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# Damage costs due to automotive air pollution and the influence of street canyons

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**Abstract**

Using the methodology of the ExternE Project of the European Commission, we have evaluated the damage costs of automotive air pollution by way of two case studies in France: a trip across Paris, and a trip from Paris to Lyon. This methodology involves an analysis of the impact pathways, starting with the emissions (e.g., g/km of particles from tailpipe), followed by local and regional dispersion (e.g., incremental $\mu g/m^3$ of particles), calculation of the physical impacts using exposure-response functions (e.g., cases of respiratory hospital admissions), and finally multiplication by unit costs factors (e.g., € per hospital admission). Damages are aggregated over all affected receptors in Europe. In addition to the local and regional dispersion calculations carried out so far by ExternE, we also consider the increased microscale impacts due to the trapping of pollutants in street canyons, using numerical simulations with the FLUENT software. We have evaluated impacts to human health, agricultural crops and building materials, due to particles, $NO_x$, CO, HC and $CO_2$. Health impacts, especially reduced life expectancy, dominate in terms of cost. Damages for older cars (before 1997) range from 2 to 41 Euro cents/km, whereas for newer cars (since 1997), the range 1–9 Euro cents/km, and there is continuing progress in reducing the emissions further. In large cities, the particulate emissions of diesel cars lead to the highest damages, exceeding those of gasoline cars by a factor of 7. For cars before 1997 the order of magnitude of the damage costs is comparable to the price of gasoline, and the loss of life expectancy is comparable to that from traffic accidents. 

*Keywords:* Air pollution; Cars; Health impacts; Damage cost; External cost; Urban canyons

## 1. Introduction

Around the world the transport sector, which includes automobiles, trucks, trains, ships and airplanes is contributing ever-increasing shares of the total air pollution burden. There are many policy options to reduce the resulting damages, such as tightening emission limits, shifting to cleaner fuels, developing new propulsion systems, providing financial incentives for retiring older vehicles or restricting the use of cars, although the later option tends to impose severe direct or indirect costs (one should not forget that cars also provide great benefits). Before implementing any of these measures, it is advisable to estimate their benefits and costs.

For environmental policies, one needs to know which source of pollution causes how much damage. Therefore, one needs to begin at the pollution source rather than at the measured ambient concentration of a particular pollutant (bottom-up analysis). In recent years, much progress has been made in such a quantification of environmental damages and costs (Delucchi and McCubbin, 1996, ExternE, 1998). The present paper, by participants in the ExternE Project

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("External Costs of Energy"), presents an overview of the methodology and an application to the case of automotive emissions in France. It also examines a potentially important issue that has not been adequately covered in the dispersion calculations carried out so far, namely the increase in concentrations due to trapping of pollutants in urban street canyons.

The burdens to human health, agricultural crops and building materials are assessed by an analysis of impact pathways: starting with the emission of the pollutants (e.g., kg/km of particles from tailpipe), followed by local and regional dispersion (e.g., resulting increase in $\mu g/m^3$ of particles in the air) and calculation of the physical impacts using exposure response functions (e.g., cases of respiratory hospital admissions). Damage costs are calculated by multiplying the impacts by monetary unit costs factors (€ per impact case). Damages are aggregated over all affected receptors; the spatial boundary of the analysis extends up to thousands of km from the point of emission for the classical pollutants, while it is global for greenhouse gases.

This involves a difficult multidisciplinary system analysis. The ExternE Project has been going on since 1991, with the participation of many tens of scientists in all countries of the EU (dispersion modelers, epidemiologists, ecologists, economists, etc). The uncertainties are large because the required data are often incomplete or missing, and the present knowledge about the effects of air pollutants is not sufficiently detailed. Rabl and Spadaro (1999) have analyzed the uncertainties and found that the true values could be a factor four smaller or larger than the central estimates. Despite the uncertainties, the damage estimates are valuable: they are certainly far better than the infinite uncertainty in the absence of analysis. In addition, most policy decisions do not require great accuracy of the damage estimates, and in many cases, firm recommendations can be made despite the large uncertainties (ETSU, 1996, Krewitt et al., 1999, Rabl et al., 1998).

In this paper, we calculate the damage costs of automotive emissions by considering two case studies in France: a trip across a large metropolitan area (the city of Paris) and an intercity journey between Paris and Lyon (approximately 500 km southeast of Paris). Results are obtained for diesel and gasoline cars equipped with different pollution control technologies. We have quantified the impacts to human health, agricultural crops and building materials due to emissions of PM (particulate matter), $NO_x$, CO and HC (hydrocarbons). For global warming, we cite the ExternE (1998) cost estimates for $CO_2$, $CH_4$ and $N_2O$ (all expressed in terms of $CO_{2eq}$, using global warming potentials). Cost estimates for greenhouse gases, including those species that provide a cooling effect (such as sulfate aerosols), have large uncertainties, and a detailed discussion is beyond the scope of this paper.

In the ExternE Project, topography is taken into account indirectly by considering different surface roughness coefficients and the effect on meteorological data due to the presence of natural or man-made obstacles. However, there is another effect not taken into account: a pollutant emitted in a street canyon takes some time to rise to the freely flowing air above roof level. Consequently, people in the street are exposed to higher concentrations than if the terrain were flat. The degree to which canyons trap pollutants depends strongly on the geometry of the canyon, the speed and direction of the wind relative to the canyon axis and local meteorological conditions, particularly diurnal variations. In this paper, we provide a first order estimate of the influence of street canyons on the total damage cost. Two approaches have been used: (1) a simple box model and (2) a numerical treatment using the computational fluid dynamics software FLUENT (CREARE, 1991). To our knowledge, this is the first attempt to integrate microscale effects into local and regional dispersion calculations to estimate the total impact to the public exposed to automotive emissions in large cities.

## 2. Assessing the impacts and damage costs

### 2.1. The steps of the impact pathways analysis

#### 2.1.1. Source characterization

The impact pathways analysis begins by specifying the source: location (rural or urban site), physical characteristics of the source (stack height, temperature, velocity) and inventory of the emissions. Automotive emissions occur at ground level and are expressed in terms of emission factors (mass of pollutant per distance traveled), which depend on driving conditions (cold start, urban, highway, etc.) and vehicle characteristics (make, age, engine parameters, pollution control technology, etc.).

#### 2.1.2. Atmospheric dispersion and chemistry

Over the years, numerous dispersion models have been developed. Usually separate models are used for the local and the regional domains. In the local domain, less than 50 km from the source, pollutant deposition and aerosol formation by chemical transformation are relatively insignificant and concentrations are influenced primarily by meteorological parameters, such as wind speed and wind direction. Beyond 50 km, chemical reactions and deposition, both dry and wet, play an important role in depleting (removing) the pollutant from the air.

In this analysis, we use a Gaussian plume for local scale concentrations as implemented in the ROADPOL model (Vossiniotis et al., 1996), whereas regional values are calculated using the Lagrangian trajectory models in EMEP (Sandness, 1993; Simpson, 1993) and the Windrose Trajectory Model developed for the ExternE (1995) Project (Krewitt et al., 1995).

### 2.1.3. Impact

Impacts are quantified using Dose-Response functions, also known as Concentration-Response or Exposure-Response functions (ERF). The latter term is used here. ERFs relate the pollutant concentration to the resulting impact on a receptor (human health, crop, etc.). Impacts on human health include asthma attacks, hospital admissions, chronic bronchitis, restricted activity days, and premature deaths. ExternE calculates mortality impacts as a reduction in life expectancy, expressed as years of life lost (YOLL). That is necessary to allow meaningful comparisons with other causes of death, for instance accidents for which the YOLL per death are much higher than for air pollution.

For health impacts, the ERFs are derived from a survey of epidemiological studies (ExternE, 1998). In view of the available epidemiological evidence, we assume that the ERFs for health are straight lines, with no safe threshold below which there is no effect (at least not on a population wide level and for current ambient concentrations).

For crops and building materials, the ERFs have nonlinear shapes. In fact, there is even the possibility of a small benefit for some agricultural crops when the background concentration of $SO_2$ and/or $NO_x$ is sufficiently low (fertilizer effect). For crops, one calculates the losses or gains in yield, and for building materials, the surface area that is damaged by pollution.

### 2.1.4. Monetary valuation

To obtain the damage costs, one multiplies the number of impacts (ex., cases of asthma attack) by the unit cost per case (ex., € per asthma attack). For health impacts, the unit costs include the cost of illness, wage and productivity losses, which are market based factors, as well as non-market costs that take into account an individual's willingness-to-pay (WTP) to avoid the risk of pain and suffering. Economists have developed several techniques for valuing non-market goods. In recent years, contingent valuation has become the method of choice, which obtains WTP estimates by asking individuals how much money they are willing to pay to achieve a benefit. For mortality impacts, one needs to determine the value of a life year (VLY), which is based on the so-called value of statistical life (VSL), the amount of money that society is willing to pay to avoid an anonymous premature death. In this work, we have used the VSL and VLY values adopted by ExternE (1998), with VSL equal to 3.1 million $€_{1995}$ and VLY equal to 84330 $€_{1995}$ for chronic mortality. The unit cost values for crops and building materials are based on market and material replacement costs. As crop damages are relatively small, they are estimated simply on the basis of quantity times constant price, without consideration of induced effects (compensatory producer behavior).

Monetization is a convenient method for aggregating health impacts and environmental burdens having different physical units into a single damage estimate or indicator. Moreover, an economic assessment is advisable when comparing the benefits and costs of abatement measures, technological choices or policy regulations, for example, legislation in favor of mass transit and/or electric cars instead of conventional automobiles. Without economic evaluation, one risks making decisions that may lead to substantial welfare losses or improper allocation of resources.

Since health impacts dominate the damage costs (accounting for 90% or more of the total) and the ERFs for health effects are simple to state, we summarize them in Table 1, along with the corresponding monetary unit costs.

## 2.2. A simple formula for damage estimation: the uniform world model

Since we are interested in the expectation value of the damage, rather than the damage for a particular pollution episode, we calculate the concentration for a steady emission source and express the damage per unit mass (kg of pollutant). The damage cost $D$ is the product of the receptor density $\rho(\mathbf{r})$, the exposure response function $f(\mathbf{r}, c(\mathbf{r}, Q))$ and the unit cost $U_V(\mathbf{r})$, integrated over all points $\mathbf{r} = (x, y)$ where receptors are affected by the pollutant

$$D = \int_{\text{Area of impact}} \rho(\mathbf{r}) f(\mathbf{r}, c(\mathbf{r}, Q)) U_V(\mathbf{r})\, \mathrm{d}x\, \mathrm{d}y. \tag{1}$$

Here $c(\mathbf{r}, Q)$ is the incremental ground level concentration at location $\mathbf{r}$ due to a source at $\mathbf{r} = 0$ that emits at the rate $Q$. For the most general case, the ERF may depend on the concentration $c$. In the situation, where the ERF is linear with slope $f_{\text{ER}}$ and the unit cost is constant, mass conservation allows Eq. (1) to be rewritten as (Curtiss and Rabl, 1996; Spadaro, 1999)

$$D = \frac{\rho_{\text{av}} f_{\text{ER}}\, Q U_V}{k} R \tag{2}$$

with

$$R = \int_{\text{Area of impact}} \frac{\rho(\mathbf{r})}{\rho_{\text{av}}} \frac{M(\mathbf{r})}{Q}\, \mathrm{d}x\, \mathrm{d}y,$$

where $\rho_{\text{av}}$ is the receptor density averaged (land and water) over a radius of 1000 km from the source; $M$ is the pollutant depletion flux due to deposition (dry and wet) and chemical transformation; while $k = M/c$ is the depletion velocity relating the depletion flux and the ground-level concentration. The depletion velocity $k$ can

Table 1
ERFs and unit costs for health impacts in Europe (ExternE, 1998)

| Health endpoint | ERF[a] | Pollutant | Unit cost (€$_{1995}$/case)[b] |
|---|---|---|---|
| Long-term or chronic mortality (YOLL) | $4.1\times10^{-4}$ | (Nitrates) | 84,330 |
| | $6.8\times10^{-4}$ | (PM, Sulfates) | |
| Short-term or acute mortality (YOLL)[c] | $5.3\times10^{-6}$ | ($SO_2$) | 155,000 |
| Chronic bronchitis (adults) | $3.7\times10^{-5}$ | (Nitrates) | 105,000 |
| | $5.9\times10^{-5}$ | (PM, Sulfates) | |
| Restricted activity days (adults) | $1.9\times10^{-2}$ | (Nitrates) | 75 |
| | $3.2\times10^{-2}$ | (PM, Sulfates) | |
| Asthmatics (adults)[d] | $1.0\times10^{-1}$ | (Nitrates) | 7.5 |
| | $1.7\times10^{-1}$ | (PM, Sulfates) | |
| Respiratory hospital admissions | $2.1\times10^{-6}$ | (Nitrates) | 7870 |
| | $3.5\times10^{-6}$ | (PM, Sulfates) | |
| | $2.0\times10^{-6}$ | ($SO_2$) | |
| Cerebrovascular hospital admissions | $5.0\times10^{-6}$ | (Nitrates) | 7870 |
| | $8.4\times10^{-6}$ | (PM, Sulfates) | |
| Congestive heart failure (elderly) | $2.4\times10^{-6}$ | (Nitrates) | 7870 |
| | $4.0\times10^{-6}$ | (PM, Sulfates) | |
| | $7.0\times10^{-8}$ | (CO) | |
| Bronchitis (children) | $3.9\times10^{-4}$ | (Nitrates) | 225 |
| | $6.5\times10^{-4}$ | (PM, Sulfates) | |
| Chronic cough (children) | $5.0\times10^{-4}$ | (Nitrates) | 225 |
| | $8.3\times10^{-4}$ | (PM, Sulfates) | |
| Asthmatics[d] (children) | $1.9\times10^{-1}$ | (Nitrates) | 7.5 |
| | $3.2\times10^{-1}$ | (PM, Sulfates) | |

[a] ERF has units of cases/yr per person per concentration (micrograms/m$^3$).
[b] In 1995, 1 € was equal to US $1.25.
[c] YOLL = Years Of Life Lost.
[d] Asthmatic impacts include bronchodilator usage and incidences of coughing and wheezing.

vary somewhat with **r**, however here we have assumed a uniform value across the entire impact domain. We have calculated values of $k$ for typical European conditions and found that for $PM_{2.5}$ $k$ is approximately 0.7 cm/s (range 0.5–2 cm/s) (Spadaro and Rabl, 1999).

For primary pollutants the quantity $R$ in Eq. (2) can vary strongly with receptor distribution and source parameters, particularly the stack height. $R$ is unity when the depletion velocity and the receptor distribution are uniform throughout the domain. For an elevated source (stack heights greater than 25 m), the value of $R$ is typically less than 10, but when emissions occur close to the ground in a large city $R$ can have values as high as 100 for cars in Paris (Spadaro, 1999). For secondary pollutants the value of $R$ is close to unity; local conditions have little influence on the damage costs because secondary pollutants form at some distance (typically tens of kilometers) from the point of emission of the primary pollutants (Spadaro and Rabl, 1999; Spadaro, 1999).

## 3. Damage costs of transport

### 3.1. Case study description and emissions

Two case studies have been considered: (1) an urban trip across the city of Paris in France and (2) an intercity journey between Paris and Lyon, the two largest cities in the country. The results reported here are limited to calculating the damages due to tailpipe emissions. The damage costs from up- and down-stream processes, such as fuel and material extraction and processing, vehicle production and disposal, have been quantified in Spadaro (1999).

To obtain representative damage estimates for Paris, we consider a trip between the two main Parisian airports of Charles-de-Gaulle, northeast of Paris, and Orly, in the south. The journey traverses the center of the city and covers a distance of 43 km. The population density for Paris and its metropolitan area are, respectively, 22,300 and 1000 persons/km$^2$.

For our second case study, we have chosen a journey from the center of Paris to the center of Lyon. Most of the journey is highway driving across sparsely populated country. The travel distance is 481 km. The local population (persons living within 50 km of the road) is around 15 million, two-thirds of whom live in the Greater Paris area, 10% in the Lyon metropolitan area and 20% in the region separating the two cities; the average receptor density is 375 persons/km$^2$.

The following cars have been considered: (1) a diesel car before 1997 (Standard Euro I), (2) a diesel car since 1997 (Standard Euro II), (3) a gasoline car without a catalytic converter (required in France only since 1993), (4) a gasoline car with a three-way-catalytic converter compliant with Standard Euro I, and (5) a gasoline car compliant with Standard Euro II (required since 1997).

The emissions are evaluated by subdividing the journey into several segments according to local meteorological conditions and driving conditions (urban vs. highway, cold vs. warm driving cycle, etc.). Separate dispersion calculations are carried out for different segments. The emission factors have been taken from Jourmard et al. (1995). The emission factors recommended by these studies are considered representative averages for the fleet of automobiles in use in France in 1995 (Euro I). For newer cars in compliance with Euro II standards, we use the emission factors estimated by the MEET Project of the European Commission (MEET, 1999). Table 2 summarizes the aggregate emissions, expressed in grams, for the two case studies. Note that particles from gasoline and diesel engines are $PM_{2.5}$ (smaller than 2.5 μm diameter).

### 3.2. Discussion of case study results

The damage costs normalized per ton of pollutant (€/ton) and per vkm (vehicle-kilometer) (Euro cents/vkm) are presented, respectively, in Figs. 1 and 2. These costs cover human health, agricultural crops and building materials, with health impacts accounting for at least 90% of the total.

As shown in Fig. 1, particles have the highest damage costs per unit mass, especially in Paris. Values for primary pollutants in Paris exceed those for the intercity journey by an order of magnitude; this is consistent with differences in the local receptor density. For driving in

Table 2
Aggregate emissions in grams[a]

| Transport mode | CO | HC | $NO_x$ | $CO_{2eq}$ | $PM_{2.5}$ |
|---|---|---|---|---|---|
| Diesel car | | | | | |
| *Trip across Paris (43 km)* | | | | | |
| Euro I (before 1997) | 41 | 7.3 | 32 | 9280 | 7.5 |
| Euro II compliant (since 1997) | 11 | 1.4 | 18 | 6550 | 1.5 |
| *Paris–Lyon journey (481 km)* | | | | | |
| Euro I (before 1997) | 268 | 54 | 299 | 84,740 | 72 |
| Euro II compliant (since 1997) | 157 | 157 | 265 | 83,780 | 31 |
| Gasoline car | | | | | |
| *Trip across Paris (43 km)* | | | | | |
| Euro I, no catalyst (before 1993) | 1123 | 126 | 122 | 8850 | 1.5 |
| Euro I, TWC[b] (since 1993) | 123 | 11 | 30 | 11,230 | 0.8 |
| Euro II compliant (since 1997) | 95 | 3.0 | 8.5 | 7390 | 0.1 |
| *Paris-Lyon journey (481 km)* | | | | | |
| Euro I, no catalyst (before 1993) | 6341 | 780 | 1617 | 81,810 | 14 |
| Euro I, TWC (since 1993) | 622 | 59 | 379 | 97,380 | 7.2 |
| Euro II compliant (since 1997) | 879 | 23 | 104 | 73,160 | 1.0 |

[a] Sources of emission factors: (a) Euro I results based on measured emissions during standard driving cycles of the EU (Jourmard et al., 1995); (b) Euro II data from MEET (1999), except for $PM_{2.5}$ results for gasoline vehicles for which we use measured data by CONCAWE (1998).

[b] TWC = Three-Way-Catalytic converter.

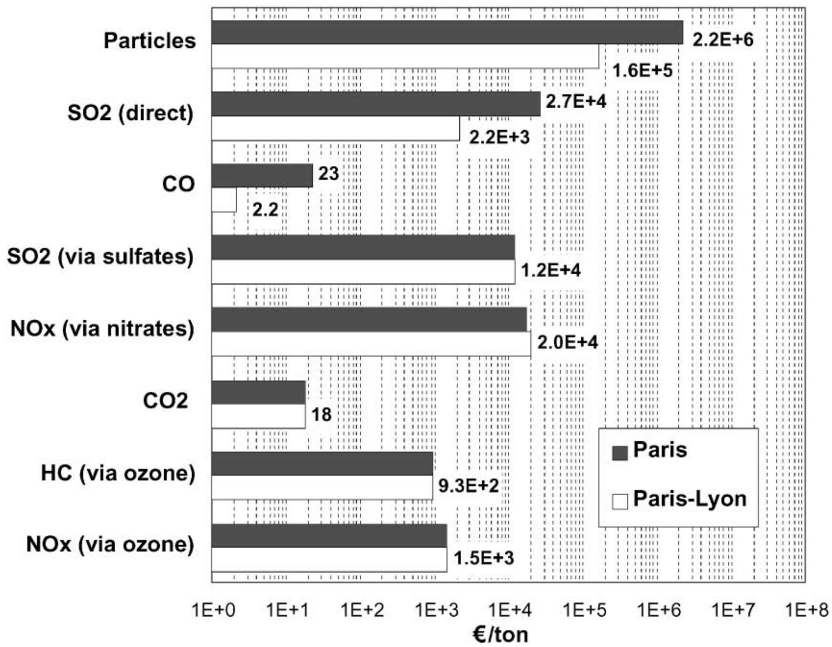


Fig. 1. Damage costs in €$_{1995}$ per ton of pollutant. Estimates include damages to human health, crops and buildings. Cost of global warming and ozone are those of ExternE (1998) and Rabl and Eyre (1998), respectively.

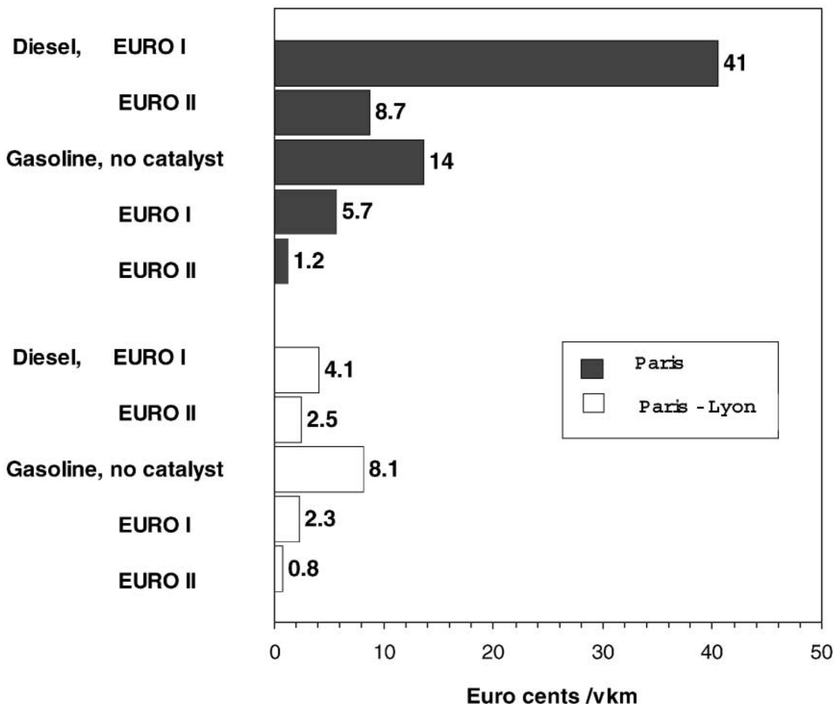


Fig. 2. Damage costs of transport, for emissions of Table 2. (1 € was equal to US $1.25 in 1995, and about $0.9 in 2000).

entirely rural areas, additional calculations with a local receptor density of 60 persons/km$^2$ indicate that the damages for primary pollutants are lower than the Paris–Lyon values by a factor of 10 (Spadaro, 1999).

Damages via secondary pollutants, by contrast, are rather insensitive to the local population density, hence urban and intercity costs are quite similar. The costs for particles, nitrates and sulfates are large because they include long-term (also called total or chronic) mortality, whereas for CO and for direct impacts of $SO_2$ only short-term (acute) impacts are included.

Results per vkm are presented in Fig. 2. For Paris, the diesel car has the largest damage cost, 41 cents/vkm for cars before 1997 and 8.7 Euro cents/vkm for cars since 1997 (Euro II). For diesel cars before 1997, more than 95% of the overall cost is due to PM and 3% due to $NO_x$. The gasoline car without catalytic converter has a damage cost of 14 cents/vkm, more than double the corresponding estimate for the Euro I gasoline car (5.7 cents/vkm), and more than an order of magnitude larger than for Euro II gasoline cars (1.2 cents/vkm). A closer look at the damage distribution by pollutant (not shown here) reveals that despite the very low emission rate of particles from gasoline cars, the contribution to the overall cost remains significant, around half of the total, because of the high unit damage cost for particles.

For the intercity trip, the gasoline car without catalytic converter has the highest damage, 8.1 cents/vkm, twice that of the diesel car before 1997, nearly four times that of the Euro I gasoline car or Euro II diesel car, and eight times that of the Euro II gasoline car. For diesel cars, damages are once again dominated by health impacts due to particles, whereas $NO_x$ impacts dominate the costs for gasoline cars, 88% of the total for the gasoline car without catalyst. 90% of the impacts from particles occur locally (50 km of the road). Paris alone accounts for a quarter of the impact, Lyon 5%, Ile-de-France (Greater Paris) for half and the region separating the two cities accounts for about 20%.

For comparison, we note that the price of fuel is about 3.9 cents/vkm for diesel and 7.6 cents/vkm for gasoline (taxes being 80% of the fuel price (DHYCA, 1997)). Another comparison is with the years of life lost (YOLL) due to traffic accidents, in Fig. 3; this comparison is especially interesting because it avoids the uncertainties and controversies surrounding the value of a YOLL. The YOLL due to air pollution can be extracted from the damage costs by noting that approximately 85% of the total health damage are due to mortality, which has been evaluated at 84,330 €/YOLL. For traffic accidents, about 9000 fatalities per year in France in 1995, we take an average loss of life of 40 YOLLs. Allocating this to the average distance driven, 5850 vkm/yr per person (OECD, 1997), one finds approximately 1 YOLL/million km (for a population of 58.7 million). This estimate is of the same order

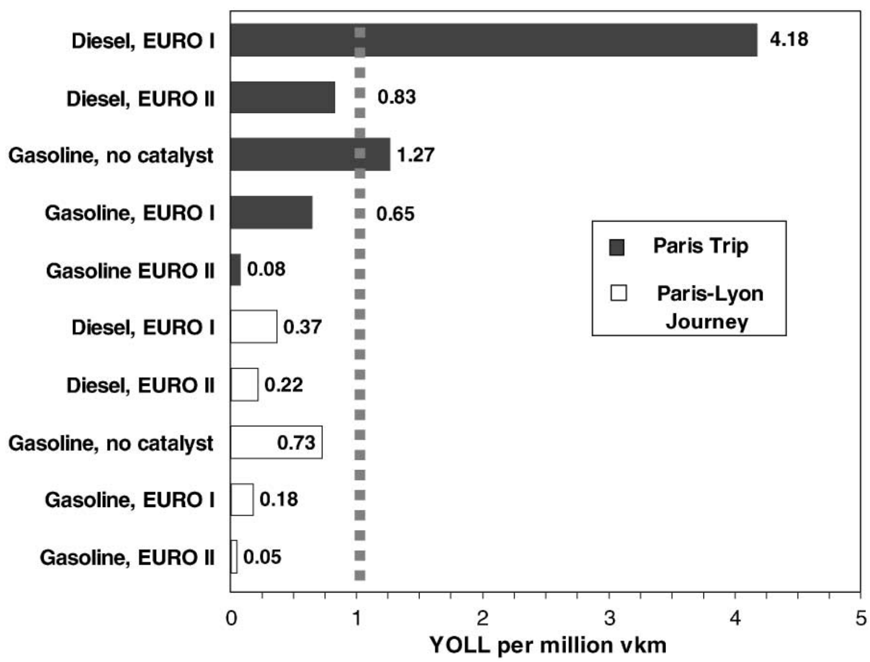


Fig. 3. Comparison of mortality impacts due to traffic accidents (dashed line) and automotive air pollution (bars) for French population. YOLL = Years Of Life Lost.

of magnitude as most of the numbers shown in Fig. 3. Only for new gasoline cars (since 1997) are pollution mortality risks very much lower than accident risks.

## 4. Street canyons

### 4.1. General remarks

Since automotive emissions take place at ground level, street canyons can have a significant influence on the dispersion of pollutants near the source. Canyons affect the dispersion in two ways. Firstly, they enhance vertical mixing above the canyons. Secondly, they tend to trap pollutants within their geographical or physical boundaries, thereby delaying pollutant transport to the freely moving air above the canyons. Whereas the first effect has implicitly been accounted for by choosing urban dispersion conditions as input to the Gaussian plume model, here we address the second effect.

Qualitatively, this effect of canyons on the impact estimates is obvious: higher local concentrations result in higher impacts and damage costs. The effect could be particularly important for large European cities, where local receptor densities are high and street canyons fairly narrow and deep. The canyon effect is relevant only for primary pollutants. For secondary species, it is much less important because the canyon residence time is small compared to the time needed for chemical reactions.

One of the difficulties of this work lies in the great variability of possible local conditions and in the explosion of detail, were one to try to take every street of a large city into account. Modeling the dispersion to such level of detail would not be feasible with current computational means, to say nothing of the labor needed to input all the detail into the program. On the other hand, for most policy decisions such detail is not even relevant because they should be based on typical driving patterns rather than a particular car in a particular street. Since our work is directed towards policy applications, a coarser modeling approach is appropriate.

Fig. 4 shows flow streamlines and concentration contours for a steady source of a chemically inert pollutant located in the middle of a street canyon. These results were obtained using the commercial software FLUENT, version 4.0 (October 1991); it is a computational fluid dynamics solver that calculates pressures and flow velocities using the SIMPLE algorithm of Patankar (1980). Profiles are shown for two different aspect ratios: 1 : 1 and 6 : 1 (height : width). For square canyons, a single vortex is formed, with its center in the middle of the canyon volume; whereas, two stacked vertices exist inside the 6 : 1 canyon. The size and strength of the vortices vary strongly with canyon volume and wind speed.

In Fig. 5, we consider a single, steady source located in a two-dimensional (2D) street canyon, which is part of a series of 33 equally, spaced canyons, each having an aspect ratio of 1 : 1. The source is in Canyon #16. The wind blows from left to right at a speed of 3 m/s. Concentration values at selected points are also indicated, normalized by the concentration at the bottom left-hand corner of Canyon #16. Concentration variations for Canyon #16 are consistent with measured data by Gally et al. (1991) for street canyons in Paris.

The average concentration along the right wall of Canyon #16 is approximately five times higher than that along the left wall of Canyon #17. This explains why, for instance, the surface of a building exposed to traffic emissions on one side is much dirtier than another surface facing a yard. Indeed this was the case (before the recent cleaning) for the Ecole des Mines building in Paris, where the authors work(ed): one side of the building faces the heavily traveled Boulevard St-Michel and the other side the Luxembourg gardens. Furthermore, the average concentration in Canyon #16 is an order of magnitude higher than in Canyon #17, about 18 times higher than in Canyon #18, and nearly 25 times higher than in canyon #19. Reducing the wind speed to 1.5 m/s increases the absolute values of the concentrations by about 30%.

### 4.2. The canyon effect: a simple box model

The canyon effect occurs only in the canyon where the source is located; once the polluted air has emerged from the canyon, it will not be reconcentrated if it descends into neighboring canyons. Thus, its dispersion beyond the first canyon can be modeled by a conventional Gaussian plume, provided the presence of buildings is accounted for by means of appropriate surface roughness parameters. This suggests that the total damage can be calculated as a sum of two terms: (1) the damage within the emission canyon itself, and (2) the damage beyond.

As first, very rough estimation of the canyon effect, let us consider a steady source with emission rate $Q$ inside a 2D rectangular street canyon of height $h_C$ and area $A_C$, modeling the canyon as a well-mixed box with volume $V = h_C A_C$ and air exchange rate $\dot{V}$. The time constant $\tau$ for air exchange is

$$\tau = \frac{V}{\dot{V}}. \tag{3}$$

Let $c_C$ be the concentration in the canyon. Since under steady state the emission rate $Q$ equals the rate $\dot{V}\, c_C$ at which the pollutant leaves the canyon, we have

$$c_C = \frac{Q\tau}{A_C h_C}. \tag{4}$$

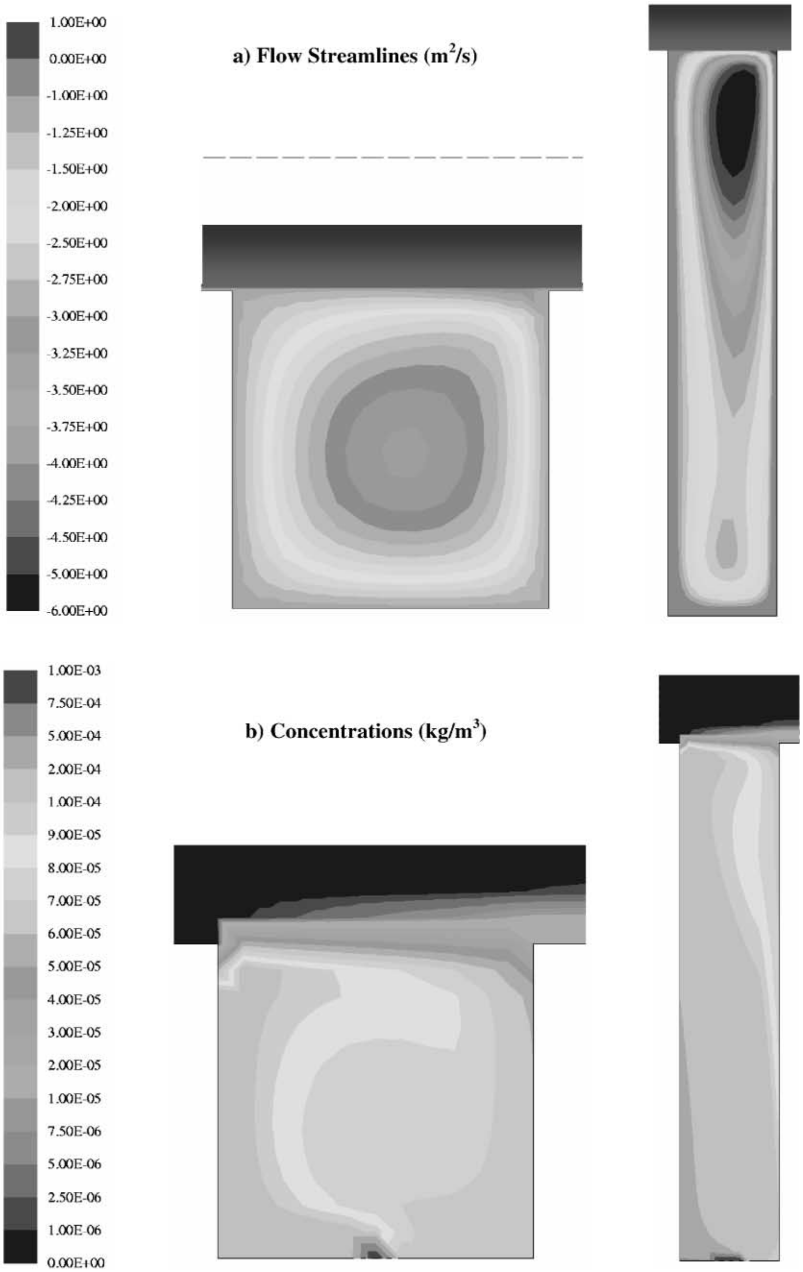


Fig. 4. FLUENT profiles of streamlines (a) and concentrations (b) for an emission source located inside 2D city canyons, of aspect ratios 1 : 1 and 6 : 1. The emission rate is steady at 0.23 g/s and the flow is turbulent. Wind speed is 5 m/s, from left to right. (Colorized version: http://www-cenerg.ensmp.fr/english/themes/index.html).

The damage in the canyon $D_C$ is the product

$$D_C = f_{ER} A_C \rho_C c_C U_V, \tag{5}$$

where $\rho_C$ is the receptor density in the canyon, $f_{ER}$ the slope of the ERF and $U_V$ the unit cost. Inserting Eq. (4) this can be written as

$$D_C = f_{ER}\, \rho_C \frac{Q\tau}{h_C} U_V. \tag{6}$$

To get an idea of typical values of the time constant $\tau$, we have carried out a large number of simulations with FLUENT. Most of them were 2D, with wind transverse to the canyon. A few were three-dimensional (3D), varying the angle between wind and canyon (Spadaro, 1999). For the present purpose it suffices to summarize the results in terms of the time constant: we found it to be in the range 100–1000 s for situations that should be

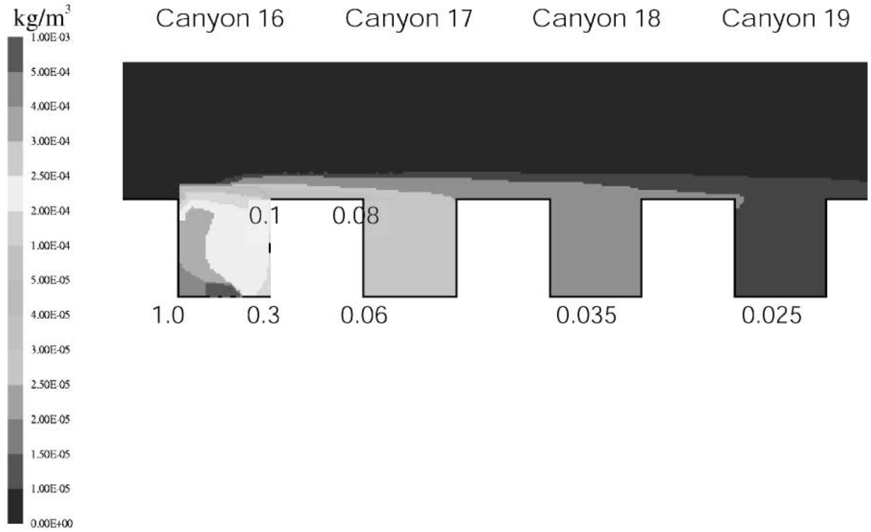


Fig. 5. FLUENT concentrations for a single source located at the bottom center of canyon 16 in a two-dimensional configuration of 33 canyons. The emission rate is steady at 0.35 g/s and the flow is turbulent. Wind blows from left to right at 3 m/s. The numbers indicate some concentration values relative to bottom left of source canyon. (Colorized version: http://www-cenerg.ensmp.fr/english/themes/index.html).

representative of real cities. This is in good agreement with simulations reported by Lee and Park (1994), as well as with the data measured by DePaul and Sheih (1984) as cited by Lee and Park. The larger values correspond to infrequent conditions such as low wind speeds or wind transverse to deep canyons.

If Eq. (2) is to yield the damage costs of Fig. 1, the value of $\rho_{av}\ R$ must be

$$\rho_{\text{eff, Paris}} = \rho_{av}\ R = 6240\ \text{persons/km}^2. \tag{7}$$

Substituting Eq. (7) into Eq. (2) and comparing with $D_C$ of Eq. (5), we obtain the canyon effect as

$$\frac{D_C}{D_{\text{PARIS}}} = \left(\frac{k}{v_C}\right)\left(\frac{\rho_C}{\rho_{\text{eff, Paris}}}\right), \tag{8}$$

where $v_C = h_C/\tau$ is the velocity at which the pollutant is removed from the canyon. Taking $h_C = 20$ m, typical of street canyons in Paris, and $\tau = 100-1000$ s, one finds $v_C = 2-20$ cm/s. For the depletion velocity of particles, we assume $k = 0.7$ cm/s. The receptor density in the center of Paris is approximately $\rho_C = 22000\ \text{person/km}^2$. Thus, the ratio $D_C/D_{\text{Paris}}$ could lie in the range 0.1–1.

### 4.3. A series of canyons modeled by FLUENT

Treating buildings as simple rectangular boxes, we have used the FLUENT software to investigate a large number of 2D and 3D configurations, varying, for example, the aspect ratio of the canyons and the wind speed and wind direction relative to the axis of the canyon. However, the most instructive simulations, and the ones used for this analysis, involve the 2D configurations shown in Fig. 6, where a city is represented by a series of regularly spaced canyons with aspect ratio 1 : 1. The wind blows from left to right at a constant speed of 3 m/s. At the bottom of each canyon (in Figs. 6b and d) or at top of each building (in Fig. 6c), covering 35% of the ground (or roof) are pollutant sources, all of equal strength. Thus, we now have distributed sources, by contrast to the source in a single canyon of the previous section.

The population distribution is assumed uniform over all the horizontal surfaces. This may appear a gross simplification since few people live on the roof. However, a large fraction of windows face courtyards. And while different floors are exposed to different concentrations, as first approximation we assume that the average exposure is the same as if one half of the population were exposed to the concentration at street level and the other half to that at roof top. A more detailed distribution does not seem appropriate for a first analysis, as the effect is relatively minor. We recall that only average exposure matters because the ERFs for health are linear.

Our first thought had been to compare the total exposure of the population for the configuration of Fig. 6b, with the one in Fig. 6a corresponding to flat terrain. However, a direct comparison between these two configurations is not appropriate because the turbulence and the resulting plume spread are much greater in the presence of building than over flat terrain (even though we specify turbulent flow for all of the FLUENT simulations).

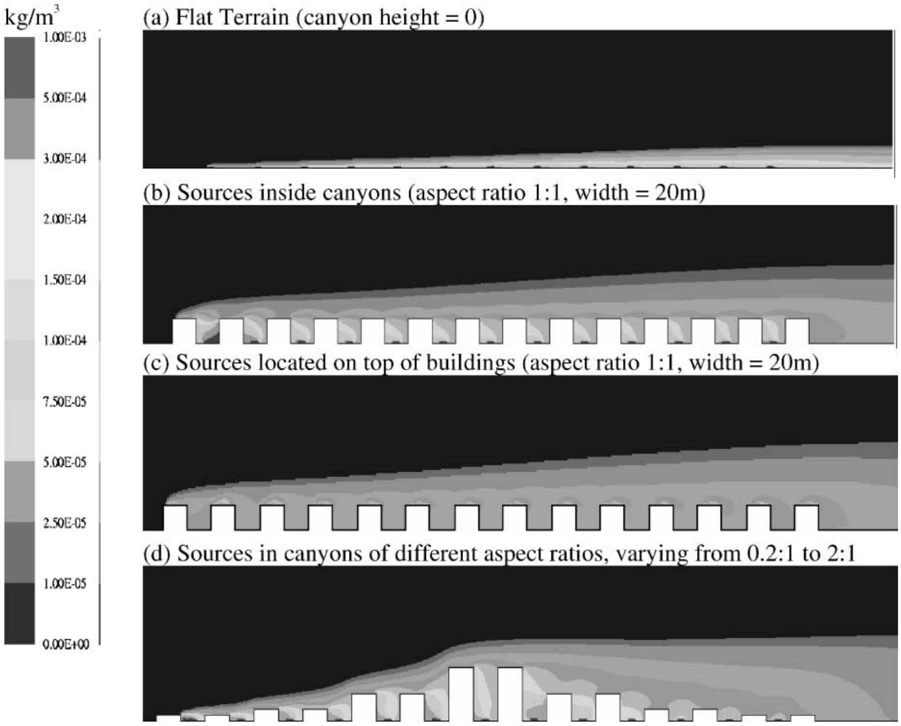


Fig. 6. FLUENT concentration contours for various canyon and source configurations (turbulent flow). Wind blows from left at 3 m/s. Canyons are two dimensional, with the emission sources located at the bottom center of each one (emission rate 0.47 g/s per canyon). The canyon effect is the ratio of concentrations for cases (b) and (c). (Colorized version: http://www-cenerg.ensmp.fr/english/themes/index.html).

Instead, we compare Fig. 6b with Fig. 6c, of the same geometry, but where all the sources are on top of the buildings. This corresponds roughly to the flat terrain Gaussian plume calculations in urban settings. Since damage is proportional to concentration, it suffices to compare concentrations between the two configurations. The concentrations for a series of 33 equally spaced canyons are plotted in Fig. 7. Each value represents the average over the horizontal surfaces of one canyon unit (building top plus adjacent canyon).

Ideally, we would have preferred a much larger number of canyons with aspect ratios decreasing towards the edge of the city (as in Fig. 6d). Unfortunately, our version of FLUENT is not very convenient for inputting such complex geometries, and our results are limited to only 33 canyons, which cover a distance of roughly 1 km (about 10% the size of Paris). Except for the first canyon, the ratio of the concentrations varies slowly with increasing canyon number, suggesting that this ratio might indeed give a fair indication of the importance of the canyon effect. For the individual canyons, the ratio varies from 1.11 to 1.34 between canyons 2 and 33. Averaged over the entire city, the ratio is 1.26 excluding and 1.27 including the first canyon. If we extrapolate the results up to 100 canyons ($\sim$3 km), the ratio would increase to 1.5. This assumes, of course, that the observed rate of growth does not change, but in reality, we would anticipate that the ratio would approach an asymptotic limit. Furthermore, as we move further away from the center of the city, the canyon aspect ratio decreases, and the additional mixing with less polluted background air would limit the concentrations inside the canyons as already indicated in Fig. 6d. If these results were indeed typical of real cities, they would imply that the canyon might contribute about 25%, and most likely no more than 50%, of the total damage. Repeating the calculations at different wind speeds, we find that the ratio 1.26 would go up to 1.5 when the wind speed is reduced by a factor of two, namely 1.5 m/s. These numbers are in agreement with the estimate using the simple box model. Therefore, we conclude that the canyon effect is of relatively minor importance, potentially increasing the damages by 25% or so.

We have not included the canyon effect in our damage estimates for Paris because much of the journey involved open suburban roads for which pollutant confinement is negligible. However, for policy applications that concern driving within large cities that canyon factor of about 1.25 should be included.

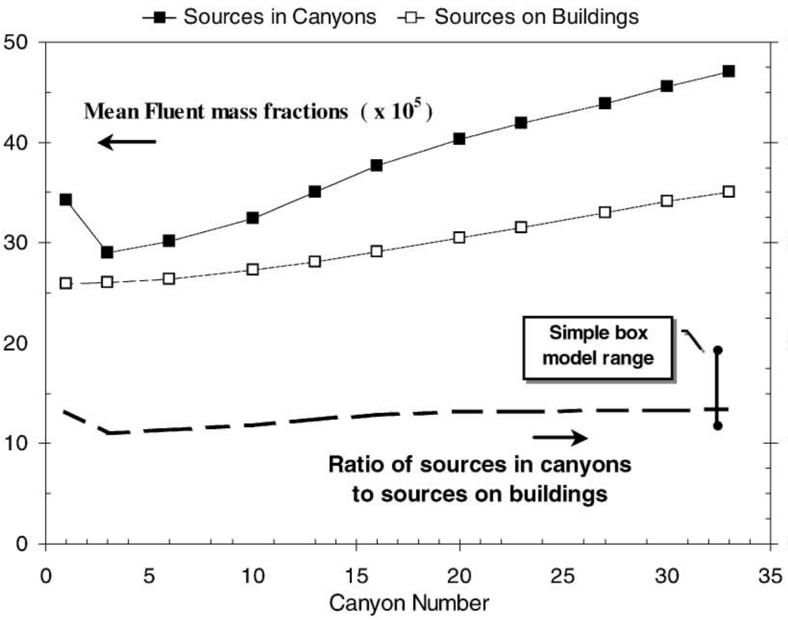


Fig. 7. The effect of street canyons on local concentrations. Wind speed = 3 m/s, emission rate = 0.35 g/s m, aspect ratio = 1 : 1, and canyon width = 14 m. Left scale shows concentrations (thin solid lines), right scale the ratio of sources in canyons to sources on buildings (dashed line).

## 5. Summary and conclusions

Following the impact pathways methodology, we have estimated the damage costs associated with automotive emissions. The impact pathways analysis traces the fate of a specified pollutant from emission to its final impact on human health, agricultural crops and man-made environments. The dispersion calculations cover all of Europe. Once the physical impacts are estimated, the costs are obtained by multiplication with a monetary unit cost factor. For mortality and morbidity impacts, these factors take into account both market and non-market costs (e.g., willingness to pay to avoid the pain and suffering of illness).

We considered two case studies in France: one a trip across the city of Paris and the other an intercity journey from Paris to Lyon. Damage estimates were presented for pre-1997 diesel and gasoline cars without and with catalyst (required in France only since 1993), as well as for Euro II compliant cars (since 1997). For urban transport, the diesel cars have the highest damage costs, 8.7 and 41 Euro cents/vkm for Euro II and Euro I, respectively. Older gasoline cars without a catalyst, on the other hand, have the highest damage for the intercity trip (8.1 Euro cents/vkm). There has been impressive progress in reducing car emissions. In fact, Euro II gasoline vehicles have much lower damages, 0.8 and 1.2 Euro cents/vkm for the intercity and urban journeys, respectively. Similar performance can be achieved by fitting diesel cars with a particle filter.

Our analyses indicate that the damages for diesel cars are dominated by health impacts due to particle emissions, whereas health impacts due to $NO_x$ emissions contribute the lion's share in the case of gasoline-powered vehicles. Per ton of pollutant, the damage of primary pollutants is approximately ten times higher for Paris than for the trip from Paris to Lyon, by contrast to secondary pollutants for which there is little difference.

To put the damages in perspective, we have compared damage costs with fuel price and mortality risks from air pollution with traffic accidents. For older automobiles and Euro II type diesel vehicles, the damage costs are comparable to the price of fuel in France. Furthermore, the loss of life expectancy from exposure to traffic emissions is similar in magnitude to the loss of life due to transport accidents, except for the Euro II gasoline cars.

For urban traffic emissions, we have combined local and regional dispersion analyses with curbside dispersion (microscale effects) to account for the trapping of pollutants in street canyons. FLUENT simulations indicate that concentrations inside the source canyon can be several times higher than those observed in neighboring canyons. This explains why the facade's of buildings exposed to transport emissions can be much darker (dirtier) than surfaces not exposed to the traffic. However, our preliminary assessment of the canyon effect suggests that it is unlikely to increase the total damage by more than about 50%.

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