lane of a three lane motorway and lowest for the lane closest to the centre strip. However, model calculations and subsequent comparison with measurements showed that the difference between one direction and individual lanes was minimal.

Figure 1: Illustration of the impact of considering different lengths of the motorway in model validation. Upper 1,483 m and lower 10,000 m. All wind directions are included. 'Obs' is measurements and 'OML' is modelled values.





Apart from a slightly different emission distribution and larger area sources, the emission contribution from the entire motorway network is now considered. For model development and validation only 742.5 m of the motorway section was considered in each direction around the monitor stations. To test the impact of considering a larger part of the road network model calculations were carried out including 5,000 m of the motorway section in each direction around the monitor stations (Figure 1). It showed that model calculations overestimate measured concentrations by approx. 7-13% for NOx and 12-17% for NO_2 depending on the distance to the motorway. This means that a larger part of the road network should be been taken into consideration in the model development and validation.

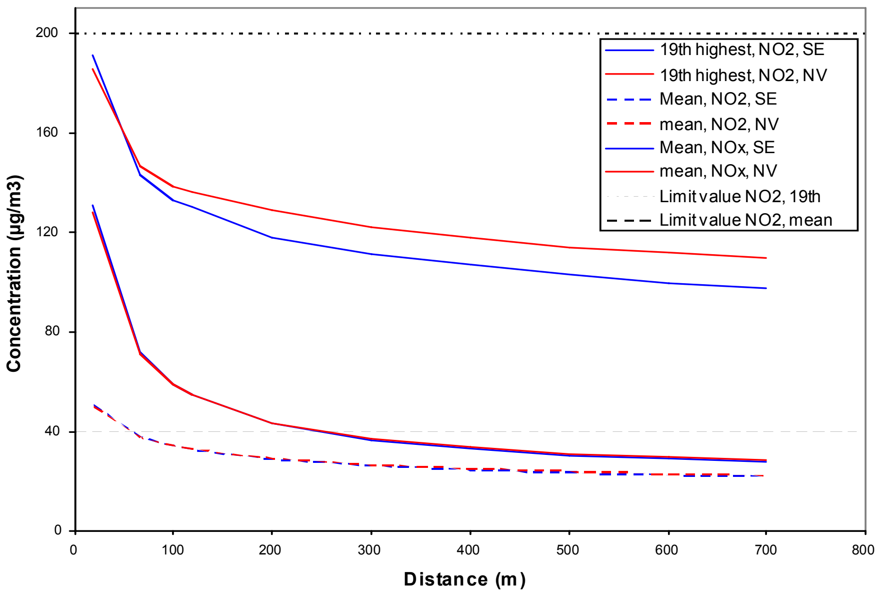
2. Dependence of air quality on distance to the motorways

As expected the concentration levels of NOx and NO_2 decline fast with distance to the motorway, and NO_2 levels diminished slower than NOx due to chemical processes involving ozone. NO_2 decreases slower than NOx with distance since NO_2 is formed due to reactions between NO and O_3. These reactions take some time. The major part of NOx is emitted as NO as only 15% of NOx is assumed to be emitted directly as NO_2. Non-reactive species like CO, benzene and PM_{10} are assumed to show a similar pattern as NOx.

The measurements show that concentrations continue to decrease to the farthest off monitor station located 259 m from the motorway indicating that the background concentration levels have not been reached at this distance. This is also supported by the fact that the regional background station (Lille Valby) measures lower levels than the monitor station at the distance of 259 m.

In Figure 2, the dependence of NOx and NO_2 concentrations on distance to the motorway is showed for modelled data for the entire year of 2003 at distances up to 700 m from the motorway.

Figure 2: Dependence of NOx and NO_2 modelled concentrations in relation to distance from the motorway for 2003. Same place as the measurement campaign. 'SE' is South East and 'NV' is North Vest in relation to the motorway. Note that the concentration profile is almost similar for both sides of the motorway since the frequency of wind directions and speeds are almost similar on both sides of the motorway.



3. Air quality along motorway network in 2003

In Figure 3, NO_2 concentrations are showed on all addresses within 1,000 m of the motorway network.

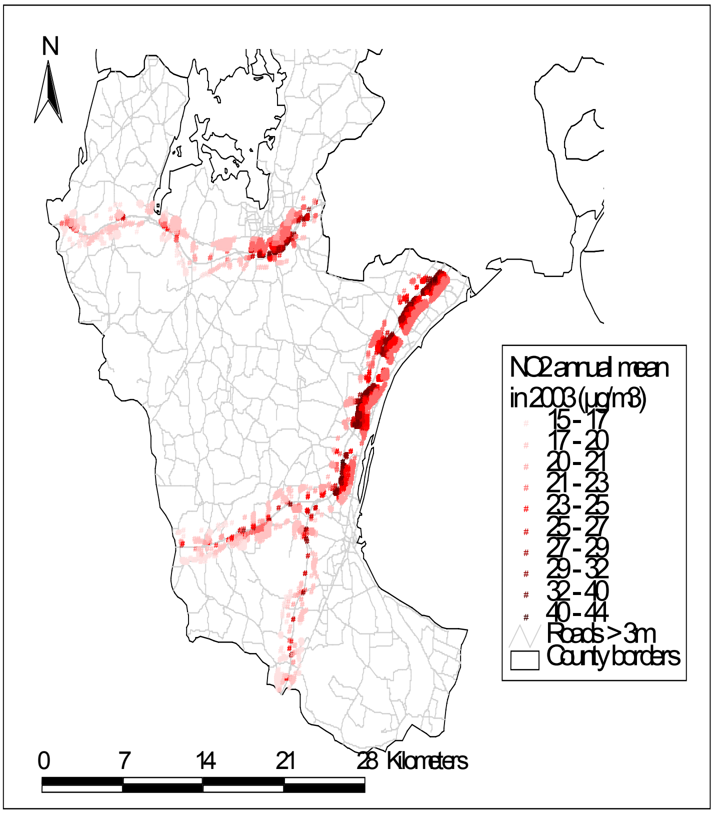
The highest concentrations were found where the traffic levels were highest. However, a relatively high percentage of trucks also results in relatively high levels of NO_2 concentrations although traffic levels were moderate.

The modelled background concentrations were in the range of 12-15 µg/m^3 and the motorway contributed with 3-29 µg/m^3. This means that there is a small contribution from the motorway even at a distance of 800 m that is the farthest off calculation point representing the exposure zone at a distance of 600-1,000 m.

Only three addresses (11 persons or 0.2%) in 2003 exceeded the European Union limit value of 40 µg/m^3 for NO_2 to be met in 2010. The limit value plus margin of tolerance (54 µg/m^3) in 2003 was not exceeded. The highest calculated NO_2 level at an address was 44 µg/m^3. These levels are quite small compared with NO_2 levels observed in urban areas (Jensen et al., 2005b).

A rough assessment was made of the expected development of NO_2 concentration towards 2010 where the limit value of 40 µg/m^3 for NO_2 has to be met. The Danish Road Directorate expects traffic levels on motorways to increase by 17% from 2003 to 2010. On the other hand, NOx emissions are expected to decrease in the order of 20% (Jensen et al. 2005b). Since increase in traffic levels and decrease in emissions in 2010 seems to balance one another out NO_2 concentration levels in 2010 can be expected to be as in 2003. The model used for mapping also slightly overestimates concentrations. Therefore, only few addresses can be expected to be close to or violate the NO_2 limit value in 2010 based on these rough estimates.

Figure 3: Annual mean of NO2 in 2003 for all residential addresses within 1,000 m of the motorway network in the County of Roskilde.



4. Human exposure along motorway network in 2003

Approximately 56,000 people live along the motorway network in the County of Roskilde within a distance of 1,000 m from the motorway. Only 225 people live very close the motorways (within 25-75 m). No difference could be observed in the age or gender distribution depending on the distance to the motorway. An accumulated distribution function of population exposure to NO_2 in 2003 for people living within 1,000 m of the motorway showed that e.g. 10% of the people live at addresses exposed to annual NO_2 levels above 28 µg/m^2.

5. Assessment for PM10

Since PM_{10} (particles less than 10 micrometer) was not measured during the measurement campaign it has not been possible to validate the model against PM_{10}

measurements. Therefore, assessment of PM_{10} has only been based on model results. Model results showed that the contribution from the motorways was in the range of 0.2 to 4.8 µg/m^3. However, the contribution is uncertain due to the limited knowledge about particle emission factors especially at high travel speeds. The regional background concentration in 2003 was about 24 µg/m^3. PM_{10} concentrations along the motorway network is therefore between 24 and 29 µg/m^3 in 2003. The contribution from the motorway is relatively small compared to the dominant regional background contribution. Modelled PM_{10} concentrations in 2003 are well below the European Union limit of 40 µg/m^3 to be met in 2005. PM10 concentrations are monitored at Jagtvej in Copenhagen, an urban street canyon with Average Daily Traffic of about 29,000. Average NOx concentrations are similar to that measured close to the motorway with vehicle levels at about 100,000 per day illustrating the influence on dispersion of the building topography. At Jagtvej, the local vehicle PM10 emission contributes with about 10 µg/m^3 out of an average concentration level of 33 µg/m^3 in 2003. Close to the motorway, it would be expected that the contribution from the motorway would be at the same level but it is only about half. The reason is that the present PM10 emission module estimates less non-exhaust for higher speeds as a combined effect of less brake and tire wear, and re-suspension is kept constant. Furthermore, the exhaust decreases for heavy-duty vehicles with higher speed whereas it increases for light-duty vehicles. These relations are still uncertain. However, it is likely that the present emission module underestimates PM10 emissions at higher speeds. Measurements from the German BAB II study give some indication that the estimated levels may be underestimated since the study showed that the contribution to PM10 from a motorway with traffic levels of about 62,000-65,000 vehicles per day is about 5 µg/m^3 out of 27 µg/m^3 measured at a distance of 60 m, see Rosenbohm et al. (2005).

Conclusion

An air quality model was developed to describe the conditions along motorways. The model is based on the Danish OML model and integrates the parameterisation of traffic turbulence from the Danish OSPM model. The model was validated against a three month measurement campaign of NOx and NO_2 at a busy section of a motorway. The validation showed good agreement between modelled and measured data. Modelling was carried out for the year 2003 for all residential addresses up to a distance of 1,000 m from the motorway network in the County of Roskilde, Denmark. As expected the concentration levels of NOx and NO_2 declined fast with distance to the motorway, and NO_2 levels diminished slower than NOx due to chemical processes involving ozone. Mapping of the air quality along the entire motorway network in the County of Roskilde showed that only 3 addresses (11 persons out of 56,000) were exposed to annual NO_2 concentrations above 40 µg/m^3. This is the European Union limit value to be met in 2010. The limit value plus margin of tolerance in 2003 (54 µg/m^3) was not exceeded as the highest calculated value was 44 µg/m^3.

Acknowledgement

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How to estimate roadworks emissions factors from traffic and air quality monitoring measurements - A methodological approach

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Abstract

Impact assessment studies are imposed by regulations for important constructions, and, in the case of roads, it is generally focused on the prediction of the effects of traffic, but few data are available on pollutant airborne emissions from roadworks. A road repairing has occurred in 2002 near a traffic air quality monitoring station in Dunkerque, France. Different pollutant air concentrations and meteorological parameters were daily collected by "Atmo Nord Pas de Calais", before, during and after the roadworks. With the help of Dunkerque city, activity data on vehicle flows, work schedule, work devices were put together with pollutants data in order to elaborate a database. This paper aims to extract some quantitative elements about the atmospheric pollution related to conventional species around a road work place. Non-parametric regressions were used to select the data suitable for linking air concentrations to pollution emitted from road. A dispersion model was used to calculate the meteorological function gouverning pollutant dispersion for road emissions. It is then used to find out road maximal impact on concentrations measured at sensor. Regression between traffic and concentrations is used to assess road works and traffic emissions during repairing. Road works contributions are found weak, and they are discussed in terms of representativeness and fiability.

***Keys-words**: roadwork, air monitoring, pollutant emissions, pollution pictures,*

Introduction

Decisions for rehabilitation of road sections in urban area can not only rely upon technical criteria. Notably, road works can generate nuisances and pollutions. Many studies are interested in estimating impacts of traffic on air pollutions, but few investigate around road works themselves, although some publications from the US-EPA (2003) shows that air pollutions due to global non road equipments are far from being immaterial. In June 2002, in Dunkerque city, the rehabilitation of a road section pavement, has been run very close to an air quality monitoring station. This road work has been the departure point of collaboration willing to use available data to estimate the possible effects of the rehabilitation operations on air quality. Initiated by LCPC, the collaboration regroups several partners : Atmo Nord Pas de Calais supplied average pollutants concentrations (CO, NO, NO_2, SO_2, O_3 and PM10) and meteorological data from their monitoring stations with an hour time interval, before and during the road works period ; Dunkerque city associated LCPC to the road works weekly reports and detailed maps, provided technical specifications of the new road, and also provided traffic flows nearby the road section ; CEREA and LCPC used available data with the objective of extracting the contribution of the road works to air pollution, from the one of the traffic.

Figure 1: Positioning of road works and traffic a) before, b) after 2002 dec. 16th

Presentation of the case study and notations

The quai des Hollandais rehabilitation road works consisted in passing from 2 lanes per direction to one lane per direction. Vehicules circulation was modified according to the road works progression as shown on Figure 1. During the first phase (from september 16th 2002 to december 16th 2002), vehicles were circulating nearby the docks while the ancient road was deconstructed and the new road was built, and during the second phase (from december 16th 2002 to end of june 2003), vehicles were circulating on the new road. Three traffic counters were situated nearby the quai des Hollandais as shown on Figure 2: n°7 on the pier itself, n°6 and n°8 at each extremity of the Fusillés Marins street. Although the traffic counter n° 7, located on quai des Hollandais, was the only one representing the traffic on the pier, it had to be removed during road works. Counter n°6 is the only one that has been functioning during the whole road works period. A significant correlation (CT7 = 0.75 x CT6, r^2=0.97) has been established between counters 6 and 7 for April and May 2002. The same daily pattern was observed for both counters. This correlation has been assumed to be representative of the whole period, but it slightly under predicts traffic flow during nights . The vehicles flow is noted N_{veh} and expressed in $véh.hr^{-1}$. For meteorological parameters, The wind direction angle given by the monitoring station is noted θ and corresponds to 90° for the East. The notations used in the following are given here. A wind direction angle α corresponding to □30°, has been used in some programs, order to avoid including periodicity into calculations. The wind velocity is noted u. The hour of measurements, expressed in universal time unit, is noted H. Concentrations measurements, noted C_i for pollutant i, are stored in matrixes noted $[\theta,H]$ or $[\alpha,H]$, and $[\theta,N_{veh}]$, $[\alpha,N_{veh}]$. Extreme pollutants concentrations were rejected and only values under the percentile 99 were kept.

Figure 2:Spatial configuration of monitoring equipment

Studies of local dispersion of pollutants :

The relationship between pollutant concentrations and traffic flows has been studied for street canyon geometry by Palmgren (1999) and Berkowicz (2005). Concentrations were written as the sum of the urban background concentration (an upwind concentration in fact) and local contribution as shown in Eq. 1 :

$$C_{sensor} = C_{background} + Q(N_{veh}).f(met) \qquad \text{Eq. 1}$$

where *f(met)* is the meteorological function depending upon the wind velocity and direction, as well as atmosphere stability, and $Q(N_{veh})$ is a function of the vehicle flow.

In the Quai des Hollandais case, the street geometry is not a canyon : they are no buildings on the north side of the street, therefore the Palmgren (1999) model can not be used. UCD2001 is a Gaussian model developed by Held (2003) for road pollutant dispersion similar to our site. Road link is represented by series of finite point sources in the mixing zone above the traffic lanes. Each source contributes to the pollutant concentration measured at (x,y,z) according to the following solution for point source (Eq. 2):

$$C_s = dQ(N_{veh})\frac{1}{\sigma_y\sqrt{2\pi}}\exp\left[-\frac{(y-y_s)^2}{2\sigma_y^2}\right]\frac{(zz_s)^{\frac{(1-n)}{2}}}{xb\alpha} * \exp\left[-\frac{a(z^\alpha+z_s^\alpha)}{xb\alpha^2}\right] I_{-v}\left[\frac{2a(zz_s)^{\frac{\alpha}{2}}}{xb\alpha^2}\right] = Q(N_{veh}).f_s(met) \quad \text{Eq. 2}$$

$$\text{where } \alpha = 2+p-n, \quad v = \frac{1-n}{\alpha}, \quad u = az^p = az^{0.25}, \quad \sigma_y = c+dx^e, k = bz^n = 0.28z^{0.81},$$

The sensor concentration is the sum of these contributions (Eq. 3)

$$C_{sensor} = C_{background} + \sum_s C_s = C_{background} + Q(N_{veh})\sum_s f_s(met) = C_{background} + E_{veh}.N_{veh}\sum_s f_s(met) \quad \text{Eq. 3}$$

where E_{veh} is the emission factor of vehicles.

To apply the model, it is necessary to provide the background concentration, the source intensity and the meteorological data (u, θ) as well as road geometry. As measurements do not bring any information about the background concentrations, the UCD 2001 model can not fit either with the collected database. But used with zero background values and uniform emission, it provides information about the road pollution seen by the sensor for a given meteorology. This kind of model provides concentration prognostics made on an hourly basis: pairing prognostics and measurements allows estimating the required emission minimizing their difference. Due to the lack of background values, it was not possible to analyse measurements on an hourly basis. We used therefore the regression method to average concentration measurements during the periods with and without road works. Averaging smoothes the temporal variability of pollutant concentrations and let appear the picture of roadwork impact.

Methodological approach

Non parametric regressions method was used by Yu (2004) to identify the impact of an airport on a air quality monitoring site in Los-Angeles, as a function of wind velocity and wind direction. It was found to be relevant to understand pollution picture, especially when used with periodic variables like wind direction and hour of measurements, and to observe non linear relations between pollutant concentrations and other variables. Furthermore, this method ensured data smoothing and avoided blank values in the concentration matrix, that is calculated using Eq. 4 :

$$C(a,b) = \sum_{i=1}^{N} K_1(x_{1,i})K_2(x_{2,i})C / \sum_{i=1}^{N} K_1(x_{1,i})K_2(x_{2,i}) \quad \text{Eq. 4}$$

σ is the smoothing coefficient, increasing the fuzziness of results . No systematic way was found to choose the value of σ, that was determined using testing sets. K_1 and K_2 are weight functions, called kernel functions, affected to the observation i.

To avoid the necessity of background concentrations, values of meteorological parameters were searched, where the Quai des Hollandais has a maximal impact on the measured concentrations of the monitoring station. This is done by using non parametric regression to elaborate a pollution picture of the site. To observe the relationship between traffic and concentration without any assumptions about the shape of this link, the Yu (2004) non parametric method was extended for that study to other variables like traffic vehicles flows N_{veh}. K_1 and K_2 were chosen as gaussian kernel, affecting a high weight if (G_1,G_2) are simultaneously closed to (a,b). K_1 is expressed by Eq. 5 (also available for K_2) :

$$K_1(x_{1,i}) = K_1\left(\frac{a-G_1}{\sigma_1}\right) = (2\pi)^{-1/2} exp\left(\frac{-(a-G_1)^2}{\sigma_1^2}\right) \quad \text{Eq. 5}$$

G_1 and G_2 of are the descriptive variables chosen in the analysis (N_{veh}, H, α, θ in our case).

After selection of wind directions angles corresponding to the Quai des Hollandais contribution, the Yu (2004) non parametric regression was used to compare pictures with and without road works.

The UCD2001 model was used to check the interpretation of the picture. Maximal impact zone, defined by Eq. 6 was found from both the model and from regression pictures.

$$\theta_{\max} / \Sigma f_s(met) = \Sigma f_s(\max) \quad \text{Eq. 6}$$

Non parametric regression for suitable wind direction angle θ_{max}, provided an average concentration given by Eq. 7.

$$C_{sensor}(N_{veh}, \theta_{\max}) = g(N_{veh}) \quad \text{Eq. 7}$$

For high traffic flows and θ_{max} wind direction, it was assumed that the background contribution was negligible compared to the local contribution of road. Road works are very closed to the road and they are considered as road emissions. The road works emissions are therefore expressed as an additional pollutant flux as follow :

$$N_{veh}(P_{work}) = N_{veh}(P_{without}) + N' \quad \text{Eq. 8}$$

Estimate of roadwork contribution is then given from the numerical integration of the curve (Eq. 9):

$$\overline{g(N_{veh})} = C(N_{veh}, \theta_{\max}) \quad \text{Eq. 9}$$

leading to Eq. 10 :

$$N' = [C(P_{work}) - C(P_{without})] / E_{veh} \Sigma f(\theta_{\max}) = [C(P_{work}) - C(P_{without})] / \overline{g(N_{veh})} \quad \text{Eq. 10}$$

Results

1. Determination of suitable wind direction angles for quai des Hollandais contribution

Concentration maps during the total studied period are shown on Figure 3. Around the 270° /North direction High concentrations are observed, all the day long and for quite all the pollutants. This θ line of high concentration levels corresponds to industrial activities located in the harbour area, confirmed by high SO_2 concentrations. Low concentrations are observed for all pollutants in a large range around 200°/North. Regional pollution is generally small for these wind directions, that are distributed along the coast and perpendicular to industrial area. At local scale, the sensor is also upwind to the road. Around the 120°/North direction, NOx, PM10 and CO concentrations are found to be intermediate all the day long, with a slight increase at the morning and evening peak hours. This is related to wind coming from the city with low velocities and pollution tends to stagnate due to unfavourable dispersion conditions. Around the 50°/North direction, high concentrations of NOx, PM10 and CO are also observed, particularly between 8 h and 18 h. This is the typical signature for road emissions, downwind from the Quai des Hollandais. Finally, most of the road impact was found to be observed for measurements between 330 and 70 °/North directions.

Concentration matrix $[\alpha, N_{veh}]$ with is computed with non parametric regression using selected wind directions (α in [0, 100] for θ in [330,70] °/North) to keep only situations where traffic of quai des Hollandais seems to have its maximal impact on the monitoring station. As seen on Figure 4, the SO_2 and NOx concentrations are high for α values near 0° and 100°. For SO_2, concentrations increase with the traffic flow N_{veh} for α being comprised between 0 and 40°. This relationship between traffic and concentrations disappears for α being comprised in [40, 100], where higher SO_2 concentrations are observed higher for intermediate traffic. This was also seen for PM10 and CO, although not shown here. This indicates other pollution sources acting for α comprised between 70 and 100°. Thus, surprisingly, SO_2 concentration seems more properly related to traffic although it is considered in general as a poor tracer of automotive activities. Regression with traffic allows to identify the wind directions corresponding to the minimal concentrations for a same traffic count. But we are not able to check the angular value for the maximal impact because of discrepancies between pollutants.

Figure 3: Concentration at the monitoring station versus θ wind direction and H hour of day: a) NOx, b) SO_2 (for SO_2, only concentration under the percentile80 were retained)

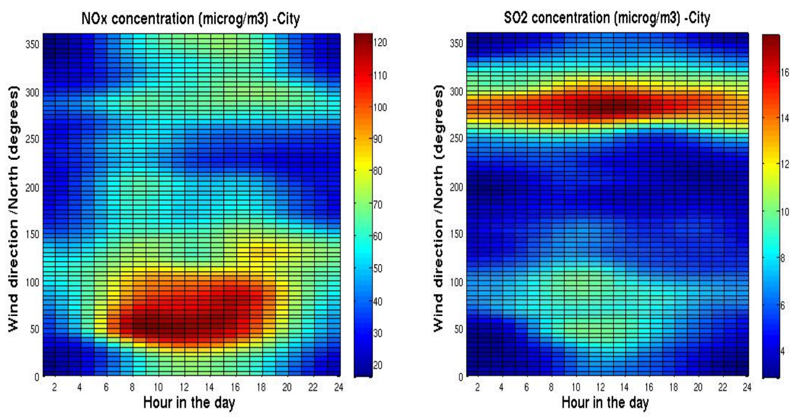
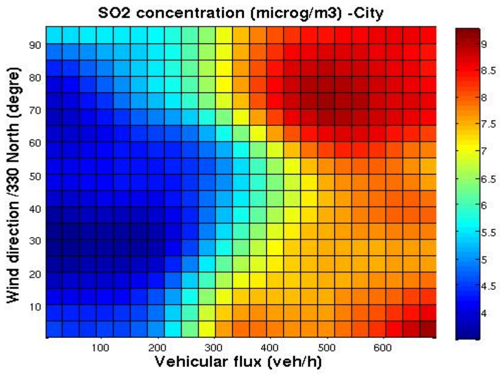


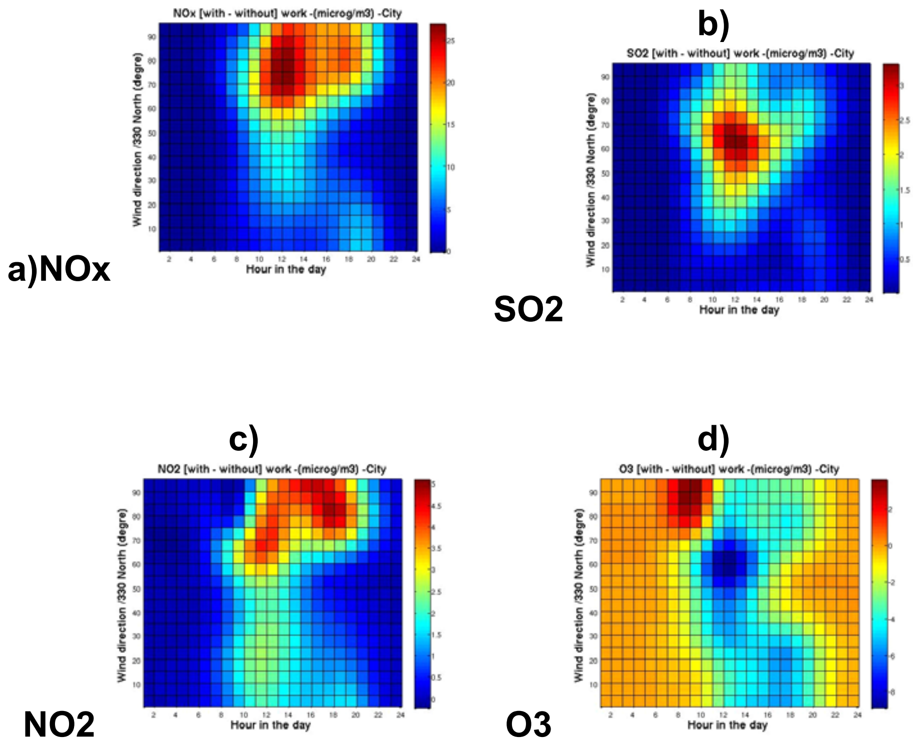
Figure 4: SO_2 concentrations at the monitoring station versus modified wind direction α (= θ + 30°) and vehicle fluxes Nveh



2. Comparison of periods with and without roadworks by non parametric regression

The concentrations matrixes [α,H] and [α,N_{veh}] are computed with non parametric regression, for α being comprised between 0 and 100°, separately for P1 and P2 periods (with and without roadworks respectively). Figure 5 shows (P1 - P2) concentration matrix as a function of [α,H] for all the pollutants. SO_2 (Figure 5b) gives the most clear picture of roadwork impact. The maximal influence of roadworks is seen for α ranging from 60° to 70° in the middle of the day, which is in agreement with the daily period for works and is in the road axis. This angle value is defined as θ_{max} ranging from 30° to 40°. The O_3 (Figure 5d) is a secondary pollutant which is removed by conversion of NO into NO_2 near traffic sources, see Kukkonen (2001). An area where ozone is removed is located at the same place where the supposed roadwork impact is seen for SO_2. This supports the conclusion for a very near pollutant source in the plume of whom rapid chemical reactions are modifying concentrations. For NOx and NO_2 (Figure 5a and c), maximum differences between P1 and P2 concentrations also occur for an area located around θ_{max} (at α = [60,70]) and for daily hours between 9h and 17h. An increase of concentration is also observed in the evening, H between 12h and 20h and for α above 70°. The angular position of this rise is to bring together with the previous observations of Figure 3. This area may correspond to a more distant source, perhaps the traffic junction with rue Jean Jaurès (see Figure 2): roadworks on Quai des Hollandais may influence the traffic in the junction, what could also explain differences between with and without period seen on Figure 5.

Figure 5: Difference between concentrations with and without roadworks at the monitoring station versus modified wind direction α (= θ + 30°) and hour of the day H



3. Interpretation of pictures using ucd 2001 model

The set α[70,100]*H[12,18] are, in average, more windy situations, which could bring some pollution from a more distant source. The UCD 2001 model was used to calculate the meteorological function from measured concentrations, using Eq. 7 and for several wind direction angles as shown on Figure 6. It can be observed that for small wind velocities, the meteorological function is weak. For θ_{max} = 35°, the maximum value is observed around 4 m/s of wind velocity, after what the function value decreases. Between 2 and 8 m/s wind velocities, variations of the impact coefficient stay in a ± 10% range. For wind direction, previously supposed to correspond to with rue Jean Jaurès traffic junction (θ = 75°), the meteorological function also reaches a maximum value, which is equivalent to the one for θ_{max}, but shifted in the 6 to 8 m/s range of wind velocity. This fits with the assumption that pollutants in that area come from a farer location.

The Figure 7 represents the average concentrations of SO2, CO, NOx and PM10, for θmax = 35°, distinguishing P1 (with roadworks) and P2 (without roadworks) periods. Obviously, the general trend of P2 curves shows that concentrations increase when the traffic increase for all pollutants. The P2 period is mainly related to night hours : the local contribution of road should be small during nights compared to the urban background values. It is also noticeable that the linear link between traffic flows and concentrations is not verified for high traffic flows values. Without any data about vehicle speed, it is difficult to estimates if this change is related to vehicular emission changes or to dispersion condition changes. Similar values of concentrations are found for P1 and P2 periods at low traffic flows. It reveals comparable background concentrations for the two series, except for SO2. The average hourly traffic was also found similar during the P1 and P2 periods : between 200 and 600 veh.hr-1 during the working hours.

4. Assessment of roadworks contribution to air pollution

From Figure 6, the meteorological function was assumed to remain constant for the considered θ_{max} value, and calculated from Eq. 9. An estimate of the concentration rise per vehicle was calculated from Eq. 10 using curves obtained in Figure 7 and for N_{veh} ranging from 200 to 600 veh.hr^{-1}. The average estimate for work emissions is therefore expressed in equivalent vehicle. It gives the number of additional vehicles required to reach the concentration measurements during repairing. Results are presented in the Table 1.

Figure 6: Meteorological function calculated from the UCD2001 model for different θ values

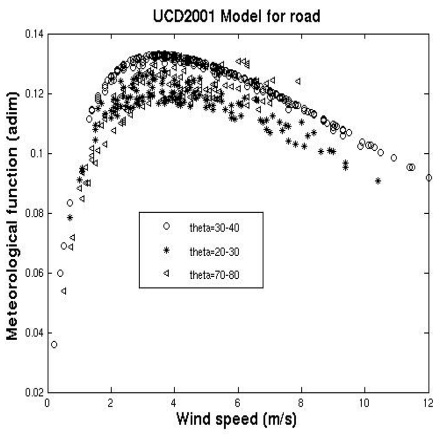
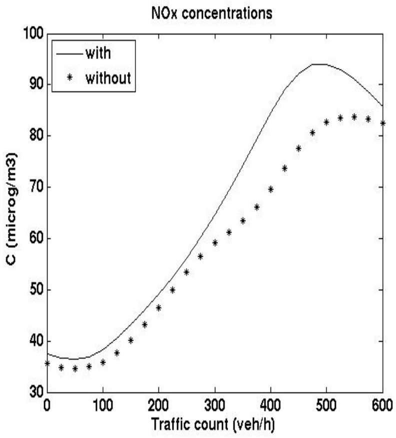


Figure 7: NOx average concentrations with and without roadworks as a function of the vehicle fluxes Nveh



It is found that road works emissions are equivalent to an additional traffic of 50-60 veh.hr^{-1} for CO and NOx, and of 175-200 veh.hr^{-1} for SO_2. The higher SO_2 value may be related to the fuel used for road works equipment engines, that contains higher sulphur rate than commercial fuels for vehicles. The PM10 is surprisingly the smallest emission : equivalent to 35 veh.hr^{-1} . This may be related to road emission of dust during the repairing: a bad state of road could lead to high emission of particles due to resuspension by vehicle movement. Such an emission occurs during the whole period of measurement, whatever engines are working or not. This emission would affect the 2 periods and the comparison fails to give an estimate for this phenomena. Roadwork equipment engines and operations may also emit a lot of larger particles that are not collected within PM10 measurements. Uncertainties about the estimated emission are high, due to the small number of concentrations and averaging methods. Due to the care needed to select record, the sample size is small. A second estimate is also added for a larger θ interval. The impact coefficient decreases moderately for θ value around 25°/North. Due to the lack of vehicle counter during the repairing period, the proposed over-emission includes work engine emission as well as traffic changes. As road changes from 2-lanes to one lane within the work period, it is supposed that the reduction of the vehicular flows may compensate engine emissions and lead to these quite low values. Further researches with actual traffic on the one-lane road are needed to isolate roadwork engines from traffic emissions.

Table 1: Road and roadwork impact on the monitoring station

wind direction angle	θ = [30,40] °/North				θ = [20,40] °/North.			
pollutant	*SO2*	*CO*	*PM10*	*NOx*	*SO2*	*CO*	*PM10*	*NOx*
P1 sample size	101	135	95	133	280	383	272	389
P2 sample size	71	106	68	106	224	328	221	337
ΔC(sensor) / 100 veh (microg/m3)	1.44	0.11	8.25	21.4	1.12	0.11	8.00	18.0
Roadwork emissions (eq veh)	174	67	35	59	195	42	34	62

Conclusion

A database was elaborated with measurements of vehicular activities, meteorological parameters and air pollution monitoring around a place where roadwork were conducted. Due to the lack of background pollution, it was not possible to use directly dispersion models to predict pollution on the site nor deduce emissions from inverse modelling techniques. Therefore non parametric regressions analysis were used to give a comprehensive picture of the pollution measured at this place. Situations corresponding to the maximal impact of road on the sensor were found using the regression and agreeement was found with the dispersion model, used as an indicator for meteorological effects. Non parametric regression allowed to reasonably estimate the emission related to road repairing. Emissions were found more detectable for SO_2 than for other pollutants. The estimation includes road repairing engine emission and vehicle emission changes related to the modified driving conditions on the road during works. Further research are need to better explain the link between concentration and traffic intensities, to take into account for vehicle speed .and congestion in the model.

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Cartographie de la qualité de l'air en agglomération : comment intégrer pollution de fond et pollution de proximité

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Résumé

La cartographie de polluants atmosphériques au niveau de l'agglomération repose fréquemment sur des mesures de « fond » de la qualité de l'air, les éventuelles mesures de trafic étant ignorées en raison de leur faible représentativité spatiale. L'importance des phénomènes de proximité routière pour certains polluants limite donc l'usage d'une telle cartographie.

L'article présente une méthodologie flexible et innovante de cartographie intégrant les phénomènes de fond et de proximité. Elle consiste à cartographier la pollution de fond à l'aide des mesures de fond et de variables auxiliaires (cadastre des émissions, occupation du sol) puis à spatialiser, à partir d'un modèle de rue, les concentrations liées au réseau routier en cohérence avec sa distance d'impact.

La méthodologie, illustrée pour le dioxyde d'azote sur l'agglomération de Toulon, conduit à une cartographie qui intègre de façon pertinente l'ensemble des mesures (fond et proximité).

***Mots-clefs :** qualité de l'air, proximité, cartographie, émissions, modèle de rue, géostatistique.*

Abstract

Mapping air quality in urban areas: how to integrate background pollution and proximity pollution.

Mapping atmospheric pollutants in urban areas frequently relies on a number of air quality measurements. In this context, a classical approach consists in mapping background urban pollution from background measurement sites, ignoring proximity measurements. Proximity phenomenon being of importance for pollutants like nitrogen dioxide or benzene, ignoring this information restricts the use of such mapping, in particular for characterizing air pollution nearby roads and evaluating the potential exposure of populations. This article describes an innovative method that overcomes these issues. Based on pollutant measurements and a physico-chemical road model, the method consists in:

- *mapping background pollution from background measurements and emission inventory, using classical geostatistical techniques (Cressie, 1998; Deraisme & Bobbia, 2003; Jeannée, 2004),*
- *correcting potential bias between measured and modeled concentrations nearby roads,*
- *spatializing the corrected concentrations known on the traffic network consistently with the information available upon the type of decrease and the distance of impact.*

The method is illustrated for mapping nitrogen dioxide on the conurbation of Toulon. By a pertinent incorporation of all available data (103 NO_2 measurements by diffusive samplers, an emission inventory and the road model STREET), it provides a globally consistent map.

Keys-words: *air quality, proximity, mapping, emissions, road model, geostatistics.*

Introduction

La cartographie à fine échelle de la pollution atmosphérique au sein de l'agglomération répond à plusieurs objectifs : compréhension des phénomènes, communication sur les niveaux de pollution, évaluation des populations potentiellement exposées à des niveaux élevés de pollution. Ces objectifs sont notamment à l'origine, en France, du groupe de travail Air-ProCHE. Coordonné par l'AFSSET (Agence Française de Sécurité Sanitaire de l'Environnement et du Travail) et l'IFEN (Institut Français de l'Environnement), Air-ProCHE vise à coordonner et développer une démarche harmonisée de cartographie fine permettant d'identifier les portions du territoire national où les populations sont affectées par les niveaux de pollution atmosphériques les plus élevés. Il s'inscrit notamment dans le cadre du Plan National Santé Environnement (PNSE) 2004-2008.

Afin d'établir une telle cartographie de la qualité de l'air au niveau de l'agglomération, il est possible de spatialiser les mesures de la qualité de l'air acquises en quelques points de l'agglomération, grâce à des analyseurs automatiques ou à la mise en œuvre de campagnes de mesures. Les techniques géostatistiques sont ainsi classiquement utilisées en qualité de l'air depuis plusieurs années pour cartographier des pollutions atmosphériques à l'échelle urbaine (Cressie, 1998), régionale (Roth, 2001) ou nationale (Jeannée, 2003). L'approche géostatistique permet en outre d'intégrer d'autres informations pertinentes pour la cartographie du polluant : concentrations issues d'une modélisation physico-chimique, cadastres d'émission ou occupation des sols, mesures plus nombreuses d'autres polluants ayant un comportement similaire (Bobbia et al, 2001).

La cartographie géostatistique de données de la qualité de l'air en agglomération se focalise usuellement sur la pollution de fond. En dépit de leur importance, les phénomènes de pollution de proximité ne sont en général pas reproduits, étant délicats à appréhender par des mesures dont la représentativité spatiale est faible. Cela pose des problèmes :

- de communication, les cartographies diffusées n'illustrant qu'une partie du phénomène de pollution atmosphérique, excluant pour des polluants tels que le NO_2 la source principale de pollution, même si leurs niveaux de fond rendent compte en partie des sources liées au trafic ;
- de caractérisation de la pollution, en particulier si la cartographie doit servir de base à une évaluation de l'exposition potentielle des populations, sachant qu'une proportion de cette population réside à proximité d'axes routiers.

Les niveaux de concentration observés en proximité routière peuvent aujourd'hui être modélisés grâce à des outils numériques tels que STREET (Targeting, http://www.targeting.fr), ADMS-Roads (CERC, http://www.cerc.co.uk/software/admsroads.htm) ou SIRANE (Soulhac, 2002). En effet, à partir de différents paramètres (émissions, typologie de l'axe, météorologie, ...), ces modèles de rue évaluent les concentrations du polluant le long des principaux axes (Rouïl, 2004).

Afin d'intégrer cette information de proximité, une première approche consiste à réaliser une cartographie de la pollution de fond, puis à superposer le long des principaux axes les concentrations issues du modèle de rue. Bien que plus satisfaisante qu'une cartographie exclusivement de fond, cette méthodologie présente deux inconvénients :

- les concentrations modélisées le long des axes de circulation ne sont pas spatialisées, en dépit de leur impact reconnu sur la qualité de l'air jusqu'à 100 ou 200 m des axes routiers,
- la cohérence entre les résultats du modèle de rue et les mesures de proximité est - dans le meilleur des cas - évaluée qualitativement.

La méthodologie proposée permet de résoudre ces problèmes. Elle est présentée puis illustrée pour la cartographie de la concentration moyenne annuelle du dioxyde d'azote sur l'agglomération de Toulon pour l'année 2001. Cette agglomération a fait l'objet de plusieurs campagnes de mesure ces dernières années, menées par l'association AIRMARAIX, en raison de l'existence de populations potentiellement exposées à des niveaux significatifs de polluants tels que le dioxyde d'azote. Les résultats obtenus sont finalement discutés en regard des objectifs attendus d'une telle cartographie.

Matériels & Méthodes

En France, l'étude et le suivi de l'air ambiant sont confiés à des Associations Agréées pour la Surveillance de la Qualité de l'Air (AASQA). En agglomération, les mesures réalisées sont qualifiées de « proximité » lorsqu'elles ont pour but de documenter la qualité de l'air au voisinage de certaines sources de pollution (industries, axes routiers) et de mesures de « fond » (urbaines, périurbaines) lorsqu'elles sont destinées à caractériser la pollution atmosphérique moyenne sur ces zones (ADEME, 2002). Les mesures de fond et de proximité correspondent à des populations statistiques bien distinctes, à la fois en terme de niveaux de concentration et de représentativité spatiale, ce qui justifie leur traitement dissocié. La méthodologie de cartographie fond/proximité repose ainsi sur deux étapes consistant : (1) à cartographier la pollution de fond, puis (2) à lui ajouter une cartographie des « sur-concentrations » de proximité routière à la fois mesurées et modélisées par STREET (version 4.1).

La résolution d'une telle cartographie fond/proximité doit être suffisamment fine pour capturer les phénomènes de proximité, sans pour autant laisser illusoirement penser que l'on peut atteindre, en terme de cartographie de la qualité de l'air, une précision de l'ordre du mètre. Une résolution de l'ordre de 25m constitue un compromis intéressant au regard des données d'entrée : topographie, îlots de population INSEE au 1/25000ème, occupation du sol CIRGE 1999 à 2,5ha.

1. Cartographie de la pollution de fond

La cartographie de la pollution de fond est obtenue par l'application de techniques géostatistiques classiques aux mesures de fond disponibles : krigeage des concentrations mesurées de fond, cokrigeage intégrant des variables auxiliaires telles qu'un cadastre d'émissions, etc. (voir par exemple Deraisme et Bobbia, 2003, ou Jeannée, 2004, pour une description de la méthodologie géostatistique appliquée à la qualité de l'air). Il est important d'exclure du jeu de données de fond les mesures impactées par la proximité routière. Outre les sites de type trafic ou industriel, on exclura les points de mesure qui, bien qu'étant typés de fond, présentent un niveau de concentration supérieur aux autres points de mesures de fond proches, niveau explicable par la proximité d'un axe routier. Ces points doivent être retirés du processus de construction de la cartographie de la pollution de fond, afin d'éviter d'y intégrer des points exposés à des phénomènes de proximité.

2. Estimation des concentrations sur le réseau routier

Disposant d'une cartographie de la pollution de fond et de sites de proximité exclus de sa construction, l'objectif est à présent de combiner ces informations à celles liées à la pollution modélisée en proximité routière, afin d'obtenir une cartographie intégrée fond/proximité pertinente.

Les modèles physico-chimiques de rue requièrent en entrée une information relative à la pollution de fond, qui doit être cohérente avec la cartographie de pollution de fond réalisée au préalable afin de ne pas donner lieu à des résultats incohérents. Il importe ensuite de garantir la cohérence entre concentrations modélisées sur le réseau routier et la pollution de fond (la concentration modélisée sur le réseau routier doit lui être supérieure) ainsi que les mesures de proximité (mesures de proximité et concentrations modélisées sur les axes avoisinants doivent être corrélées).

Cette analyse est menée sur les « sur-concentrations » de proximité, obtenues en soustrayant la pollution de fond cartographiée aux concentrations de proximité, à la fois mesurées et modélisées ; en effet, en soustrayant les tendances de fond de la pollution, l'analyse se trouve simplifiée.

Les concentrations issues du modèle de rue sont disponibles sous forme de données linéiques correspondant aux tronçons du réseau routier modélisés. Afin de permettre leur spatialisation, ces données linéiques sont discrétisées en un chapelet fin de points couvrant le réseau routier. L'observation de biais entre mesures de proximité et concentrations modélisées sur le réseau routier pourra être corrigée par exemple par krigeage avec dérive externe, dont l'idée consiste intuitivement à garder l'allure générale des concentrations modélisées tout en les recalant aux niveaux de concentrations mesurés en proximité routière.

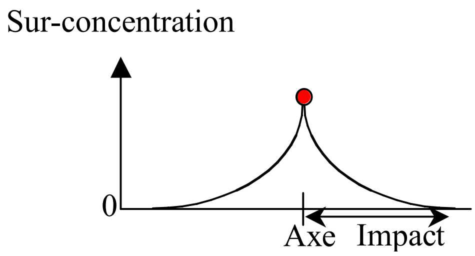
3. Spatialisation des concentrations de proximité routière

Les concentrations de proximité routière, connues exclusivement sur le réseau routier, doivent finalement être spatialisées. L'expérience acquise pour le dioxyde d'azote montre que la décroissance des niveaux de pollution autour des axes est exponentielle (ASPA, 2003, ADEME 2002). En outre, le tissu urbain environnant semble déterminant pour la distance d'impact direct de l'axe routier, de 100m maximum en milieu urbain fermé (rugosité importante, hors rues de type « canyon »), et 200m maximum en milieu ouvert (routes nationales et autoroutes). La densité de bâti permet de distinguer ces deux cas et de particulariser la spatialisation en fonction du type de milieu.

Les « sur-concentrations » de proximité routière sont connues le long du réseau routier. Pour chaque type de milieu, ces « sur-concentrations » sont spatialisées par krigeage simple à l'aide d'un modèle de variogramme exponentiel de portée égale à la distance d'impact direct de l'axe. Le krigeage simple, ou krigeage à moyenne connue, nécessite la connaissance de la moyenne des sur-concentrations en l'absence de mesure. Lorsque le réseau routier conduisant à des niveaux de concentrations significatifs est exhaustivement modélisé, la moyenne des sur-concentrations attendue en dehors de ce réseau est simplement égale à 0. Le modèle de variogramme exponentiel contribue à une décroissance exponentielle des concentrations autour des axes (figure 1).

Figure 1 : Principe de la spatialisation par krigeage simple des concentrations le long d'un axe routier, jusqu'à la distance d'impact de l'axe sur la qualité de l'air environnant.

Figure 1: llustration of how concentrations modeled along the traffic network are spatialized using simple kriging. The exponential decrease is justified by several transects studies performed along roads. The distance of impact (« Impact ») of the road on the neighboring air quality classically varies between 100m and 200m depending on the density of urban fabric, known via Corine Land Cover land use database.



4. Construction de la cartographie finale

La cartographie finale est obtenue en sommant cartographie de la pollution de fond et cartographie des sur-concentrations en proximité routière. Par construction, la cartographie ainsi obtenue respecte les données mesurées, prend en compte les informations de fond disponibles (émissions) et intègre, en les spatialisant, les niveaux de pollution modélisés en proximité routière.

Résultats

1. Données d'entrée

L'illustration a pour objectif la cartographie sur l'agglomération de Toulon (France) de la concentration moyenne annuelle en dioxyde d'azote en 2001. Ces concentrations ont été évaluées par AIRMARAIX en 103 points, à partir de plusieurs campagnes de mesures réalisées sur Toulon depuis 1997 (mesures passives ou actives, statistiquement adaptées à l'année d'étude). On distingue parmi ces 103 points : 73 sites de fond et 30 sites de proximité.

Un cadastre kilométrique d'émissions en oxydes d'azote est disponible sur le domaine d'étude (inventaire régional PACA 1999 – AIRMARAIX 2004). Des données d'occupation du sol existent également à différentes échelles, dont 50m pour la plus fine. Le modèle de rue STREET a permis de modéliser les concentrations de NO_2 sur le réseau principal de l'agglomération (636 brins).

2. Cartographie de la pollution en NO_2 de fond

Classiquement, la cartographie du NO_2 gagne à intégrer la connaissance d'un cadastre d'émissions. En raison de la corrélation entre ces émissions et les mesures de NO_2, le cadastre améliore la qualité de la cartographie du polluant, en particulier dans les zones sous-échantillonnées. Le cadastre est transformé (lissage et calcul du logarithme) afin d'améliorer la corrélation avec les mesures du polluant. Les figures 2 et 3 illustrent le cadastre ainsi obtenu et la corrélation entre émissions transformées et concentrations de fond en NO_2. L'analyse des 73 sites de fond a conduit à en écarter 7 en raison de niveaux de concentrations qui, étant notablement plus élevés que les sites de fond voisins, sont en outre explicables par la proximité du réseau routier. La cartographie finale de la pollution de fond est obtenue par krigeage avec dérive externe (figure 4). Les modélisations géostatistiques sont réalisées à l'aide du logiciel ISATIS (Geovariances, 2005).

Figure 2 : Agglomération de Toulon : cadastre d'émissions en NOx transformées, position des mesures en NO_2 (carrés : sites de fond en noir, sites de proximité en rouge) et du réseau modélisé par STREET (trait noir).

Figure 2: Toulon conurbation: transformed nitrogen oxides emissions, location of NO_2 background (black squares) and proximity (red squares) measurements. Traffic network modeled by the road model STREET (thin black lines).

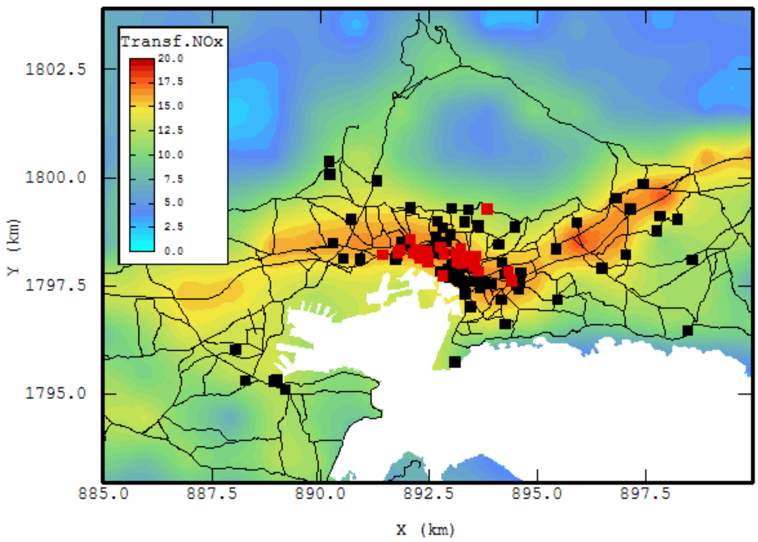
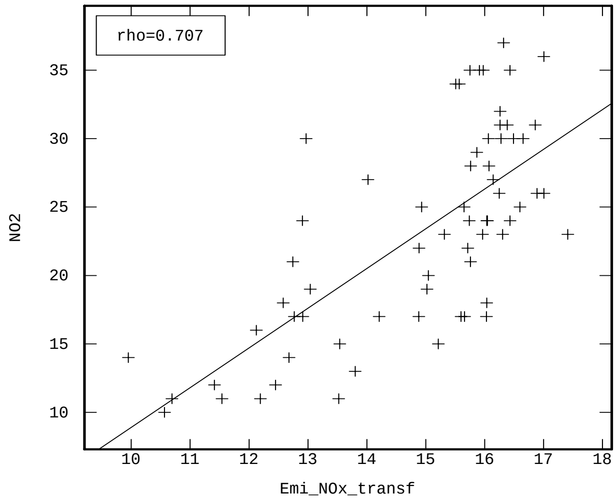


Figure 3 : Nuage de corrélation entre émissions en NOx transformées et concentrations de fond en NO_2. Indication de la droite de régression linéaire.

Figure 3: Scatter diagram between transformed nitrogen oxides emissions and measured background NO_2 concentrations. Linear regression line indicated. The quality of the correlation justifies the integration of such emissions as an auxiliary variable during the mapping of background NO_2.



3. Spatialisation des sur-concentrations de proximité routière et cartographie finale

Les concentrations modélisées par STREET, discrétisées tous les 20m, sont superposées à la cartographie de la figure 4. Leur corrélation moyenne avec les mesures de proximité conduit à les corriger par krigeage avec dérive externe : la forme générale de la modélisation de STREET est conservée, mais les niveaux sont corrigés afin d'être cohérents avec les mesures de proximité. Ainsi corrigées, les sur-concentrations modélisées par STREET sont spatialisées par krigeage simple à l'aide d'un modèle de variogramme exponentiel. La portée du variogramme, qui correspond à la distance d'impact direct de l'axe routier, est égale à 100m en milieu fermé (défini par une proportion de bâti supérieure à 50%) et 200m en milieu ouvert (bâti inférieur à 50%). La cartographie finale du NO_2 sur l'agglomération de Toulon, obtenue en sommant la cartographie de fond et celle des sur-concentrations en proximité routière, est illustrée à la figure 5.

Figure 4 : Cartographie de la pollution de fond en NO_2 (µg/m^3). Superposition des concentrations issues de STREET, avec la même échelle de couleur.

Figure 4: Background annual average NO_2 concentration map in µg/m^3. NO2 concentrations provided by the road model STREET are overlaid with the same color scale.

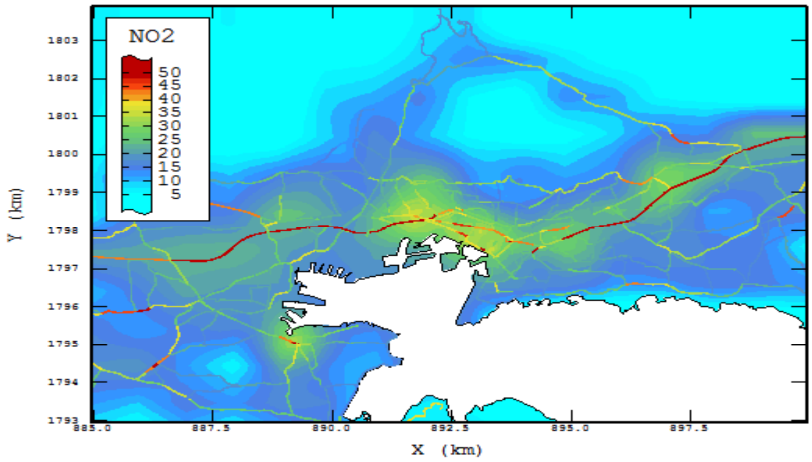
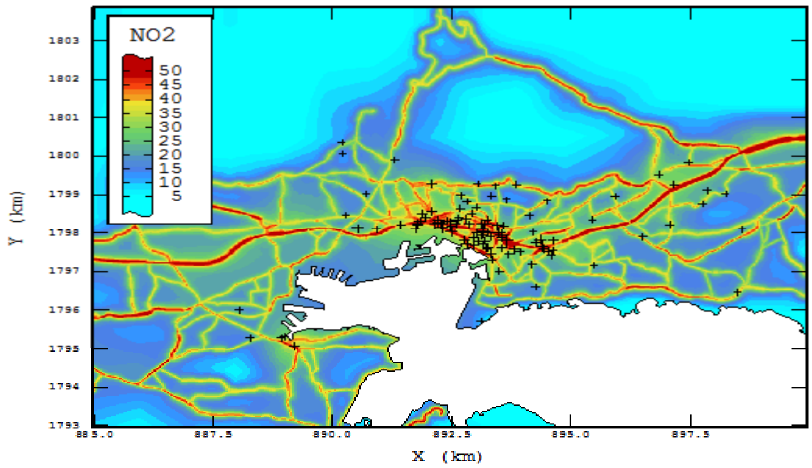


Figure 5 : Cartographie finale de la pollution en NO_2 (µg/m^3) sur l’agglomération de Toulon, obtenue par sommation de la cartographie de fond et de la spatialisation des « sur-concentrations » de proximité routière.

Figure 5: Final NO_2 map (µg/m^3), obtained from the summation of the background map and of the spatialized concentrations provided by the road model, corrected in order to be consistent with proximity measurements.



Discussion

1. Pollution atmosphérique en NO_2 sur l'agglomération de Toulon

La première approche consistant à superposer les concentrations issues du modèle de rue à la cartographie de fond est illustrée à la figure 4. On constate l'absence de spatialisation des concentrations de proximité routière, bien que de nombreuses études le long de transects routiers aient montré que les niveaux atteints sur le réseau conduisent à des sur-concentrations significatives jusqu'à 100 voire 200 m des axes. En outre, la cohérence entre les résultats du modèle de rue et les mesures de proximité n'est pas validée dans cette approche. Ces arguments ont motivé la méthodologie présentée dans cet article. La cartographie qui en découle (figure 5), présente un réalisme accru, intégrant dans sa construction l'information sur l'existence d'une zone d'influence des axes routiers sur la qualité de l'air environnant. Par ailleurs, la confiance que l'on peut avoir dans cette cartographie est meilleure, la cohérence entre les différentes informations combinées ayant été validée expérimentalement.

Reposant sur des algorithmes d'interpolation géostatistique, la méthodologie s'appliquera plus aisément en présence de mesures de polluants suffisamment nombreuses et représentatives des différentes configurations rencontrées sur la zone d'étude : à la fois de fond afin de cartographier correctement les différentes situations (centre ville dense, zones périphériques, contraintes de relief et de littoral) et des mesures de proximité afin de pouvoir notamment évaluer la cohérence du modèle de proximité routière (STREET) et le cas échéant corriger un éventuel biais de ce dernier. Bien qu'un nombre moins élevé de sites de fond eut probablement suffi pour le NO_2, dont la pollution de fond est relativement lisse et homogène, l'ensemble des mesures a été conservée pour cette première application sur l'agglomération de Toulon.

2. Validation de la cartographie et incertitude associée

Bien que la géostatistique permette de quantifier l'incertitude associée à une cartographie, cette évaluation est délicate dans le cas présent, l'incertitude associée aux concentrations modélisées par STREET étant méconnue. En outre, seul le réseau routier principal a été modélisé par STREET dans cette étude. La non prise en compte de certains axes constitue une incertitude supplémentaire, notamment en zone péri-urbaine, même si la corrélation avec les émissions peut compenser ce biais.

Il est également délicat de différencier avec précision milieux « fermés » et « ouverts ». L'utilisation du pourcentage de bâti comme critère de sélection introduit une incertitude supplémentaire délicate à quantifier dans le cadre de cette étude. Ce critère pourra être affiné avec une classification précise des axes selon des catégories explicites et l'introduction notamment d'un type de rue « canyon » pour mieux approcher les concentrations de proximité en milieu urbain très dense.

La cohérence globale de la cartographie obtenue a néanmoins été vérifiée par comparaison avec une modélisation numérique réalisée à l'aide d'ADMS-Urban sur une portion de l'agglomération.

3. Applicabilité de la méthodologie à d'autres cas

La flexibilité de la méthodologie la rend aisément transposable à d'autres cas. Ainsi, elle a déjà été appliquée dans le cadre du groupe de travail Air-ProCHE sur la Communauté Urbaine de Strasbourg. Elaborée à partir d'analyseurs automatiques uniquement, la cartographie obtenue s'est avérée cohérente avec une modélisation numérique réalisée à l'aide du modèle ADMS-Urban.

Moyennant certaines hypothèses, la méthodologie est en outre applicable en l'absence de modèle de rue, en exploitant alors les émissions sur le réseau, voire juste la position du réseau. La méthodologie conduit alors à une cartographie qui, bien que sujette à des incertitudes plus élevées, reste préférable à une cartographie de la pollution exclusivement de fond dont le biais est évident.

D'autres polluants primaires influencés par le trafic routier, tels que le benzène, peuvent également être cartographiés à l'aide de cette méthodologie moyennant une modification des hypothèses de dispersion du polluant autour des axes routiers (distance d'impact) ainsi qu'une validation de la corrélation avec l'inventaire des émissions.

Conclusion

La cartographie à fine échelle de pollutions atmosphériques est fréquemment réalisée exclusivement à partir de mesures de fond en raison du manque de représentativité spatiale des mesures de proximité, cela en dépit de l'importance de ces phénomènes de proximité. La méthodologie présentée apporte une solution à ce problème, en proposant une cartographie qui repose sur l'existence de mesures à la fois de fond et de proximité pour le polluant, d'un cadastre d'émissions classique et d'une modélisation des concentrations sur le réseau routier issue d'un modèle de rue. Elle exploite de façon pertinente ces informations, tout en garantissant leur cohérence.

Pour l'agglomération de Toulon, la méthodologie conduit à une cartographie pertinente de la concentration moyenne annuelle en NO_2, à une résolution de 25m. Le résultat final, s'il est cohérent avec l'ensemble des points de mesures à la fois de fond et de proximité disponibles, gagnerait néanmoins à être validé en quelques points non intégrés dans la construction de la cartographie et affiné du point de vue de la classification des axes en fonction de leur environnement.

La méthodologie de cartographie intégrée fond/proximité est aisément généralisable et applicable à d'autres cas : cartographie d'autres polluants, cartographie à partir d'analyseurs automatiques, cartographie focalisée sur certains axes routiers. Finalement, le cadre proposé correspond tout à fait à des besoins de cartographie de la qualité de l'air intégrant la pollution le long d'axes routiers importants, problématique fréquemment évoquée par les populations riveraines et pré-requis incontournable pour aborder la question de l'exposition de la population.

Remerciements

Le travail présenté a été partiellement développé dans le cadre du groupe de travail Air-ProCHE. Il a ainsi bénéficié d'un financement de l'AFSSET et de l'IFEN, que les auteurs tiennent à remercier.

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Parcs et trafics routiers en France, 1870-1970 : Caractérisation et principaux enjeux environnementaux

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Résumé

Le texte aborde deux questions complémentaires, l'une historique, l'autre rétrospective : quels ont été les effets des systèmes de transports passés sur leur environnement ? Quels sont les effets cumulés des systèmes de transport ?

Le transport routier et le cas français ont été retenus à travers l'analyse des parcs et trafics des années 1870 aux années 1970, pour un travail qui se situe à la charnière de l'histoire des techniques et de l'écologie industrielle. La première permet une analyse de l'évolution du système de transport routier, la seconde conduit à la détermination d'indicateurs des interactions transports-environnement selon le principe des bilans de matière. Les quantités de matières mobilisées par un système donné constituent en effet une première approche de ses impacts environnementaux.

Mots-clefs : *transport routier, parcs de véhicules, trafic, bilan de matières, consommation de carburant, France, XIXe-XXe siècles.*

Abstract

Vehicle fleets and road traffic in France, 1870-1970: characterisation and main environmental stakes.

This paper aims to contribute towards the characterisation of the French transport system and its evolution in the (relatively) long-term and identify decisive elements regarding environmental stakes and impacts, both historically for the period determined (What consequences does the transport system have on the environment? In the light of these, what stakes can be identified, etc.?) and retrospectively (What are the major trends? What are the cumulated and deferred effects of the transport system, etc.?).

Road transport has been chosen in particular given the complexity of the changes occurring at present, and the fact that relatively little is known about them or the stakes they represent. Although the environmental impact of infrastructures is far from negligible, we have decided to focus our attention to analysing vehicle fleets and traffic. The period studied stretches from the 1870s to the 1970s: this choice was made in order to begin before the advent of mechanisation in order to obtain better understanding of the related changes.

Concepts and methods from both history of technology and industrial ecology have been used. The first allows the analysis of the considered transport system, the second the determination of its interaction with the environment thanks to material balance. The results are far from definitive, but demonstrate the growing impact of transport system during the considered period and the relevance of the chosen indicators.

***Keys-words**: road transport, fleet, traffic, material balance, fuel consumption, France, 19th-20th centuries.*

Introduction

Si la prospective semble de règle en matière d'analyse des interactions entre systèmes de transport et environnement, force est de constater que l'on sait assez peu de l'évolution de ces interactions (citons néanmoins De Vooght et coll. (2003) pour la Belgique). Dans ce contexte, il s'agit d'aborder deux questions complémentaires, l'une historique, l'autre rétrospective : quels ont été les effets des systèmes de transports passés sur leur environnement ? Quels en sont à une date donnée les effets cumulés ?

Le transport routier et le cas français ont été plus particulièrement retenus compte tenu de la complexité des évolutions à l'œuvre, de leur relative méconnaissance et des enjeux qui y sont aujourd'hui associés. Bien que l'impact environnemental des infrastructures soit loin d'être négligeable, nous avons centré le travail sur l'analyse des parcs et trafics. La période étudiée court des années 1870 aux années 1970 : débuter avant la mécanisation permet de mieux comprendre les évolutions qui y sont associées et de ne pas négliger l'impact de la traction animale.

Le travail se situe à la charnière de l'histoire des techniques et de l'écologie industrielle. La première permet une analyse de l'évolution du système de transport routier, en particulier des parcs et des trafics, la seconde conduit à la détermination d'indicateurs des interactions transports-environnement selon le principe des bilans de matière (Ayres, Ayres, 2002 ; Brunner, Rechberger, 2004). Il nous semble en effet que les quantités de matières mobilisées par un système donné constituent une première approche de ses impacts environnementaux.

Les parcs puis les trafics seront abordés. Dans chaque cas, on précisera les sources utilisées ainsi que leurs limites, puis les méthodes adoptées pour constituer les chroniques et déterminer les indicateurs ; on donnera enfin quelques éléments de quantification. La démarche adoptée étant exploratoire, les résultats présentés doivent être considérés comme provisoires et préalables à des analyses plus approfondies.

Les parcs

1. Sources et méthodes

Les sources concernant les parcs s'avèrent hétérogènes, pas entièrement compatibles et pas toujours exhaustives (Barles et coll., 2004). Pour le transport individuel, et jusqu'en 1933, ce sont les sources fiscales qui sont les plus fiables ; elles concernent l'ensemble des véhicules, qu'il s'agisse des voitures suspendues et des chevaux, mules et mulets destinés au transport des personnes (depuis 1871), des véhicules automobiles (depuis 1889) ou des deux roues avec ou sans moteur (depuis 1893). Les parcs sont alors connus de façon assez précise (Ministère des Finances, 1942 ; INSEE, 1952). En revanche, les véhicules entrants et sortants (neufs d'une part, mis au rebut d'autre part) ne sont pas recensés. À partir de 1934, la taxe de circulation est remplacée par une taxe sur les carburants. Les parcs ne sont plus connus que de façon indirecte, sauf pour les bicyclettes, taxées jusqu'en 1949, et les véhicules à traction animale qui échappent désormais à tout recensement (Mahieu, s. d.). Simultanément, la centralisation des états mensuels établis par les préfectures délivrant les cartes grises permet la constitution d'une nouvelle série statistique : celle des immatriculations. De 1934 à 1939, les parcs automobiles annuels sont ainsi établis à partir de celui de 1933 augmenté des immatriculations nouvelles et réduit des véhicules usagés (estimés). Le parc est très mal connu au cours du second conflit mondial et jusqu'à la création, en 1950, du fichier central de l'automobile, géré par l'INSEE. Le système des immatriculations est modifié, les cartes grises entièrement renouvelées, ce qui permet de poser les bases d'une nouvelle chronique du parc, fréquemment critiquée mais maintenue (CAC, 770444 article 3).

Les véhicules utilitaires quant à eux ne font l'objet d'aucun recensement avant 1920 où ils sont soumis à la taxe de circulation. La loi du 28 février 1933 soumet par ailleurs les véhicules commerciaux à des taxes au poids et à l'encombrement, tandis que les remorques font l'objet d'un impôt sur la circulation, si bien que les effectifs de véhicules utilitaires sont bien mieux connus que ceux des véhicules individuels à partir des années 1930 (Bertrand, 1941 ; Mahieu, s. d.).

Les chroniques de composition des parcs, d'effectifs de véhicules neufs (entrants) et de véhicules usagés (sortants) ont été établies sur ces bases incertaines, dans un premier temps uniquement pour le transport individuel. Lorsque nous disposions de chiffres relatifs aux parcs, mais pas aux immatriculations, nous avons d'abord estimé un nombre annuel de véhicules sortants. Schématiquement, la mise au rebut peut être due soit au vieillissement du véhicule, soit à un accident quelconque, soit à la combinaison des deux. Un calcul basé sur une loi de mortalité (courbe en cloche autour d'une durée de vie moyenne de dix ans) donne une première estimation des sorties. Une seconde estimation est basée sur un taux de casse annuel de 5 % du parc (fréquemment rencontré dans la littérature). Le nombre de sorties finalement retenu est la moyenne des deux estimations (la sortie accidentelle et la sortie par vieillissement sont supposées de poids égal). La méthode a été appliquée aux deux roues motorisés ou pas et aux voitures automobiles jusqu'en 1930. Pour ces dernières et à partir de 1931, nous avons déduit les sorties des parcs et des immatriculations, sur la base, pour les parcs, des chiffres fournis par la Chambre syndicale des constructeurs automobiles, plus fiables que ceux établis dans un premier temps par l'INSEE et d'ailleurs plus tard adoptés par lui (INSEE, 1990). Nous avons donc fait à rebours le calcul de nos prédécesseurs sans le remettre en cause, son détail étant rarement donné. Ces estimations pourraient bien sûr être affinées — estimation des sorties, analyse critique de la méthode de la Chambre syndicale des constructeurs automobiles, recours à des modèles de parcs (Hugrel et Joumard, 2005), etc. — ; cependant, l'utilisation des diverses méthodes envisageables ne conduit pas à des écarts majeurs dans les tendances observées.

Afin d'approcher de façon macroscopique l'impact environnemental des parcs, les bases d'un bilan de matières brutes ont été posées (Eurostat, 2001). Il s'agit dans un premier temps de réaliser un bilan annuel, comptabilisant les importations de matières dans le pays étudié (les importations étant comprises comme les entrées dans le système socio-industriel, c'est-à-dire à la fois les importations en provenance d'autres pays et les productions locales), puis les exportations vers la nature (rejets divers du système), ici ramenées aux mises au rebut, et enfin l'augmentation ou la diminution des stocks (ici constitués par les parcs de véhicules). Les effets cumulés de la mécanisation sont dans un second temps analysés. La somme des poids des véhicules neufs mis sur le marché depuis la fin du XIXe siècle traduit ainsi la pression sur les ressources, tandis que celle des poids des véhicules mis au rebut donne une mesure des déchets produits. Ces indicateurs sont grossiers puisqu'ils ne prennent pas en compte l'éventuel recyclage des matériaux, ni les effets de la réparation, moins encore la quantité réelle de matériaux extraits de la biosphère en vue de la fabrication d'un véhicule quelconque, assurément très supérieure au poids final du véhicule (notion de flux caché) ; ils nous semblent néanmoins utiles pour alimenter une première réflexion. La quantification repose sur plusieurs hypothèses simplificatrices facilement révisables. La principale concerne le poids unitaire des véhicules, supposé constant tout au long de la période : 400 kg pour les véhicules hippomobiles, 12 kg pour les bicyclettes, 40 kg pour les deux roues motorisés et 600 kg pour les voitures particulières. Ces chiffres ont été trouvés dans la littérature (en particulier Picard, 1906). L'analyse a donc été limitée aux véhicules individuels, les sources relatives au poids des véhicules utilitaires, très variable, n'ayant pas encore été dépouillées.

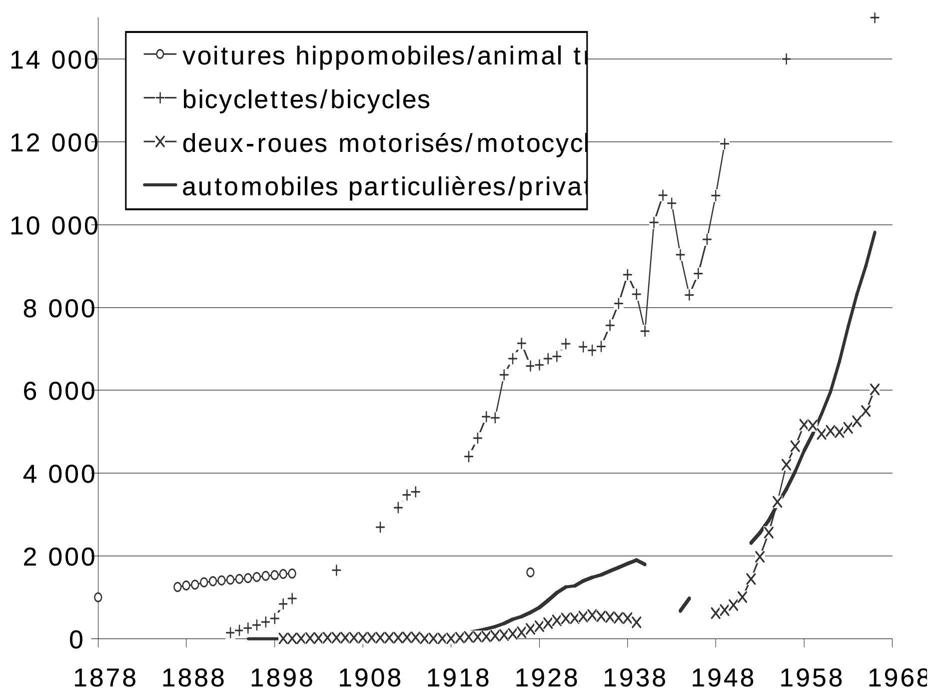
2. Résultats

La figure 1 montre l'évolution du parc français de véhicules individuels entre 1878 et 1968. Au total, il passe d'un million de véhicules en 1878, soit moins de 3 véhicules pour 100 habitants, à 2,6 millions de véhicules en 1900 soit un peu plus de 6 pour 100 habitants, 9 millions en 1927 soit 22 pour 100 habitants et enfin 31 millions en 1966 soit 63 pour 100 habitants. À titre de comparaison, le parc de véhicules utilitaires passe de 80 000 à 2 250 000 unités entre 1920 et 1966.

Malgré un accroissement réel du parc de véhicules hippomobiles particuliers à la fin du XIXe siècle, la progression de l'équipement individuel est d'abord celle des cycles sans moteur dont on voit le très rapide essor à la veille de la Première Guerre Mondiale. L'entre-deux-guerres est marqué par la prépondérance de la bicyclette — 80 % du parc de véhicules individuels en 1938. Cependant on constate un premier essor des véhicules motorisés — automobiles à quatre roues et dans une moindre mesure deux roues. Au lendemain de la Seconde Guerre Mondiale, après une phase de reconstitution, les parcs reprennent leur croissance avec une plus grande rapidité encore. En 1966, le parc automobile représente déjà un tiers du parc de véhicules individuels, tandis que les bicyclettes en fournissent la moitié, le reste étant composé des deux roues motorisés. Une analyse plus précise de la composition de ce dernier parc montrerait que les grosses cylindrées ont tendance à régresser (report vers l'automobile) tandis que les petites augmentent (report de la bicyclette).

Figure 1 : Parcs de véhicules particuliers, France, 1878-1966 (milliers de véhicules).

Figure 1: Private vehicle fleets, France, 1878-1968 (thousands of vehicles).

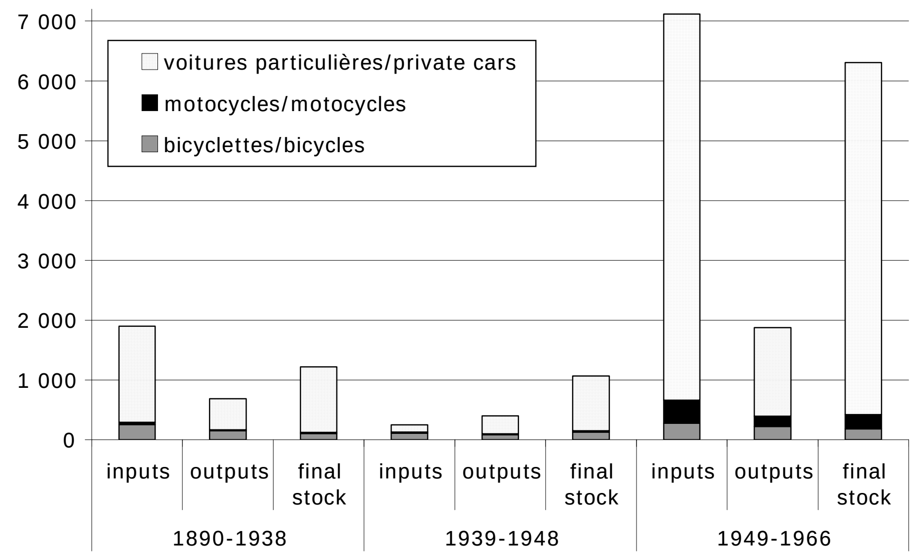


La figure 2, relative au bilan de matières, résume les résultats obtenus pour trois périodes homogènes du point de vue des tendances observées pour les parcs mécanisés : 1890-1938 (les quantités mentionnées concernent en fait essentiellement l'après-guerre), 1939-1948 et 1949-1966. Elle indique pour chacune d'entre elles les poids 1) des véhicules ajoutés au stock initial, 2) des véhicules mis au rebut, 3) du stock en fin de période. Elle montre l'augmentation considérable du stock de matière dans le système de transport, puisque le stock initial (non représenté) constitué par les véhicules hippomobiles s'élève à environ 500 000 tonnes (il varie peu et culmine aux environs de 640 000 tonnes à la fin des années 1920). La très forte augmentation des parcs de véhicules mécanisés dans l'entre-deux-guerres s'accompagne de la constitution d'un stock d'un peu plus d'un million de tonnes à la veille de la Seconde Guerre mondiale, stock à peu près égal à celui de 1948, très affecté par la guerre. Entre 1949 et 1966, le stock sextuple et atteint 6,3 millions de tonnes.

De même, le poids des entrées et, dans une moindre mesure, des sorties, est-il beaucoup plus important pour la dernière période. Dans tous les cas, et malgré son effectif toujours inférieur à celui des bicyclettes, c'est la voiture particulière qui contribue pour l'essentiel aux flux de matières considérés. Au total, entre 1890 et 1966, ce sont plus de 9 millions de tonnes de véhicules neufs qui sont entrées dans le système, et le tiers qui en est sorti sous forme de véhicules mis au rebut. Le parc de véhicules, hors traction animale, représente 3 kg/hab en 1919, 29 kg/hab en 1938, 26 kg/hab en 1949 et 128 kg/hab en 1966 (il dépasse actuellement 400 kg/hab). La prise en compte des voitures hippomobiles le porterait à environ 18 kg/hab pour 1913, 39 kg/hab pour 1938.

Figure 2 : Poids des véhicules entrants (inputs), des véhicules mis au rebut (outputs) et du parc en fin de chaque période (final stock), France, 1890-1938, 1939-1948, 1949-1966 (milliers de tonnes).

Figure 2: Weight of vehicle inputs, of vehicle outputs and of vehicle fleet at the end of each period (final stock), France, 1890-1938, 1939-1949, 1950-1966 (thousands tons).



Les trafics et la consommation énergétique

1. Sources et méthodes

Il n'existe pas, à l'échelle nationale, de chronique des trafics routiers portant sur une longue durée. Ce n'est qu'à partir de 1951 qu'une estimation annuelle a été réalisée, estimation indirecte basée sur les consommations de carburants (CAC, 770444 article 3). En revanche, les recensements de la circulation effectués sur le réseau routier national depuis 1844 constituent une source précieuse de connaissance des trafics et plus encore de leur évolution. Destinés à programmer les opérations d'entretien et de réfection des chaussées (puis, à la toute fin de notre période, d'élargissement), ils sont réalisés assez fréquemment (19 recensements entre 1844 et 1970) et portent sur le réseau divisé en sections de trafic supposé homogène, soit 5 500 puis 4 500 puis 4 100 points d'observation. Les comptages, manuels jusqu'en 1960, partiellement automatiques en 1965, entièrement en 1968, sont effectués au cours de plusieurs journées (28 puis 15) et nuits (14 puis 7) réparties dans l'année (CAC, 780264 article 2).

La principale discontinuité de cette source réside dans le changement de variable opéré à l'occasion du recensement de 1928. Jusque-là, l'unité de compte était le collier, correspondant à un animal attelé à une voiture et l''intensité de la circulation désignait le nombre de colliers par jour en un point donné (Moullé, 1910). Une typologie des usagers de la voirie (voitures vides, voitures chargées, animaux non attelés, bétail…, plus tard véhicules à traction mécanique) permettait de convertir les colliers bruts en colliers réduits, chaque type étant affecté d'un coefficient prenant en compte son impact plus ou moins grand sur la chaussée. Une estimation départementale du tonnage transporté permettait de déterminer le tonnage brut supporté par les chaussées ainsi que le tonnage utile (Moullé, 1910, Rénouard, 1960). Cependant, le développement de la traction mécanique, le rôle croissant des automobiles et de la vitesse dans la détérioration des chaussées ont conduit, au cours des années 1920, à l'abandon du collier au profit de l'unité de véhicule et d'une nouvelle typologie qui évoluera d'un recensement à l'autre (Delemer, 1926), comme le tonnage brut uniformisé (CAC, 770444 articles 1 et 2). Le bétail disparaît en 1955, la traction animale en 1960.

Ces comptages ne sont donc pas entièrement compatibles. Ils ne sont pas non plus exhaustifs, puisque ne portant que sur le réseau routier national. Sa définition varie considérablement en 1930 avec le reclassement d'une partie des routes départementales et des chemins de grande communication, désormais routes nationales (Mahieu, s. d.) : la longueur du réseau double brusquement, passant de 40 000 km à 80 000 km environ sur un total de 630 000 km, hors chemins ruraux et voirie urbaine (INSEE, 1961). Malgré ces limites, les recensements constituent l'unique source de connaissance du trafic routier jusqu'aux années 1950, et reflètent avec fidélité les tendances de son évolution (Barles et coll., 2004).

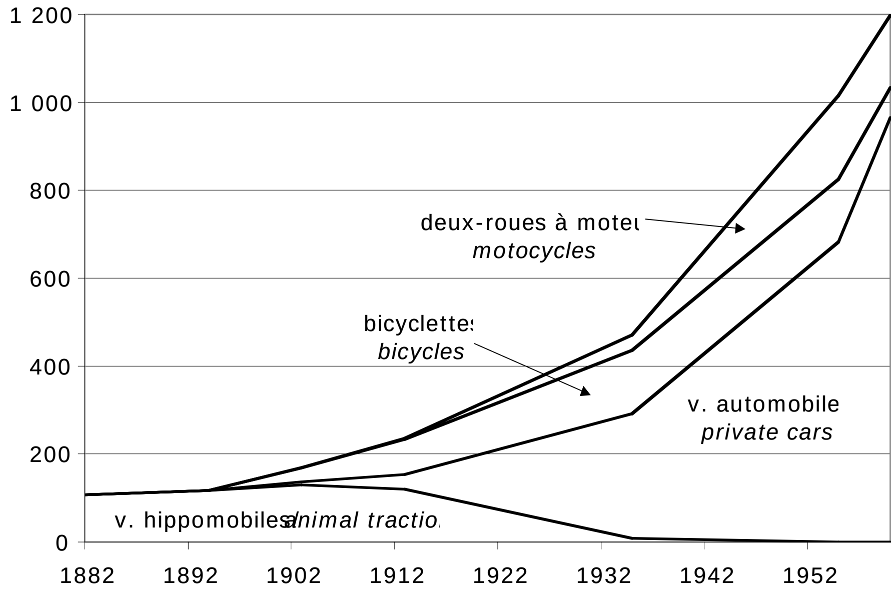
Ces données ont été complétées par les statistiques établies par le Comité professionnel du pétrole (1955-1969). Les enquêtes annuelles réalisées auprès des transporteurs permettent en effet de connaître les parcours et consommations moyens de leurs véhicules, dont on déduit leur consommation. Celle-ci est soustraite à la consommation totale de carburants, le reste étant affecté aux transports individuels (770444 article 3). Nous n'avons pas eu accès à l'ensemble des séries statistiques, qui seront donc à compléter.

2. Résultats

L'examen des recensements de la circulation sur les routes nationales montre dans un premier temps l'essor général du trafic, qui quadruple entre 1882 et 1960, passant de 353 animaux/jour en 1882 (nombre de chevaux attelés ajouté à celui des animaux non attelés et au menu bétail) à 1 480 véh/j en 1960. Mais bien plus marquantes sont à la fois la part grandissante des véhicules particuliers dans l'ensemble du trafic — 30 % environ en 1882 et 1894, 40 % en 1903, 50 % en 1913, 70 % en 1935, 80 % en 1955 et 1960 — et l'intensification de ce trafic particulier, multiplié par onze entre 1882 et 1960.

Figure 3 : Trafic cumulé des véhicules individuels sur les routes nationales, France, 1882-1960 (colliers et véh/j).

Figure 3: Cumulated traffic of private vehicles on national roads, France, 1882-1960 (draught horses and vehicles/day).



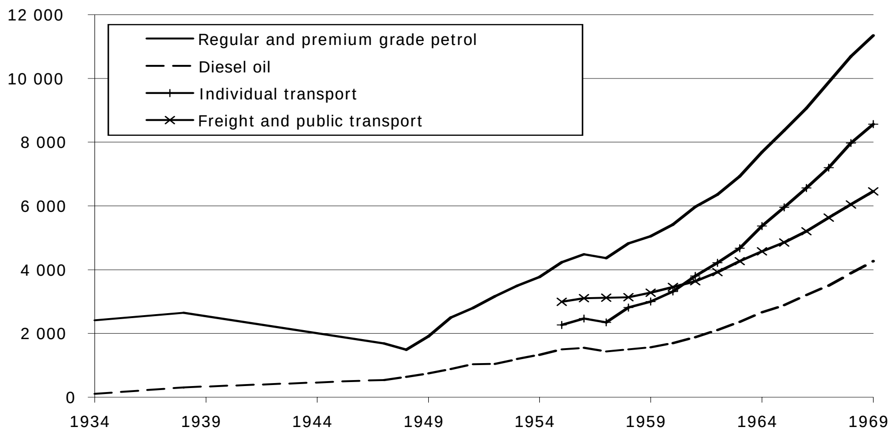
La figure 3 en présente le cumul. Dès le tout début du XXe siècle, la traction animale connaît une légère régression tandis que l'augmentation du trafic est principalement assurée par les bicyclettes. Dans l'entre-deux-guerres, la traction animale diminue, la croissance de la circulation étant aussi bien due aux bicyclettes qu'aux voitures particulières (automobiles) ; cependant le trafic des premières semble plafonner dès 1935. Après la Seconde Guerre Mondiale, l'intensification du trafic provient de l'automobile pour l'essentiel, avec un appoint des deux roues à moteur.

Une analyse rapide pourrait conduire à conclure que seule le trafic motorisé produit un impact environnemental remarquable. La traction animale ne doit cependant pas être négligée. Le transport individuel routier de personnes repose, à l'aube de la mécanisation, sur un nombre non négligeable de chevaux : 1 100 000 en 1888, 1 200 000 en 1920 (Foville, 1890 ; Bertrand, 1931 ; chiffres concernant les seuls chevaux destinés au transport des personnes et imposés). Compte tenu des rations couramment admises (15 l d'avoine/cheval/jour, 4/3 de botte de foin de 5 kg), les chevaux de 1888 ont consommé environ 60 millions d'hectolitres d'avoine et 530 millions de bottes de foin. Considérant les rendements moyens de 1888 (Statistique agricole, 1889), il aura fallu 2,6 millions d'hectares pour produire la seule avoine destinée à ces chevaux. Ils ont certes fourni un fumier recherché, mais représentent un prélèvement de matières considérable sur les systèmes agricoles.

La figure 4 montre l'évolution des consommations de carburants — essence (ordinaire et super) et diesel — de 1934 à 1969. Indépendamment de leur augmentation générale (sextuplement), elle montre l'essor du diesel, uniquement destiné aux véhicules utilitaires jusqu'en 1960, puis touchant les véhicules particuliers. En outre, la part des transports individuels dans la consommation de carburant dépasse celles des transports de marchandises et collectifs à partir de 1961. Les voitures particulières dominent largement dans la consommation des véhicules individuels pour la période renseignée (1955-1969), puisqu'elles en représentent 90 à 95 %. L'analyse de la décennie précédente montrerait probablement que les deux roues motorisés occupaient alors une part plus importante de la consommation du fait de leur importance relative dans le parc de véhicules individuels et de leur cylindrée plus importante, associées à des parcours annuels plus longs.

Figure 4 : Consommation de carburants par le trafic routier, France, 1934-1969 (milliers de tonnes).

Figure 4: Road transport fuel consumption, France, 1934-1969 (thousands tons).



Au total, la consommation de carburant représente 58 kg/hab/an en 1934, 97 kg/hab/an en 1955 et 225 kg/hab/an en 1969 (quatre fois plus qu'en 1934). Pour ces deux dernières années, la consommation des transports individuels s'élève respectivement à 52 kg/hab/an et 170 kg/hab/an.

Conclusion

Ces quelques lignes montrent l'importance croissante prise par le système de transport routier dans la société conçue dans sa dimension la plus matérielle : les véhicules sont de plus en plus présents dans le paysage, qu'ils soient neufs ou usagés, en service ou au rebut. Les quantités de matières mobilisées — matériaux constitutifs des véhicules, carburants — ne cessent d'augmenter avec les parcs et les trafics, de même que les quantités stockées, consommées et rejetées. Ce résultat n'est pas surprenant ; il montre néanmoins qu'il est difficile d'ignorer l'entre-deux-guerres comme de laisser de côté les deux roues dans l'analyse de l'évolution des interactions transport-environnement.

Les indicateurs simples tels que stocks, entrées et sorties de matières qui ont été établis résument de façon à la fois synthétique et démonstrative les évolutions à l'œuvre. Leur détermination semble prometteuse dans une perspective historique, rétrospective et comparative. Il reste néanmoins tant à poursuivre la collecte des données qu'à affiner les nombreuses hypothèses qui jalonnent ce travail qui peut être considéré comme la base de futures et plus précises analyses de bilan de matières voire d'analyses de cycle de vie.

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La mobilité des Français et leurs émissions de CO_2

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Abstract

The commitment made by the French government to reduce greenhouse gas emissions by a factor of 4 by 2050 thus calls for reflection now on the technological progress that can be made, the organisation of our society and the lifestyles that generate transport and emissions. In this context, the objective of this presentation is to take stock of CO_2 emissions (more than 95% of greenhouse gas emissions from transport) generated by individual mobility in France by asking the following questions: what socio-demographic groups require transport? What types of mobility are involved (local, long distance). What modes are used and for what activities? What are their contributions to the global evaluation?

To answer this series of questions, we have made use of the Transport and Communication survey performed in 1993-1994 by the INSEE on 14,213 households. We have estimated the CO_2 emissions related to each trip recorded by the survey as a function of distance, type of vehicle and its characteristics and the road taken in the case of private cars. A typology of individuals was then established to explain the emission levels obtained, highlighting in particular the lifecycle, the residential localisation and the motorization rate relating to local mobility, and revenue relating to long distance mobility.

Key-words : *CO_2 emissions, greenhouse effect, socioeconomic analysis, individual behaviour, national transport survey 1993-94, local mobility, long distance mobility*

Résumé

Cette communication propose un bilan des émissions de CO_2 générés par la mobilité des résidents du territoire français. Après une présentation de la méthode de calcul de ces émissions, deux grands types de mobilités aux dynamiques très différentes sont distingués, locale et à longue distance. Les facteurs socioéconomiques explicatifs des comportements individuels et des émissions qui en découlent sont analysés.

Mots-clefs : *émissions de CO_2, effet de serre, analyse socio-économique, comportements individuels, enquête nationale transport 1993-94, mobilité locale, mobilité longue distance.*

Introduction

Les transports sont à l'origine de 16% des émissions de gaz à effet de serre liées aux activités humaines dans le monde. En France, ils sont même responsables de 23% de ces émissions hors aérien, du fait d'une part d'un niveau de vie élevé permettant un haut niveau de mobilité et d'autre part d'une forte production d'énergie nucléaire qui diminue d'autant le poids en CO_2 des activités utilisant l'électricité[2]. L'engagement du gouvernement français d'une division par 4 des émissions nationales de gaz à effet de serre à l'horizon 2050 nécessite dès lors de s'interroger tant sur les progrès technologiques réalisables que sur l'organisation de notre société et les modes de vie qui génèrent déplacements et émissions.

Le CO_2, très lié au carbone contenu dans les énergies fossiles, représente plus de 95% des GES dans le secteur des transports qui nous intéresse plus particulièrement ici. Ce sont donc les émissions de ce gaz qui servent d'indicateur dans cet article. Des inventaires sont régulièrement établis au niveau national pour connaître ces émissions ainsi que la part imputable à chaque secteur d'activité (en France, voir Citepa, 2005, pour les inventaires intersectoriels globaux ; Ademe, 1998, ou Hugrel, 2004, pour les transports). De nombreux autres travaux existent également pour évaluer les émissions de CO_2 à des niveaux internationaux (GIEC, 1997).

Dans le domaine des transports, ces calculs reposent sur des statistiques de trafics par mode, croisées avec les résultats de travaux sur les émissions des véhicules (exemple des modèles COPERT III en Europe ou MOBILE 6 en Amérique du Nord). Par ailleurs, des travaux s'intéressant à la dynamique d'évolution des transports soulignent les risques d'une très forte montée des émissions de CO_2 par ce secteur, avec l'accès à la voiture dans les pays émergents et une exigence de vitesse qui amène à un recours de plus en plus fréquent à l'avion dans les pays riches (Schafer, 2000). Cependant, les estimations de trafics sur lesquelles reposent ces chiffres ne nous disent rien sur qui les réalisent, ni pour quelles raisons. Les travaux portant sur l'analyse des mobilités individuelles peuvent enrichir la compréhension des émissions de CO_2 par la mise en lumière des facteurs socioéconomiques situés en amont, permettant également de mieux comprendre l'impact social que peuvent avoir des mesures destinées à limiter les gaz à effet de serre émis par les transports.

L'objectif de cette présentation est de proposer un bilan des émissions de CO_2 générées par la mobilité individuelle en France : quels groupes sociodémographiques se déplacent, pour quels types de mobilité (locale, longue distance), avec quels modes et pour réaliser quelles activités ? L'enquête

[2] Ces chiffres prennent en compte l'ensemble des gaz à effet de serre retenus par le GIEC, pondérés de leur pouvoir radiatif à 100 ans. Ils ont été calculés à partir des données du GIEC et de l'AIE pour les émissions mondiales, du Citepa au niveau national. J.-M JANCOVICI, http://www.manicore.com

Transports et communications réalisées en 1993-1994 par l'INSEE a été exploitée pour répondre à ces questions. Les émissions de CO_2 des déplacements recueillis par l'enquête ont été estimées et une typologie des individus a été établie pour expliquer les niveaux obtenus. Après une présentation des options méthodologiques retenues, cet article fait ainsi ressortir les principaux facteurs socio-économiques explicatifs des émissions de CO_2 liés à la mobilité des résidents du territoire français.

Estimer les émissions de CO_2 liées à la mobilité des Français

La dernière enquête nationale transports et communications, qui donne une image de la mobilité quotidienne et à longue distance des résidents du territoire français en 1994, fournit une base cohérente et exhaustive pour l'analyse des facteurs socioéconomiques qui influent sur les mobilités individuelles (§1). Sa construction fait notamment ressortir les logiques différenciées des mobilités locale et longue distance (§2). Elle fournit également des informations qui permettent d'estimer les émissions de CO_2 au niveau de chaque déplacement (§3), avec cependant un niveau plus ou moins fin suivant le type de déplacement (§4).

1. L'enquête nationale transports et communications comme base de données déplacements

Réalisée par l'INSEE entre mai 1993 et avril 1994 auprès d'un échantillon final de 14 213 ménages, cette enquête apporte plusieurs éclairages sur la mobilité des Français (RTS, 1997) :

- un carnet de bord a été rempli pendant une semaine pour une voiture du ménage ;
- une personne, tirée au sort parmi les individus du ménage de plus de 6 ans présents au moment de l'enquête, a été interrogée sur ses déplacements de la veille et du week-end précédent et 92 925 déplacements ont ainsi été recueillis ;
- un second tirage d'une personne du ménage a également permis de recueillir 41 774 déplacements réalisés à longue distance (*ie* à plus de 80 km à vol d'oiseau du domicile) au cours des trois derniers mois précédents l'enquête. Leur recueil nécessite cette procédure particulière pour permettre une exploitation fiable. En effet, ils sont moins fréquents et leur nombre est statistiquement insuffisant avec une simple enquête sur les déplacements de la veille. Dans le même esprit, le tirage au sort favorisait la personne du ménage susceptible d'avoir le plus de longs déplacements, les déformations induites étant ensuite corrigées par le jeu des coefficients de redressement.

Tous les déplacements, à motifs privés comme dans le cadre du travail, sont pris en compte. Seuls les déplacements professionnels des personnes dont le métier est de se déplacer (chauffeurs routiers, livreurs, taxis, etc.) sont omis. Les caractéristiques socioéconomiques du ménage et des personnes interrogées sont par ailleurs saisies lors de l'enquête. De même, une description détaillée des véhicules du ménage est recueillie.

Compte tenu de notre problématique, presque tous les éléments de l'enquête ont été exploités : les informations fournies par le questionnaire spécifique sur les déplacements à longue distance ont complété celles issues des déplacements de la veille, permettant de disposer d'une bonne image de la mobilité des Français ; les caractéristiques socioéconomiques ont constitué la toile de fond analytique des comportements de déplacements ; enfin la description détaillée des véhicules a permis un calcul plus précis des émissions de CO_2.

2. Mobilités locale et à longue distance

Les informations fournies par les déplacements de la veille et ceux à longue distance tout à la fois se recoupent et sont disjoints. Il a donc été nécessaire de faire des choix méthodologiques pour combiner les deux sources d'information de manière satisfaisante.

Tout d'abord, les deux bases se recoupent : il est tout à fait possible que certains déplacements réalisés la veille ou le week-end précédent soient sortis du rayon de 80 km du domicile. Pour éviter les doubles comptes, ces déplacements ont été retirés de l'exploitation. La base « longue distance » a été privilégiée puisque plus performante pour représenter ce type de déplacements. Mais par ailleurs, les deux bases sont disjointes. Un tirage au sort séparé est effectué au sein du ménage pour chacun des deux questionnaires et ça n'est donc pas forcément la même personne qui est interrogée.

Deux possibilités sont ouvertes pour l'exploitation des résultats. On peut travailler sur le sous-échantillon des 7729 individus qui ont été tirés au sort pour les deux questionnaires (Orfeuil, Soleyret, 2002). L'avantage réside dans la cohérence apportée pour analyser les liens entre mobilité locale et longue distance ; l'inconvénient est un échantillon deux fois plus petit, qui se trouve donc plus vite limité pour des analyses fines. Mais on peut également travailler sur les deux bases séparément et cumuler les résultats de distances parcourues et d'émissions de CO_2 à un niveau agrégé ou semi-agrégé, par catégories de personnes aux caractéristiques et aux comportements proches. Le lien entre les deux types de déplacements est perdu au niveau individuel, par contre toute l'enquête peut-être exploitée et des analyses plus détaillées peuvent être menées au sein de sous groupes d'individus. C'est cette seconde option qui a été retenue pour ce travail.

3. Un "budget CO_2" des déplacements

Chaque déplacement enregistré par l'enquête est caractérisé par plusieurs variables qui rendent possible une estimation de ses émissions de CO_2 (cf. partie suivante). On peut ainsi partir du niveau le plus fin, correspondant au déplacement, ce qui fournit une très grande souplesse pour les analyses ultérieures. Les résultats peuvent en effet être réagrégés ensuite au niveau d'analyse désiré, par mode, par type de déplacement ou, en ce qui nous concerne ici, par type de personne.

Le principe de ce type d'analyse a été posé dans le cadre des enquêtes déplacements locales pour faire le bilan des consommations énergétiques dues aux mobilités au sein d'une agglomération (Orfeuil, 1984). Il a été progressivement élargi aux émissions de polluants atmosphériques (Gallez, Hivert, 1998), avant d'être systématisé dans sa méthode par la mise en oeuvre d'un logiciel adapté aux enquêtes françaises locales (Lanquar, 2005). Le principe en est repris ici, avec des adaptations en matière de calcul des émissions pour pouvoir être appliqué à l'enquête nationale.

4. Mesurer les émissions de CO_2

Tous les calculs d'émission de CO_2 réalisés reposent sur la méthodologie proposée dans le programme européen MEET (European Communities, 1999). Les choix et hypothèses de calcul précis sont présentés dans (Raux et *alii*, 2005).

Mobilités locale et longue distance confondues, l'automobile est prédominante puisqu'elle représente 75% du total des distances. Pour ce mode, MEET fournit des courbes d'émissions en fonction de la vitesse moyenne sur le parcours, de la cylindrée, de l'âge et du type de carburation des véhicules. L'enquête transports et communications fournit directement l'information sur ces deux dernières caractéristiques. La puissance fiscale des véhicules est précisée, à partir de laquelle, âge et carburation aidant, la cylindrée a pu être inférée (Gallez, Hivert, 1998). Pour chaque déplacement, on connaît la distance parcourue sur les différents réseaux, pour lesquels une vitesse moyenne a été fixée. Connaissant l'heure de départ et d'arrivée, il était également possible de savoir si le démarrage s'opérait à froid ou non, compte tenu du déplacement précédent. Dans l'affirmative, le calcul du coefficient de surémission proposé dans MEET a été mis en oeuvre. Enfin, les émissions de CO_2 ainsi établies ont été attribuées à la personne enquêtée au prorata du nombre d'occupants du véhicule, qui est également connu dans l'enquête.

Tableau 1 : émissions moyennes de CO_2 des différents modes (g/pers.km)

Table. 1: average CO_2 emissions from different modes (g/pers.km)

Automobile	111
Avion	169
Train	15
Autocar	24
Bus urbain	41
2 roues motorisés	41

Les autres modes apparaissent plus marginaux en termes de distances parcourues : l'avion (11%), le train (4,5%), l'autocar (3,5%) puis les autres modes (6%). Ils ont également été traités à partir de la méthodologie proposée par MEET. Dans le cas de l'avion et du train, des émissions moyennes par voyageur.kilomètre ont été estimées à partir du détail des distances annuelles totales parcourues par mode, et rapportées au trafic total de voyageurs, connus par ailleurs dans les statistiques nationales (Castro Ortega, 2001). Ces coefficients moyens ont ensuite été appliqués uniformément à chaque déplacement concerné. Cette méthode fournit un résultat juste au niveau global, mais elle conduit à lisser les variations qui existent d'un déplacement à l'autre, notamment dans le cas de l'avion pour lequel les émissions unitaires sont très différenciées suivant la distance (poids important du décollage et de l'atterrissage), le type d'aéronef et son taux de remplissage. Le tableau 1 donne un ordre de grandeur des émissions moyennes de CO_2 finalement obtenues par voyageur.kilomètre (David, 2005), compte tenu des émissions de chaque mode et des taux de remplissage moyens correspondants (pour 1994, date de référence de l'enquête).

Des caractéristiques socio-économiques qui conditionnent fortement les émissions de CO_2

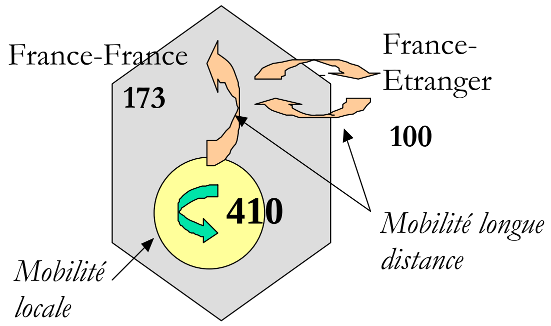
Il reste maintenant à se pencher sur les déterminants socioéconomiques des personnes qui peuvent apporter une compréhension de leur mobilité et des émissions de CO_2 qui s'y rapportent. Deux grands segments de mobilité se distinguent nettement avec la mobilité locale d'une part et la mobilité à longue distance de l'autre (§1). Ils sont donc séparés dans la présentation et des analyses distinctes des déterminants individuels sont proposées (§2).

1. Mobilité locale, mobilité longue distance, des pratiques bien différentes

Une première analyse permet de distinguer rapidement les logiques des mobilités locale et longue distance. La première représente en moyenne 19,6 déplacements et 159 kilomètres par semaine (soit environ 7 900 km/an). La seconde est beaucoup plus exceptionnelle, avec 6 déplacements par personne et par an à plus de 80 kilomètres du domicile, représentant 5350 km/an. Si la mobilité locale correspond à 99% des déplacements pour 60% des distances parcourues, la mobilité à longue distance fait respectivement 1% des déplacements pour 40% des distances.

Fig. 1 : Champs spatial couvert par l'enquête et milliards de voyageurs km correspondants

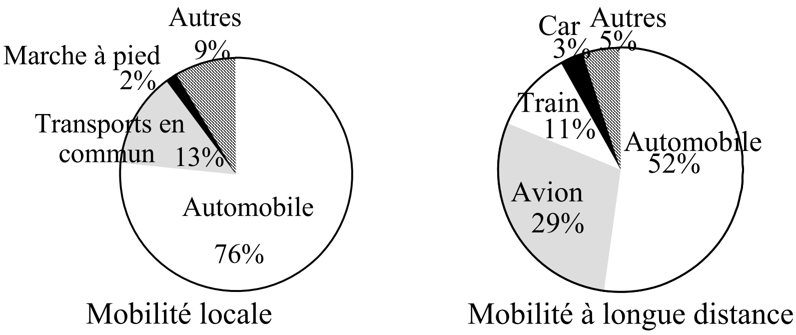
Fig. 1: Spatial level covered by the survey



Les modes utilisés varient en conséquence. La voiture reste majoritaire dans les deux cas, mais chute de 76 à 52% des kilomètres parcourus entre le local et la longue distance, au profit notamment de l'avion (29%) et du train (11%). En termes d'émissions de CO_2, les surémissions de l'avion sont compensées par les excellentes performances du train et de l'autocar, et sur 1440 kg émis en moyenne par personne et par an pour se déplacer, 857 kg (60%) proviennent du local et 583 kg (40%) de la longue distance, proportionnellement aux distances parcourues.

Fig 2 : Répartition modale des distances parcourues localement et à longue distance (David, 2005)

Fig. 2:Modal share of local and long distance mobility relatively to distance covered (David, 2005)



2. Statut, revenu, localisation et taille de l'agglomération : les 4 variables clé explicatives des mobilités et des émissions de CO_2

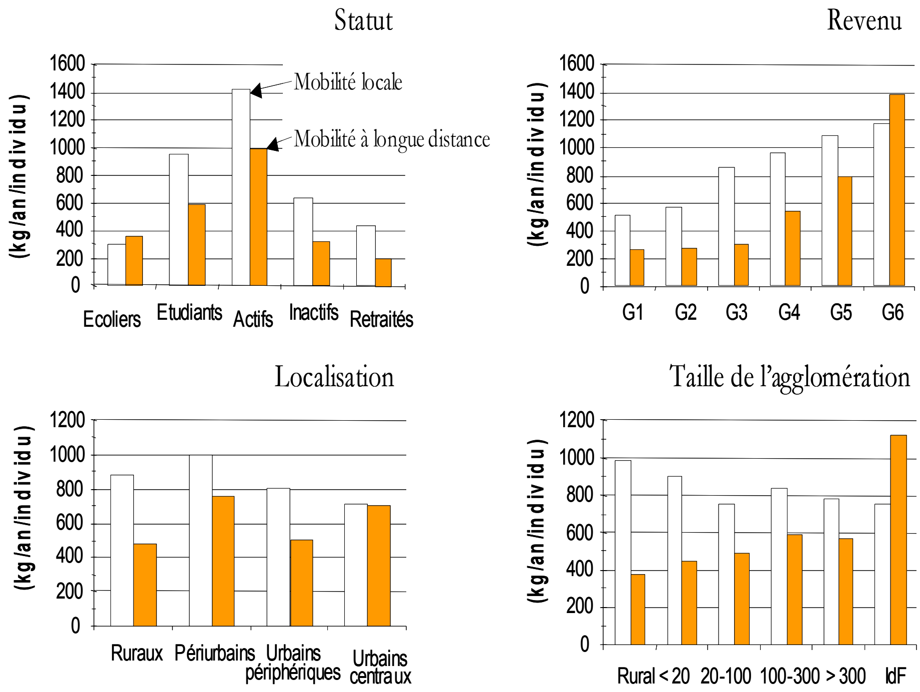
De précédents travaux (Nicolas et *alii*, 2001, au niveau local ; Orfeuil, Soleyret, 2002, ou Mézière, 2003, sur l'enquête transports et communications) ont permis de faire ressortir les principaux facteurs explicatifs des mobilités individuelles qui ont ensuite servi de grille d'analyse des résultats d'émissions. Quatre d'entre eux se dégagent plus particulièrement :

- Le statut (scolaire, étudiant, actif, chômeur, au foyer ou retraité), se recoupe largement avec l'âge et le cycle de vie, et joue sur les activités qui structurent et rythment la vie quotidienne.
- Le niveau de revenu du ménage, ramené à l'unité de consommation[3], est découpé ici en 6 classes d'effectifs égaux. Il facilite toujours l'accès à la voiture particulière des personnes en âge de conduire, même si son usage est bien généralisé aujourd'hui (Dupuy, 1999) ; il ouvre également de larges possibilités en matière de mobilités de loisirs à longue distance.
- La localisation résidentielle du ménage (ville centre, banlieue, périurbain, rural) affecte les distances parcourues au quotidien et joue, en conséquence, sur les modes utilisés.
- La taille de l'agglomération (Ile-de-France, plus de 300 000 habitants, 100-300 000, moins de 100 000, rural), enfin, permet de faire ressortir le cas particulier la région parisienne, avec son réseau de transports collectifs très développé et le haut niveau de revenu de sa population.

[3] Du fait des économies d'échelle au sein du ménage, l'Insee propose de compter la première personne pour 1, puis tous les autres adultes pour 0,5 et les enfants de moins de 15 ans pour 0,3.

Figure 3. :Emissions moyennes de CO_2 (kg/an) et caractéristiques individuelles (David, 2005)

Figure. 3: average CO_2 emission (kg/year) and individual characteristics (David, 2005)



En mobilité quotidienne locale, ce sont d'abord le statut et la localisation résidentielle qui jouent. L'accès au volant constitue également un facteur notable parmi les personnes en âge de conduire.

Les actifs, contraints par leurs trajets domicile-travail, parcourent des distances beaucoup plus longues que les autres et tendent, du coup, à plus recourir à la voiture (228 km/semaine, dont 89% en voiture, pour 28,6 kg de CO_2 émis). La distinction homme/femme au sein de ce groupe permet de souligner le fait que le lieu de travail de ces dernières reste en moyenne plus proche du domicile, tendant à diminuer leurs émissions globales de 20 à 30% par rapport à leurs homologues masculins. La localisation est bien sûr déterminante, même si les distances restent de toute façon importantes : un homme actif motorisé du centre parcourt 217 km/semaine pour 33 kg de CO_2 contre 303 km et 39 kg pour un rural, les rapports étant du même ordre parmi les femmes actives. Au sein de ce groupe, le niveau de revenu affecte peu les distances parcourues localement et les émissions afférentes. On peut ainsi montrer sur le cas lyonnais qu'à localisation identique, dès lors qu'ils sont motorisés, un actif de ménage aisé et un actif de ménage modeste parcourent les mêmes distances et recourent à leur automobile avec la même intensité (Nicolas et *alii*, 2001; Paulo, 2005). Par contre, le fait d'être non motorisé fait chuter les distances à 125 km/semaine et les émissions à 7,2 kg.

Chez les jeunes, l'autonomisation et l'éloignement progressif des lieux d'enseignement conduisent à un fort écart entre les écoliers urbains du primaire (77 km/semaine pour 4,3 kg de CO_2) et les étudiants non centraux (215 km pour 21,3 kg de CO_2).

Chez les inactifs, chômeurs ou personnes au foyer, c'est l'accès au volant qui apparaît déterminant (81 km et 5,3 kg de CO_2 pour un chômeur, 75,6 km et 4,7 kg pour une personne au foyer lorsqu'ils ne sont pas motorisés, contre 132 km et 12,7 kg de CO_2 en moyenne au sein du groupe). La localisation devient secondaire. Le revenu global du ménage apparaît donc comme une variable clé pour permettre aux personnes de ce groupe d'accéder à la mobilité automobile. Les retraités apparaissent moins mobiles que les autres (86 km/semaine et 8,6 kg de CO_2). L'âge est déterminant au sein de ce groupe, avec des seniors jeunes, motorisés et mobiles d'un côté, et des personnes plus âgées (la limite est classiquement posée autour de 75 ans), qui utilisent moins la voiture, tout à la fois pour des raisons de santé et par effet de génération (Pochet, 2003).

Enfin, on peut se pencher sur le cas particulier de l'Ile-de-France. En effet, alors que la taille de cette agglomération induit des distances à parcourir nettement plus longues qu'ailleurs, le recours important à un réseau de transports collectifs fortement électrifié conduit à des émissions locales dans la moyenne des autres agglomérations. Par exemple, les Franciliens parcourent 70% de distance en plus que les Lyonnais, quelle que soit leur localisation dans leur agglomération respective, alors que leurs émissions de CO_2 sont identiques (Nicolas et *alii*, 2001).

La mobilité à longue distance ne correspond pas aux mêmes logiques et ne repose pas sur les mêmes facteurs explicatifs individuels. Le niveau de revenu est ici premier, suivit de loin par la position dans le cycle de vie. On peut associer au revenu des opportunités plus nombreuses de déplacements professionnels à longue distance pour les actifs. Mais au delà, en matière de loisirs, les possibilités "d'escapades" et de voyages apparaissent sans commune mesure entre aisés et modestes. Ainsi, entre le sixième le plus modeste et le sixième le plus aisé de la population, les émissions de CO_2 passent de 262 à 1 380 kg/an. Le cas particulier des Franciliens, qui émettent deux fois plus de CO_2 que les habitants de Province pour leur mobilité à longue distance (1 121 kg/an contre 477 kg/an) s'explique essentiellement par ce facteur (Orfeuil, Soleyret, 2002).

L'activité/inactivité apparaît ici absorbée par cette variable de revenu, car le taux d'inactifs du ménage fait baisser le revenu par UC, qui sert ici d'indicateur. Par contre l'âge et la position dans le cycle de vie restent explicatives. Les jeunes adultes, plus souvent célibataires et sans enfants, se déplacent plus que les personnes plus mûres plus souvent installées dans des ménages avec enfants. Par ailleurs, les retraités tendent à être moins mobiles que les autres, ceci étant encore plus net pour les plus de 75 ans (Mézière, 2003).

Conclusion

L'analyse des caractéristiques socioéconomiques individuelles s'avère tout à fait intéressante pour comprendre les émissions de CO_2 liées à la mobilité des personnes.

Deux segments de mobilité bien distincts apparaissent. D'une part, une mobilité structurée par les activités du quotidien s'exprime sur un espace local et se trouve fortement conditionnée par le statut social de la personne, qui joue sur les activités qu'elle réalise, et par sa localisation résidentielle, qui affecte les distances qu'elle doit parcourir et le mode de transport utilisé. D'autre part, une mobilité à longue distance s'exprime de manière plus exceptionnelle puisqu'elle ne représente que 1% des déplacements; elle correspond cependant à à 40% des distances parcourues et des émissions de CO_2, et c'est cette mobilité qui croît le plus. Le revenu joue ici de manière déterminante et la baisse des prix relatifs l'a portée ces dernières années.

Ainsi, le débat sur l'étalement urbain, le type d'urbanisme et les localisations des activités économiques et résidentielles, constitue un vrai enjeu au niveau local, qui reste encore le lieu où est généré le plus fort niveau d'émissions. Par contre, une hausse des coûts d'usage de l'automobile, (renforcement des taxes ou renchérissement du prix du pétrole), sera beaucoup plus ressentie par les plus modestes, notamment les actifs pour lesquels elle constitue aujourd'hui un outil obligé pour relier domicile et travail. Et la forte croissance des prix de l'immobilier laisse, à ce niveau, peu de possibilité d'ajustement par la mobilité résidentielle.

La mobilité à longue distance est celle qui a le plus augmenté ces dernières années. Ainsi par exemple, la consommation de carburant par le transport aérien a augmenté en France de 50% entre 1994 et 2003 (DAEI/SES-Insee, 2004). Ainsi, même si elle ne représentait "que" 40% des émissions de CO_2 de la mobilité des Français en 1994, elle n'en constitue pas moins un enjeu fort lorsque l'on observe les tendances d'évolution à long terme. Une hausse des prix apparaît ici moins génératrice d'inéquités que dans le cas précédent, en ce qu'elle concerne des déplacements moins contraints et touche plutôt la partie plus aisée de la population. Elle peut avoir un impact sensible lorsque l'on observe le lien fort qu'elle entretient avec la croissance des revenus. Concernant cette mobilité à longue distance, ses déterminants et ses évolutions, les bases statistiques existantes dans le tourisme mériteraient sans doute d'être exploitées de manière plus approfondie en France. De même, le suivi statististique plus fin et généralisé qui se met en place aujourd'hui apportera sans doute aucun les bases d'une meilleure connaissance des dynamiques qui sous-tendent cette mobilité.

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Driving cycles and pollutant emissions estimation

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Abstract

Significant works have been conducted within the European research project ARTEMIS to analyse the influence of the driving cycles as regards the estimation of the emissions. These works included:

- The review of a large range of cycles, the building-up of a set of contrasted cycles, and the measurement on chassis dynamometer of the pollutants emissions of 9 passenger cars. These data and a complementary dataset of 30 vehicles tested using both the Artemis cycles and specific cycles for the high and low-powered cars, were analysed to characterize the influence of the cycles and of the kinematic parameters on the emission.

- The analysis of the Artemis hot emission database for cars, its harmonization as regards test cycles, the elaboration of reference emissions, and the development of a specific method to compute the emissions at a low spatial scale, i.e. the so-called traffic situation approach.

These results – briefly described in this paper - have been implemented in the European emission model Artemis for the light vehicles.

Keys-words*: driving cycle, emission factor, atmospheric pollutant, passenger car, measurement, vehicle bench, method, modelling*

Résumé

Des travaux conséquents ont été menés dans le cadre du projet de recherche européen sur l'influence du cycle de conduite sur les estimations des émissions de polluants. Ces travaux incluent:

- Le recueil d'un grand nombre de cycles, la construction d'un jeu de cycles contrastés, et la mesure des émissions de 9 voitures utilisant ces cycles sur banc d'essai. Ces données, ainsi que celles mesurées sur 30 voitures testées sur les cycles Artemis et sur des cycles dédiés aux véhicules fortement et faiblement motorisés, ont été analysées dans le but de caractériser l'influence des cycles et des paramètres cinématiques sur les émissions,

- L'analyse des données d'émission de la base de données Artemis, leur harmonisation selon le cycle d'essai, et l'élaboration d'émissions de référence, et le développement d'une approche spécifique pour calculer les émissions à l'échelle de la rue – ou approche situations de trafic.

Ces résultats sont brièvement récapitulés dans ce papier. Ils ont été mis en œuvre dans le cadre des nouveaux outils européens Artemis, pour l'estimation des émissions des véhicules légers.

***Mots-clés**: cycle de conduite, factors d'émission, polluant atmosphérique, voiture particulière, driving cycle, banc d'essai, méthode, modélisation.*

Introduction

The driving or test cycle is used on a chassis dynamometer to measure the pollutant emissions (and the fuel consumption) from the road vehicles. As a basic element of the method, its advantages (representativeness, completeness, reproductibility, reliability, etc.) are fundamental as regards the emissions data, the derived factors and finally the pollutant emission estimation process. Furthermore, the cycle is the only link with the driving conditions and the willingness to estimate emissions on a very local scale (i.e. one street for a given traffic condition) increases dramatically the need to understand and model the link between emissions and very local driving conditions through "microscopic" kinematic parameters.

Therefore, works have been conducted in the ARTEMIS[4] research project, in order to (i) review the existing driving cycles as regards their kinematics and representativeness, and select contrasted cycles to measure the emission of a cars sample, and (ii) analyse the sensitivity of emissions as regards test cycles and identify the main kinematic parameters (speed, acceleration, stops, etc.) that would enable a good estimation of the emissions as a function of the driving conditions.

These aims were finally extended to two additional directions:

- The harmonization as regards the test cycle needed for the large amount of emissions data collected in ARTEMIS and measured on a large range of driving cycles,
- The building-up of an approach for the assessment of the emissions at a street level, and of emissions figures for hundreds of "traffic situations" (per road type and traffic conditions), these figures being calculated from emissions measured under dozens of test conditions.

[4] ARTEMIS: Assessment and reliability of transport emission models and inventory systems. Project funded by the European Commission within the 5th Framework Research Programme, DG TREN.

Method and data

These works rely on 3 different experimentations and datasets briefly recapitulated hereafter:

- the WP3141 experimentation was specifically designed to analyse the influence of the driving cycles in the Artemis project, through the selection of contrasted cycles and measurement of the emissions of 9 cars.
- the PNR-Ademe experimentation aimed at studying the incidence when using common test cycles for all vehicles rather than cycles depending on the motorisation. Both specific cycles (designed for high and low powered cars) and Artemis cycles (André, 2004) were used on a sample of 30 representative French cars.
- the Artemis cars emission dataset was collected amongst the European laboratories and covered measurements from 1980 up to now (André J.M., 2005). It accounts for about 2,800 cars covering most of the European categories (fuel, technologies and emission standard), 800 different cycles or sub-cycles and 27,000 emission tests. From this database, 20,000 hot emission tests were analysed, covering 217 relevant cycles and 158 sub-cycles.

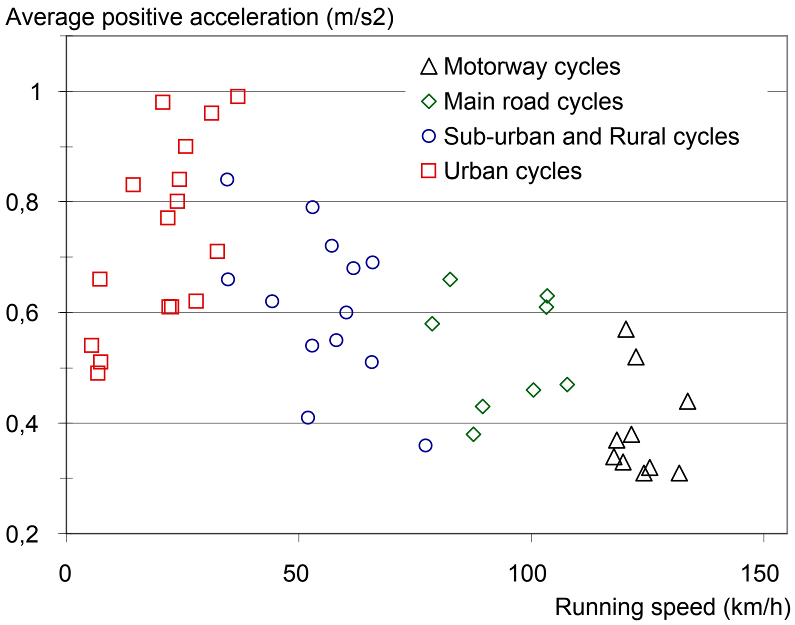
1. Driving cycles

To analyse the influence of test cycles, we envisaged a limited number of cycles, maximising their contrasts in terms of driving condition. I.e., 213 driving cycles / sub-cycles were collected (André et al, 2006), of which US and European test cycles (FTP, Highway, US.06, ECE15, NEDC), Swiss Handbook cycles representing the Swiss driving (De Haan et al. 2001), Neapolitan driving patterns (Rapone et al. 1995) and numerous European representative cycles.

The selection of cycles was based on the analysis of the kinematical content of the cycles, through the 2-dimensional time distribution of the instantaneous speed and acceleration. A factorial analysis (Binary correspondences) followed by an automatic classification enabled establishing a typology of test cycles into contrasted classes. Statistical criteria then allowed the selection of contrasted cycles, while preserving the representativity of the initial dataset. Due to the high heterogeneity of the driving conditions, a pre-classification into urban, suburban/rural, main roads and motorway was performed. These last 2 categories were analysed together due to their low number of cycles and relative similarity. As mentioned below, the contrasted emission between these driving conditions confirms the relevancy of an analysis by driving type. A further classification enabled the definition of contrasted groups of cycles for each driving type. Cycles or sub-cycles were selected amongst these groups, then offering an optimisation between representativity and contrast. Where possible, entire sets of sub-cycles (within a cycle) were privileged. The Artemis and Handbook cycles were part of this selection. Since Neapolitan driving patterns and modem sub-cycles were selected, 2 composite cycles were built-up using these data. Figure 1 highlights the dispersion and large coverage of the selected cycles with respect to running speed (stops excluded) and average positive acceleration (during the acceleration phases).

Figure 1: Selection of the cycles /sub-cycles and coverage as regards running speed and acceleration

Figure 1: Sélection des cycles et sous-cycles et leur couverture selon vitesse et accélération



The second experimentation used 2 sets of cycles dedicated to high and low powered cars. These cycles were derived using the same database and principles than the Artemis cycles, but considering distinctly the two classes of vehicles (André et al., 2005). Although they are similar in terms of structure, these cycles reproduce the statistics of car use and driving conditions observed for each car category. They offered a good contrast with respect to dynamic.

Table 1: Vehicles tested in the 2 experimentations (in brackets, cases of high emitting vehicles)

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	PNR-Ademe				WP3141				Total
Emission standard	Diesel		Petrol		Diesel		Petrol		
Pre-EURO	2								2
EURO1	3	*(1)*	3		2				8
EURO2	10	*(2)*	6		2	*(1)*	1	*(1)*	19
EURO3	2		4	*(1)*	1		3	*(1)*	10
Total	17		13		5		4		39

2. Test vehicles

The vehicles tested are recapitulated in Table 1. Six cars were used in both experimentations and therefore used both cycles sets. The samples per vehicle category were somehow limited. The most significant concerned the EURO2 and diesel vehicles. Several vehicles were identified as "abnormal" emitters (for one pollutant, the figure exceeded 100% of the average emission of the vehicle category). Although such gaps are quite usual in emission measurements (high variability between vehicles), their strong incidence on the following analyses led us to exclude these “high emitters” in several cases. However, these data were used at a later stage to compute actual emissions.

Influence of the driving cycle

Although the 2 datasets showed slightly different characteristics (PNR-Ademe: 52 km/h, 0.8 stops per km - WP3141: 58 km/h, 0.6 stops / km) we observed good consistency for the vehicles tested with the two protocols. This allowed us to consider together the 2 datasets for the analyses. Emission was analysed with respect to factors such as: fuel, driving type and cycle, dataset, vehicle category, etc., and to a large range of kinematic parameters: speed, acceleration, stops, time distribution of speed, acceleration, etc. A variance analysis identified the level under which analyses can be conducted, and then assessed the relative influence of factors and parameters.

1. Emissions parameters

Considering the whole dataset, the Fuel Type (petrol, diesel), Emission Standard, Driving Type (i.e. urban, rural, motorway/main roads) and cycles, and Vehicle (variability between vehicles) were identified as significant factors. The variation induced by the driving type was more significant than the variation induced by the fuel type (for HC, CO2), the emission standard (NOx, CO2), or even between the vehicles (CO2). This highlights the importance of driving conditions.

Considering petrol and diesel cars separately, it appeared that Driving Type, Driving Cycle and Vehicle were the main factors for Diesel cars, while Vehicle and Emission Standard were significant for petrol cars. The emitter status (high/normal) was almost always significant. This emphasizes the need to analyse the data by vehicle category (fuel, emission standard) and driving type. However, the similarity between EURO2 and EURO3 enabled us to associate these categories to get sufficient samples. The results also demonstrated that: (i) for Diesel cars, the urban driving systematically leads to higher emissions, while the rural and motorway driving leads to low emissions, (ii) for Petrol cars, the urban driving implies high CO2, HC and NOx while CO emission is associated with the motorway driving, and the rural driving leads to low emissions.

2. Influence of the driving cycles and kinematic parameters

The analysis of the EURO2-EURO3 normal emitters demonstrated that the urban congested driving (with a lot of stops, i.e. Artemis.urban_3 sub-cycle and similar ones) produces high CO2 (petrol and Diesel) and NOx Diesel. On motorways, the very high speeds (Artemis.motorway_150_3 and similar) generate high CO2, while the unstable high speeds (Artemis.motorway_150_4 and similar) increase the NOx Diesel and CO Petrol emissions.

For Diesel cars, a) in urban driving, (i) all the pollutants increase with the stop frequency and duration (associated with low speed); (ii) all pollutants decrease when the speed increases, except the CO emission which is sensitive to high speeds (60-100 km/h); (iii) NOx and CO2 are sensitive to the frequency of accelerations and strong accelerations.

b) On motorways and main roads, (i) Diesel NOx and CO2 are sensitive to high speeds (120-140 km/h) and speed variability, and decrease at intermediate speeds (60-100 km/h); (ii) CO increases with the occurrence of intermediate speeds, stops and accelerations, and is low at high speed.

c) On rural roads, all the diesel pollutants increase with the stop frequency and duration, decrease when the speed increases, and are sensitive to low speeds (20-40 km/h or less) and accelerations (average positive acceleration, std dev., accelerations frequency). However, the CO emission seems rather sensitive to strong acceleration / deceleration.

For petrol cars, a) in urban driving, (i) all the pollutants are sensitive to acceleration parameters (frequency of accelerations and strong accelerations, average acceleration, time spent at high acceleration). (ii) CO and HC emissions are sensitive to high speeds (60-100 km/h) and strong accelerations. (iii) CO2 and HC increase with the stops, CO2 decreases when the speed increases.

b) On motorway and main roads, all the petrol pollutants are sensitive to accelerations occurring at high speeds. CO2 and CO are furthermore high at high speeds (120-140 km/h and above) and low at intermediate speeds (60-100 km/h).

c) On rural roads, as for urban, all the petrol pollutants are strongly sensitive to acceleration parameters. CO2, HC and NOx increase with the stops (duration or frequency); CO2 and NOx decrease when the speed increases.

In conclusion, we observe quite contrasted behaviours for Diesel (rather sensitive to speed and stop parameters) and Petrol cars (rather sensitive to accelerations). There is also a certain similarity between urban and rural driving for both the categories of vehicles.

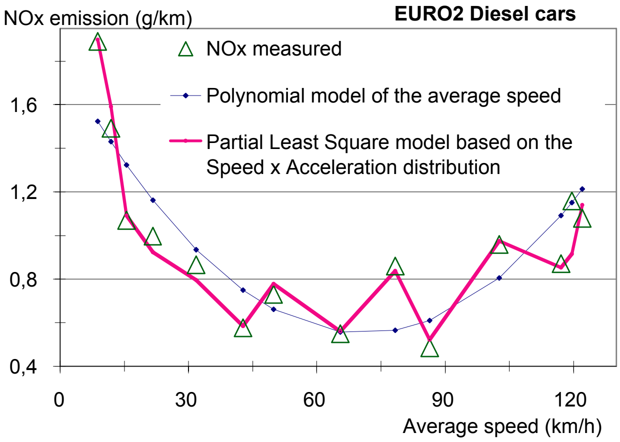
Emission model combining Partial Least Square regression approach

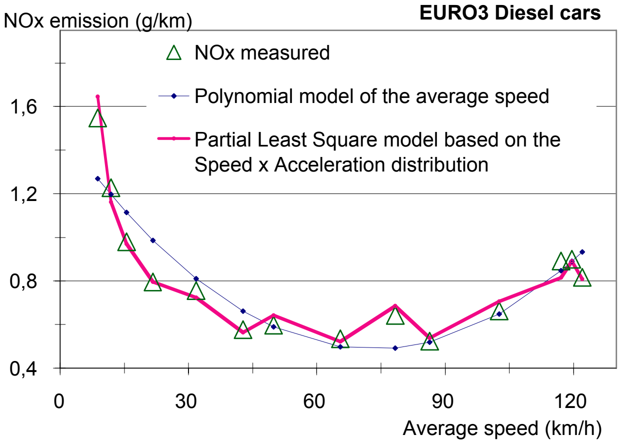
The Artemis emissions data (Artemis cycles only) were analysed by fuel type, emission regulation and engine size. Due to the data number, we considered 3 Diesel cases (EURO1, 2, 3) and 7 gasoline cases (EURO1, EURO2 and EURO3 by engine size, and EURO4). A hierarchical model was built-up to explain the logarithm of the total emission per cycle. This high-level model combined two individual Partial Least Square regression models based on principal components analysis (P.C.A., i.e. orthonormal factors were built-up from the correlations between the initial kinematic variables).

The first model (MG) relied on dynamic related parameters, i.e: average speed, square and cubic speed, idling and total running durations, average of the speed x acceleration product (positive), plus the inverse of the cycle distance. The second model (MVA) consisted in a P.C.A. approach on the 2-dimensionnal distribution of the instantaneous speed and acceleration (Rapone et al., 2005).

Figure 2: NOx Diesel emissions through Polynomial and Partial Least Square models

Figure 2: Émissions NOx Diesel selon modèles polynomial et modèle PLS





The driving cycle was identified again as a main factor for most pollutants. The engine size was significant for CO2 (petrol cars). Most often, the best fit between observed and predicted emissions was obtained using the speed x acceleration model. The dynamic related model was satisfying for CO2 Euro1 Diesel. The high level model (combining the 2 previous ones) enhanced slightly the prediction (Figure 2). The average speed model (through a parabolic trend) was unable to predict the "tooth-shaped trend" emissions due to critical test cycles (acceleration factor at different speeds) and led in some cases to a significant over-estimation at high speed.

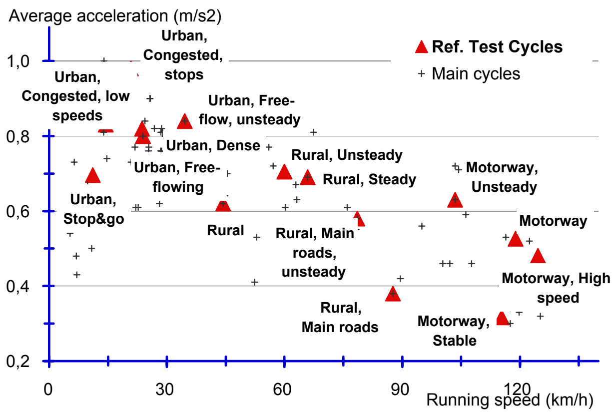
The model fit (which is not a validation criteria) was generally good for CO2 and less significant for the other pollutants due to a large variability between the vehicles and "high emitting" gasoline cars. Further investigations should be conducted in that direction as well as for a model validation.

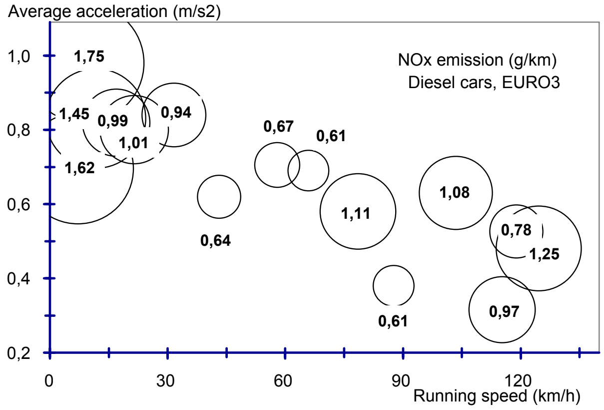
Emission correction and elaboration of reference emissions

The Artemis emission database accounts for a large number of car emission data, issued from today's and ancient experimentations, and using a high number of different cycles. In this heterogeneous dataset, each sub-sample (Artemis/non-Artemis, etc.) covers different vehicles categories, but never the whole range. In spite of its richness and due to the strong influence of the test cycles, this data cannot be used without a correction as regards the driving cycle. An approach was developed in that aim, considering similar cycles as different observations of a same driving condition. It consists in 3 main steps: 1-Classification of the cycles and building-up of a typology of test patterns, 2- Selection of pertinent cycles to represent each pattern, 3- Selection of the cycles to be considered for the calculation of reference emissions, which should be used then for the elaboration of emissions factors.

Figure 3: References test cycles (left) and NOx EURO3 Diesel reference emission (right)

Figure 3: Cycles de référence (gauche) et émissions de référence NOx, EURO3, Diesel (droite)





1. Typology of the test cycles

We considered the most significant driving cycles (i.e. 98 representative cycles / sub-cycles, for which there are a significant number of emission data) to elaborate a typology through an automatic classification based on the speed and acceleration distribution. The other cycles were then classified according to this typology. The resulting classes or Reference Test Patterns (RTP) then included sub-sets of homogeneous driving cycles (as regards kinematic conditions, Figure 3). For each test pattern, one or several Reference Test Cycle(s) (RTC) were selected amongst the most significant (in terms of representativity and number of emission data). Thirteen of these cycles were combination of Artemis cycles / sub-cycles. The 2 other cases represented the very congested driving and the stabilized motorway driving in the range of 100 km/h (Table 2).

2. Elaboration of reference emissions

We considered 20,000 data (hot cars emission, vehicle x test) by emission standard and fuel. The variability within a test pattern can be high (relative emission ranging from 0.2 to 10 for NOx and CO, from 0.4 to 2 for CO2 as regards test cycle, the variability between vehicles being also high). We then analyse the consistency of the emission as regards cycle and throughout the vehicle categories to select correctly the data for the reference emissions.

In most cases, the emissions of both the reference test pattern (i.e. the whole class) and the reference test cycles were very similar and the variability for the most important cycles was generally low. Therefore, considering all the data did not significantly affect the results. However, some deviating cycles showed quasi-systematic under- or over-estimation. When these ones did not represent a high quantity of tests, the corresponding data was cancelled. When the difference was neither systematic nor understandable, the cancellation of the related data was unavoidable. The relative evolution observed between pre-EURO, EURO1, 2, 3 and 4 was also examined, as it should be – theoretically - consistent for different cycles.

Table 2: Definition and characteristics of the test patterns and reference test cycles

Tableau 2: Definition and characteristics of the test patterns and reference test cycles

Test pattern number and Characteristics			Reference test cycle	Average speed (km/h)	Average acceleration (m/s2)	Stop duration (%)	Stop /km
7		Stop&go	**OSCAR.H1,2 &3, TRL.WSL_CongestedTraffic**	7	0,70	35	**16,3**
3	Urban	**Congested, stops**	Artemis.urban_3	9	0,98	**58**	10,2
2	Urban	Congested, low speeds	Artemis.urban_4	12	0,83	19	16,7
1	Urban	Dense	Artemis.urban, & urban_1	17	0,82	29	5,2
4	Urban	Free-flowing	Artemis.urban_5	22	0,80	10	4,3
5	Urban	Free-flow, unsteady	Artemis.urban_2	32	0,84	9	2,3
6	Rural		Artemis.rural_3	43	0,62	3	0,5
11	Rural	Unsteady	Artemis.rural, Artemis.rural_1	**58**	0,71	3	0,3
9	Rural	Steady	Artemis.rural_2	66	0,69	0	0,0
10	Rural	Main roads, unsteady	Artemis.rural_4	79	0,58	0	0,0
8	Rural	Main roads	Artemis.rural_5	88	0,38	0	0,0
14	Motorway	Unsteady	Artemis.motorway_150_2	104	0,63	0	0,0
15	Motorway	Stable	**EMPA.BAB, modemHyzem.mway,TRL.MotorwayM113**	115	0,32	0	0,0
13	Motorway		Artemis.motorway_130, & 150_1	119	0,53	0	0,0
12	Motorway	High speed	Artemis.motorway_150, 150_3, & 150_4	125	0,48	0	0,0

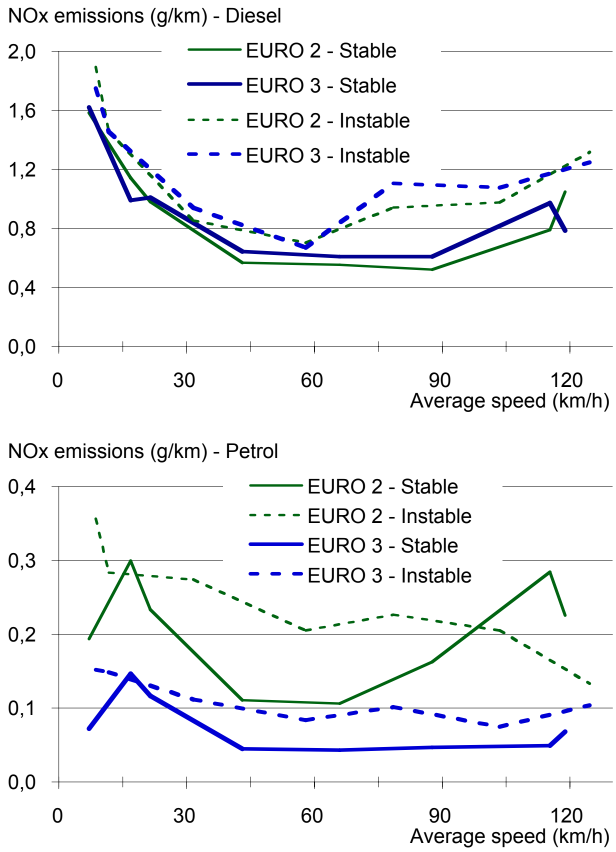
About 11,000 coherent data were retained (3,100 diesel and 7,700 petrol) and enabled the computation of the reference emissions for Diesel and petrol cars, from pre-Euro to Euro4 (Figure 3). For several cases which were insufficiently covered, extrapolation - of the rate Euro4/Euro3 (resp. Euro3/Euro2, etc.) observed on a similar test pattern (urban, rural or motorway) -, and equivalence - between close vehicle categories (i.e. Euro4 and Euro3, etc.) - were implemented.

Implications as regards the emissions modelling

The typology of cycles and computation of the emission per test pattern is a robust approach: indeed, prior to any computation, a certain equilibrium was obtained between the different driving conditions, taking into account the cycles according to their quality. Then it seems relevant to build-up emissions functions while starting from this basis. Furthermore, the typology of test cycles constitutes a good mapping of the driving conditions with respect to both average speed and acceleration, i.e. the dynamic of the traffic conditions. Indeed, we clearly identified two classes of driving along the speed scale, i.e. the stable driving with low acceleration and stop frequencies on one side, and the unsteady driving on the other side. Therefore, such an approach should enable later a better consideration of the traffic dynamic, as this one is already clearly identified for certain pollutants (NOx and CO2) and vehicle categories (Figure 4).

Figure 4: Dynamic influence on the NOx pollutant emissions.

Figure 4: Influence de la dynamique des conditions de conduite sur les émissions de NOx



The previous concepts were also implemented to estimate the emissions at a street level (Fantozzi et al. 2005). Traffic situations were defined, considering the existing road types and traffic conditions (from free-flow to stop-and-go). By computing the speed x acceleration distribution of a representative speed curve, we analyse / project the traffic situation as regards the factorial space elaborated from the driving cycles. By measuring distances from one traffic situation to the 15 reference test cycles, we then compute its emission by a linear combination of the reference emissions of the closest reference cycles. This interpolation approach then enables the emission estimation for a high number of traffic situations as regards the reference emissions.

Conclusion

These works have enabled a better understanding of the link between the emissions and the driving cycles, kinematic parameters and driving conditions. The 2-dimensional time distribution of the speed and acceleration, largely used to characterize driving cycles, also constitutes a good basis for the emission modelling. It was successfully used to harmonize the Artemis emission database as regards the test cycles and to develop an emission estimation approach at a local scale. Reference emissions have also been elaborated. The question of the high emitters was raised-up and should require further investigations to improve the emission estimation.

Acknowledgments

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Development of Driving Cycles under Special Traffic Conditions: Parking Driving Cycle in Athens

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Abstract

The aim of this project was to develop a methodology to estimate the vehicle's driving parameters during parking procedure in comparison with the similar parameters of regular vehicle motion in traffic. Modern electronic equipment was applied for the collection and processing of the driving data. On-the-road measurements covered almost all days of the week and representative hours of the day. Parking spots were randomly selected. The position of the vehicle was recorded using a GPS. Statistical evaluation of the driving data gave useful parameters such as the average time needed to find a parking spot, the average speed and the average distance travelled for each leg. Further processing of the data resulted in representative driving profiles (Driving Cycles) for both the parking procedure and the regular driving in traffic conditions. These driving cycles finds broad applications on exhaust emissions and fuel consumption studies when applied on a chassis dynamometer

***Keywords**: Driving cycles, exhaust emissions, atmospheric pollution, parking procedure, GPS*

Introduction

Densely built areas and enormous blocks of flats with insufficient parking lots for the residents, as well as the rapid increase in the car fleet the past few years, have lead to a huge parking problem, in many of the residential areas in the Athens basin. The unnecessary roving of vehicles searching for a parking spot results on heavier traffic conditions. The impact is both short and long term, on individuals, communities, transportation systems, transit systems and efficiency of traffic circulation in down town areas.

There are mainly two types of parking: public which is divided to curb-side and off-street and private. Furthermore there are other vehicles that compose the traffic (passing by vehicles) in those same areas. The target of this research is the monitoring of the driving patterns during the search for randomly selected parking spots in unregulated curb-side parking areas. In Athens, these areas constitute the majority of parking areas in almost all the residential and commercial areas, even down town. The result of this approach is the development of a Parking Driving Cycle - PDC, which represents the special driving conditions of the vehicles. The PDC finds broad applications on fuel consumption and exhaust emission studies, emission prediction models, traffic monitoring systems and decision making policies.

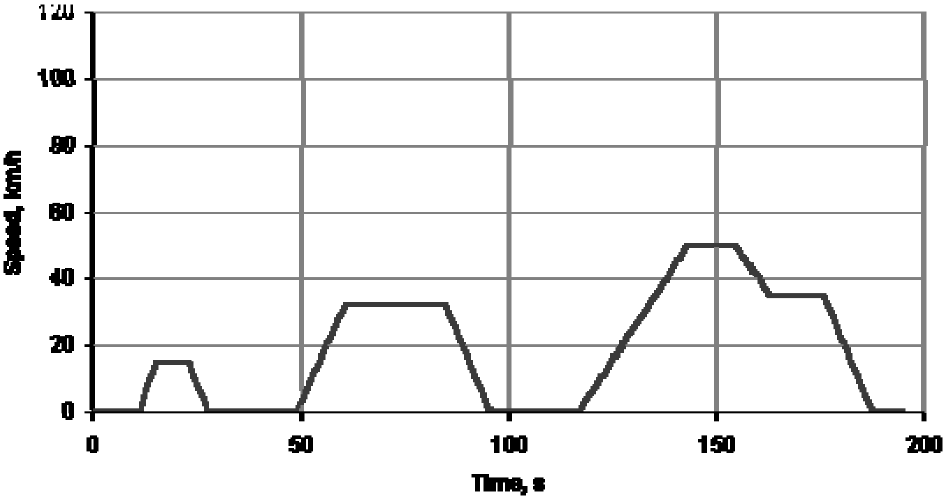
Traffic is one of the main sources of atmospheric pollution in cities around the globe and Athens is no exception. The tremendous increase of transportation vehicles resulted in serious traffic problems consisting of frequent stops and intense driving (vigorous accelerations and decelerations). Vehicles are responsible for almost all of the Carbon Monoxide (CO) emissions, for about the 75% of the Hydrocarbon (HC) emissions and volatile organic compounds (VOC), and for about the 65% of the Nitrogen Oxide (NOx) emissions, Ministry of Environment Physical Planning & Public Works (1989).

Emissions from vehicles are affected by the driving patterns which mainly depend on traffic conditions. "Driving Cycles" have been developed to provide a speed-time profile that is representative either of urban, rural or extra urban driving (Andre, 2004), (Lyons et ali, 1986). They are used to assimilate driving conditions on a laboratory chassis dynamometer for the evaluation of fuel consumption, the exhaust emissions and emission coefficient, (Andre, 2005, Simanaitis, 1977, Ergeneman et ali, 1997).

There are two major categories of driving cycles, legislative and non-legislative. According to legislative driving cycles, Exhaust Emission Specifications are imposed by governments for the car Emission Certification. Such cycles are the FTP-75 used in the USA, the European Driving Cycle used in Europe which is shown in figure 1 and the "10-15 modes" used in Japan. Non-legislative cycles, such as the Hong Kong driving cycle developed by Tong et ali (1999), the Sydney driving cycle and the recently presented Athens Driving Cycle (ADC) by Tzirakis et ali (2005), find broad application in research for energy conservation and pollution evaluation.

There are two ways of developing a driving cycle. One is composed from various driving modes of constant acceleration, deceleration and speed (like the European Driving Cycle and Urban Driving cycle or ECE-15), and is referred as modal or polygonal, (Kuhler and Karstens, 1978). The other type is derived from actual driving data and is referred as "real world" cycle (Figure 2) like those presented by.

Figure 1: A modal driving cycle (ECE-15).



Methodology

1. Equipment

Several types of equipment have used in the past for driving data crystallization (measuring and logging) (Andre et ali, 1996). Two different packages of modern electronic equipment were applied for the creation of the driving patterns database. Files were saved in text file format. The first package included the following:

An OBD II reader (On Board Diagnosis) which is a diagnostic tool, used in most passenger cars and light trucks today, providing complete engine control. It was used along with its data recording ability software to monitor the vehicle's driving characteristics that were essential for this project (eg vehicle speed, engine speed, engine load).

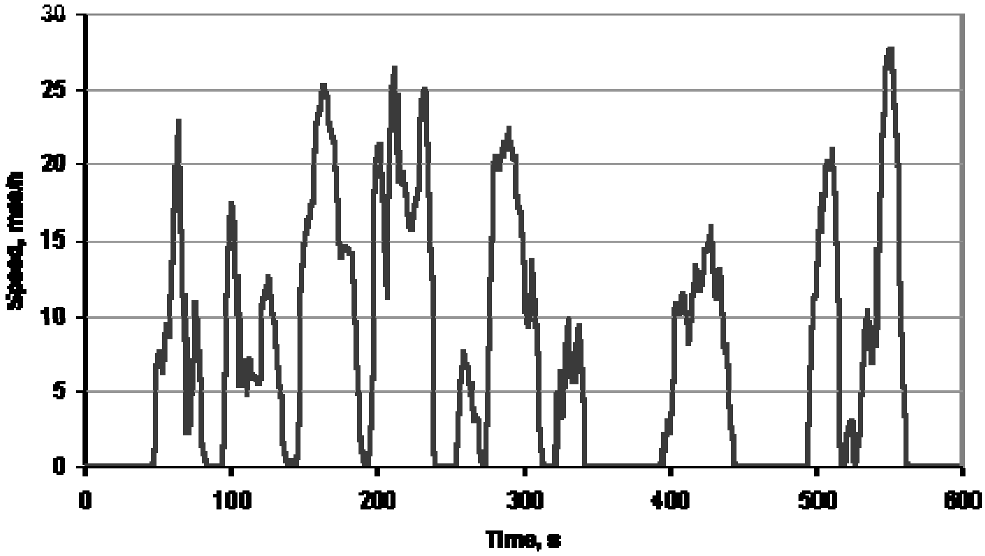
The use of a GPS (Global Positioning System) receiver unit helps on defining the geographic position of the vehicle.

Also accelerometers were considered to be essential for the estimation of road gradient, which affects fuel consumption and emissions.

All the above apparatus was connected to a Laptop Computer in order to be able to synchronize the data.

The second package was more easy to use and included a VBOX (Racelogic Ltd, UK) which is a powerful gps unit for measuring vehicle's dynamics. Peripheral equipment connected delivered more information (vehicle speed, engine speed, throttle position, etc) from the vehicle's CAN Bus. All data needed were stored in a compact flash card and then copied to the computer for further processing.

Figure 2: A real world driving cycle (EPA NYCC).



2. Development of driving database

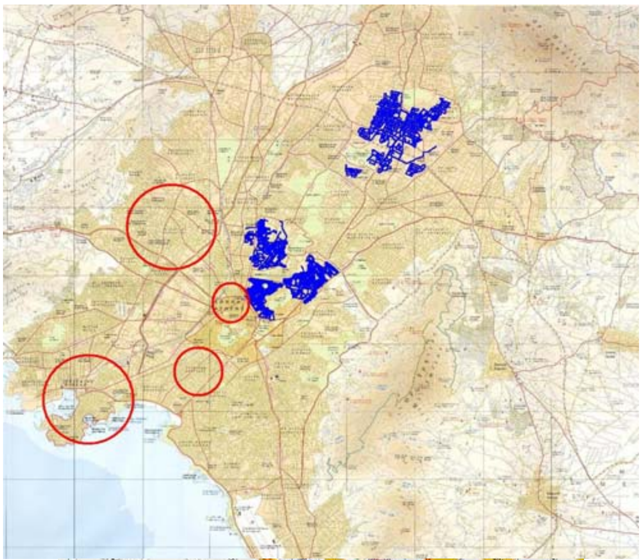
According to Papacostas and Prevedouros (2005), there are various types of studies for Parking. However, references about the influence of driving characteristics of vehicles looking for a parking spot on traffic conditions cannot be found.

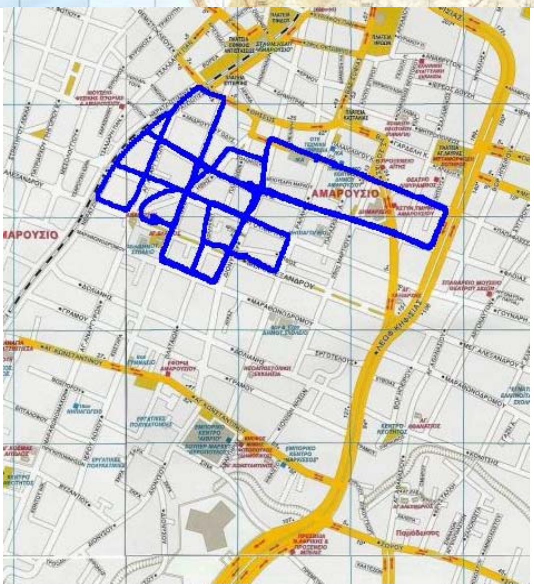
It was difficult to obtain a methodology for the collection of driving data as the research in this field is poor. In most cases the published time values of duration per parking spot search, are based on polls by transport expert scientists and not in actual on road data. Even though the time per search is recorded by those polls, no research about how vehicles are driven (driving patterns), during the procedure of parking spot search, has been recorded.

As far as regular traffic data collection is concerned, several methods have already been used. Boulter et ali (1999), describes some methods of crystallizing data, which however cannot be all adopted due to the nature of this research that holds up a limit for the adaptation of those methods. For example "chase car" method used by Ergeneman et ali (1997), could not be adopted as the test vehicle driver does not know the intentions of the vehicle driver ahead. Because of that, the method used suggested that private passenger cars were equipped and driven by their owners on the search for a parking spot in the aforementioned areas of Attica Basin.

Areas of the Attica Basin, well known of their serious parking problems were the field of this research. Those included urban and suburban areas which were residential areas of the Athens city center, residential areas near by the city center and areas which combine trade centers, shopping centers with residencies. Driving data was collected during the years 2004 and 2005, Monday to Saturday. The collection time periods included morning hours (8 to 11 am), afternoon hours (2 to 6 pm) and night hours (8 to 11 pm). In this first approach six passenger cars of various categories were used to simulate the Athens car fleet. The total distance recorded from the equipment was 771.9 kilometres. Human resources of the University were used for the research, i.e., students working on their diploma thesis and with the cooperation of the teaching staff and the administration assistants. They were driving mainly for the research and as a part of their normal life driving. Maps of the research area were used divided in blocks and numbered. The parking targets were randomly selected using programming.

Figure 3: Left: Athens map showing the research areas. Right: Focusing on a specific route that vehicle follows while searching for a parking spot as recorded by the GPS.





Researches have confirmed the drivers thinking when searching for parking using various methods (Lambe, 1996) and techniques (Bonsall and Palmer, 2004). Minimisation of driving distance and walking distance and parking fee are mainly the issues. Fee is not an issue for this research so the instructions to the drivers were simple: data log commence according to driver's judgement depending on the parking area, minimizing the distances mentioned above. It was usually started after the first pass in front of the target spot without having found anywhere to park. Figure 3 shows the areas of Athens that the research took place (blue line) and those to be investigated (red circles) as well as a single route followed by a test vehicle during the procedure of a parking spot search. It should be mentioned that the distance of the parking spot from the target could not be more than 5 minutes walk. This was clarified to the drivers of the test vehicles in order to avoid extreme situations (some drivers are satisfied only with a nearby parking spot, in opposition to some others who do not bother to park very far from the spot).

Results

The characteristics of the Parking Driving Cycle for Athens were the result of a statistical process of the driving data collected. In figure 4 the frequency that instant speeds of the vehicles can be observed for normal driving mode. As far as parking is concerned, 232 runs performed each one corresponds to one parking spot. Many parameters were taken into account such as the number of phases, the duration, the number of stops as well as the mean and the maximum speed of each run. Generally, cruising speeds do not vary greatly as observed from the measurements. The distance traveled and time needed to find a parking spot vary in significant relation to each other (Figure 5). The special traffic profile prevail and repeat it self greatly except some rare occasions where the driving patterns are affected by other parameters such as heavy traffic at a specific point of the route while searching or great difficulty on finding parking. The speed-time profile that was derived is shown in Figure 6. It consists of four phases of total duration 356 s. The average speed is 13.36 km/h and the maximum speed is 33.67 km/h.

A driving cycle of normal mode driving has been developed for comparison purposes. For the development of normal driving mode cycle data were collected from equipped passenger vehicles that were been driven through the same areas in the same time zones as the vehicles used for the data collection for the present research. Even though those vehicles have covered almost the entire Attica basin (road data were used for the development of a New Athens Driving Cycle from Tzirakis et ali (2005)), in order to produce a driving mode for the same areas and time zones where the parking mode took place, the corresponding road data were isolated. Therefore the test vehicles were covering the same area in the same time bracket for both modes. Files that were isolated for this research were broken down into smaller runs with durations equal to the parking driving mode.

Although the duration of the two cycles was chosen to be the same for comparison purposes (the travelling speed can be compared more easily), the duration of parking driving cycle is according to the (statistically evaluated) mostly appeared parking mean time recorded during the measurements.

Figure 4: Repeatability of instant speeds over 1 km/h for regular driving mode.

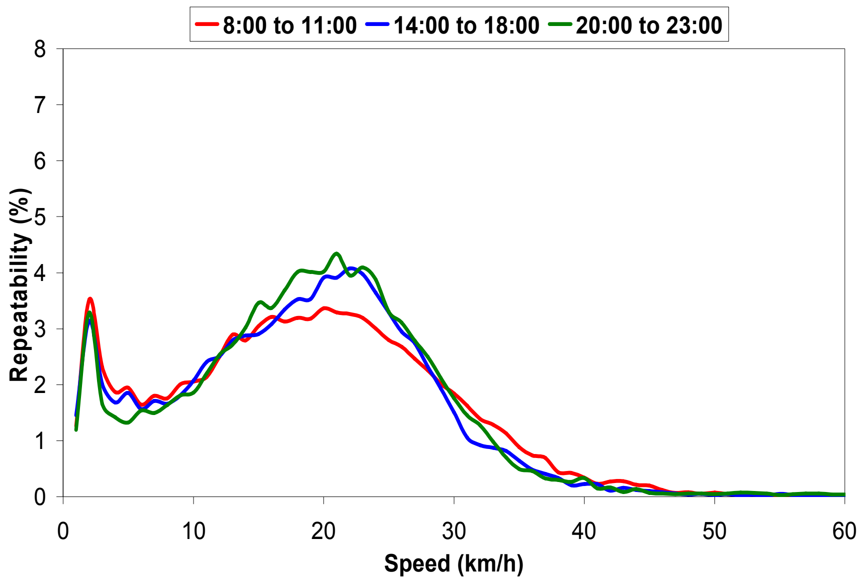


Figure 5: Distance travelled versus time needed for all the parking spots found.

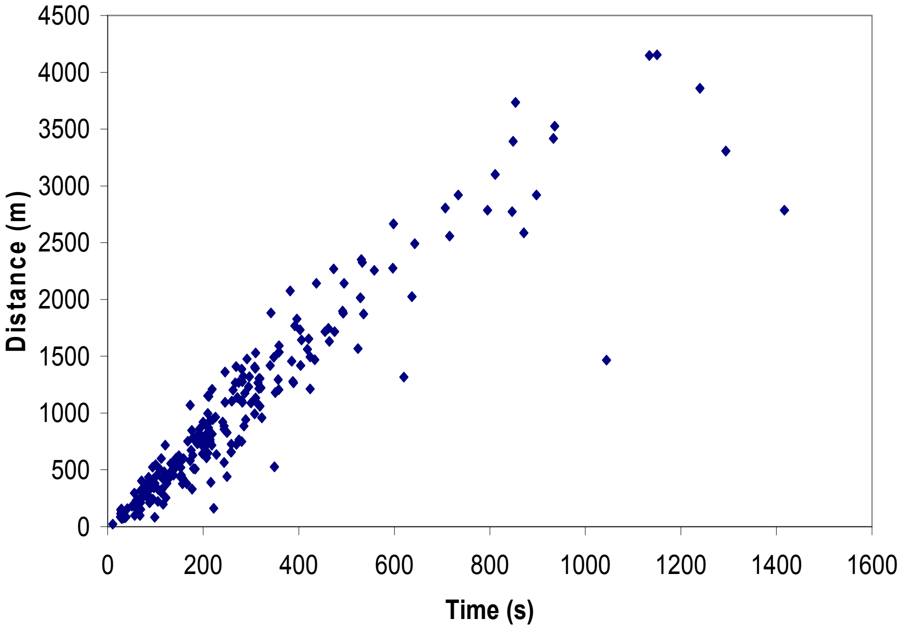


Figure 6: Driving Cycle for vehicle motion during parking search procedure.

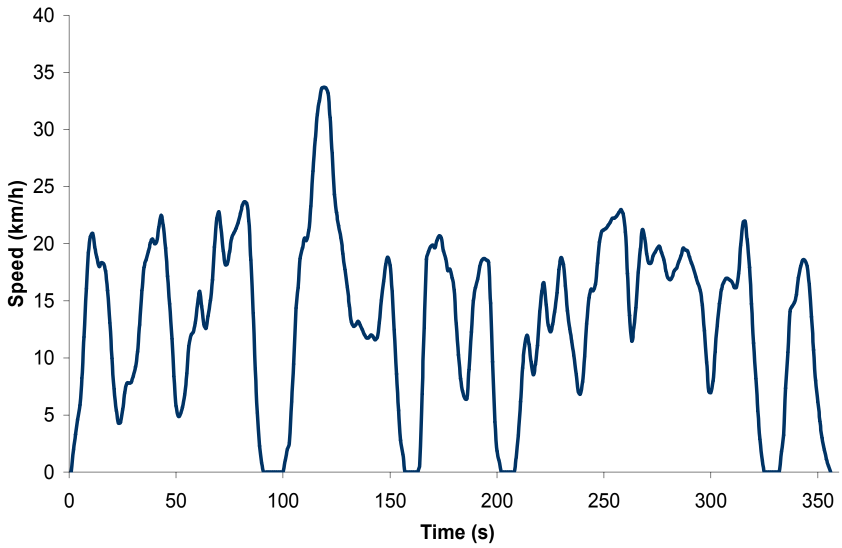
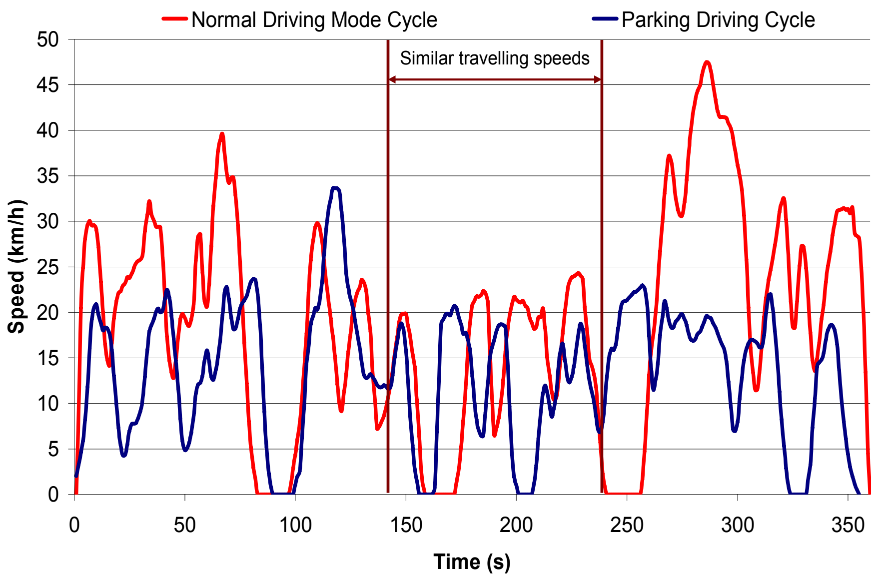


Figure 7 reveals the differences between the two driving modes. Those differences can also be observed by the comparison of their basic characteristics presented in table 1. Normal driving mode is described by the more rapid accelerations and deceleration, the higher traveling speeds and the longer idle time which are evidence of a more aggressive driving. More specifically, the average speed of normal driving mode cycle is 30% higher than the parking driving cycle. The corresponding figure for maximum speed is 48% and for acceleration 31%. Idle time is 37% higher in normal driving mode. However, it is obvious that vehicles that are in the procedure of searching for a parking spot block the passing by vehicles (see Figure 7, from 140s to 240s), contributing on the creation of traffic.

Table 1: Basic characteristics of the parking driving cycle against the normal driving mode cycle.

	Normal Driving Mode Cycle	Parking Driving Cycle (PDC)
Duration (s)	**360**	**356**
Average Speed (km/h)	**19,18**	**13,36**
Maximum Speed (km/h)	**47,50**	**33,67**
Average positive acceleration (m/s2)	**0,498**	**0,38**
Idle time (%)	**13,06**	**9,55**

Figure 7: Comparison of parking driving cycle with normal driving profile.



Conclusion

The purpose of this research is a first approach on monitoring the differences in the driving behaviour during the search of parking spot (curb-side and unregulated) and to compare it with typical driving patterns, in specific problematic areas of Attica. The driving profile that was derived is an additional motion for a considerably large number of passenger cars in those areas, which differs from normal driving, creating a significant amount of extra pollution.

According to a previous research for Athens Driving Cycle 2002 development, (Tzirakis et ali, 2005) the mean travel time of a passenger car from cold start is about 20 minutes (1160 s). Parking search increases the mean travel time of a passenger car in the city, by 24%. The surcharge is not only on the traffic conditions but also on fuel consumption and exhaust emissions in a degree higher that 24% if we consider the characteristics of Parking Driving Cycle. More rarely this value is over 100% which means that the vehicle spends more time on parking search than the time spend in the journey. Application of Parking Driving Cycle on a chassis dynamometer for emission and fuel consumption evaluation will reveal any changes on those fields from typical driving cycles and prediction models.

Parking Driving Cycle could be a useful tool for decision makers in organisations or the government. It could force on optimisation of traffic conditions in modern cities by developing flexible taxation and motivation systems which will induce people on using the improved transportation system.

This research is continued to more problematic areas of Attica, data base is enriched with more road data, in order to create a better image of the situation.

Acknowledgments

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Detection and quantification of aircraft emissions in air pollution time series data

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Abstract

Plans to build a third runway at London Heathrow Airport (LHR) have been held back because of concerns that the development would lead to annual mean concentrations of nitrogen dioxide (NO_2) in excess of EU Directives, which must be met by 2010. The dominant effect of other sources of NO_X close to the airport, primarily from road traffic, makes it difficult to detect and quantify the contribution made by the airport to local NO_X and NO_2 concentrations. This work presents approaches that aim to detect and quantify the airport contribution to NO_X at measurement sites close to the airport. First, a graphical technique using bivariate polar plots that develops the idea of a pollution rose is used to help discriminate between different source types. The sampling uncertainties associated with the technique have been calculated through randomization methods. Second, the unique pattern of aircraft activity at LHR enables data filtering techniques to be used to statistically verify the presence of aircraft sources. It is shown that aircraft NO_X sources can be detected to at least 2.7 km from the airport, despite that the airport contribution is very small at that distance. Using these approaches, estimates have been made of the airport contribution to long-term mean concentrations of NO_X and NO_2. At the airport boundary we estimate that approximately 28 % (34 $\mu g\ m^{-3}$) of the annual mean NO_X is due to airport operations. At background locations 2-3 km downwind of the airport we estimate that the upper limit of the airport contribution to be less than 15 % (< 10 $\mu g\ m^{-3}$).

Keys-words*: London, urban air quality, Heathrow Airport, aircraft emissions, source apportionment.*

Introduction

The aviation sector in the UK has grown five-fold in the past 30 years and is expected to increase by another 2-3 times by 2030 compared with current day estimates (DfT, 2003). Currently, the future development of London Heathrow (LHR) is supported by the UK Government, including the building of a third runway, but only if it can be shown that the development does not exceed EU Limit Values for ambient air pollution. The government White Paper called for an urgent programme of work to tackle the air quality problems at Heathrow. The principal concern is whether the annual mean nitrogen dioxide (NO_2) limit of 40 g m^{-3}, which must be met by 2010 as part of the EU Daughter Directive (1999/30/EC), can be achieved. In the context of air pollution, the contribution made by the airport and its operation to concentrations of NO_2 and NO_X is therefore of key importance. However, to determine this impact, a detailed knowledge of the sources of NO_X close to LHR is essential, together with their contribution to measured concentrations, if current and future assessments of annual mean NO_2 concentrations are to be reliable. Comparatively little source apportionment work has been undertaken in the vicinity of airports to determine the extent to which aircraft emissions affect local air quality. Yu et al. (2004) used a nonparametric approach based on kernel smoothing to identify aircraft sources at Los Angeles International Airport (LAX) and Hong Kong International Airport. Sulphur dioxide (SO_2) was found to be a useful tracer of aircraft emissions at both airports. The additional insight provided by incorporating wind speed into the nonparametric approach greatly enhanced the method by providing some discrimination between ground-level and elevated plumes. The analysis by Yu et al. (2004) additionally showed that CO and NO_X concentrations at LAX were dominated by road traffic sources. This paper aims to develop methods to discriminate between road traffic and aircraft sources of NO_X in air pollution data sets. Of particular interest is whether aircraft emissions can be detected and the contribution quantified in hourly data sets of NO_X from routine monitoring sites in the vicinity of the airport. Because the airport is situated within Greater London, it is embedded in a region of high emissions of NO_X, which makes the analysis of its impact on local NO_X and NO_2 concentrations problematic because of the confounding influence of other sources of NO_X.

Method

1. Description of site and data used

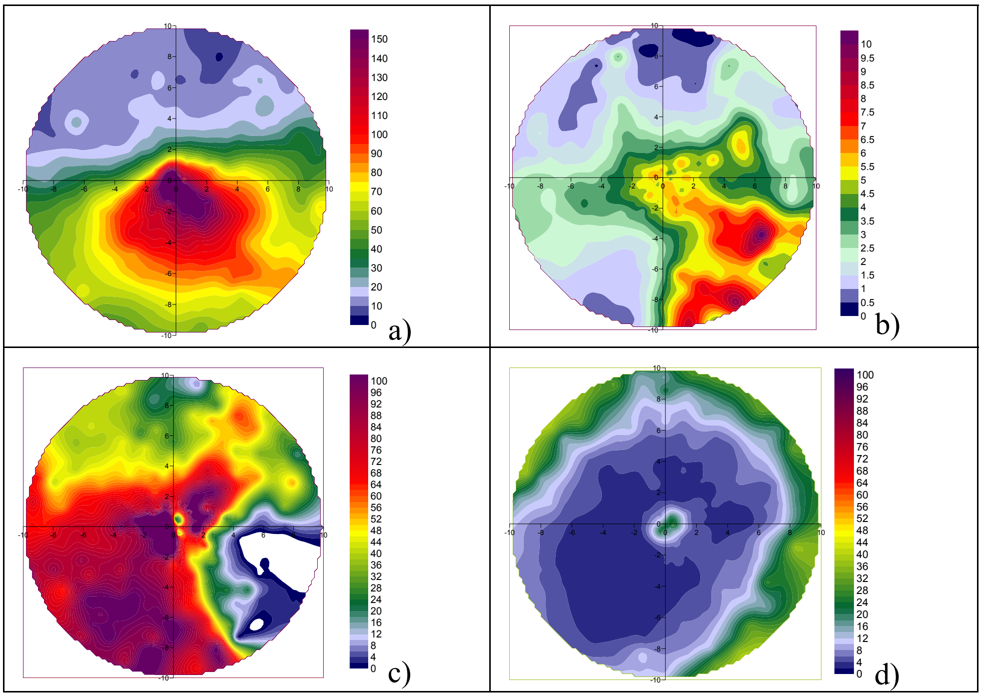
Heathrow Airport is situated within the Greater London Authority boundary in west London approximately 25 km from central London. Heathrow Airport has two runways: a northern and a southern runway separated by approximately 1.4 km. Heathrow operates a ‘runway alternation’ system of operation for noise mitigation reasons. During daytime westerly operations (taking off and landing into the prevailing westerly wind), landing aircraft use one runway from 07:00 until 15:00 and switch to the parallel runway from 15:00 until 23:00. Runway operation also operates on weekly basis so that communities in west London situated under the final approach tracks may benefit from predictably quieter periods at certain times of the day. Heathrow also operates a 'westerly preference'. The preference provides for westerly operations to continue when there is a light easterly following wind up to 5-knots (2.5 m s^{-1}), if the runways are dry and any cross-wind does not exceed 12-knots. These features of runway use at LHR provide an important and unique activity profile, which is very different to other major NO_X sources such as road transport. Data from National Air Traffic Services (NATS), made available as part of the project, were used to provide information on runway alternation. These data provided hourly information on the number of aircraft movements during each hour including: runway used, whether the aircraft was arriving or departing and the direction of take off or landing.

Table 1: NO_X and NO_2 monitoring sites used in the analysis.

Site	Data period	Distance to northern runway (m)	Distance to southern runway (m)	Mean NO_X ($\mu g\ m^{-3}$)	Mean NO_2 ($\mu g\ m^{-3}$)
LHR2	Jul. 2001-Dec. 2004	180	1600	127	55
Harlington	Jan. 2004-Dec. 2004	1230	2670	71	38
Hounslow	Jul.2001-Dec. 2004	1600	2580	65	36
Oaks Road	Jul.2001-Dec. 2004	2070	650	67	34
Main Road	Jul.2001-Dec. 2004	1060	550	81	39
Green Gates	Jul.2001-Dec. 2004	370	1770	76	38
Slough	Jul.2001-Dec. 2004	1390	2360	68	36
Hillingdon	Jul.2001-Dec. 2004	2060	3460	120	48

There are several routine monitoring sites close to LHR that belong either to national, London or British Airways Authority networks. Data from these sites undergoes quality assurance and control procedures consistent with that of the national network in the UK (AQEG, 2004). In total there are eight monitoring sites within 2 km of the airport boundary. Table 1 summarises the data available from these sites together with their distances from the northern or southern runway. Of principal importance is the LHR2 site situated 180 m north of the northern runway. This site is a few metres within the airport boundary, close to the eastern end of the northern runway. The LHR2 site is therefore ideally placed for considering airport sources when the wind is from a southerly or south-westerly direction i.e. the prevailing wind direction. The airport boundary road is approximately 15-20 m north of LHR2. With the exception of the Hillingdon site, most of the other sites are located either close to minor roads or in background locations, as borne out by the measured NO_X and NO_2 concentrations, which are typical of background concentration in London (Fuller, 2005). Hourly meteorological data were obtained from the Met Office Heathrow site.

Figure 1: a) Bivariate polar plot for NO_X ($\mu g\ m^{-3}$) at the Hillingdon monitoring site located approximately 40 m north of the M4 motorway, b) bivariate polar plot for SO_2 (ppb) at the Thurrock background monitoring site located close to areas of industrial activity c) bivariate polar plot for NO_X ($\mu g\ m^{-3}$) at LHR2 with background NO_X concentrations subtracted from Oaks Road for 06:00-23:00 on an hourly basis, d) estimated error surface (at 2σ) for (c) based on the re-sampling approach described in section 2.3. In each plot the wind speed increases radially outwards towards the circumference to 10 m s^{-1}.



2. Graphical analysis for source apportionment

Yu et al. (2004) showed how accounting for wind speed in addition to wind direction yielded information on the types of source in the vicinity of a monitoring site. Here, a similar approach is used, with several modifications. First, data were averaged into different wind speed (0-1, 1-2 m $s^{-1}$$_{\ldots}$) and wind direction (0-10, 10-20°…) categories (cells) and the mean concentration of NO_X calculated. The choice of the cell size affects the bivariate surface generated. An inappropriate cell size can lead to unnecessary imprecision: too small a cell causes the plot to become excessively noisy due to a small sample size, and too large a cell leads to an excessively coarse partition and a loss of information. The choice used here was in part determined by the meteorological data, which were provided rounded to the nearest 10°. Section 2.3 considers the affect of cell sample size on the estimated error of the mean concentration in a cell, which provides additional information on the most appropriate choice of cell size. This process yielded a surface in Cartesian coordinates, which was then converted into polar coordinates. Henry et al. (2002) did not favour converting to polar coordinates because data are compressed close to the centre. However, it will be shown later that when data from several monitoring sites are available, the polar coordinate system is a very effective one.

Figure 1a shows an example of a NO_X bivariate polar plot for a monitoring site located approximately 40 m north of a motorway. There are several points that should be noted. First, the highest concentrations are recorded when the wind blows from the south. This is entirely expected because of the dominant motorway source to the south of the site. Second, as the wind speed increases from any direction, the concentration of NO_X decreases. This pattern of decrease is what would be expected from a ground level source where the concentration takes the form of a function that is inversely proportional to the wind speed. Figure 1b shows the bivariate polar plot for SO_2 at a monitoring site in east London, which is affected by industrial sources. For SO_2 there are three clear regions where a source has an influence (approximately 60, 120 and 160°). Unlike Figure 1a, the concentration of SO_2 increases with increasing wind speed. In fact, a consideration of potential sources highlights an oil refinery source at 12 km, a power station source at 6 km and other industrial sources at 4 km.

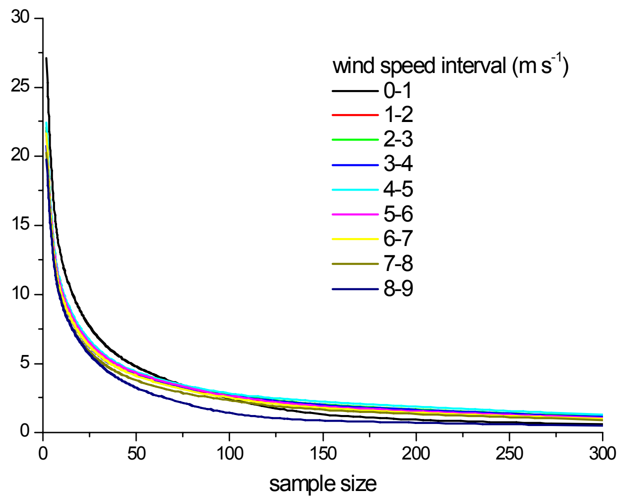
Increases in concentration with wind speed are indicative of a buoyant plume from a source such as a chimney stack, where the plume is brought down to ground-level when the wind speed increases. These features can be shown by considering the basic Gaussian plume dispersion equation (see Seinfeld and Pandis, 1998). In the absence of plume rise, the ground-level centreline concentration, c(x, 0, 0) is proportional to $\bar{u}^{-1}$, where $\bar{u}$ is the mean wind speed. In the presence of plume buoyancy, c(x, 0, 0) is a function of $\bar{u}^{a}$, where a is a constant > 0, such that the lower the wind speed, the greater the plume rise. As the wind speed increases for a buoyant plume the ground-level concentration increases to a maximum and then decreases.

The bivariate polar plot approach has been applied to the LHR2 site. The availability of monitoring sites around LHR allows the subtraction of a background NO_X concentration for certain wind sectors. Figure 1c shows the effect of subtracting the Oaks Road NO_X from LHR2 on an hourly basis with the purpose of highlighting the effect of airport sources of NO_X from the south. The Figure shows the presence of a large source of NO_X south, which does not decrease in concentration as the wind speed increases. Figure 1c also shows that there is a sharp decrease in NO_X concentration at about 150°. The angle between LHR2 and the end of the runway is approximately 110°, which suggests that aircraft plumes should be detected from 110-150°. The reason for the sharp change at 150° is that aircraft take off in an easterly direction (on the southern runway) for easterly wind conditions. Figure 1c on its own does not provide conclusive proof of the presence of an aircraft source. However, it does highlight a very different pattern compared with a road source (Figure 1a).

3. Sources of uncertainty and estimation of errors

The pattern of concentration shown in Figure 1c is influenced by many factors related to meteorology and emissions. Furthermore, as shown by Yu et al. (2004), the choice of wind speed and direction interval size is also a factor. However, the principal influence on the uncertainty associated with Fig. 1c is the number of data points that exist in any wind speed – wind direction cell. Even with several years of data, some cells only have a few data points; most notably those at high wind speeds (≥ 9 m s^{-1}) with an easterly component. As the sample size decreases in a cell, the representativeness of the mean concentration in that cell will become more uncertain. Estimates of the sampling errors were made using a re-sampling technique. For concentrations affected most by airport sources, a wind angle from 180-220° was considered. By considering several wind direction sectors affected by the same dominant source together, enough data points were available to test the effect of sample size on the mean concentration. Within each wind speed category (e.g. from 1-2, 2-3 m s^{-1}…) different sample sizes from 2 to 300 were randomly selected without replacement 500 times. For each ensemble, the standard deviation of the mean was calculated. Figure 2 shows the effect of sample size on the error at LHR2. It shows that for cells with relatively few measurements the standard deviation of the mean NO_X concentration is high. For example, for a cell with only 10 measurements, the standard deviation is approximately 10 μg m^{-3}.

Figure 2: Dependence of sample size on the standard deviation of the mean NO_X concentration for a wind direction 180-220° and different wind speed intervals at LHR2.

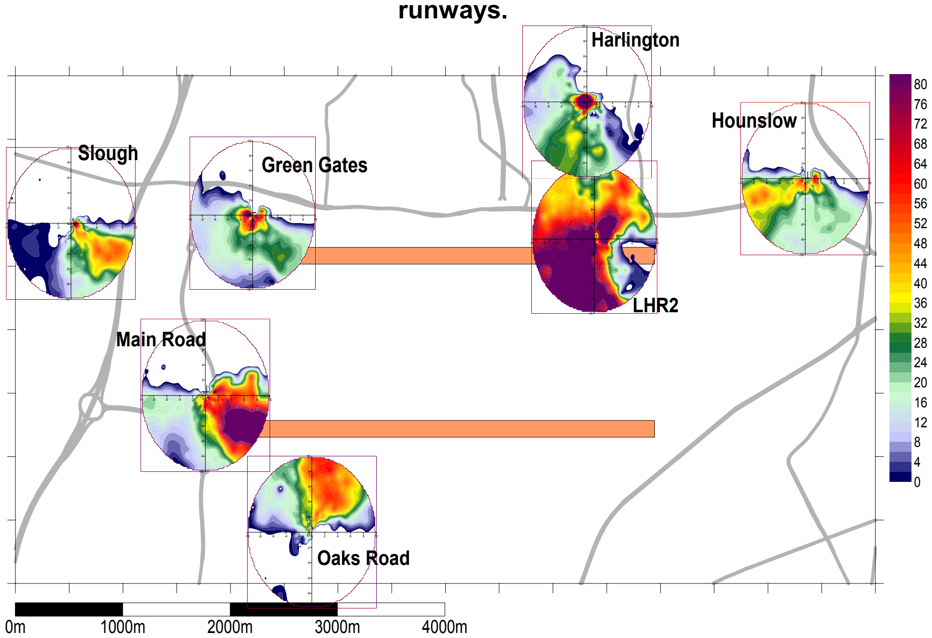


In Figure 1d, the largest errors were calculated for wind speeds > 8 m s^{-1} from all wind directions except the south-west sector. The lower estimated error at high wind speeds from the south-west is due to the high proportion of wind angles from that direction, which is the prevailing wind direction. Overall, the pattern of concentration shown in Figure 1c does not change much if estimated sampling errors are accounted for. In addition to the sample population errors, errors are associated with the interpolation routine used. However, these errors are small in comparison with the sample population errors because over 95 % of the wind speed/direction cells are populated with one or more measurements. Increasing the resolution of the grid spacing led to a smoother plot rather than a plot with a different distribution of concentration. Nevertheless, care would need to be exercised when applying this approach to sparse data sets.

Results and Discussion

1. Spatial analysis

Figure 3: Bivariate NOX polar plots for monitoring sites close to Heathrow Airport with background concentrations subtracted (μg m-3). These plots are for hours from 06:00–22:00. The grey lines highlight major roads and the two rectangles show the location of the northern and southern runways.



Bivariate polar plots for NO_X have been derived for seven monitoring sites shown in Figure 3. Each of these plots considers hours from 06:00-22:00, to maximise the NO_X signal from aircraft sources. Appropriate background NO_X concentrations were subtracted, consistent with the assumptions shown in Table 2, with the aim of highlighting potential airport sources. Most plots highlight elevated NO_X concentrations even at wind speeds up to 10 m s^{-1} when the wind direction is from the airport. At Harlington and Hounslow NO_X concentrations of between 30-50 μg m^{-3} are observed for wind speeds around 5-6 m s^{-1}. For the sites to the east of LHR (Slough, Green Gates and Main Road), and for wind speeds typically > 3 m s^{-1}, NO_X concentrations will be detected for 'easterly operations' i.e. take off east on the southern runway and landing on the northern runway. This potentially explains the relatively high concentrations of NO_X recorded at Main Road (550 m from the southern runway) due to aircraft taking off on that runway. Similarly, at Green Gates, lower NO_X concentrations might be expected because aircraft land on the northern runway during easterly operations. On this basis, the NO_X concentrations shown in the Slough plot look anomalously high, which might suggest the influence of other sources. Finally, the site at Oaks Road, also highlights a potential aircraft NO_X source, which is most apparent in the direction from 340-80°. Taken together, these plots appear to show the presence of a NO_X source in the direction of LHR that does not have the characteristics of typical ground-level sources. However, the analysis is qualitative and does not provide statistically robust proof of the presence of aircraft sources. Section 3.3 seeks to quantify the presence of aircraft by applying a statistical test.

Table 2: Results of the Mann-Whitney test applied to filtered data at monitoring sites and estimated upper limit of airport NO_X and NO_2 contribution to measured NO_X and NO_2 concentrations.

Location	Wind direction[2]	Background site	P	Z	Upper limit for airport NO_X contri-bution ($\mu g\ m^{-3}$)	Upper limit for airport NO_X contri-bution (%)	NO_X range ($\mu g\ m^{-3}$)*	Upper limit for airport NO_2 contri-bution ($\mu g\ m^{-3}$)
LHR2	150-260	Oaks Road	0.000	64.8	33.9	26.7	21.5-**33.9**	15.0
Harlington	160-260	Oaks Road	0.000	10.2	9.9	14.0	**5.7**-9.9	6.6
Hounslow	200-260	Oaks Road	0.000	6.2	9.5	12.0	**5.7**-9.5	6.5
Green Gates	340-80	Hounslow	0.000	5.3	3.0	4.0	**1.1**-3.0	1.5
Main Road	80-170	Oaks Road	0.002	3.1	7.1	8.8	**3.3**-7.1	4.2
Slough	100-170	Oaks Road	0.000	3.5	1.8	2.6	**1.7**-1.8	1.5
Oaks Road	100-170	Oaks Road	0.803	0.3	5.9	8.9	**2.2**-5.9	2.0
Hillingdon	130-230	Oaks Road	0.268	1.1	-	-	-	-

[2]Data have also been filtered by hour of day (06:00-22:00); wind speeds > 3 m s^{-1}.

*numbers in bold are considered to be closest to the actual contribution.

2. Detection of an aircraft NO_X signal

There exist certain conditions where the detection of aircraft emissions at monitoring sites close to LHR would be most likely and these can be used to apply a statistical test to determine whether they can be detected in the ambient data sets. As discussed previously, these conditions include: filtering data by wind direction such that emissions from LHR are upwind of the monitoring sites, selecting higher wind speed conditions where the effect of ground-level sources such as roads are diminished, selecting hours of the day (06:00-22:00) when aircraft activity is high, and removing a local background concentration to highlight airport sources. Filtering for these conditions does not, however, result in the unambiguous identification of airport or aircraft emissions because of the remaining dominant effect of road traffic NO_X sources.

A quantitative approach for detecting aircraft sources at LHR is to exploit the unique activity of aircraft movements and in particular the pattern of runway alternation together with the data filtering described above. A statistical test can be constructed that compares measurements of NO_X when aircraft take-off on the northern runway (and land on the southern runway) with take-off on the southern runway (and land on the northern runway) for westerly operation. These two modes of operation yield two independent hourly data sets that can be analysed for a statistical difference between them because aircraft take-off emissions are many times higher than taxing or landing emissions (see the International Civil Aviation Organization (ICAO) emissions databank at http://www.caa.co.uk/). This unique activity profile of aircraft movements at Heathrow therefore results in emissions that vary spatially and temporally in a way that is different from other sources such as road transport and thus distinguishes them from other emission sources. Here, use is made of the nonparametric Mann-Whitney U test for two independent samples. Table 2 summarises the results of the test applied to the monitoring sites and highlights the value of *p*, the probability that there is no difference in the means, and the test statistic *Z*.

The strongest signal of aircraft emissions is at LHR2, which is not surprising given the proximity of this site to the airport. At this site there was also a large difference in NO_X concentration between aircraft taking off or landing on the northern runway (136.2 vs. 44.8 $\mu g\ m^{-3}$). At other sites the difference in measured NO_X due to aircraft operation is much less significant. At the Hounslow site, for example, the difference was 29.8 vs. 29.2 $\mu g\ m^{-3}$ and yet there is a highly statistically significant difference between the two modes of aircraft operation. Only two of the monitoring sites did not show an indication of an aircraft source: Hillingdon and Slough. The Hillingdon site is about 2 km from the northern runway and is dominated by a nearby motorway source of NO_X. Because runway alternation is not used for easterly operation, it is difficult to use this approach to detect aircraft sources to the east of the airport. It is not possible to say with confidence therefore, whether the concentration pattern shown for Slough in Figure 3 is due to the airport or other sources of NO_X. These results show that the unique pattern of aircraft operation at LHR2 is a very effective characteristic that can be used to detect the influence of the airport even at locations where the contribution to NO_X is small and dominated by other sources of NO_X. It is possible that some of the results in Figure 3 could have arisen by chance because of variations in meteorology and emissions that are not associated with the airport. Therefore, as an additional check, the Mann-Whitney test was also applied to a sample of 10 other NO_X monitoring sites across London more than 5 km away from LHR (5 background and 5 roadside) using the same data filtering techniques to determine whether there was a statistically significant difference due to aircraft operation. At all of these sites $p > 0.1$ suggesting that there was no statistically significant difference in NO_X concentration due to different runway operation modes. These results provide strong evidence that aircraft emissions can only be detected at monitoring sites within a few km of LHR.

3. Quantification of airport contribution to concentrations of NO_X

Estimates of the upper limit of the airport contribution to NO_X can be made by considering the wind sector where the airport is likely to have an effect and by removing a background contribution. The choice of background site and wind sector is shown in Table 2. The estimate is an upper limit because of the presence of other sources of NO_X between the background site and the site analysed. In the case of LHR2, the upper estimate should also be a best estimate because there are very few other sources between LHR2 and Oaks Road. For all the other sites, there are other sources (principally roads) that would also contribute. A better estimate of the airport contribution is likely to be made by filtering for wind speeds > 3 m s^{-1}. The filtering has the effect of reducing the influence of road sources while maximising aircraft sources. This approach is, however, an approximation and dispersion modelling would be needed to reduce the uncertainty.

An upper limit to the airport contribution to NO_2 concentrations has also been estimated, as shown in Table 2. At LHR2 it is estimated that the airport contributes 15.0 μg m^{-3} (27.3 %) of the total measured NO_2, which is similar to the NO_X contribution. This estimate yields a NO_2/NO_X ratio of 0.44. Estimates at Harlington and Hounslow were similar, and contributed 17.4 and 18.0 % of the total NO_2 measured as an upper limit at these sites, respectively. Using the 3 m s^{-1} wind speed filtering suggests that a contribution of about 10 % is probably closer to the actual contribution at these two sites. These results also show that NO_2 accounts for a greater proportion of the total NO_2 than for NO_X at Harlington and Hounslow (ca. 10 vs. 18 %). This is probably because the airport plume is “aged” and that there is enough time for the plume to be well-mixed and for NO to react with ozone to form NO_2.

Conclusion

A graphical technique has been developed, extending the work of Yu et al. (2004), which can help discriminate between sources of NO_X emitted at ground level with little or no buoyancy (e.g. road traffic) and sources of NO_X with significant amounts of buoyancy (e.g. aircraft and large point sources). Bivariate polar plots have been derived that extend the idea of pollution roses by also accounting for wind speed. It is shown that for a small network of monitoring sites, where enough sites exist to subtract a background concentration, that bivariate polar plots are effective at highlighting the presence of aircraft NO_X emissions. By removing a local background contribution a much clearer indication of airport source characteristics can be gained, such as the wind speed dependence of aircraft jet plumes. Although bivariate polar plots have been used in this work to distinguish between sources where plume buoyancy is important or not, they are useful in other situations where there is a complex relationship between the concentration of a species, wind speed and direction. Examples include complex flows in street canyon locations, where the presence of building can affect the flows in a complex manner and particle sources where wind-blown re-suspension is important and where particle concentrations can increase with wind speed. This work has also highlighted the contrasting wind speed-dependence of road traffic and aircraft plumes. In the case of aircraft it is found that approximately 180 m from the runway, concentrations of NO_X vary little across a wind speed range from 2-12 m s^{-1}. These results indicate that the buoyant nature of aircraft plumes is an important characteristic that should be accounted for in dispersion modelling studies. These results should also provide an effective additional means of validating dispersion models used for the prediction of concentrations in the vicinity of airports.

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Several remote sensing methods for direct determination of aircraft emissions at the airports Zurich and Budapest

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Abstract

Objectives:

Emission indices for NO_x and CO for each engine are listed in a data base of the International Civil Aviation Organisation (ICAO) for four different thrust levels (idle, approach, cruise and take-off). But real emissions of aircraft at airport are not well known. These emissions are to be determined for different operational scenarios. Another source for air pollutants are ground supporting equipments whose emission rates are not yet well studied under real in-use conditions. Two methods on the basis of remote sensing measurements are applied for these tasks.

Methods:

One method to determine emission indices of aircrafts is used, where concentration measurements of CO2 together with other pollutants within the aircraft exhaust plumes are needed. During intensive measurement campaigns, concentrations of CO_2, NO, NO_2 and CO were measured. The measurement techniques were Fourier-Transform-Infrared (FTIR) spectrometry and Differential Optical Absorption Spectroscopy (DOAS). The advantage of these methods is that operations on the airport are not influenced during the measurements. Knowing the emission index of CO2 from total combustion of kerosene the emission indices of the other compounds can be determined with the concentration measurements. Together with detailed observations of taxiway movements real-in-use emissions become available.

A second method to this problem for the surrounding of terminal buildings is presented. Making assumptions about the source geometry on an airport and estimating the activity of a source by observations, a dispersion model can be used to describe the relationship between the source and a specific concentration measurement. The examination of such relationships results in a dispersion matrix which is an input to inverse methods. The other input values are concentration measurements upwind and downwind of the source. Two inverse techniques are used to solve the problem - the pseudoinverse matrix using the Singular Value Decomposition (SVD) and a Bayesian approach. Both techniques have the capability to solve for many source strengths simultaneously.

Results and conclusions:

Field studies were conducted on two airports (Zurich airport ZRH and Budapest international airport BUD). The two studies focused on the emission rate estimation of CO and NO_x. The results are presented here as well as a comparison with ICAO emission indices.

***Keys-words**: inverse methods, remote sensing, aircraft emissions, airport air quality*

Introduction

The impact of air traffic on the atmosphere was subject to several works in the last years (an overview can be found in Rogers et al. 2002). Emission sources on the airport can be subdivided into 5 parts: aircrafts, point sources, cars, ground support emissions and others (e.g.: painting, maintenance of aircrafts). Nevertheless, only a few scientific works have been done concerning airport air quality. All airport related sources contribute up to 50 % to the concentration of air pollutants around the airport Athens (Moussiopoulos et al. 1997). The dominant sources of air pollutants at the Hong Kong International Airport are ground vehicles while at the Los Angeles International Airport also aircrafts have a considerable influence on air quality (Yu et al. 2004). Unal et al. (2005) investigated the influence of different emission estimation methodologies on the pollution levels of ozone and PM2.5 for the Atlanta International Airport. According to them, aircraft emissions dominate the influence of the airport on ambient concentrations. These investigations emphasise the relative importance of different kind of sources upon air quality and demonstrate the difference that may arise regarding different airports.

Objectives

The quantification of aircraft related sources was investigated with a focus on aircraft emissions, but also considering emission on aircraft stands and other sources. With a top-down approach, starting with ambient concentration measurements, source strengths were determined with different methods: The proportion method is based on simultaneous measurement of CO_2 and other pollutants in engine plumes to determine emission indices of aircrafts. Inverse methods are based on a combination of dispersion modelling, concentration measurements at different locations and mathematical methods to determine simultaneously the emission rate of different sources. Two measurement campaigns were conducted on the airport Zurich and Budapest to gather the concentration data. With these, several methods were used to determine source strength of different sources.

Methods

1. Determination of Emission indices

Simultaneous concentration measurements of CO_2 and other pollutants in the plume of an aircraft engine allow the determination of emission indices (Popp et al. 1999, Schäfer et al. 2003 and Herndon et al. 2004). Concentrations reach peak values in the plume. Due the relatively well known emission index of CO_2, a determination of the emission indices of CO is possible, if the concentration show a pronounced peak in the plume:

$$EI(CO)=EI(CO_2)*A(CO)*(c_{fo}(CO)-c_{bg}(CO))/(c_{fo}(CO_2)-c_{bg}(CO_2)) \quad (1)$$

with $EI(CO_2)$: emission index of CO_2 (3.150 g/kg, Schäfer et al. 2003), c_{pl}: concentration measurement within the aircraft plume, c_{bg}: concentration measurement before the aircraft exhaust plumes were detected and A(CO) the ratio of molecular masses of CO to CO_2 (0.6364).

2. Inverse methodology

Another method to estimate emission rates are inverse methods in combination with dispersion models. These techniques are used in several studies to determine emission rates (e.g.: Hashmonay et al. 1999, Siefert et al. 2004). Almost all of these works considered only one source. On airports this is usually not the case. Inverse methods can easily handle more than one source, if enough concentration measurements are performed. Therefore the use of inverse methods has been chosen to investigate the source strengths on airports.

Emission source strengths and concentration measurements in the atmosphere are related through the process of air transport and dispersion, chemical reactions and removal from the air (deposition, wash out). If only transport and dispersion is considered, the problem is linear:

$$\underline{c}=\mathbf{F}*\underline{q}+\underline{b} \quad (2)$$

The problem consists of a measured vector of concentration $\underline{c}$, of a dispersion matrix **F**, of a vector of source strengths $\underline{q}$ and of a vector of background concentrations $\underline{b}$. The concentration $\underline{c}$ and the background concentration $\underline{b}$ were measured, the dispersion matrix **F** is determined with a dispersion model as described below and the source strengths $\underline{q}$ are the unknown to be determined.

To determine **F**, a known measurement and source geometry has to be given. Then for each source, the dispersion model Austal2000 (Janicke 2004) is used with a known source strength q_j. The modelled concentrations c_{ij} are then evaluated at each measurement location i. The dispersion matrix **F** is given as:

$$\mathbf{F}_{\mathbf{ij}} = c_{ij}/q_j \tag{3}$$

A solution to problem (2) can be found with the pseudoinverse matrix $\mathbf{F}^+$:

$$\underline{q}=\mathbf{F}^{+}*(\underline{c}-\underline{b}) \tag{4}$$

In case of a higher amount of measurements than sources, the use of equation (4) leads to a solution with a minimum of the least-square difference of modelled and measured concentrations. The evaluation of F^+ is done by the singular value decomposition (SVD):

$$\mathbf{F}=\mathbf{USV^T} \Leftrightarrow \mathbf{F}^{+}=\mathbf{VS^{-1}U^T} \tag{5}$$

Where U, V, S are the matrices of the singular value decomposition. For the calculation of the solution $\underline{q}$ and of the singular value decomposition, algorithms presented in Press et al. (1986) are used.

If more unknown source strengths $\underline{q}$ than concentration measurements $\underline{c}$ are given, the pseudoinverse matrix gives a minimum length solution (Press at al. 1986). In this case an infinite amount of solutions exists. Therefore, more information has to be provided to find one single solution. Following Tarantola (1987) this is done with a Bayesian approach with the use of a-priori information about the source strength. The following cost function has to be minimised:

$$J=(\mathbf{F}\underline{q}-(\underline{c}-\underline{b}))^{T}\mathbf{C_M^{-1}}(\mathbf{F}\underline{q}-(\underline{c}-\underline{b}))+(\underline{q}-\underline{q}_{prior})^{T}\mathbf{C_{prior}^{-1}}(\underline{q}-\underline{q}_{prior}) = \min \qquad 6)$$

with J the cost function, $\mathbf{C_M}$ the covariance matrix of the model and measurement, $\underline{q}_{prior}$ an a-priori estimation of the source strength and $\mathbf{C_{prior}}$ the corresponding covariance matrix. The minimisation is done with the following algorithm:

$$\underline{q}=\underline{q}_{prior}-(\mathbf{F^T C_M^{-1} F}+\mathbf{C_{prior}^{-1}})^{-1}\mathbf{F^T C_M^{-1}}(\mathbf{F}\underline{q}_{prior}-(\underline{c}-\underline{b})) \tag{7}$$

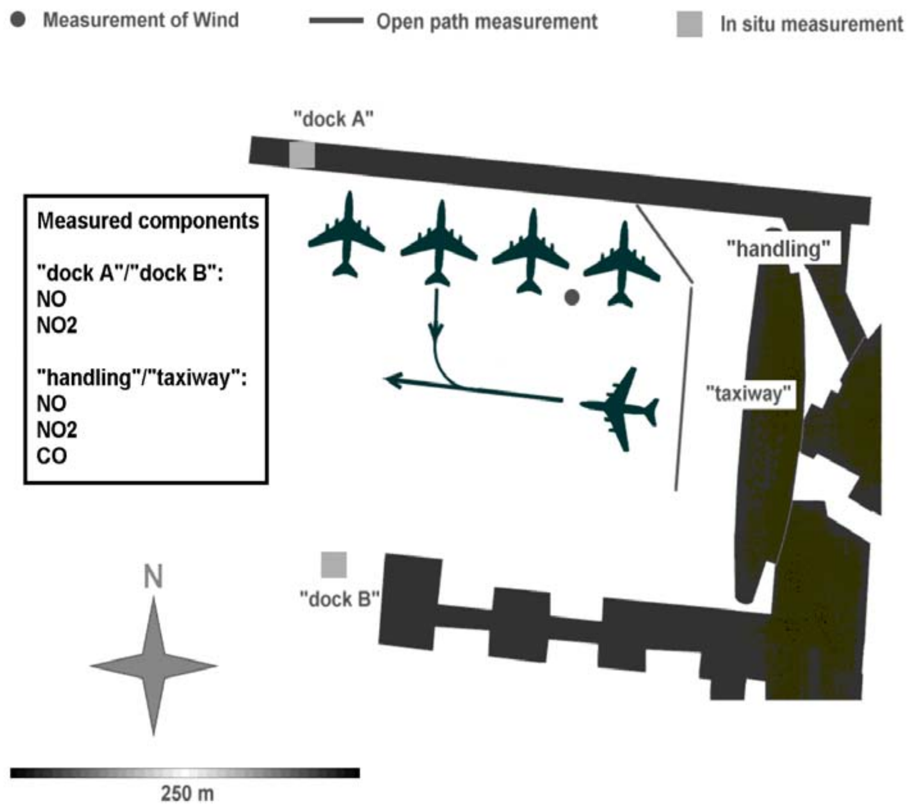
3. Measurement Campaigns

The application of the above described methods was applied to concentration measurements on the airport Zurich and Budapest. The study area on the airport Zurich (ZRH) was located between dock A and B of airport Zurich. In this study area, 4 measurement sites were operated for two weeks in 2004 (30 June - 15 July). While mainly westerly winds were observed during the measurement campaign in Zurich, the measurement stations “dock A” and “dock B” (figure 1) were assumed to measure background concentrations. A detailed description of these measurements is given elsewhere (Schürmann et al. 2005). The pseudoinverse method was used for the inverse calculation with half hourly mean concentrations.

In Budapest the study area was located around terminal 2 of airport Budapest (BUD). Overall four locations were equipped with measurement devices for CO, NO and NO_2 from 16 April 2005 until 27 April 2005. Compared to the measurement setup in Zurich, the investigated area in Budapest was around 10 times bigger. Therefore, a high amount of unknown sources has to be determined and a Bayesian approach was used. The used concentrations were hourly averages.

Figure 1: Measurement set-up on the airport Zurich.

Figure 1: Installation de mesure sur l'aéroport Zurich.



Concentration measurements were accomplished with open-path- and in-situ-devices. They are based on the physical effect that many trace gases in ambient air and exhaust compounds absorb the radiation in the UV, visible and infrared spectral range. Two different open-path devices were used. Fourier Transform Infrared (FTIR) Spectroscopy is performed with an infrared radiation source (glowbar). The gas concentrations are determined by the differential absorption method and least squares fitting of measured and reference spectra using the software code EVAL from Kayser-Threde (Haus et al. 1994). Spectrometers from Kayser-Threde with telescope were used. With FTIR spectroscopy, CO concentrations were determined.

NO and NO_2-concentration can be detected in the uv/visible spectral region with higher sensitivities than in the infrared due to greater absorption coefficients. The measurements were performed with a Differential Optical Absorption Spectroscopy

(DOAS) system and software from OPSIS AB in mono-static configuration with a xenon lamp to emit UV/visible radiation.

In-situ measurement of CO was conducted with a device of Aero-Laser GmbH (type designation AL5001). NO/NO_2 concentrations were measured with different devices from Thermo Electron Corporation (42 CTL) and Antechnika GmbH (AC 30 M, AC 31 M).

In addition, ultra sonic anemometers and cup anemometers were used to gather wind and turbulence parameters. Information about aircraft movements were provided by the airport authorities.

Results

1. Emission indices

At the Zurich airport, CO emission indices were determined for a total of 44 aircrafts with 8 different engines. A maximum emission index for CO of 80.5 g/kg fuel was measured, while the minimal emission index was 7.9 g/kg. A pronounced difference from aircraft to aircraft is observable (figure 2), because on different aircrafts also different engines are used. The variability within the same aircraft (and the same engine) is most likely caused by different engine settings during its operation. These settings weren't observed during the campaign and they are controlled by the pilot and depend on a variety of things (e.g.: weight of the aircraft).

Figure 2: Results of emission index calculation with the proportion method. Crosses indicate one single measurement, while columns stand for the same aircraft type. The two columns with "A319" stand for the same aircraft type, but from different airlines.

Figure 2: Résultats de calcul d'index d'émission avec la méthode de proportion. Les croix indiquent une seule mesure, alors que les colonnes représentent le même type d'avion. Les deux colonnes avec "A319" se tiennent pour le même type d'avion, mais de différentes lignes aériennes.

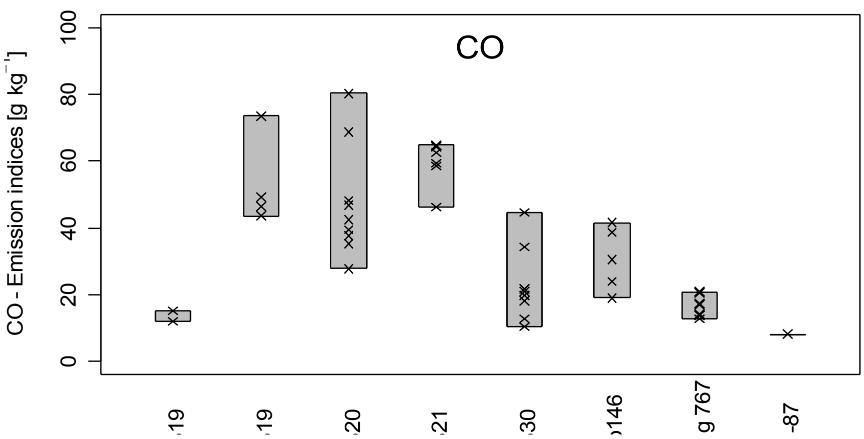
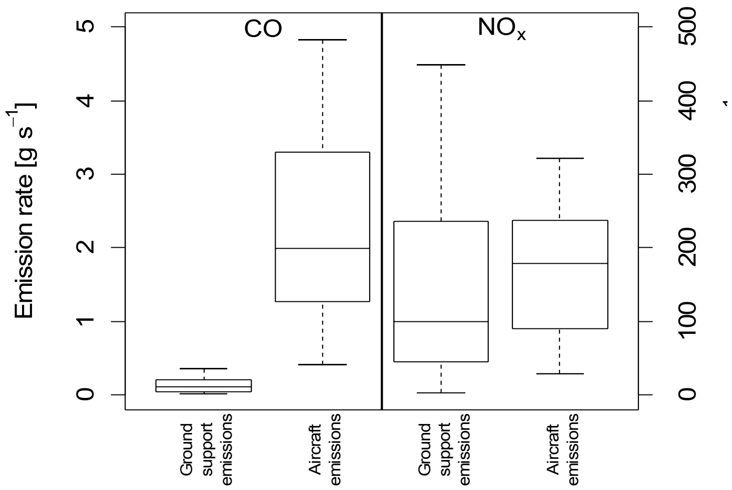


Figure 3: Emission rates as a result of the inverse methods for the airport Zurich for CO and NO_x.

Figure 3: L'émission évalue avec des méthodes inverses sur l'aéroport Zurich pour la CO et le NO_x.



2. Inverse procedure – ZRH

Inverse methods were applied for the airport Zurich with a definition of two sources: Aircrafts and ground support emissions. Aircraft emissions were considered when an aircraft was present on the taxiway, while ground support emissions were assigned to an aircraft stand according to the specific activities on the aircraft stands. Ground support emission rates were found to be 2.8 - 448.1 mg/s for NO_x and 0.02 - 0.36 g/s for CO (figure 3). This wide range reflects the variety of emission events happening on an airport. The lowermost values are found, when only few ground support activities were performed, while higher values correspond to high ground support activities. The determined emission rates for aircrafts on the taxiway were 28.6 - 322.5 mg/s for NO_2 and 0.4 - 4.8 g/s for CO. The time resolution of these results is half an hour. This is especially important for the aircraft emissions, because usually an aircraft was only approximately 5 minute on the taxiway and hence the true emission rate is higher.

3. Inverse procedure - Budapest

For the inverse procedure in Budapest, another approach as in Zurich was used, because more unknown sources have to be estimated than measurements were available. This leads to the use of the Bayesian approach. Overall 3 runways, 7 aircraft stands, one road and one freight area were defined as source regions. Therefore 12 emission rates have to be considered. The results presented here are given for emission rates of NO_x. As a-priori information, the results of the Zurich campaign for aircrafts and aircraft stands were adapted for Budapest and adjusted with the number of aircrafts present in the source region. For taxiways emission rates of 111 and 160 mg/s per aircraft were used, while for aircraft stands an emission rate per aircraft of 100 mg/s was used. A-prior emission for roads were estimated very coarse on the basis of an existing emission inventory guidebook (EEA 2001) and were set as 270 mg/s.

Preliminary results show emission rates for taxiways from 61 mg/s to 603 mg/s per aircraft and ground support emissions from 51 - 358 mg/s per aircraft. The emission rate of the road varies from 100 to 881 mg/s and is highly dependent on the hour of the day what indicates the dependence of these emissions on the amount of traffic. For the freight area, only one case was found in this preliminary study, which is influenced not only by a-priori information. This emission rate is 186 mg/s. It has to be stated here, that the inverse method based on a Bayesian approach is dependent on the a-priori knowledge. How important the a-priori knowledge and the choice of its covariance matrix are with respect to the found results has to be figured out in further investigations. Therefore this results for Budapest needs to be interpreted with caution.

Discussion

According to the results presented in this work, emissions of aircrafts are high for CO and low for NO_x. This is characteristic for taxiing aircrafts. During taxiing, engine thrust is low compared to take-off conditions. Therefore combustion is not complete and doesn't reach maximum temperature. This leads to high CO emissions and to low production of NO_x. During take-off, a converse behaviour is found and NO_x-emissions reach maximum values, while CO emissions are very low (ICAO, 1993). Emissions on aircraft stands are mainly caused by combustion engines of ground support vehicles and they are characterised by low CO emissions which are a factor of 10 smaller than emissions of taxiing aircrafts. On the other hand NO_x emissions of ground support activities reach similar levels as those of taxiing aircrafts.

The proportion method and the inverse method have been applied for the same aircrafts at the airport Zurich. Even though, a comparison of both methods is not straightforward, because the results of the two methods have different units. A conversion needs assumptions about the fuel flow and the operating time of an aircraft on the taxiway. The fuel flow is given in the emission database of ICAO (ICAO, 1993). The time of operation was taken from observations during the measurement campaign. With these assumptions, the overall emission of one aircraft can be calculated on the basis of emission indices. The comparison of these data with the results of the inverse method is given in table 1. According to these results, the results of the inverse method lie within 30 % of the emission indices measured with the proportion method.

The inverse procedure based on a Bayesian approach as it was presented here for the airport Budapest gives encouraging results. The estimated emission rates seem reasonable even though the effects of different a-priori knowledge on the solution have to be investigated in more detail.

Table 1: Comparison of emission rate estimations with the inverse method and the proportion method.

Tableau 1: Comparaison des évaluations d'émission avec la méthode inverse et la méthode de proportion.

Date	Aircraft registration	Time of operation [min]	Inverse method - emitted mass [g]	Proportion method - emitted mass [g]
1. July	HB-IJK	5	6642 ± 414	3870 ± 1565
1565	G-EUOI	4	990 ± 126	864 ± 212
3. July	HB-IQH	4	5670 ± 144	5481 ± 677
	HB-IQP	4		
6. July	N69154	4	2826 ± 232	2105 ± 1000
9. July	HB-IQH	5	3774 ± 234	2328 ± 453
13. July	HB-IOH	1	1944 ± 144	2850 ±944
	HB-IJQ	4		

Conclusion

The determination of source strengths of CO and NO_x with different methods was presented for two measurements campaigns on the airport Zurich and Budapest. Measurements of emission indices with the proportion method is a easy to use method which is possible, if simultaneous measurements of air pollutants and of CO_2 are available and distinct concentration maxima are found in the aircraft engine plume. Inverse methods are more complex to use because they require the application of a dispersion model. In exchange, emission rates of several sources are calculated simultaniously and the method is not restricted to aircraft emissions.

Measured emission indices show a high variability from aircraft to aircraft but also within the same aircraft type. This indicates the importance of thrust settings on real in-use emissions of aircraft engines. These variations are up to now not incoorporated in emission inventories of airports which are based on the emission database of ICAO (ICAO 2003).

The simultaneous estimation of aircraft emissions and ground support emissions with inverse methods allowed an assessment of the importance of these sources in the vicinity of aircraft stands. CO emissions are dominated by aircraft engines, while ground support emissions are a factor of 10 lower. NO_x is emitted in equal amounts from both sources. It is therefore not appropriate to consider only aircraft emissions on airports, but also emissions from ground support activities have to be taken into account for air quality investigations.

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New Emission Inventory for Non Road Machinery

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Abstract

This paper explains the new Danish 1985-2020 emission inventory for non road machinery. Stock and operational data are from different statistical sources, research institutes, relevant professional bodies and machinery manufacturers. Updated fuel use and emission factors are from the German IFEU institute and originate from various measurement programmes. Future factors are tailored to the current EU emission legislation. Beyond the basic calculation approach, the emission computations take into account the effects from engine deterioration, transient loads and gasoline evaporation. The major source of NO_x and TSP emissions are diesel engines. Most of the VOC and CO emissions come from gasoline machinery. From 1985 to 2020, the total fuel use and the emissions of VOC, NO_x and TSP decrease by 7, 43, 62 and 87%, respectively, whereas the CO emissions increase by 7%. The availability of new non road emission factors is very useful for other European countries, and work should be done to include these data in the European EMEP/CORINAIR guidebook.

***Keys-words**: NO_x, VOC, CO, TSP, Diesel, Gasoline, Agriculture, Forestry, Industry, Household.*

Introduction

The emission contributions from non road mobile working machinery and equipment are significant and the non road shares of total transport NO_x, TSP, VOC and CO emissions where 22, 24, 29 and 36%, respectively, in 2004. Countries are obliged to make annual emission estimates for international bodies such as UNFCCC (United Nations Framework Convention of Climate Changes) and the UNECE CLRTAP (United Nations Economic Commission for Europe Convention of Long Range Transboundary Air Pollutants) conventions and the EU Monitoring Mechanism.

Until now, the National Environmental Research Institute of Denmark (NERI) has used background data from separate national studies (Dansk Teknologisk Institut, 1992 and 1993; Bak et al., 2003) and European fuel use and emission factors from

EMEP/CORINAIR (2003), to make the non road emission estimates for Denmark. However, the relative importance of the non road emission sources and the fact that much of the operational data and fuel use/emission information is outdated, points out a strong need for a complete inventory revision.

This paper explains the new Danish fuel use and emission inventory of NO_x, VOC, CO and TSP for non road mobile machinery from 1985-2020. Fuel use and emission results are grouped into the subsectors agriculture, forestry, industry and household/gardening (supporting national emission reports), and are further analysed by single source and as totals in cross-sectoral comparisons.

Method

1. Emission legislation and emission factors

The emission directives agreed by the EU relates to both diesel and gasoline fuelled non road machinery, and list specific emission limit values for NO_x, VOC (or NO_x + VOC), CO and TSP. The limit values (g/kWh) depend on engine size (kW for diesel, ccm for gasoline) and date of implementation (referring to engine market date).

For diesel engines, the EU directives 97/68 (emission stage I and II) and 2004/26 (emission stage IIIA, IIIB and IV) regards non road machinery other than agricultural and forestry tractors, whereas for tractors the relevant directives are 2000/25 (emission stage I and II) and 2005/13 (emission stage IIIA, IIIB and IV). For gasoline engines, the EU directive 2002/88 (emission stage I and II) distinguishes between hand held (SH) and not hand held (NS) types of machinery.

The emission factors used in the Danish inventory are grouped into EU emission legislation categories. However, for engines older than directive first level implementation dates three additional emission level classes, <1981, 1981-1990 and 1991-stage I, are added so that a complete matrix of fuel use and emission factors underpins the inventory.

Fuel use and emission factors for stage II engines and prior technologies come from various emission measurement programmes and type approval tests (gasoline stage I and II), see IFEU (2004). The latter source also suggests factors for deterioration, transient engine loads and gasoline evaporation which are used in the present inventory.

The determination of emission factors for future diesel machinery is based on own judgement, taking into account today's emission factors for new machinery and future EU emission legislation limits. If the emission factor constructed as 90% of the emission legislation value is higher than the stage II value, the stage II value is used. Otherwise, the 90% figure of the legislation value is used.

Table 1 shows the basis factors for fuel use, NO_x and TSP in g/kWh used for diesel engines.

Table 1: Basis factors for fuel use, NO_x and TSP for diesel engines (g/kWh)

	Engine size [kW]	<1981	1981-1990	1991-Stage I	Stage I	Stage II	Stage IIIA	Stage IIIB	Stage IV
Fuel use	P<19	300	285	270	270	270	270	270	270
	19<=P<37	300	281	262	262	262	262	262	262
	37<=P<56	290	275	260	260	260	260	260	260
	56<=P<75	290	275	260	260	260	260	260	260
	75<=P<130	280	268	255	255	255	255	255	255
	130<=P<560	270	260	250	250	250	250	250	250
NO_x	P<19	12.0	11.5	11.2	11.2	11.2	11.2	11.2	11.2
	19<=P<37	18.0	18.0	9.8	9.8	6.5	6.2	6.2	6.2
	37<=P<56	7.7	8.6	11.5	7.7	5.5	3.9	3.9	3.9
	56<=P<75	7.7	8.6	11.5	7.7	5.5	4.0	3.0	0.4
	75<=P<130	10.5	11.8	13.3	8.1	5.2	3.4	3.0	0.4
	130<=P<560	17.8	12.4	11.2	7.6	5.2	3.4	3.0	0.4
TSP	P<19	2.8	2.3	1.6	1.6	1.6	1.6	1.6	1.6
	19<=P<37	2	1.4	1.4	1.4	0.4	0.4	0.4	0.4
	37<=P<56	1.8	1.2	0.8	0.4	0.2	0.2	0.02	0.02
	56<=P<75	1.4	1	0.4	0.2	0.2	0.2	0.02	0.02
	75<=P<130	1.4	1	0.4	0.2	0.2	0.2	0.02	0.02
	130<=P<560	0.9	0.8	0.4	0.2	0.1	0.1	0.02	0.02

In all years, most fuel is used by 75<=P<130 kW engines; in 2004 their fuel use share is 45%. However, during the inventory period relatively more and more fuel is being used by the two largest engine groups. From 1985 to 2020 their total fuel use share increase from 40 to 67%, mainly due to engine size increases for tractors and harvesters.

Table 2 shows the basis factors for fuel use, VOC and CO in g/kWh used for the most commonly used gasoline engines in the inventory. Most fuel is used in the SN4 (around 50%; predominantly riders) and the SN3 (around 25%; mostly lawn movers and cultivators) groups.

The deterioration and transient factors used are not shown. Deterioration effects are assumed for diesel machinery (all size classes) and for gasoline equipment except not hand held 4-stroke machinery. Transient factors (diesel only) are used for engines prior to stage IIIB, since the EU type approval test procedure for stage IIIB and IV takes into account transient engine loads.

For diesel engines the total impact from deterioration and transient corrections is marginal for fuel use and NO_x whereas corrected TSP emissions are generally 50-60% higher than baseline emissions. The deterioration adjustment of fuel use is marginal for gasoline engines, whereas corrected VOC and CO emissions are 30-50% and 60-80% higher, respectively, than the baseline emissions. The difference between baseline and adjusted emissions increase during the time period, since stage I and II deterioration factors are generally the highest.

Table 2: Basis factors for fuel use, VOC and CO for gasoline engines (g/kWh)

	Engine type	SH/SN	Engine size [cm3]	<1981	1981-1990	1991-Stage I	Stage I	Stage II
Fuel use	2-stroke	SH2	20<=S<=50	882	809	735	720	500
		SH3	S>=50	665	609	554	529	500
	4-stroke	SH3	S>=50	496	474	451	406	406
		SN1	S<66	603	603	603	475	475
		SN3	100<=S<225	601	573	546	546	546
		SN4	S>=225	539	514	490	490	490
VOC	2-stroke	SH2	20<=S<=50	305	300	203	188	44
		SH3	S>=50	189	158	126	126	64
	4-stroke	SH3	S>=50	33	27.5	22	22	22
		SN1	S<66	26.9	22.5	18	16.1	16.1
		SN3	100<=S<225	19.1	15.9	12.7	11.6	9.4
		SN4	S>=225	11.1	9.3	7.4	7.4	7.4
CO	2-stroke	SH2	20<=S<=50	695	579	463	379	379
		SH3	S>=50	510	425	340	340	340
	4-stroke	SH3	S>=50	198	165	132	132	132
		SN1	S<66	822	685	548	411	411
		SN3	100<=S<225	525	438	350	350	350
		SN4	S>=225	657	548	438	438	438

2. Stock and operational data

The types of machinery comprised in the Danish non road inventory are shown in Table 3.

For agricultural tractors and harvesters, historical fleet numbers and new sales/engine size figures are provided by Statistics Denmark (1965-1981; 2005) and The Association of Danish Agricultural Machinery Dealers, respectively. The latter organisation has also provided new sales numbers for the most important types of construction machinery. Data regarding fork lift new sales/lifting capacity are provided by The Association of Producers and Distributors of Fork Lifts in Denmark (IFAG). For household and gardening equipment, total stock numbers and engine sizes per machinery type have been established through detailed discussions with relevant professional bodies, large engine manufacturers, research institutes etc.

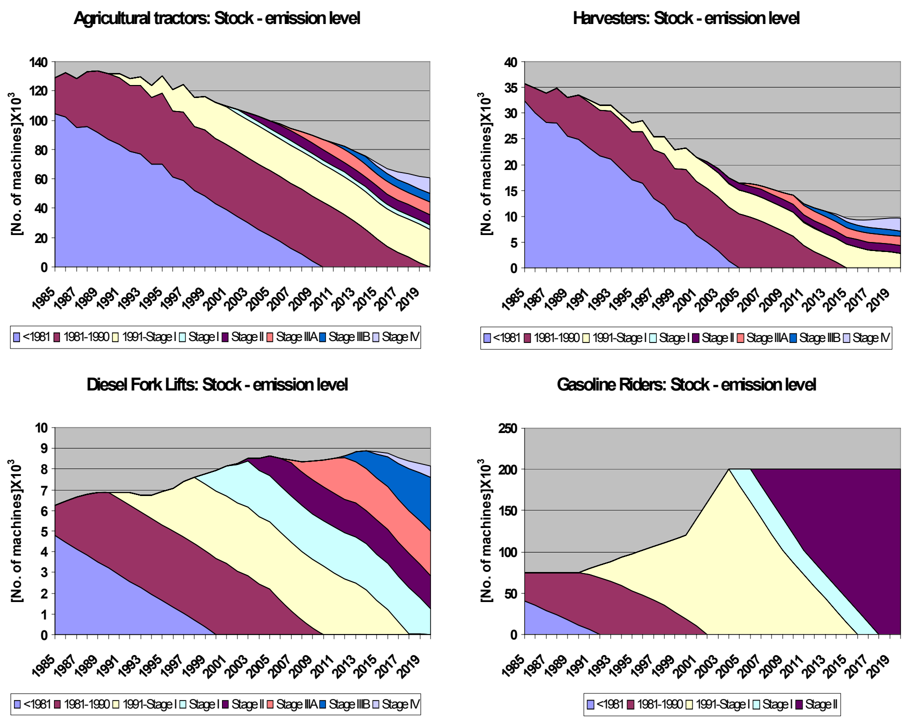
The engine types for which new stock data has been obtained, cover most of the fuel use and emissions from Danish non road machinery. Stock data for the remaining machinery types, and data for load factors, annual working hours and engine lifetime are repeated from the previous inventory. In some cases, however, data have been updated and/or new data added after discussions with external non road experts. Future year's (2005+) stock data have been produced by assuming new sales or total stock data as in 2004, and engine lifetimes control the phase-out of old technology.

Table 3: Machinery types comprised in the Danish non road inventory

Sector	Diesel	Gasoline/LPG
Agriculture	Tractors, harvesters, machine pool, other	ATV's (All Terrain Vehicles), other
Forestry	Silv. tractors, harvesters, forwarders, chippers	-
Industry	Construction machinery, fork lifts, building and construction, Airport GSE, other	Fork lifts (LPG), building and construction, other
Household/ gardening	-	Riders, lawn movers, chain saws, cultivators, shrub clearers, hedge cutters, trimmers, other

Figure 1 shows the stock development in emission level groups from 1985 to 2020 for the three most important types of diesel machinery and the largest single source of gasoline fuel use and emissions. For diesel tractors, harvesters and fork lifts the new sales year determines the emission level for each vehicle. For tractors and harvesters the total stock has decreased considerably since the beginning of the 1990's, due to structural changes in the agricultural sector. A gradual shift is made towards fewer vehicles with larger engines, and this trend is likely to continue in the future.

Figure 1: 1985-2020 stock per emission level for tractors, harvesters, diesel fork lifts and riders.



For gasoline riders (as well as for many other types of gasoline working machinery) the total stock has increased significantly in the later years, mainly because of economic growth. The stock curves for gasoline riders (and other gasoline types) appear more artificial since the only data available are total stock estimates from key experts. Subsequently, emission level distributions are made by assuming equal shares of yearly new sales during the machinery lifetime.

3. Calculation method

Prior to adjustments for deterioration effects and transient engine operations, the fuel use and emissions in year X, for a given machinery type, engine size and engine age, are calculated as:

$$E_{Basis}(X)_{i,j,k} = N_{i,j,k} \cdot HRS_{i,j,k} \cdot P \cdot LF_i \cdot EF_{y,z} \quad (1)$$

Where E_{Basis} = basis fuel use/emissions, N = number of engines, HRS = annual working hours, P = average rated engine size (kW), LF = load factor, EF = fuel use/emission factor (g/kWh), i = machinery type, j = engine size, k = engine age, y = engine size class and z = emission level.

The deterioration factor for a given machinery type, engine size and engine age in year X, depends on the engine size class (only for gasoline) and the emission level. The deterioration factors for diesel and gasoline 2-stroke engines are found from:

$$DF_{i,j,k}(X) = \frac{K_{i,j,k}}{LT_i} \cdot DF_{y,z} \quad (2)$$

Where DF = deterioration factor, K = engine age, LT = lifetime.

For gasoline 4-stroke engines the deterioration factors are calculated as:

$$DF_{i,j,k}(X) = \sqrt{\frac{K_{i,j,k}}{LT_i}} \cdot DF_{y,z} \quad (3)$$

No deterioration is assumed for fuel use (all fuel types) or for LPG engine emissions, and hence DF = 1 in these situations.

The transient factor for a given machinery type, engine size and engine age in year X, only rely on emission level and the load factor, and is denominated as:

$$TF_{i,j,k}(X) = TF_z \quad (4)$$

No transient corrections are made for gasoline and LPG engines, and hence TF_z = 1 in these cases.

The final calculation of fuel use and emissions in year X then becomes:

$$E(X)_{i,j,k} = E_{Basis}(X)_{i,j,k} \cdot TF(X)_{i,j,k} \cdot (1 + DF(X)_{i,j,k}) \quad (5)$$

The evaporation of gasoline hydrocarbon emissions is also estimated from the fuelling procedure and because of tank evaporation. The tank loading emissions are calculated as the product of total gasoline fuel use and evaporation factors (g

NMVOC/kg fuel), whereas tank evaporation emissions are found as the product of engine numbers and evaporation factors (g NMVOC/year).

Results

The calculated fuel use and NO_x, VOC, CO and TSP emission results for 2004 are shown in Table 4 per sector and fuel type, together with the sectoral 2004-2020 percent reduction figures.

The diesel fuelled machinery in agriculture and industry are the most important sources of fuel use and emissions of NO_x and TSP in 2004. Agricultural tractors is the most dominant single source, with fuel use and NO_x and TSP emission totals of around one third of the grand totals for non road machinery. Most of the VOC and CO emissions come from gasoline fuelled working machinery. The largest single VOC emission share is calculated for chain saws used in forestry and household/gardening (25%), and for CO the emission share for riders (private/professional) is 53%. From 1985 to 2004 the total non road fuel use and VOC and TSP emissions decrease by 1, 4 and 52%, respectively, whereas the emissions of NO_x and CO increase by 3%.

From 2004 to 2020, the total energy use decreases by 8% (20% for LPG, 6% for diesel, 5% for gasoline). The diesel fuel use decline is mainly due to a 16% reduction of the fuel use for tractors. This is visible from Figure 2, where the course of the fuel use curve for tractors is the result of the development towards fewer tractors, with larger and more fuel efficient engines. The total 2004-2020 emissions decreases of 70 and 75% for NO_x and TSP, respectively, are mainly driven by the emission reductions for agricultural and industrial diesel machinery. The gradually strengthened diesel emission standards during the period (Table 1) bring down the total emission loads. For TSP this is also the case before 2004, whereas for NO_x, the general emission lift in the 1990's is caused by the somewhat higher 1990-stage I emission factors compared to older technologies, for the engine size groups with large fuel consumptions.

Table 4: Fuel use and NO_x, VOC, CO and TSP emission results for 2004 (sector and fuel type) and sectoral 2004-2020 percent reduction figures.

	Subsector	Fuel type	Fuel use [PJ]	NO_x [tons]	VOC [tons]	CO [tons]	TSP [tons]
2004	Agriculture	Diesel	13.4	11811	1367	6393	991
		Gasoline	0.3	27	362	8649	7
	Forestry	Diesel	0.2	131	11	58	7
		Gasoline	0.1	4	500	1233	6
	Industry	Diesel	11.2	9297	1297	5372	1029
		Gasoline	0.2	32	261	2116	2
		LPG	1.1	1415	164	112	5
	Household	Gasoline	4.1	317	9022	114073	87
	Total		30.5	23033	12983	138005	2135
%-change 2004-	Agriculture		-10	-72	-63	-42	-82
	Forestry		-4	-86	-45	-1	-49

2020						
	Industry	-3	-55	-53	-34	-70
	Household	-4	19	-34	12	3
	Total	-8	-70	-38	6	-75

The growth in gasoline fuel use is 39% from 1985 to 2004, and is more powerful after 2000, especially due to the increased use of riders (Figure 2). For the same reason, significant fuel use inclines are, however, also noticed for lawn movers and chain saws in the same time period. After 2004, the gasoline fuel use is almost constant since total stock figures and operational data remain unchanged in the forecast period. Though, the 5% reduction in gasoline fuel use from 2004 to 2020, is the benefit from the improved fuel efficiency for 2-stroke engines.

CO and VOC emissions predominantly come from gasoline engines, and between 1985 and 2004 their gradual emission factor improvements (Table 2) more than outbalances the emission deterioration, giving smaller growth rates for emissions compared to fuel use. This is also the case for riders where significant emission increases are found. Between 2004 and 2020, the total emissions of VOC and CO decrease by 38% and increase by 6%, respectively (Table 4). Small or zero emission factor reductions for stage I and II engines in combination with higher deterioration factors cause the CO emissions for gasoline machinery to increase even after the time of stage I (2005) and II (2007) engines entering the market (most visible for riders). For VOC, the same explanation applies for chain saw and lawn mover stage I engines; their emissions continue to increase until the emission efficient stage II engine technology enter into the market in 2008.

Figure 3 shows the total Danish NO_x, TSP, VOC and CO emissions from mobile sources in the forecast period 2005-2020. In the beginning of the forecast period non road machinery is the second largest emission source, in all four cases. The non road emissions of NO_x and TSP decrease by 61 and 71% from 2005 to 2020, due to the gradually strengthened diesel emission standards (Table 1). The non road emission reductions are almost the same as for road transport, (69 and 77% respectively, for NO_x and TSP), and the emission contributions from both sectors become smaller than for internal marine (national sea transport, small boats, fishing vessels).

The reduction percentage for non road VOC emissions are somewhat smaller (40%), whereas for CO the emission development is so poor, that non road machinery ends up being the largest source of emissions after 2015. This is due to small or zero CO emission factor reductions for stage I and II engines (Table 2) and higher deterioration factors for the same emission technology levels.

Figure 2: 1985-2020 Fuel use and emissions of NO_x, TSP, VOC and CO in sub sectors of diesel and gasoline non road machinery in Denmark

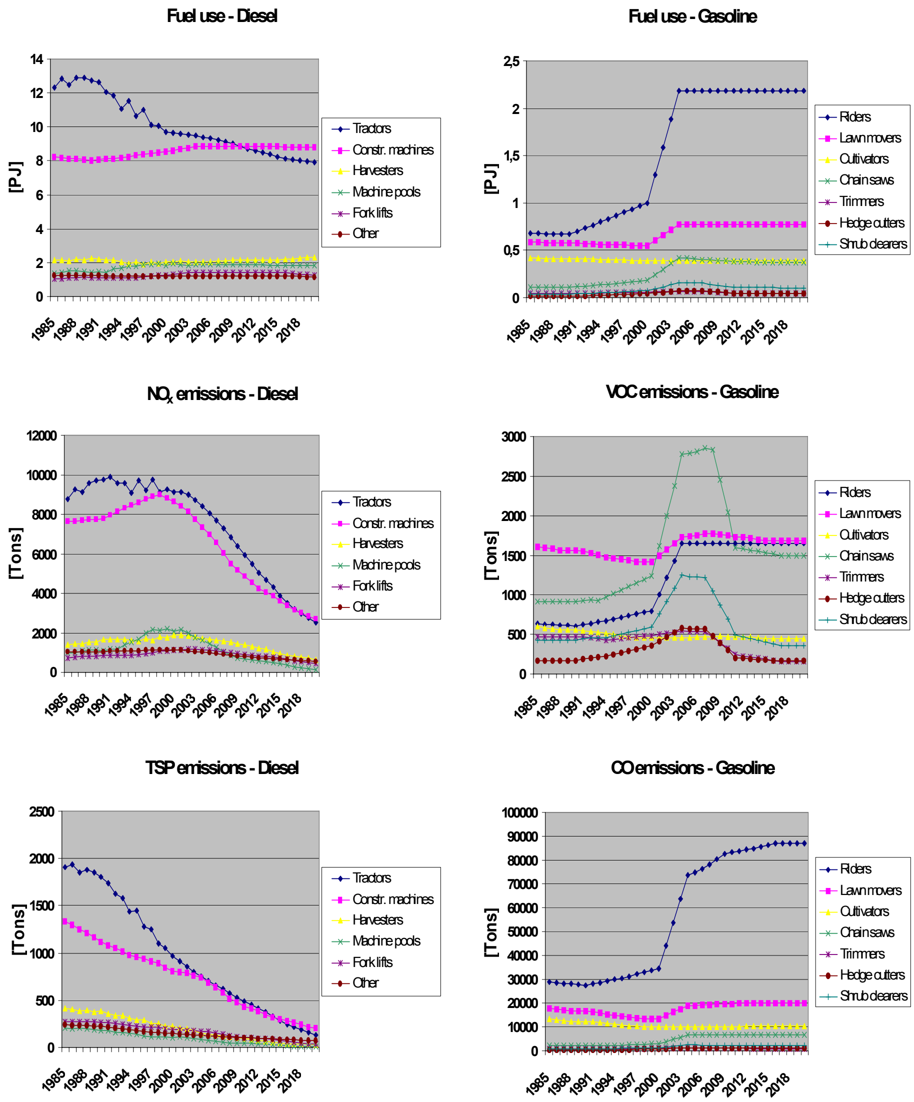
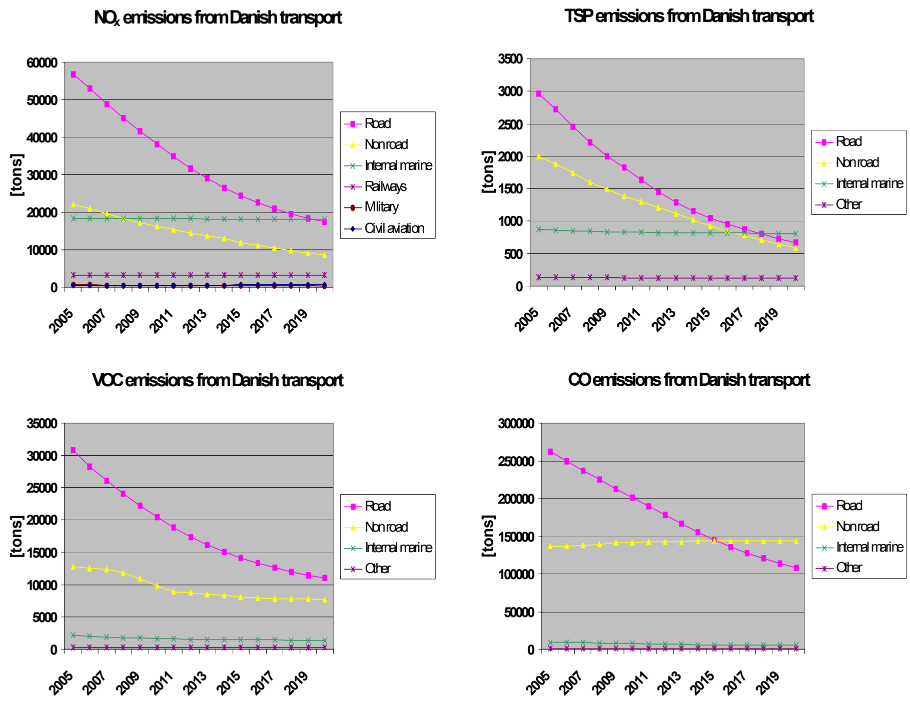


Figure 3: Total Danish NO_x, TSP, VOC and CO emissions from mobile sources from 2005-2020.



Conclusion

The trends in fuel use and emissions depend of the development in stock numbers, operational data and the estimated fuel use and emission factors. From 1985 to 2020, the total fuel use and the emissions of VOC, NO_x and TSP decrease by 7, 43, 62 and 87%, respectively. For CO, the emissions increase by 7%. The further goal is to make annual updates of the non road inventory and to the extent that Danish statistical numbers are produced, new sales numbers and total stock figures should be gathered each year for the non road machinery types included in the inventory model.

The purpose of the European EMEP/CORINAIR guidebook is to provide inventory support for country estimates. However, the guidebook data are more than ten years old and consequently the demand for new data have increased during the later years. The fuel use and emission data used in the German inventory (IFEU, 2004) and in the present Danish inventory are able to cover the data need, and work should therefore be done to include these data in the EMEP/CORINAIR guidebook.

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Quality of Emission Factors – Improvement and Validation with Road Tunnel Experiments

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Abstract

Measurement campaigns were performed in different road tunnels with the aim of determining concentrations of various gaseous species and particulate matter. From the measured parameters real-world traffic emission factors for the whole vehicle fleet as well as for the two categories light duty vehicles and heavy duty vehicles were derived and compared with emission factors from the new ARTEMIS emission model. The tunnels which were investigated showed different characteristics regarding to roadway gradient, vehicle fleet composition and traffic volume.

All three tunnel studies showed good agreement between measured CO_2 emissions and CO_2 emissions derived from calculations with emission factors. The Lundby Tunnel study showed a slight underestimation of CO emission and an overestimation of NO_x emission with emission factors from ARTEMIS emission model. For Plabutsch and Kingsway Tunnel study the CO emission calculated with ARTEMIS emission factors is much higher than the measured emission and the NOx emission calculated with emission factors is marginally lower than measured. Further investigations will be done to find the magnitude of possible inaccuracies in the calculation.

***Keys-words**: road tunnel experiments, vehicle emissions, emission factor, validation.*

Introduction

The emission behaviour of road vehicles is influenced by many different parameters, including emission standard, vehicle technology, vehicle weight, driving behaviour etc. A large number of laboratory tests are therefore required to obtain statistically reliable emission factors (EFs). However these tests are performed under controlled conditions with well known parameters. But the on-road emission behaviour looks sometimes different.

Thus it is desirable to use alternative methods to validate the emission data from vehicle and engine tests, and adjust them to real world conditions. Road tunnels can be used to determine EFs for in-service vehicles under real-world conditions. Due to the limited dispersion and dilution conditions in the tunnel environment, pollutant concentrations tend to be higher than in normal ambient air. In addition, external meteorological influences are reduced.

In the 5^{th} Framework Program within the Project ARTEMIS (Assessment and Reliability of Transport Emission Models and Inventory Systems) measurements were undertaken in 3 different street tunnels over Europe:

(i)	The Lundby Tunnel	(Gothenburg, Sweden)
(ii)	The Plabutsch Tunnel	(Graz, Austria)
(iii)	The Kingsway Tunnel	(Liverpool, United Kingdom)

The tunnels showed different characteristics regarding to roadway gradient, the vehicle fleet composition and the traffic volume. The measurements covered the standard pollutants CO_2, CO, NO_x, HC, as well as VOC profiles. Furthermore TSP, PM_{10} and $PM_{2.5}$ were monitored.

Objectives and Method

The objective was the analysis of the measured data to derive new EFs respectively to improve the accuracy of existing emission models. The obtained data had to be processed and a statistical analysis was done for each tunnel in order to disaggregate the different vehicle categories. Therefore, it was necessary to find cross-correlations between pollutants, source profiles and traffic/ventilation data.

In tunnels with longitudinal ventilation systems, pollutant concentrations increase along the length of a tunnel as the emissions from the traffic accumulate. An average EF for a pollutant i and all the traffic passing through the tunnel during a time period t can therefore be derived using Equation 1 (Weingartner *et al.,* 1997):

$$EF_i = \frac{(C_{i,exit} - C_{i,entrance}) \cdot v_{air} \cdot t \cdot A}{L \cdot N} \qquad \text{(Equation 1)}$$

EFi	=	Total emission factor for pollutant i (g vehicle-1 km-1)
Ci, exit	=	Concentration of pollutant i at tunnel exit (g m-3)
Ci, entrance	=	Concentration of pollutant i at tunnel entrance (g m-3)
vair	=	Velocity of the air in the tunnel (m s-1)

t	=	Time duration of sampling (s)
A	=	Tunnel cross-sectional area (m2)
L	=	Tunnel length (km)
N	=	Number of vehicles passing during time t

Equation 1 has been employed extensively to derive emission factors in various tunnel studies (*e.g*. Gillies *et al*., 2001; Staehelin *et al*., 1997; John *et al*., 1999). The measurement set-up depends on the tunnel ventilation system too. For a simple longitudinal ventilation system, instruments should be placed where the highest concentrations are to be expected, and the volume flow can be defined exactly – this is normally near to the exit portal. In tunnels with transverse ventilation systems, pollutant concentrations are more or less constant over a considered ventilation section. The concentration measurements have to be made in this ventilation section and at the entrance into the fresh air ducts. Equation 1 then has to be modified (Rodler *et al.*, 2000).

Multiple regression models were used to calculate averaged EFs for light duty vehicles (LDV) and heavy duty vehicles (HDV) respectively and for the air pollutants available from the different road tunnel measurements.The parameters which were considered in doing so are on the one hand the emission per time period calculated from the measured pollutant concentration and the estimated dilution with fresh air and on the other hand the number of vehicles of each category within the particular time period. The results were then EFs for aggregated vehicle categories for regulated and non-regulated pollutants and for averaged traffic situations. The data from the measurements have been compared with those from the calculations based on the existing EFs and the actual traffic load and driving characteristics recorded during the tunnel measurements.

Measurements

The three ARTEMIS measurement campaigns are summarised in this sections. For each campaign, a brief description is provided of the tunnel used, the monitoring sites, and the measurements undertaken. The monitoring methods used were broadly similar in the different tunnels, and these are listed in Table 1.

1. Lundby Tunnel

The first ARTEMIS measurement campaign was carried out in Lundby tunnel in March 2001. The measurements were conducted at three sites in the south bore of the tunnel. Figure 1 shows the tunnel, its ventilation system, and the measurement sites. During the measurement campaign the vertical ventilation system was not used. Hence, the tunnel could be treated as a simple longitudinally ventilated tunnel.

Wind speed was measured with two ultrasonic anemometers (USA) on different places. One instrument was installed in the middle of the tube; another instrument was installed at Site 4. This measurement was required to determine the flow via the vertical shaft. The number of vehicles passing through the tunnel was recorded using automatic systems. However, no distinction was made between light-duty and heavy-duty vehicles in the vehicle count, and so a video survey was conducted at the tunnel exit in order to determine the fleet composition. The speed limit in Lundby tunnel is 70 km/h but was dropped down to 50 km/h during certain times of the experiment.

Table 1: Measured pollutants and instruments used.

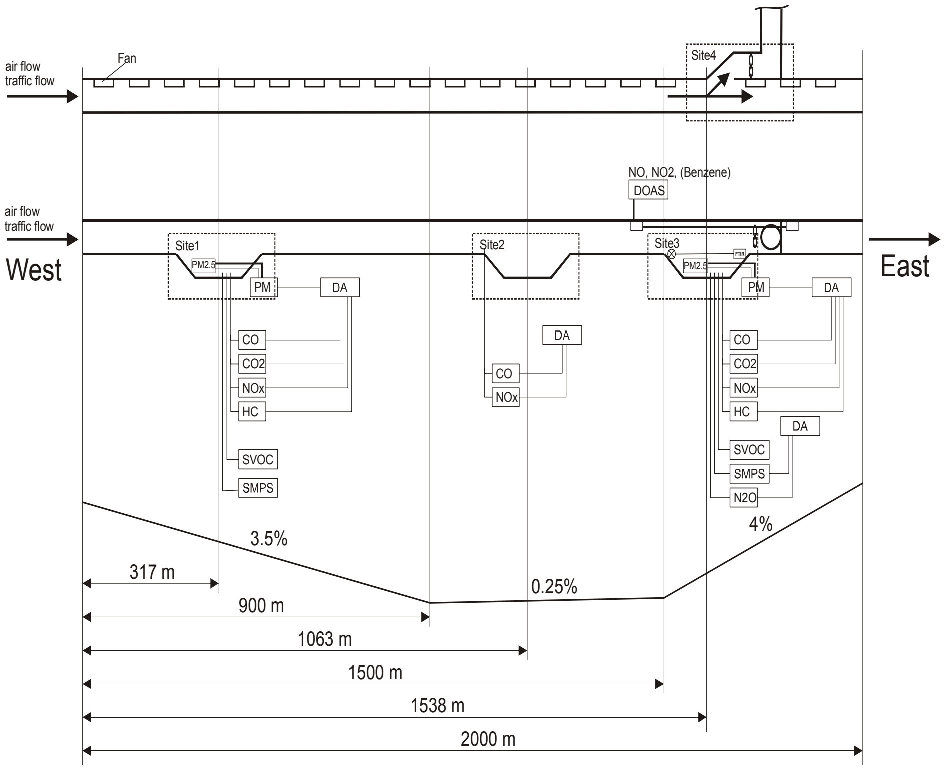
Pollutant	Method	Tunnel		
		Lundby	Plabutsch	Kingsway
NO_x	Chemiluminescence	✓	✓	✓
CO	NDIR	✓	✓	✓
CO	UV-resonance-		✓	✓
CO_2	NDIR	✓	✓	✓
CO_2	GC	✓	✓	✓
HC/NMHC	FID	✓	✓	✓
THC	FID	✓	✓	✓
NMVOC	GC	✓	✓	✓
SVOC	Tenax	✓		
N_2O	GC	✓	✓	✓
SF_6	FTIR	✓	✓	✓
SF_6	GC	✓		
SF_6	Bag samples	✓		
PM_{10}	TEOM	✓		
PM_{10}	Filter	✓		
$PM_{2.5}$	TEOM		✓	✓
$PM_{2.5}$	Filter	✓		
PM_1	Filter	✓	✓	✓
PM size distribution	SMPS	✓	✓	✓

2. Plabutsch Tunnel

The Plabutsch Tunnel has a total length of 9,755 m. The gradient of the tunnel varies from -1 % to +1 %, and the speed limit is 80 km/h. Figure 2 shows a longitudinal profile of the tunnel. It is divided into five ventilation sections and is equipped with a transverse ventilation system with a maximum air flow rate of 200 m³/s fresh/waste air in each section. Section 1 is ventilated by a station located at the northern portal. Sections 2 and 3 are ventilated by a 240 m-high shaft (north shaft), and sections 4 and 5 by a 90 m-high shaft (south shaft).

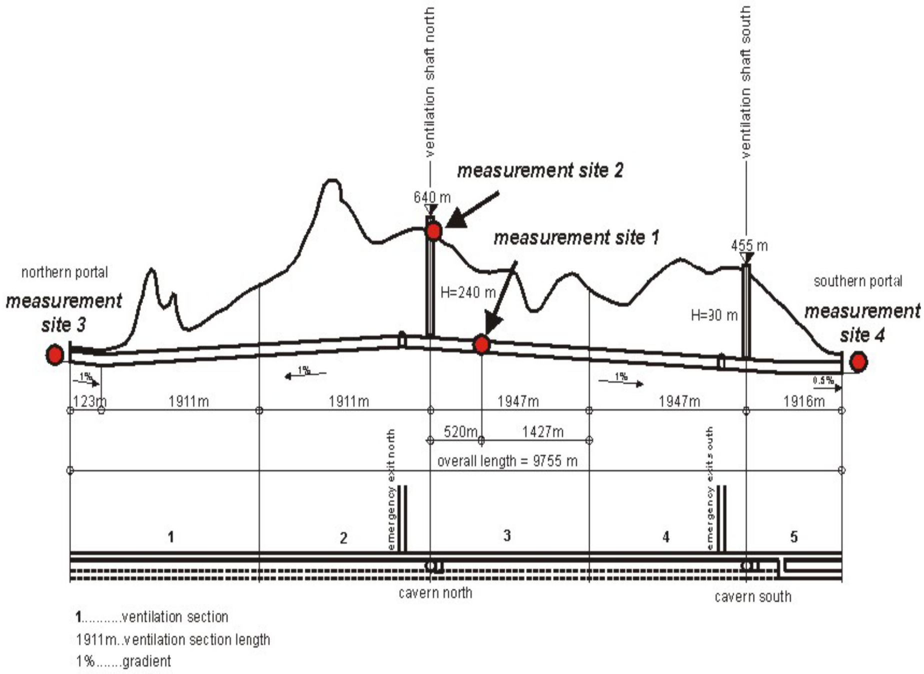
The measurement campaign in the Plabutsch Tunnel took place in November 2001. The measurements were undertaken in the east bore of the tunnel, which at the time carried bi-directional traffic. The transverse ventilation used in Plabutsch tunnel ensures an evenly distributed pollutant concentration over the whole length of each ventilation section (Sturm *et al*. 2000). In order to derive EFs for the vehicles passing through a ventilation section, pollutant concentrations have to be measured in the tunnel section as well as in the fresh air duct. The throughput of fresh air and waste air is also required. Furthermore, the tunnel measurement site has to be located where constant emission behaviour can be assumed. Given these considerations, the optimum location for the in-tunnel measurements was identified as being ventilation section 3 (Site 1 in Figure 2). Site 2 was used for the measurement of the incoming fresh air.

Figure 1: Longitudinal section and horizontal projection of the Lundby tunnel with locations of measurement sites.



Any additional data needed for the EF calculations were provided by the tunnel operator, including the traffic flow, with a distinction between light duty vehicles (LDV) and heavy duty vehicles (HDV), and the volume flows of fresh and exhaust air. As in the Lundby Tunnel, a video survey was conducted on two days of the campaign to provide more detail on the composition of the traffic.

Figure 2: Longitudinal section and horizontal projection of the Plabutsch tunnel with locations of measurement sites.

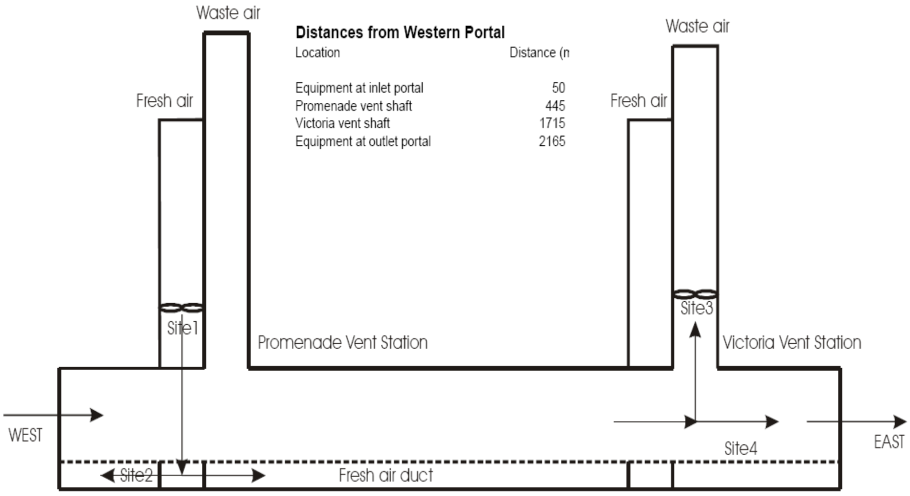


3. Kingsway Tunnel

The Kingsway Tunnel is a toll tunnel. Having opened in 1971, it is the newer of the two tunnels under the Mersey River connecting Liverpool in the east to Wallasey in the west. The tunnel incorporates two circular bores, which are 2,480 m long. Each bore carries two lanes of uni-directional traffic, with the north bore carrying traffic from Wallasey to Liverpool, and the south bore carrying traffic in the opposite direction. The speed limit in Kingsway tunnel is 40 mph (64 km/h).

The tunnel ventilation is semi-transverse. Clean air enters the tunnel via the two ventilation shafts and via the portals. The air from the ventilation shafts is fed into a sub-floor duct and permeates into the tunnel through vents along its length (Figure 3). The vents are designed to allow an even flow of inlet air along the tunnel length at maximum ventilation rate. Exhaust air is removed via the ventilation shafts, and can also leave via the portals. During the ARTEMIS experiment, the ventilation was configured in a way which would encourage the longitudinal flow of air through the north bore of the tunnel in the direction of the traffic.

Figure 3: Ventilation System in Kingsway Tunnel and location of measurement sites.



The measurement campaign in the Kingsway Tunnel took place in February 2003. The monitoring sites are shown in Figure 3. To measure pollutant concentrations in the incoming fresh air, analysers where installed inside the Promenade vent station. Pollutant concentrations in the air entering the tunnel through the inlet portal (Wallasey) were measured at Site 2. The instruments where installed in the sub-floor duct (Figure 3), and a sampling line was fed into the tunnel through a vent. $PM_{2.5}$ measurements were performed at kerbside in the tunnel. The measurements of polluted tunnel air were conducted at Site 3, which was located inside the Victoria vent station. At Site 4, located at the outlet portal (Liverpool) in the sub-floor duct, analysers were installed to measure the same pollutants. The toll information was used to calculate the total eastbound traffic flow and the proportion of heavy duty vehicles in the traffic during each hourly period. Again a video survey was conducted to determine the accurate fleet composition.

Results

As described before, multiple regression analysis was used to obtain EFs for the two vehicle categories LDV and HDV and for particular traffic situations whereby LDV refers to passenger cars (PC) and light duty commercial vehicles (LDCV). In a further step the results were compared with the new ARTEMIS EFs. The ARTEMIS emission model for road transport provides EFs for particular traffic situations or average speeds. For each of the three tunnels, EFs for the average speeds (according to the average speeds in the tunnels) from the ARTEMIS model (Samaras *et al.*, 2005) has been compared with the EFs derived from the tunnel measurements. It has to be pointed out, that for accurate comparisons detailed information on the fleet composition, actual vehicle speeds and loading factors for the HDV is required.

Former tunnel studies from the Gubrist tunnel in Switzerland (Staehelin and

Colberg, 2004) and the Plabutsch Tunnel in Austria (Sturm and Rodler, 2000; Hausberger *et al.*, 2002) showed acceptable agreement for CO and total VOC with the Handbook of Emission Factors (HBEFA Version 1.2) (INFRAS, 1999). For NO_x, the agreement was acceptable for LDV only. The emission model predicted much smaller EFs for HDV than the tunnel studies indicated.

Table 2: Emission factors for light duty vehicles (LDV) and heavy duty vehicles (HDV) derived from the tunnel measurements.

Tunnel	Road gradient	Average speed (km/h)	Period	Emission factor (g/km) with 95% confidence intervals							
				CO		VOC		NO_x		CO_2	
				LDV	HDV	LDV	HDV	LDV	HDV	LDV	HDV
Lundby (2001)	-2.7%	75	Weekday	2.0 (+0.5/-0.4)	-	-	-	0.31 (+0.13/-0.09)	1.12 (+0.95/-0.52)	-	-
			Saturday	3.2 (+1.0/-0.8)	-	-	-	0.42 (+0.17/-0.12)	1.56 (+1.85/-0.84)	-	-
			Sunday	2.7 (+1.7/-1.0)	-	-	-	0.46 (+0.21/-0.15)	1.73 (+1.89/-0.91)	-	-
	+0.6%	75	Weekday	2.0 (+0.8/-0.6)	-	-	-	0.36 (+0.07/-0.06)	11.2 (+0.7/-0.7)	-	-
			Saturday	3.3 (+1.5/-1.1)	-	-	-	0.30 (+0.05/-0.04)	9.3 (+1.3/-1.1)	-	-
			Sunday	2.7 (+1.3/-0.9)	-	-	-	0.32 (+0.05/-0.04)	10.1 (+1.1/-1.1)	-	-
Plabutsch (2001)†	-1%/ +1%	70	Weekday	0.98 (±0.11)	-	0.15 (±0.02)	-	0.75 (±0.2)	10.0 (±1.0)	-	-
			Saturday	0.67 (±0.07)	-	0.11 (±0.02)	-	0.49 (±0.01)	6.5 (±1.4)	-	-
			Sunday	0.67 (±0.08)	-	0.10 (±0.01)	-	0.37 (±0.06)	5.0 (±1.3)	-	-
Plabutsch (2001)‡	-1%		All	0.42 (±0.04)	1.39 (±0.17)	-	-	0.27 (±0.09)	8.32 (±0.025)	98.1 (±9.8)	694.1 (±29.8)
	+1%		All	0.83 (±0.08)	2.28 (±0.28)	-	-	0.39 (±0.13)	15.7 (±0.47)	135.7 (±13.6)	1397.7 (±60.1)
Kingsway (2003)	-4%/ +4%	65	All	1.73 (±0.05)	1.73 (±0.05)	-	-	0.61 (±0.05)	11.37 (±0.6)	188.5 (±14.6)	1311.1 (±173.7)

† Analysis without considering the different grades
‡ Analysis considering the different grades

Statistical analysis of the data set obtained from Lundby tunnel study was conducted by ETH-Zurich (Colberg *et al.*, 2004). The EFs derived from the tunnel measurements are shown in Table 2. No results of CO EFs are given for HDV, as the proportion of HDVs never exceeded 30 %. The EFs for HDVs were obtained by extrapolation to 100 %, and only yielded reliable estimates for NO_x because the EF is much larger than that for LDVs, which is not the case for CO and VOCs. The results show a considerable effect of day of the week, this being most pronounced for NO_x and CO. The HDV proportion in the traffic was very small during the weekend, which means that the confidence intervals of the statistical analysis are much larger. Furthermore, the fleet of HDV is different at the weekend, since it mainly consists of coaches instead of trucks as during the week.

For CO_2 in the Lundby Tunnel a regression fit to the data gave a moderate R^2 value, but a slope very close to unity (Table 3). For CO, a R^2 value of 0.55 and slope of 0.80 were obtained, indicating that the ARTEMIS EFs were, overall, lower than tunnel-derived EFs. In the case of NO_x for the Lundby Tunnel, the R^2 value and slope were 0.80 and 1.27 respectively, indicating that the ARTEMIS EFs were leading to an overestimation of emissions.

The statistical analysis of the ARTEMIS data from Plabutsch Tunnel study by Colberg *et al.*, 2004 revealed no significant differences between the EFs of LDVs and HDVs for both CO and VOCs. Hence, no HDV EFs for these two pollutants were given. Also, no distinction was made between the different road gradients. Results of the statistical analysis are shown in Table 2. From another statistical analysis of the Plabutsch data EFs were obtained for CO_2, CO and NO_x and for both directions of traffic. This was done by extracting factors for the different road gradients from the HBEFA2.1. The results of correlation analysis between emissions derived from measurement and from calculation with ARTEMIS EFs are shown in Table 3.

In the Plabutsch Tunnel there was a good level of agreement between the CO_2 emissions calculated from ARTEMIS model and the tunnel measurements, which is an important quality check of the model used to derive emissions from the tunnel measurements. However, the results for CO were surprisingly with the ARTEMIS calculation leading to an overestimation of CO of about 80 % compared with the tunnel measurements. On weekdays, NO_x emissions estimated using the ARTEMIS model were slightly lower than those derived from the tunnel measurement, whereas during weekends the calculated emissions were slightly higher. The overall R^2 value and slope (0.90 and 0.92 respectively) for NO_x emissions were indicative of a good general level of agreement.

The statistical analysis of the Kingsway Tunnel data was undertaken by TUG. Emission factors were obtained for CO, CO_2, NO_x and for an average road gradient (-4 %/+4 %). The regression fit to the CO_2 emission data for the Kingsway Tunnel, yielded an R^2 value and slope of 0.95 and 0.86 respectively. In other words, CO_2 emissions were, on average, underestimated by 15% using the ARTEMIS model. Again, CO was overestimated by the ARTEMIS model (slope = 1.58), although the correlation was high (R^2=0.94). For NO_x, during the whole period the calculated emissions were slightly lower than the emissions derived from the tunnel measurements (Table 3). As the average proportion of HDVs in the Kingsway Tunnel traffic is only around 5 %, it can probably be concluded that the ARTEMIS EFs for NO_x emissions from LDVs are too low in this case.

Table 3: Results of correlation analysis between emissions derived from measurement and from calculation with ARTEMIS emission factors.

	CO_2		CO		NO_x	
Tunnel	**R^2**	**slope**	**R^2**	**slope**	**R^2**	**slope**
Lundby	0.77	0.99	0.55	0.80	0.81	1.28
Plabutsch	0.90	0.96	0.84	1.74	0.90	0.92
Kingsway	0.86	0.95	0.94	1.58	0.97	0.89

Conclusion

Within the research project ARTEMIS three measurement campaigns were performed in different European road tunnels with the aim of determining concentrations of various gaseous species (NO_x, CO, CO_2, HC…) and particulate matter. From the concentrations and additional parameters real-world traffic EFs for the whole vehicle fleet as well as for the two categories LDV and HDV were derived and compared with EFs from the new ARTEMIS emission model.

The measurements were undertaken in different street tunnel types: two city tunnels with high traffic volume (mean speed around 50 to 65 km/h, occasional congestions) and a highway tunnel with high traffic volume (mean speed around 75 km/h). The tunnels also showed different characteristics regarding to roadway gradient and the vehicle fleet composition.

All three tunnel studies generally showed good agreement between measured CO_2 emissions and CO_2 emissions derived from calculation with EFs. This can be considered as an important “quality check” of the used calculation models. The Lundby Tunnel study showed a slight underestimation of CO emission and an overestimation of NO_x emission with EFs from ARTEMIS emission model. This is in the opposite to the results of the Plabutsch and the Kingsway Tunnel studies. But the Lundby Tunnel study mainly delivered EFs for a highly downgrade (-3.5 %) and is only limited comparable with the Plabutsch and the Kingsway Tunnel study. For the Plabutsch and the Kingsway Tunnel study the CO emission calculated with ARTEMIS EFs are obviously higher (60-70 %) than the measured emissions and the NO_x emission calculated with EFs are marginally lower than measured ones.

It can be concluded that the CO EFs for PCs seems to be much too high whereas the EF for NO_x and HDV is a little bit too low but now much better than in former used emission models. Beyond it has to be considered that NO_x emission strongly depends on the vehicle load and it is therefore very difficult to calculate HDV emission without the knowledge of this important parameter. In view of the extensively data base, some further investigations have to be done on the possible inaccuracy concerning the detailed composition of the vehicle fleet (emission regulation, age, etc.).

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Implementation and Evaluation of the ARTEMIS Road Model for Sweden's International Reporting Obligations on Air Emissions

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Abstract

The aim of the present project was to implement and evaluate the new ARTEMIS road model for Sweden's international reporting obligations on air emissions. Fleet and traffic activity data on a national level, covering all years for the period 1990-2004, were compiled and adapted to fit the indata format required by the ARTEMIS model. The model outputs for the time period 1990-2004 for greenhouse and non-greenhouse gases were compared with outputs from calculations with the national road emission model presently used in Sweden, using the same fleet and traffic activity input data as for the ARTEMIS model. Furthermore, the results from the calculations with the ARTEMIS model were compared with real-world emission data available from on-road optical remote sensing measurements. The results show a reasonable agreement between the ARTEMIS model and the national model as regards CO_2-emissions, whereas for the regulated pollutants CO, HC, NO_X some significant discrepanices between the two models were observed. Furthermore, there was in general a fairly good agreement between the ARTEMIS model and the on-road emission data. However, the differences in HDV NO_X emissions between the Euro 1, 2 and 3 classes predicted by the model, was not observed in the on-road data.

***Keywords**: Emissions, road vehicles, emission models, ARTEMIS model.*

Introduction

The UNFCCC (United Nations Framework Convention on Climate Change, the UNECE CLRTAP (United Nations Economic Commissions for Europe Convention for Long-range Transboundary Air Pollution), and the EU NEC (National Emissions Ceiling) Directive, require yearly reporting of national emissions to air of a number of pollutants for the parties concerned. The compiling of the national emission inventories follows dedicated guidelines for the reporting in order to fulfil international quality objectives such as data being transparent, consistent, comparable, complete and accurate. The compilation and reporting of the national emissions are made on a sectoral level (e.g. transport) as well as on sub-sectoral levels (e.g. road transport, passenger cars, diesel fuelled road vehicles). In order to comply with all the quality objectives of the reporting, sectoral and sub-sectoral emission models are very useful. Both national (country-specific) and international emission models may be used, as long as the quality objectives are reached. Until reporting year 2003, Sweden has used a national emission model for road traffic for the international reporting obligations. However, a national strategic decision was recently taken by the Swedish Road Administration to switch to a EU-common model, with the first choice being the new emission model for road traffic developed within the EU 5th FP project ARTEMIS (Assessment and Reliability of Transport Emission Models and Inventory Systems). Since according to the Kyoto protocol no changes in methods are allowed in the reporting to UNFCCC after 2005 for the Kyoto first commitment period, extending until reporting year 2012, the implementation of the ARTEMIS model in Sweden, at least for the direct greenhouse gases CO_2, N_2O and CH_4, needed to be completed before the end of 2005 and to cover the whole timeseries 1990-2004.

Objectives

The aim of the present project was to implement and evaluate the new ARTEMIS road model for Sweden's reporting obligations on air emissions to the UNFCCC (the Kyoto protocol), the EU NEC Directive, and the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP).

Methods

1. The ARTEMIS road model

The ARTEMIS road model has been developed within the framework of the EU 5th FP project ARTEMIS, "Assessment and Reliability of Transport Emission Models and Inventory Systems". The model provides emissions and emission factors for segments and subsegments of six main vehicle categories (PC, LCV, HCV, Urban bus, Coach, and MC including mopeds) for a large number of traffic situations, as well as for average speeds, based on emission measurements according to different sets of real-world driving cycles, representative for typical European driving (Keller et al, 2005). The model's calculated emissions are separated into hot emissions, cold start emissions and evaporative emissions. The results presented in this report are from the version 0.2T of the ARTEMIS model.

2. Fleet data

The vehicle fleet is described by means of the number of vehicles on category level, along with segment and age distributions, derived from the Swedish national vehicle register. This register is updated with new registrations and scrapped vehicles on a daily basis. Fleet data should be representative for each year 1990-2004. A simplification could be to use the vehicle register data available by June 30 for each year. However, in the Swedish case vehicle register data available by December 31 is used. This approach requires special efforts for the youngest age class.

As for segment and age distributions some problems arise when vehicle concepts or vehicle definitions according to the ARTEMIS model are not recognised in the national vehicle register. For instance, for passenger cars, information on swept engine volume is not available in the register. Swept volume for different car models sold in Sweden are however available in a fuel consumption data set provided by the Swedish Consumer Agency. This dataset has been matched against the vehicle register for 2004, with approximately 1 million hits out of the approximately 4 million passenger cars registered in the national vehicle register. Based on this matched dataset, swept volume functions were estimated for diesel and petrol cars separately. Swept volumes are expressed as a function of year of registration; engine power, and vehicle weight.

The distinction between coaches and urban buses is based on the measure p/w: p=max allowed number of passengers/w=gross vehicle weight. An urban bus is defined as p/w>3,75. All other buses are coaches.

In order to describe trucks, two segment levels are used: with and without trailers. In the Swedish national vehicle register there is no information about the use of trailers. Trucks with trailers are described by means of vehicle transformation patterns. The transformation pattern describes distribution of mileage in each weight class with and without trailer. The segment level "with trailer" is split further into different sizes of trailers expressed as the total weight class of the vehicle combination.

For motorcycles swept engine volume is available from the national vehicle register, however not the type of engine (2- or 4-stroke). Type of engine was estimated based on year of registration, swept volume, engine power and manufacturer.

For each segment in each category there will be an age distribution including 60 year classes. The definition of the age of a vehicle is based on the first date of registration according to the national vehicle register. For privately imported cars, which is a growing phenomena in Sweden, the year of first registration is replaced by the year of manufacturing when estimating age distribution.

For each vehicle segment, each year class of first registration is assigned one or more emission concept groups by means of the indata function “Introduction schemes of emission concepts” in the ARTEMIS model. The code representing a vehicle's emission concept group in the national vehicle register is sometimes missing. In such cases codes have been assigned based on the year of first registration together with dates for introduction of new exhaust regulations.

3. Traffic activity data

3.1 Mileage etc.

The ARTEMIS model requires yearly mileages per vehicle category. For Sweden these are calculated by means of a national road mileage model. Important inputs to this model are mileage on roads administrated by the Swedish Road Administration based on traffic measurements, along with the number of vehicles in different categories. The annual mileage per vehicle category is derived by dividing the total mileage per category with the number of vehicles per category. By supplying the same number of vehicles together with the derived mileage, the ARTEMIS model will provide the same mileage as the national road mileage model.

Yearly mileages per vehicle subsegment level is used to distribute the total mileage on subsegments. Statistics Sweden has developed a method which can assign all vehicles in the register an annual mileage, based on yearly odometer readings within the Swedish inspection & maintenance (I/M) programme. These data has been used for deriving both the subsegment level mileage, and for estimating mileage as a function of vehicle age.

Load patterns as a function of age is used for heavy commercial vehicles on segment level. The ARTEMIS model requires for each segment a mileage distribution on load factor 0% and 1-99% for 60 age classes. These data have been estimated based on a major national survey:"Swedish domestic road goods transport" from 1997, including detailed information about both truck and trailer loads.

In order to estimate evaporative and cold start emissions data on trip lengths, parking times, and seasonal and diurnal variation of ambient temperature are needed. Trip lengths and parking times can be derived from surveys, or from data from instrumented cars. For Sweden an average trip length according to surveys is 12 km, and from instrumented cars 7 km. Instrumented cars give the length from engine start to engine stop, i.e. the form of data requested. Even if instrumented car data just represents a few vehicles and use in few families, this data set has been considered more reliable and thus used in the Swedish application.

The function FuelQuality in the ARTEMIS model is used for correction of emissions from diesel engines and for estimations of evaporative emissions (RVP) and SO_2 emissions. The Swedish EPA used to take representative samples and analyze fuels at filling stations until 1990. From year 2000 there is again data available on an annual basis, provided by Swedish Petroleum Institute. In the ARTEMIS model diesel fuel qualities are described as classes Euro 0 to Euro 4. However, by mainly using the sulphur content for classification the deviation at least for SO_2 should be minor.

3.2 Traffic situations

The ARTEMIS model includes 276 traffic situations, i.e. combinations of 69 road categories and for each of those 4 classes of traffic conditions. Furthermore it is possible to add different level of grade, however this was not done for Sweden. The national vehicle mileage for year 1990, 1995, 1998, 2000 and 2004 had initially been estimated through the VM-Model (SIKA and VTI, 2005). Procedures were established to allocate the total vehicle mileage over 1) urban and rural roads, 2) road categories, 3) traffic flow conditions, and to fit the result to the traffic situations in ARTEMIS.

Two national GIS road databases were employed. The first, VDB, contains all state road links attached with information about: length, road function, speed limit and ADT (average daily traffic) split on light and heavy vehicles. The second, NVDB, were used for municipal and private roads links. NVDB contains road classification and length but lacks ADT. Traffic simulations were performed for four regions to represent the distribution of vehicle mileage over road categories for municipal and private roads. To separate between urban and rural road links a GIS layer with polygons for built-up areas (delivered by SCB, 2005) were utilized. Through this, the study was able to presents new figures concerning the distribution of vehicle mileage over urban and rural roads in Sweden: 41% respective 59%. Furthermore, a model for distributing the urban vehicle mileage on cities of different sizes is presented. Ranking curves for different road types from Björketun et al (2005) and Jensen (1997) were employed for the yearly variation of ADT (monthly, weekly, daily and hourly). Calculations of traffic flow and vehicle mileage at different hours (using ranking curves) for each link of the state road network was performed. Similar calculations were carried out for the municipal and private road links in the test-regions. The results, traffic flow per lane and hour at different rank classes, were related to volume-delay functions according to Matsoms (2004). Hypotheses were formulated concerning the distribution of vehicle mileage for stop & go (level of service 4), which can not be estimated from volume delay functions alone. The work resulted in a distribution of the vehicle mileage (light and heavy vehicles) over road categories and traffic conditions for the Swedish road network for the years 1990, 1995, 1998, 2000 and 2004.

Swedish road categories were translated to ARTEMIS traffic situations based on the description of road hierarchy, speed limit, function and design. Then it was possible to sum the vehicle mileage in Sweden over the traffic situations in ARTEMIS for different years. Eighty-five of ARTEMIS's 276 TS were identified in Sweden in 2004. They consisted of 33 road categories where most of them rarely exceeded traffic condition free flow or heavy traffic. As much as 94% of the vehicle mileage in Sweden is driven at free flow conditions. Stop and go (0.05%) only occurred in the cities >2000000 inhabitants. In Larsson and Ericsson (2006) is reported more details concerning the methodology and results e.g. the most common traffic situations in Sweden.

4. Verification data

4.1 Calculations with the national EMV model

The nationally developed road vehicle emission model EMV has been used since the mid 1990's for Sweden's international reporting obligations on air emissions (Hammarström & Henriksson, 1997; Hammarström & Karlsson, 1998). The EMV model is considered a top-down model, which calculates emissions of regulated and some unregulated compounds for different vehicle categories divided on mainly two traffic situations: urban och rural driving. Besides hot emissions, the EMV model calculates cold start and evaporative emissions taking into account Swedish climate, vehicle fleet etc. For light-duty vehicles the hot emissions and cold start are taken from measurements according to the US FTP driving cycle. Cold start emissions from US FTP are adjusted for Swedish conditions based on most detailed background data. For heavy duty vehicles emission data comes to a large extent from the same sources that feed into the ARTEMIS model, i.e. the COST 346 project, and for the Swedish fleet and road conditions. Thus, the main differences between the two models are the number of traffic situations available, and the driving cycles representing the emissions for light-duty vehicles. The EMV model calculates emissions down to a level formed by the combination: vehicle type, engine type, year model, emission concept level and fuel quality. A change in fuel quality parameters will result in a change of emission factors.

For the present project the activity data in the EMV model was updated to be as equivalent as possible to the activity data fed into the ARTEMIS model.

4.2 On-road emission data

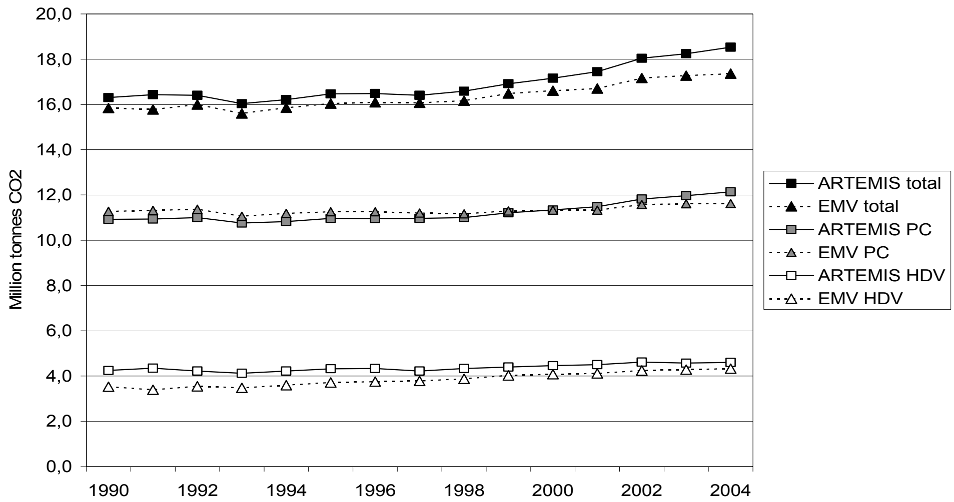
The on-road emission data were from a major remote sensing measurement campaign carried out in Göteborg, Sweden, in 2001 and 2002, originally applied for an evaluation of the COPERT III model (Ekström et al, 2004), therefore only a brief description will be given here. The dataset comprised of instantaneous emissions of CO, HC and NO expressed as grams pollutant released per liter fuel burnt for some 18,000 gasoline passenger cars, some 1,000 diesel passenger cars, and some 600 heavy commercial vehicles. The two measurement sites were classified according to the ARTEMIS traffic situation scheme as Urban Distributor with posted speed 50 km/h and slightly uphill grade of about 2% (one of the sites actually had posted speed 70 km/h, but the actual, measured average speed was closer to 50 km/h, cf. also the Results section).

Results

1. Comparison of ARTEMIS results with results of the EMV model

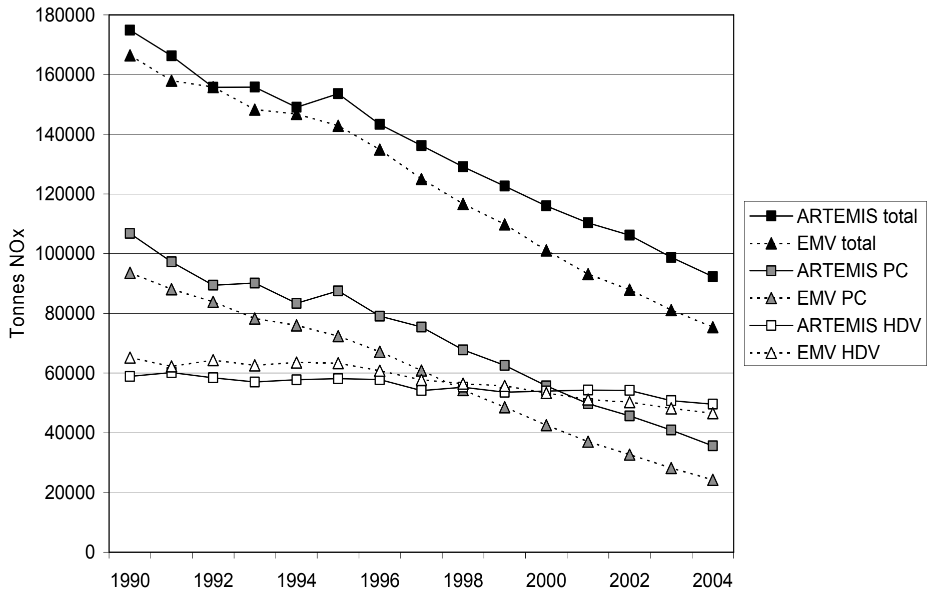
Compared to the EMV model, the ARTEMIS model yields 3 percent higher CO_2-emissions in 1990 and 6 percent higher in 2004, cf. Figure 1. The observed differences are mainly due to higher fuel consumption for HDVs according to the ARTEMIS model, especially for those older than 1990, and higher fuel consumption for new passenger cars according to ARTEMIS. The decrease in fuel consumption over the years is also smaller according to the ARTEMIS model than according to the EMV model. The fuel consumption for cars with cylinder volume larger than 2 liters is actually increasing over the years according to the ARTEMIS model, most likely because the average engine size within this vehicle segment increases.

Figure 1: Yearly national CO_2 emissions from road traffic in Sweden for the period 1990-2004 according to the ARTEMIS model and the national model (EMV), respectively.



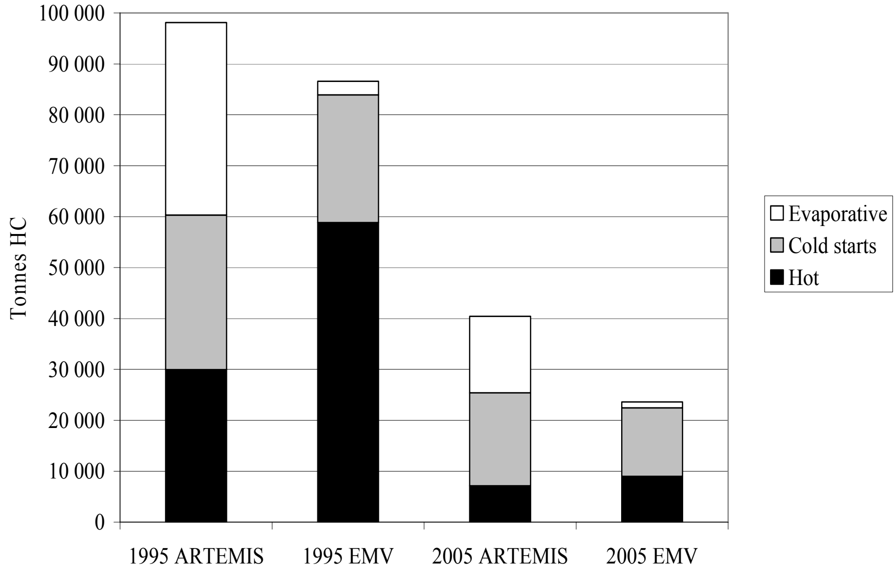
For NO_X, the ARTEMIS model yields 4 percent higher emissions in 1990 and 19 percent higher in 2004 compared to the EMV model, cf. Figure 2. The main reason is that the ARTEMIS model in general yields higher emissions for in particular gasoline passenger cars.

Figure 2: Yearly national NOX emissions from road traffic in Sweden for the period 1990-2004 according to the ARTEMIS model and the national model (EMV), respectively.



For HC, the ARTEMIS model yields markedly higher emissions than the EMV model in both 1990 and 2004: 59 and 42 percent, respectively. A comparison between the two models for gasoline PC HC emissions, separated into hot, cold start and evaporative emissions, are available for 1995 and 2005, cf. Figure 3. It can be seen that, whereas cold start emissions are fairly equal between the two models, the ARTEMIS model yields substantially lower hot emissions, and much higher evaporative emissions. The uncertainty of in particular the evaporative emissions is considered to be high, since both models in this case build on a very limited number of measurements.

Figure 3: HC emissions from gasoline passenger cars in Sweden in 1995 and 2005 according to the ARTEMIS road model and the national model (EMV), respectively.



2. Comparison with on-road data

Results from the comparison of output from the ARTEMIS model with on-road remote sensing emissions data are presented in Figures 4 and 5. For gasoline passenger cars hot emissions there was in general a good agreement between model and on-road data for all three pollutants covered (CO, HC and NO). Results for CO and NO show similar patterns as the one presented for HC in Figure 5. This is strong support for both the ARTEMIS model describing gasoline passenger cars' hot regulated emissions adequately, not only present but also historical emissions (e.g. 1990), as well as for on-road optical remote sensing being a powerful tool for verifying and evaluating road vehicle emission models. As can be seen by Figure 5, there is also a fairly good agreement between the ARTEMIS model and on-road data for HDV NO_X emissions, although the on-road data do not reveal any significant differences in NO_X emissions between the Euro 1, 2 and 3 classes.

Figure 4: HC hot emission factors (expressed as gram pollutant emitted per liter fuel burnt) for gasoline passenger cars according to on-road remote sensing measurements and according to the ARTEMIS model (UD = Urban Distributor, MWC = Motorway City, 50/70 km/h = posted speed, free flow/heavy = level of service, %-figures = applied road grade according to ARTEMIS road model traffic situations definitions).

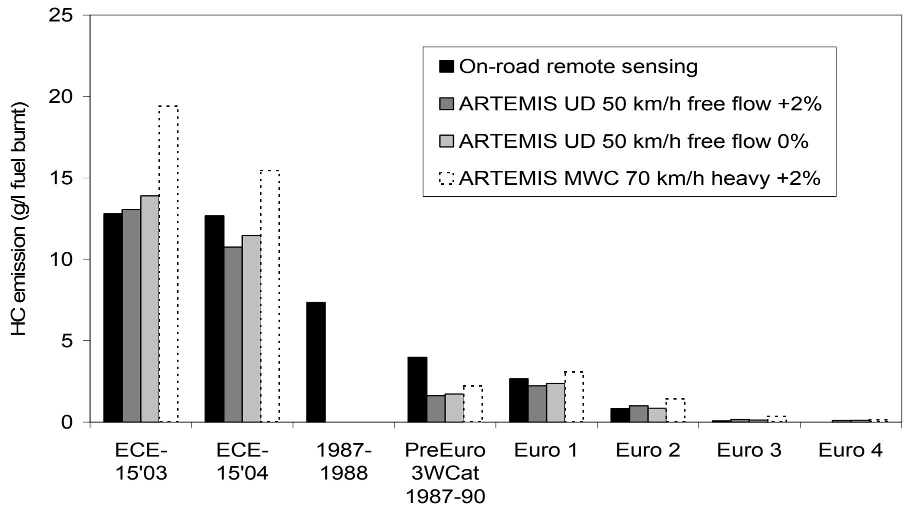
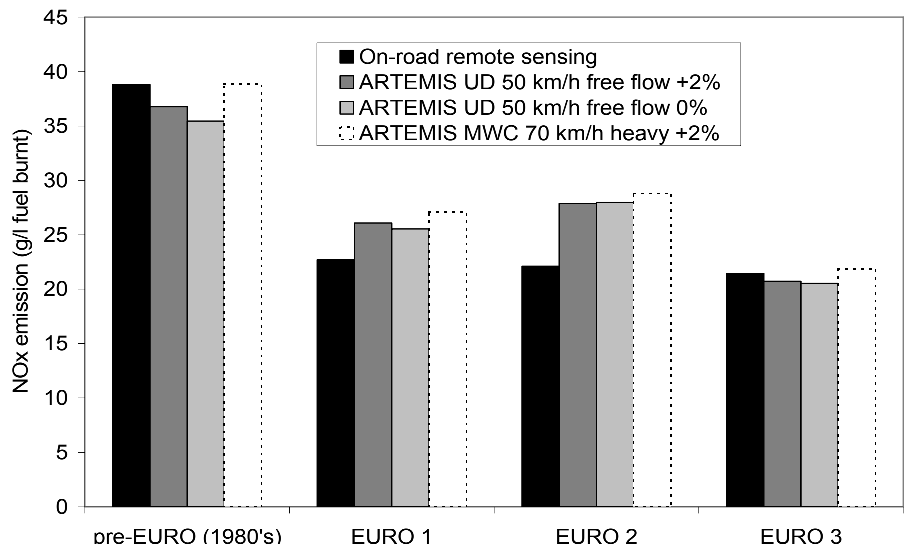


Figure 5: NO_X emission factors (expressed as gram pollutant emitted per liter fuel burnt) for heavy goods vehicles (HGV) according to on-road remote sensing measurements and according to the ARTEMIS model (UD = Urban Distributor, MWC = Motorway City, 50/70 km/h = posted speed, free flow/heavy = level of service, %-figures = applied road grade according to ARTEMIS road model traffic situations definitions).



Conclusions

This study has demonstrated a reasonable agreement between the ARTEMIS model and the national model as regards CO_2-emissions (present and historical), whereas for the regulated pollutants CO, HC, NO_X some significant discrepanices between the models were observed. In contrast, the ARTEMIS model agrees well with on-road emission data, although the differences in HDV NO_X emissions between Euro 1-3 classes predicted by the model, was not confirmed by the on-road data.

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