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Chapter xx

Impact Analysis for Air and Water Pollution: Methodology and Software Implementation

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Abstract

Rational management of the environment requires evaluation of environmental impacts and the associated costs. The logically correct way to quantify environmental impacts is the impact pathway methodology which traces the fate of each pollutant or other burden, from the source to the receptors, using dose-response functions to evaluate the damage. The present chapter describes the methodology and discusses key issues. The site dependence of impacts is examined, and a simple formula is derived that yields an order of magnitude estimate of the total damage from a pollutant source. A software implementation of the impact pathway methodology is presented. As an illustration we show results for the impacts of particulates from a coal fired power plant.

Keywords

Environmental modeling, environmental impacts, impact pathway methodology, fuel cycle analysis, dispersion modeling

1. Introduction

Rational management of the environment requires an assessment of the damage caused by pollution. The logically correct way to analyze environmental impacts is the impact pathway methodology whose principal steps are the following (see Fig.1):

- characterization of the relevant technologies and the environmental burdens they impose (e. g. kg/s of particulates emitted by the plant);
- calculation of increased pollutant concentration in all affected regions (e. g. µg/m^3 of particulates, using models of atmospheric dispersion and chemistry);
- calculation of physical impacts (e. g. number of cases of asthma due to these particulates, using a dose-response function);
- in some cases a fourth step may be called for: the economic valuation of these impacts (e. g. multiplication by the cost of a case of asthma).

Formally this can be represented as an equation for the incremental damage D of a particular type (e.g. asthma) due to an incremental quantity Q of a pollutant emitted by a source

$$D = \sum_i f_{dr,i}(f_{disp \to i}(Q)) \quad , \tag{1}$$

where

$f_{disp \to i}(Q) = c$ = increase in pollutant concentration for receptor i, and

$f_{dr,i}(c)$ = dose-response function for receptor i;

The numbers are summed over all receptors, for this particular damage type, that one wants to include in the analysis. The notation allows the possibility that the impact may be different for different individual receptors. This equation expresses the damage in functional form, hence this methodology is also known under the name damage function approach. In Sections 2 to 6 we take a closer look at the major steps of the methodology.

The receptors of concern and the evaluation of the damage depend on the circumstances. One can distinguish three kinds of situation:

1. episodic values (typically for litigation after pollution episodes);
2. peak values (typically to obtain permit for a new plant, by showing that impacts are below a damage threshold or regulatory limit);
3. expectation values (typically for policy applications such as setting of regulations, by showing that average impacts are acceptable).

For the first two the summation will typically be over a limited set of receptors, for instance the residents in a town. For the third application one will usually want to know the total damage, and the sum should cover all receptors that make a significant contribution to the total.

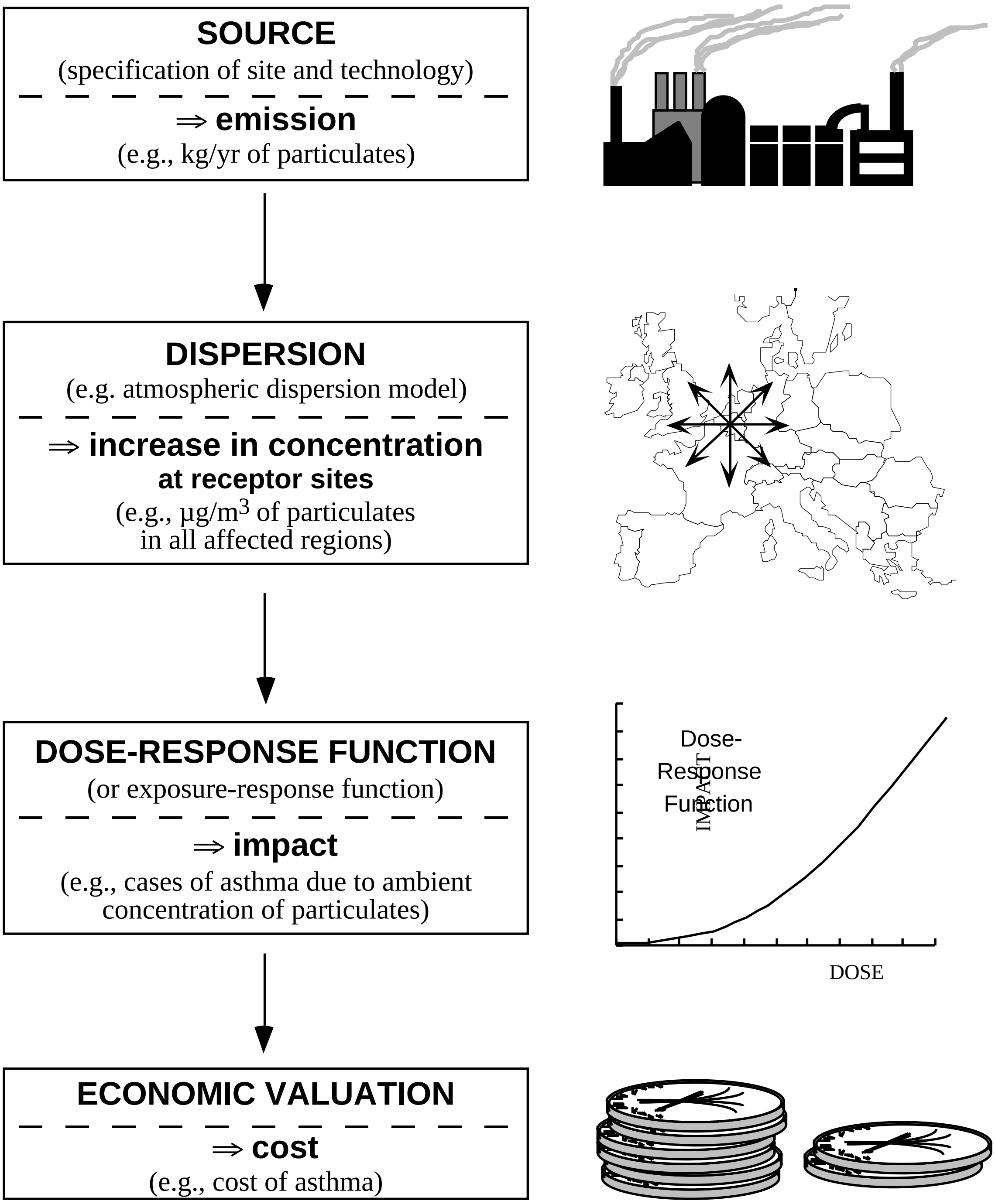

Fig.1. The main steps of the impact pathways methodology.
Impacts and costs are summed over the receptors of concern (e.g., total population).

2. The Source Term

The first step of the impact pathway analysis is relatively straightforward. One identifies the site and circumstances of a pollution source, e. g. the tons of NO per kWh_e emitted by a particular power plant. For the major air pollutants (CO_2, CO, NO, SO_2, volatile organic compounds, particulate matter) the emission rates for a given technology are quite well known. For the example of power plants the rate of CO_2 emission is especially well determined. Emissions of CO, NO, SO_2, volatile organic compounds and particulate matter are somewhat less certain, and they can vary with operating conditions. NO emissions, for instance, are likely to increase above the manufacturer's specifications if a selective catalytic reduction unit is not well maintained. There are different grades of oil and coal, and their sulfur content can differ by an order of magnitude; obviously the emissions of SO_2 depend on the quality of the fuel that will be used. Usually there are strict regulations that enforce an upper limit on the emissions; due to cost constraints power plants are unlikely to operate significantly below these limits.

The situation is less clear with regard to trace pollutants such as lead and mercury, since their content in different grades of coal can vary by much more than an order of magnitude. Furthermore, some of these pollutants are emitted in such small concentrations that their measurement is difficult. The dirtier the fuel, the greater the uncertainty of emissions. Especially with waste incineration the prime concern arises from trace pollutants that are emitted into the air.

Probably the most uncertain emissions are emissions from the disposal and storage of wastes, because they depend on events in the future. Solid waste from coal fired boilers could be dumped into a simple hole in the ground or it could be placed into an engineered landfill with water tight liners; the possible impacts will be totally different. There may or may not be a breach of containment, depending on the quality of construction and management and on natural events such as floods or earthquakes. The main risk from a landfill is the leaching of toxic minerals into ground water; in theory, at least, such risk can be kept negligible by proper construction and management.

3. Dispersion Modeling

3.1. Transport modes

Pollutants can be emitted to air, water or soil. The majority of pollutants are first emitted into the air, even if they later pass into the water or the soil. Therefore most of this section focuses on atmospheric dispersion. Transport in the soil is difficult to model because it can involve complex processes that depend on the physical and chemical properties of the soil at each site. Furthermore, for new installations such as new landfills, the emissions into the soil are not known in advance; they depend on the integrity of the containment structure over the indefinite future.

Transport by surface water, i.e. rivers, lakes and the sea, is relatively simple to analyze if fine geographical resolution is not required. Thus one can divide these bodies of water into a reasonably small number of compartments that are treated as uniformly mixed. For example, a river may be divided into 10 sections. A differential equation with empirical coefficients relates the concentration in a section under consideration to the concentration in the section immediately upstream and to the emission into this section. Sedimentation, removal and decay processes are included.

Likewise for the dispersion into marine waters, one uses a compartment model where each compartment communicates with one or several neighbors, and the volumes and flow rates are known. For instance, in a model used for the analysis of nuclear power plants [EC 1995c], the European seas have been divided into 34 compartments.

For dispersion in the atmosphere, in general both physical and chemical processes need to be considered [Seinfeld 1986, Zannetti 1990]. Some pollutants, e.g. CO_2, CH_4 and ^{133}Xe, are sufficiently inert chemically that only the physical transport need to be analyzed. Some are moderately reactive and their chemical transformation needs to be taken into account. SO_2, for instance, leads to the formation of SO_3, H_2SO_4 as well as sulfates (the latter from the interaction with NH_3 emitted by, amongst others, agricultural activities); this can have significant implications for the impact analysis on a regional and global scale [1]. Ozone is a secondary pollutant, formed by the combination of NO, VOC and light, and the chemistry is extremely complex.

Even though the modeling of the physical transport of pollutants is difficult, it is far simpler than weather modeling. The reason is that (except in the immediate vicinity of the source) pollutants can be considered a small admixture, passively transported by the currents of the surrounding medium. Such transport is linear: the increase in concentration at a receptor site is proportional to the emission (the only exception arises from secondary pollutants such as ozone whose formation depends on other variables, coupled through nonlinear phenomena).

Furthermore, for most policy applications one needs only expectation values of environmental impacts. While it is well known that chaotic phenomena in the atmosphere render the prediction of the weather impossible beyond a short time, this does not prevent the prediction of expectation values. The climate is much more certain than the weather. For expectation values of air pollution damage it suffices to know the average motion of the surrounding medium from past observations, by contrast to weather modeling where that very motion needs to be predicted in detail.

[1] For example, ammonia sulfate aerosols can reduce the impact of global warming [Charlson and Wigley 1994].

3.2. Transport in Air

A simple model for atmospheric dispersion is the gaussian plume, illustrated in Fig.2. According to this model the concentration of a pollutant is described by the product of two gaussian distributions, one for the spread in the vertical direction and one for the spread in the horizontal direction perpendicular to the prevailing wind direction. The plume width parameters are based on empirical correlations and take into account the relevant meteorological conditions.

The gaussian plume is considered adequate for the short range, up to tens of km from the source, even for episodic events [Zannetti 1990]. The use of this model at distances beyond 100 km is generally not recommended, although it is acceptable for the prediction of the average values if correction terms are included for reflection at the surface and at the planetary boundary layer of the earth, and if the depletion mechanisms (deposition, chemical transformation, radioactive decay) are correctly accounted for. As an example of dispersion software based on a gaussian plume one can cite the ISC model of the USEPA [Wackter and Foster 1987].

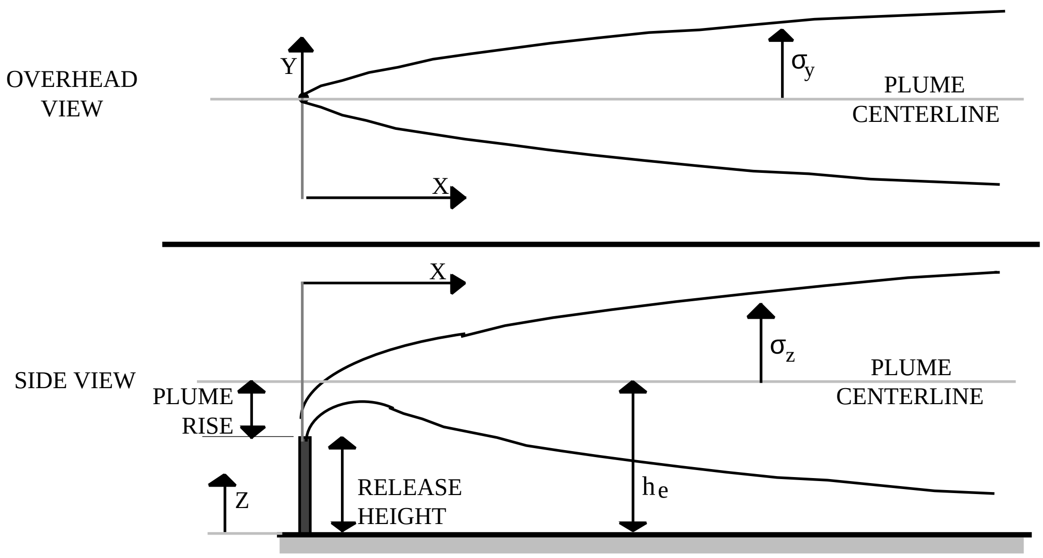

Fig.2. Gaussian plume model for atmospheric dispersion.

For regional modeling most analysts prefer to rely on more detailed computer simulations, for example the Harwell Trajectory Model [EC 1995b] or the EMEP (European Monitoring and Evaluation Programme) model of the Norwegian Meteorological Service [Barrett 1992, Sandnes 1993, Iversen 1993, Eliassen and Saltbones 1983]. The latter model is used for the official allocation of acid rain budgets among the countries of Europe.

A crucial question concerns the geographic range over which the analysis needs to be extended in order to capture most of the impacts. This involves a balance between the rates of emission, of dispersion, and of removal of a pollutant. A look at the results of long range transport models for SO_2 and NO, for instance those calculated by EMEP [Sandnes 1993], shows that these pollutants are transported over hundreds, even thousands of km. This is illustrated in Fig.3 using the EMEP data for a source at Nantes, assuming uniform receptor density (regardless of land or sea) and a linear dose-response function. The range of the analysis must be extended to at least one thousand km if one wants to capture 80 to 90% of the total impact.

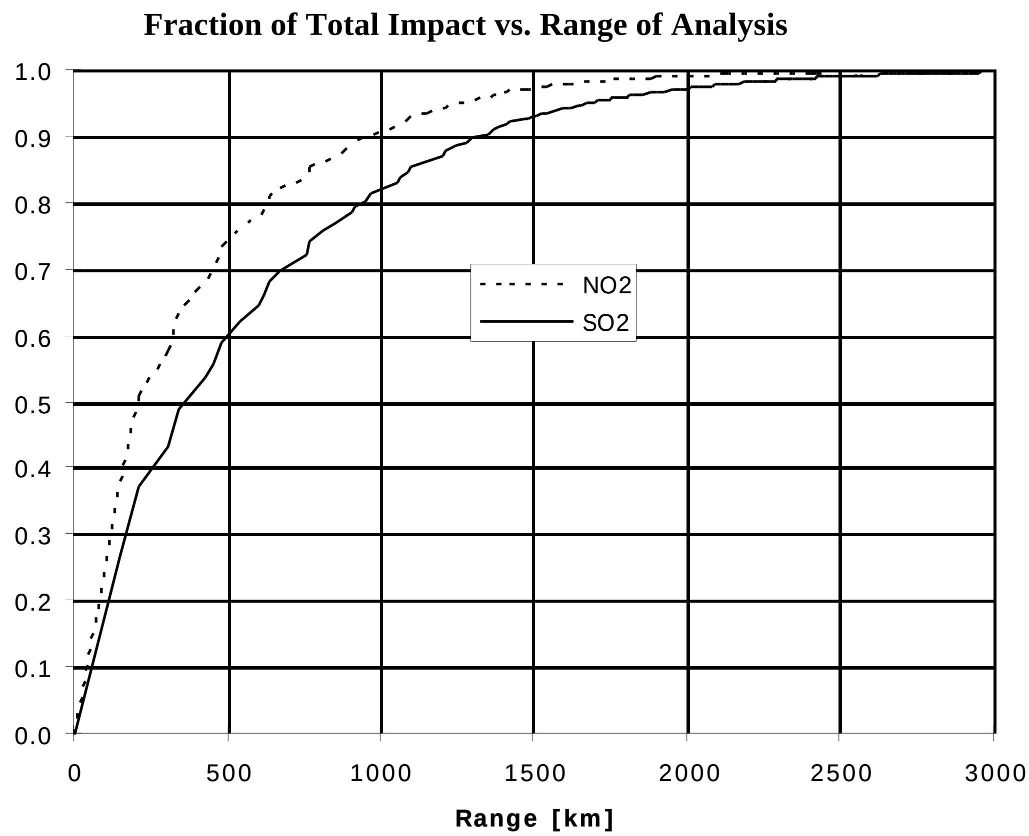

Fig.3. Fraction of total impact versus range of analysis, for uniform receptor density and linear dose-response function, based on EMEP data [Barrett 1994]. Wiggles are due to discretization.

3.3. Secondary Pollutants

Many pollutants are transformed into secondary pollutants by chemical reactions in the atmosphere. For example, the reactions shown in Fig.4 create acid rain (wet deposition of H_2SO_4) and ammonium sulfate particulates from SO_2.

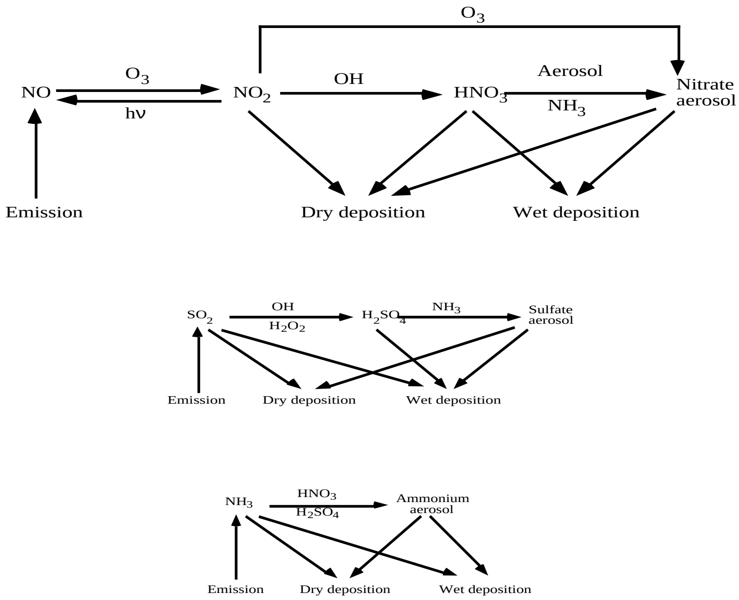

Fig.4. Chemical reactions included in Harwell Trajectory Model. From EC [1995].

Another important secondary pollutant is ozone. It is formed when several chemical reactions take place in sequence. The only reaction that forms ozone directly is

$$O + O_2 + M \rightarrow O_3 + M$$

where M is a molecule such as N_2 or O_2 whose participation is necessary to conserve energy and momentum. The oxygen atom involved in the formation of ozone is derived from photolysis of NO_2 under the action of sunlight (indicated by hν)

$$NO_2 + h\nu \rightarrow O + NO \quad .$$

The presence of VOCs (volatile organic compounds) is necessary to prevent the ozone formed from being immediately consumed by NO to produce NO_2 in the following reaction

$$NO + O_3 \rightarrow O_2 + NO_2 \quad .$$

VOCs enable the transformation of NO into NO_2 without consuming ozone. Finally, note also that NO_2 plays a double role since, while being necessary to form ozone, it consumes the radicals needed by VOCs to transform NO into NO_2.

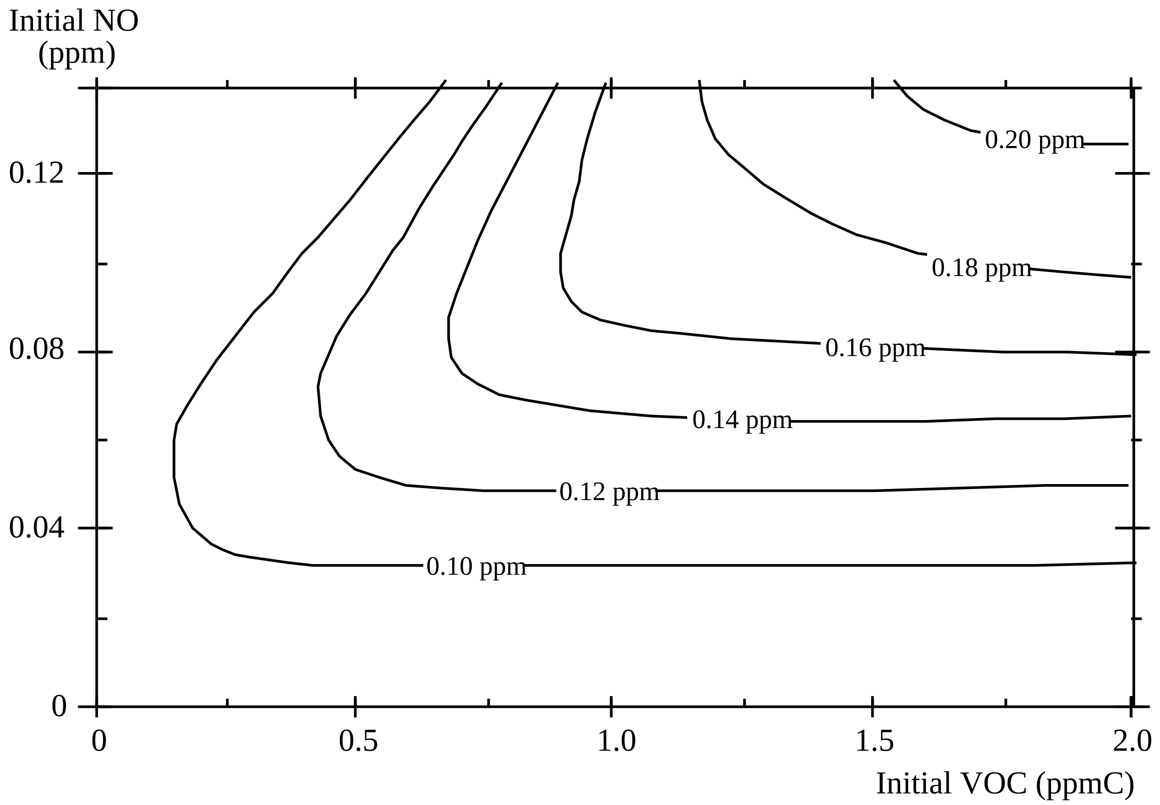

Fig.5. Isopleth plot for the maximum ozone concentration reached during a fixed length of time as a function of initial NO and VOC (volatile organic compounds) concentrations. Details of such a plot depend on site and on weather. From EPRI [1992].

In fact an equilibrium is created between these reactions. The concentration of ozone therefore is very dependent on changes in the concentrations of other products and, due to the complexity of the phenomena, it is observed for example that if VOCs are low (as in the case of an electricity power plant plume), the increase in NO may reduce O_3. Fig.5 shows the influence of the concentrations of nitrogen oxides and volatile organic compounds on the concentration of ozone. In particular we observe the phenomenon mentioned above: the consequence of an increase in NO on atmospheric ozone depends on the concentration of the organic compounds. The ozone content is also strongly dependent on the $[NO_2]$: [NO] ratio. If this ratio is low, the $[O_3]$ content will remain low.

4. Dose-response functions

4.1. The form of the dose-response function

The dose-response function

$$Y = f_{dr}(X) \qquad (2)$$

relates the quantity X of a pollutant that affects a receptor (e.g. population) to the physical impact Y on this receptor (e.g. incremental number of deaths). For impact assessment it is appropriate to define the response as the incremental effect due to the dose. Thus the dose-response function starts at the origin, and in most cases it increases monotonically with dose X, as sketched schematically in Fig.6. At very high doses the function may level off in S-shaped fashion, implying saturation.

In the narrow sense of the term, X should be the dose actually absorbed by a receptor. But often one uses, as we do in the present paper, the term dose-response function in the sense of exposure-response function where X represents the concentration of a pollutant in the ambient air; in that case $f_{dr}(X)$ accounts implicitly for the absorption of the pollutant from the air into the body. Dose-response functions for the classical air pollutants (NO_X, SO_X, O_3, and particulates) are typically of that kind. One can even define aggregated dose-response functions that include more complicated pathways, for instance dioxins passing through the food chain, if one interprets the dose-response function to include the aggregated effects of the pathways from a point at the earth's surface to all final receptors.

Dose-response functions are determined from epidemiological studies or from laboratory studies. Since the latter are mostly limited to animals, the extrapolation to humans introduces large uncertainties. Another major difficulty is that one needs relatively high doses in order to obtain observable nonzero responses in a sample of realistic size; such doses are usually far in excess of the levels one is concerned with in environmental impact studies. Thus there is a serious problem of how to extrapolate from the observed data towards low doses. Fig.6 indicates several possibilities. The simplest is the linear model, i.e. a straight line from the origin through the observed data point(s). Cancer from radioactivity is an example. Linearity also seems to be observed for mortality from fine particulates [Dockery et al. 1993, Dockery and Pope 1994, Lipfert 1994].

Another possibility is a straight line down to some threshold, and zero effect below that threshold. Thresholds occur when an organism has a natural repair mechanism that can prevent or counteract damage up to a certain limit. Many dose-response functions for non cancer toxicity are of this type.

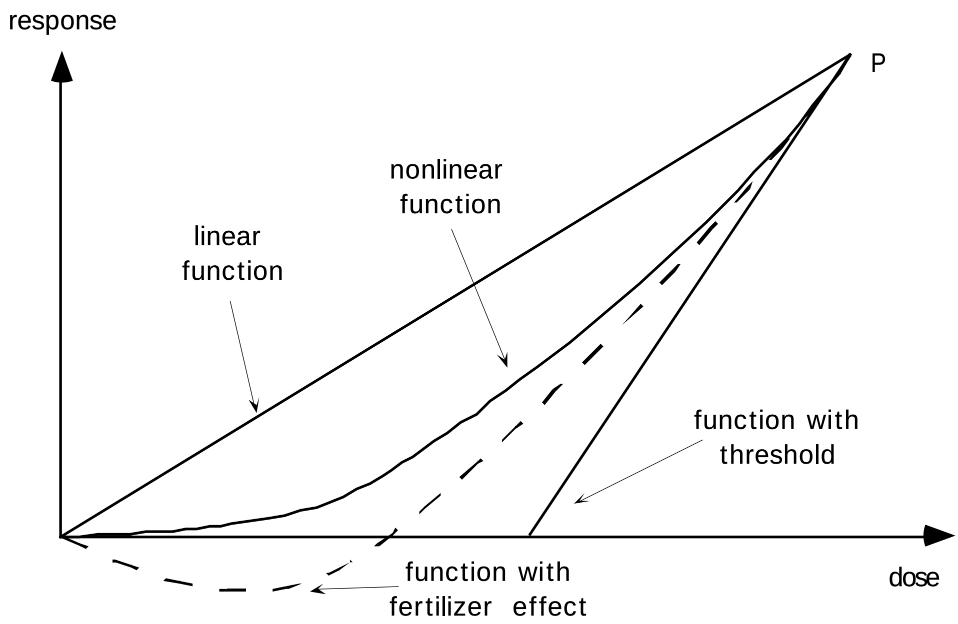

Fig.6. Possible behavior of dose-response functions at low doses:
the four functions shown have the same value at P. For the function with threshold the discontinuity in slope at the threshold is a simplification; in reality there is a smooth transition.

There is even the possibility of a "fertilizer effect" at low doses, as indicated by the dashed line in Fig.6. This can be observed, for example, in the dose-response functions for the impact of NO_X and SO_X on crops: a low dose of these pollutants can increase the crop yield, in other words the damage is negative. Such a fertilizer effect can occur with pollutants that provide trace elements needed by an organism; the effect depends on local conditions, in particular the overall balance of nutrients.

If nothing is known about a threshold and if a fertilizer effect can be ruled out, the dose-response function could be anywhere between zero and the straight line through the origin, for instance the curved solid line shown in Fig.6. A priori there is no general rule about the extrapolation to low doses, other than there being no known pollutants with a dose-response function above the straight line. There is even a case where the same substance causes different cancers according to different dose-response functions, one with and one without threshold. This was established in an experiment (sometimes referred to as the megamouse experiment) in which some 24000 mice were exposed to the carcinogen 2-acetyl-amino-fluorene at several different dose levels [Frith, Littlefield and Umholtz 1981]. The response for liver tumor is linear whereas the one for bladder tumor has a threshold.

4.2. Implications of the Threshold for the Analysis

The form of the dose-response function, Fig.6, has implications for the way an impact analysis is to be carried out. It is appropriate to distinguish two extreme cases. These two extreme cases are of great practical importance, and the corresponding analysis is relatively simple.

One extreme occurs when the dose-response function is a straight line through the origin (no threshold). In that case any incremental pollution causes an impact, and the range of the analysis needs to be extended over hundreds or thousands of km if most of the impact is to be included. This situation also pertains in the presence of a threshold, if the background concentration is everywhere above this threshold. For some air pollutants, e.g. particulates, the background in most industrialized countries is above the level where effects are known to occur [Dockery et al. 1993]. Thus the question of the precise form of the dose-response function at extremely low doses is irrelevant for these pollutants: whatever the threshold, if there is one, it is below the background concentrations of interest.

The other extreme occurs if the dose-response function has a threshold that is above the background concentration of the pollutant and if the pollution added by the source does not push the concentration above the threshold. In that case there is no impact. The analysis is simple: it suffices to verify that the resulting concentrations remain below the threshold. A short range (< 50 km) dispersion model is adequate for this purpose because the peak concentration increase certainly occurs within that region.

5. Site Dependence of Marginal Impacts

It will be convenient to write the damage as an integral over land area by introducing $\rho(\mathbf{x})$, the density of receptors at point $\mathbf{x} = (x,y)$,

$$D = \int dx \int dy \quad \rho(\mathbf{x})\, f_{dr}(\mathbf{x},c(\mathbf{x})) \tag{3}$$

where $c(\mathbf{x}) = f_{disp\rightarrow\mathbf{x}}(Q)$ is the concentration increase at $\mathbf{x}$ due to emission Q. From here on we limit ourselves to the important case where the dose-response function $f_{dr}(\mathbf{x},c(\mathbf{x}))$ can be approximated by

$$f_{dr}(\mathbf{x},c(\mathbf{x})) = d(\mathbf{x})\, c(\mathbf{x}) \qquad \text{where} \qquad d(\mathbf{x}) = \frac{df_{dr}(\mathbf{x},c(\mathbf{x}))}{dc} \tag{4}$$

is the slope of the dose-response function. With that assumption one can write Eq.3 for the damage in the form

$$D = \int dx \int dy \quad \rho(\mathbf{x})\, d(\mathbf{x})\, c(\mathbf{x}) \qquad . \tag{5}$$

This is obviously exact for any pollutant whose dose-response function is linear, or a straight line with a threshold that is everywhere below the background. It is also valid, regardless of dose-response function, for the evaluation of any marginal impacts, i.e. impacts from small pollutant increments because in that case one can linearize the dose-response function. Since c(**x**) is linear in the emission, it follows that Eq.5, and the remainder of this paper, are equally applicable to steady state situations and to emissions that vary with time.

It is instructive to relate the concentration c(**x**) to the removal rate of the pollutant. There are essentially three mechanisms by which an air pollutant can disappear from the atmosphere [Seinfeld 1986]:

1) dry deposition (uptake at the earth's surface by soil, water or vegetation)
2) wet deposition (absorption into droplets followed by droplet removal by precipitation),
3) decay or transformation (e.g. decay of radionuclides, or chemical transformation of SO_2 to $(NH_4)_2SO_4$).

When evaluating the damage of the original pollutant, this pollutant is no longer counted in the equation once it has been transformed; rather from that point on a different dose-response function comes into play for the secondary pollutant. Here we only consider primary pollutants. Our results can readily be generalized to secondary pollutants, as shown by Curtiss and Rabl [1995b].

The dry deposition rate is proportional to the concentration c(**x**) at the earth's surface, and it is customarily written in the form

$$F_{dry}(\mathbf{x}) = v_{dry}\, c(\mathbf{x}) \tag{6}$$

where
$F_{dry}(\mathbf{x})$ = deposition flux [in kg/m^2·s], and
v_{dry} = dry deposition velocity [m/s].
Wet deposition and decay or transformation can likewise be characterized in terms of fluxes $F_{wet}(\mathbf{x})$ and $F_{trans}(\mathbf{x})$, defined as the rate at which the pollutant is removed by these mechanisms per m^2 and per s. Even though in general these fluxes are not proportional to the surface concentration but rather to the average concentration in the air column above x, we can write the total removal flux

$$F(\mathbf{x}) = F_{dry}(\mathbf{x}) + F_{wet}(\mathbf{x}) + F_{trans}(\mathbf{x}) \tag{7}$$

in terms of the surface concentration c(**x**) as

$$F(\mathbf{x}) = k(\mathbf{x})\; c(\mathbf{x}) \tag{8}$$

if we allow the proportionality constant k(**x**) to vary with **x**. The units of k are m/s, and it could be called "removal velocity". Using F(**x**) and k(**x**) we can write the damage in the form

$$D = \int dx \int dy \quad \rho(\mathbf{x})\, d(\mathbf{x})\, F(\mathbf{x})/k(\mathbf{x}) \tag{9}$$

This equation is exact if we interpret Eq.8 as the definition of k(**x**).

Let us now consider a situation where the quantities in Eq.9 other than the removal flux F(**x**) are independent of site, with uniform receptor density $\rho(\mathbf{x}) = \rho_{uni}$, uniform dose-response function slope $d(\mathbf{x}) = d_{uni}$, and uniform removal velocity $k(\mathbf{x}) = k_{uni}$. In that case the integral in Eq.9 would be simply

$$D = D_{uni} = d_{uni}\, \rho_{uni}\, Q/k_{uni} \tag{10}$$

because the surface integral of the removal flux equals the emission

$$Q = \int dx \int dy \quad F(\mathbf{x}) \tag{11}$$

by conservation of matter.

Even though the assumption $k(\mathbf{x}) = k_{uni}$ may not appear very realistic, especially near a point source, the sensitivity to deviations from uniformity turns out to be surprisingly small, as we illustrate below in Fig.7. The reason is that for typical values of atmospheric dispersion parameters the total impact is dominated by regions sufficiently far from the source that the pollutant can be considered to be vertically well mixed in the planetary boundary layer, at least as far as expectation values are concerned.

Thus the simple Eq.10 can be a useful first estimate, good to an order of magnitude or better. Details of atmospheric dispersion do not matter very much. It is intuitively plausible that the damage is proportional to the slope d of the dose-response function, to the density ρ of receptors and to the emission rate Q. Furthermore, it is inversely proportional to the removal velocity k. If there were no removal mechanism, the pollutant concentration would increase without limit and the damage would be infinite.

To verify the relevance of Eq.10 we compare it with real site-dependent results, calculated with the PATHWAYS software package [Curtiss and Rabl 1995a] described in Section 8 below, which carries out an accurate numerical integration of atmospheric dispersion results over geographic data for population and other receptors; all of Europe is included in the analysis. To add substance to the results, we consider a specific impact: the increase in mortality due to particulate matter emitted by coal fired power plants. The dose-response function [based on Schwartz 1993, as cited in EC1995] is linear and can be written in the form

$$\Delta \text{deaths/yr} = 10.4 \times \Delta\, PM_{10} \text{ concentration [in g/m}^3\text{]} \quad , \tag{12}$$

where PM_{10} designates the concentration of particulates with diameter below 10µm.

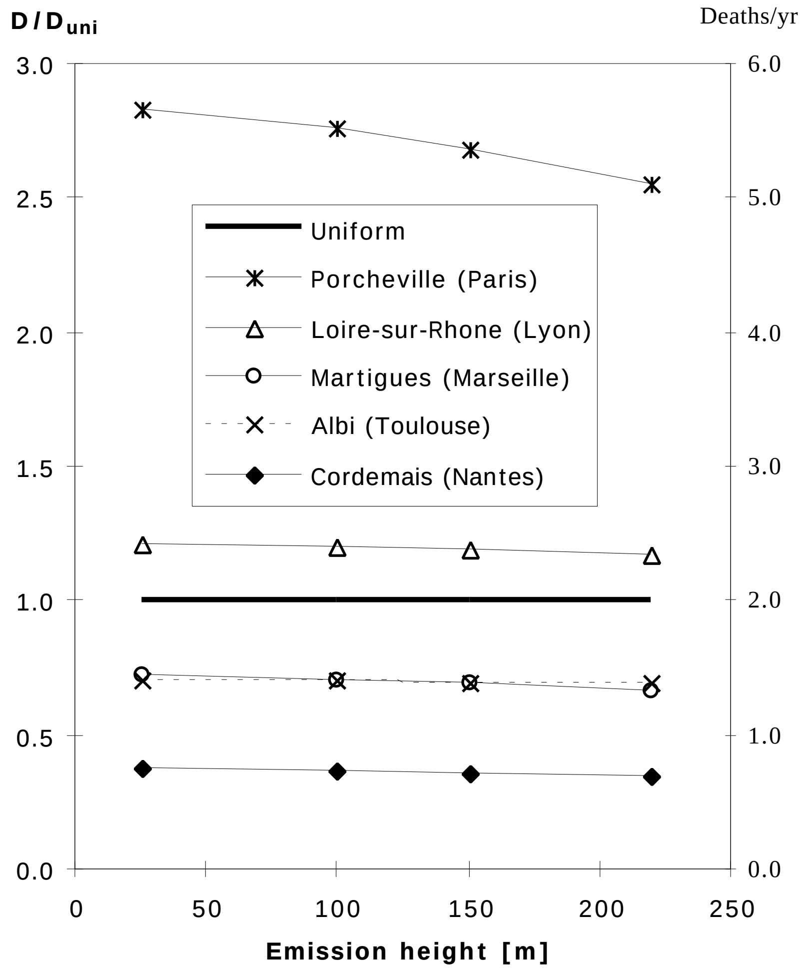

Fig.7. An example of dependence on site and on effective height of source (including plume rise): damage D from particulate emissions with linear dose-response function, for five sites in France, in units of D_{uni} for uniform world model (Eq.10). The scale on the right indicates mortality from a coal fired power plant with particulate emission 357 ton/yr (for electricity production of 2.1 × 10^9 kWh/yr).

We consider a coal fired power plant with an annual electricity production of 2.1 × 10^9 kWh/yr and a particulate emission rate equal to 0.17 g/kWh, hence an annual particulate output of Q = 357 ton/yr = 11.32 g/s [Curtiss et al. 1995b]. For the atmospheric dispersion we take a curve fit to the EMEP data for a grid cell in the center of France, for which a curve fit yields a range $1/\beta$ = 540 km, determined as the decay distance in an exponential curve fit. The parameters of the uniform world model should be averaged over the range. One can show [Curtiss and Rabl 1995b] that the range is related to the mean wind speed v, the mixing layer height H and the removal velocity k via

$$\text{range} = \frac{1}{\beta} = \frac{v\,H}{k} \quad . \tag{13}$$

Together with the mean wind speed of v = 4.2 m/s and a mixing layer height H = 800 m this implies a removal velocity $k = v\,H\,\beta$ = 0.0062 m/s. Inserting Q = 11.32 g/s, k_{uni} = 0.0062 m/s, d_{uni} = 10.4 (deaths/yr)/(g/m^3) and $\rho_{uni} = 1.05 \times 10^{-4}$ m^{-2} (for France) into Eq.10 we obtain

$$D_{uni} = \frac{10.4(deaths/yr)/(g/m^3) \times 1.05 \times 10^{-4} m^{-2} \times 11.32 g/s}{0.0062 m/s} = 2.0 \text{ deaths/yr.}$$

This number is shown as the thick solid line in Fig.7 where the damage D, in units of D_{uni}, is plotted versus effective emission height (i.e. including the plume rise). The points, connected by lines, show the damage D for five specific sites. The actual number of deaths per year is shown on the right hand scale. We have chosen these five sites because there are in fact fossil fuel power plants at these sites (the nearest big city, 25 to 50 km away, is indicated in parentheses), and because these sites appear to span the range of typical conditions. Although the real emissions of the plants at different sites are different, here we have assumed the emissions of the Cordemais plant at all sites to bring out the point of the comparison. The impact is about 2.6 times larger than D_{uni} for the site near Paris and about 0.35 times D_{uni} for Cordemais, a rural site on the Atlantic Ocean. We also see that there is little variation with stack height.

How representative are these results? Emission height dependence and deviations from uniformity are most pronounced when the receptors are concentrated near the source. A source near Paris seems like a fairly extreme example because France is highly centralized, with 19% of its population in Greater Paris. However, there are more extreme cases. In terms of geographic extent and population, France (58 million people, density 105 per km^2) and the North East census region (50 million people, density 118 per km^2) of the USA are more or less comparable. The New York metropolitan area comprises 36% of the population of the North East census region. Furthermore, the ratio local/regional impact is very roughly doubled for a source in New York, close to the ocean, compared to one in Paris, which is inland. Thus it is not surprising that the New York State study of Rowe et al [1995] found variations with emission height and emission site significantly stronger than Fig.7. Local effects could also dominate the Los Angeles Basin, an urban area surrounded mostly by ocean and desert.

Furthermore, the above results have been derived in the flat terrain approximation. Not taken into account is the canyon effect: the increase in concentration from sources in streets or valleys. That can be especially significant for the impacts of air pollution from cars.

The uniform world model can also be used for the damage due to secondary pollutants such as sulfates and ozone. Now there is an additional factor: the rate $k_{1\text{-}2}$ at which the primary pollutant is transformed into the secondary; this rate is defined analogous to Eq.8 in units of m/s. In particular for the uniform world model we obtain as generalization of Eq.10

$$D_{2,uni} = d_{2,uni}\ \rho_{2,uni}\ k_{1\text{-}2,uni}\ Q/k_{2,uni} \tag{14}$$

where the subscript 2 indicates that concentration, dose-response function and damage refer to the secondary pollutant. We have not yet performed an analogous test of Eq.14, but it is plausible that site dependence is less pronounced for secondary pollutants than for primary pollutants. Site dependence arises from correlated variations of the factors in the integral of Eq.5, in particular receptor density and concentration. Secondary pollutants are created far from the source and in diffuse manner; thus the variation of their concentration is much more gradual than the variation of a primary pollutant near its source. Eq.14 should hold even for ozone whose nonlinear creation mechanism implies that the rate $k_{1\text{-}2}$ can have different signs at different places; such sign change does not invalidate the derivation of Eq.14. More problematic is the fact that ozone formation depends on local concentrations of volatile organics and the NO/NO_2 ratio, which in turn may be correlated with population density. One needs a test with real data.

6. Economic Valuation

For the economic valuation of pollution damage a general consensus has evolved in democratic countries that one should respect individual preferences, i.e. use the willingness-to-pay for avoiding a loss rather than the direct cost. For example, for an illness one should count not only the treatment cost but also pain and suffering, as expressed by the willingness-to-pay to avoid the illness. For goods that are traded in a market, for instance agricultural output, the valuation is straightforward: there is a market price. But much pollution damage involves nonmarket goods such as health or visibility. Their valuation involves indirect valuation methods that are difficult and costly to apply, such as contingent valuation, hedonic price method, and travel cost method [see e.g., Tietenberg 1992, Braden and Kolstad 1991].

The most problematic and controversial good is loss of human life. The very idea of its monetization stirs opposition from people who perceive a value judgment. The purpose of monetization is not "... and how much for your grandmother?", but finding a consistent reference value for an equitable and efficient allocation of public expenditures for the protection of human safety. It is often referred to as Value of Statistical Life (VSL). There are three main methods for determining a VSL based on individual preferences:

- wage - risk studies (how much extra compensation do workers demand in jobs with high risk?);
- consumer market studies (e.g. how much are consumers willing to pay extra for safety measures such as air bags in cars?);
- contingent valuation (asking people directly how much they would be willing to pay if given the opportunity).

As it turns out, the value of life is the single most important parameter for the cost of air pollution, because the cost of mortality dominates if one takes a value in the range of 1 to 5 million $, for which there appears to be a consensus. For example, the recent externality studies of the European Union [EC 1995] used a value of 2.6 million ECU ($ 3.2 million). The analogous studies in the USA [ORNL/RFF 1994, and Rowe et al 1995] have taken similar values.

For the damage categories of mortality, morbidity and agricultural production, the cost per unit of physical damage is essentially constant throughout a region, and thus the last step of the analysis is a simple multiplication. That is true even in the case of agricultural production where many countries apply subsidies that introduce local distortions: a change in agricultural output is compensated by import or export on the world market, and therefore the net damage cost is determined by the world market price.

More detail on the economic valuation can be found in the references [Ontario Hydro 1993, ORNL/RFF 1994, EC 1995c, and Rowe et al 1995].

7. Analysis of Uncertainties

By contrast to the relatively small uncertainties and normal (gaussian) frequency distributions typically encountered in science and engineering, the uncertainties in impact analysis are so large that it would be inappropriate to use error intervals that are additively symmetric about the mean. Instead one should specify multiplicative intervals, in other words, intervals that are additive on a logarithmic scale. The frequency distributions are not symmetric, with implications that may appear counterintuitive.

It is helpful to think in terms of lognormal distributions because they are frequently encountered in impact analysis, analogous to the normal distributions so familiar in the more exact sciences [Morgan and Henrion 1990]. A variable x has a lognormal distribution if the variable ln(x) has a normal distribution; in other words, it is normal on a logarithmic scale. Even if the distributions of the individual errors, e.g. for the coefficient of a dose-response function, are not lognormal, the central limit theorem of statistics implies that the distribution of the damage is likely to approach a lognormal because the calculation is multiplicative [Slob 1994, Rabl 1995].

Analogous to the ordinary normal (also known as gaussian) distribution which is characterized by two parameters, the mean and the standard deviation, the lognormal distribution can be characterized by the geometric mean and the geometric standard deviation σ_G. For this distribution the geometric mean is equal to the median: half of the distribution is above, the other half below the median. The geometric standard has a simple interpretation in terms of the 67% confidence interval (a familiar number because for gaussian distributions 67% of all values are within one standard deviation of the mean): for a lognormal distribution 67% of the values are within the interval $[1/\sigma_G, \sigma_G]$. Likewise 95% are within the interval $[(1/\sigma_G)^2, (\sigma_G)^2]$. Note, however, that these values are centered around the median rather than the mean; the lognormal distribution is not symmetric. For impacts of primary air pollutants with well relatively determined dose-response functions, e.g. mortality due to particulates, σ_G may be as small as 3 [Rabl 1995]. For other impacts, e.g. cancers due to dioxins, the uncertainties could be an order of magnitude or more.

It is appropriate to note that technical or scientific uncertainties (e.g. uncertainties of emitted quantities or of dose-response functions) are not the only ones. For long terms impacts, such as cancers caused by radioactive waste, one needs to make assumptions about scenarios for the future: what quantities of radionuclides will leak into the environment and how many people will be affected by them. For the estimation of damage costs, there is also the matter of policy/ethical choice, e.g. about discount rate and value of human life.

8. Software Implementation

8.1. Structure of Data and Computations

If one wants a reasonably accurate calculation of the total damage caused by a pollution source, one needs a large quantity of data and the calculational requirements are so massive as to rule out a manual approach. In recent years several software packages have been developed for this purpose. In particular, there are three programs that have been written for the studies of fuel cycle externalities and that appear comparable in scope and function:

- ECOSENSE [Krewitt et al 1995]
- EXMOD [Rowe et al 1995]
- PATHWAYS [Curtiss and Rabl 1995a].

In this section we address some of the problems that arise in the development of software for impact pathway analysis, and we offer solutions that we have found useful. Some of the following discussion is illustrated with examples from PATHWAYS.

The first version of PATHWAYS was written in Excel4.0, a choice based on the capabilities of the macro programming language and on the attractive interface of this spreadsheet.

However, problems with file access and crashes were encountered when transferring to different machines; also the calculations were slow. Therefore PATHWAYS2.0 was entirely rewritten in Pascal. It runs on a 386 or higher PC.

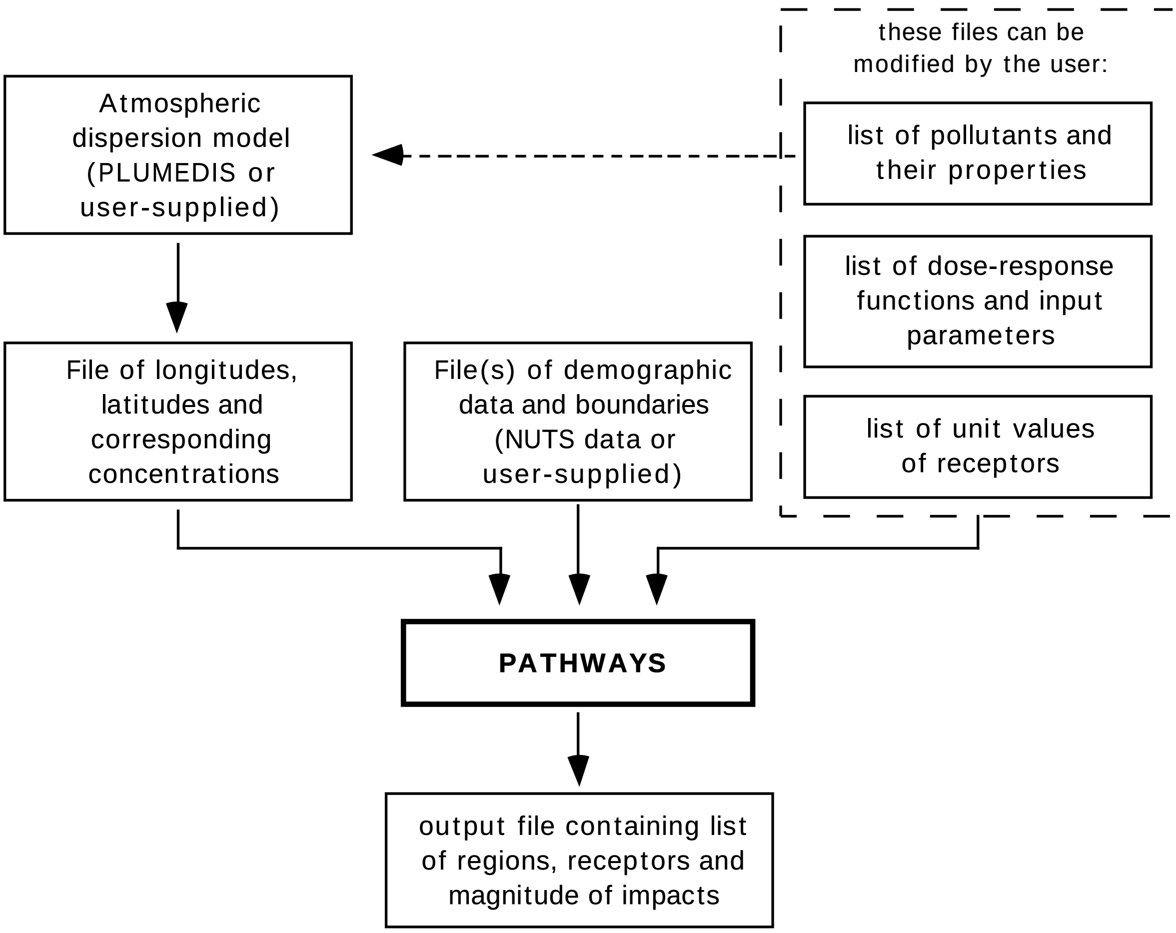

Fig.8. Flow of information in the PATHWAYS system

The PATHWAYS package consists of five files:

PATHWAYS.EXE the executable version of PATHWAYS

PLUMEDIS.EXE a gaussian plume atmospheric dispersion model (for which the user can substitute any other model)

EQUATION.LST an ASCII file containing the specifications of the dose/response functions

POLLUTAN.LST an ASCII file containing a list of pollutants and their properties

PRICES.LST an ASCII file containing a list of various receptors and their costs

In addition, there are *region files* and *configuration files*. The region files contain the geographic data with the receptor distribution. They are based on the EUROGRID [1990] and EUROSTAT [1992] data bases and cover the countries of the European Union; data for the rest of Europe have been added. The flow of information is diagrammed in Fig.8.

PATHWAYS file types

PATHWAYS uses several different kinds of files. As previously mentioned, the programs PATHWAYS.EXE and PLUMEDIS.EXE are executables and cannot be changed by the user. The format of the other files are discussed in this section.

Pollutant files

The pollution concentration data are given as values at a set of points whose coordinates can be specified arbitrarily. *Pollutant files* are simply lists of coordinates and the corresponding pollutant. There can be only one coordinate and concentration value per line, but the file may contain up to 1000 lines. The coordinates must be given in longitude and latitude and the concentrations must be in micrograms per cubic meter. An example of a pollutant file is given in Fig.9.

-1.84	43.8433333	0.00040521
-0.4	43.2133333	0.00029095
0.99333333	42.5633333	0.00023146
2.35333333	41.89	0.00019907
-0.96666667	44.9033333	0.00125568
0.5	44.2533333	0.00054421
1.91666667	43.58	0.0003257
3.29666667	42.8833333	0.0002244
-4.83666667	47.82	0.00106839
-2.83904223	47.2643531	0.0040091
-1.41748309	46.7915177	0.02068161
-0.05	45.9633333	0.00322342
1.44	45.2933333	0.00083987
2.88333333	44.5933333	0.00041522
4.28333333	43.8766667	0.0002138
5.63666667	43.1433333	0.00011544
-3.95333333	48.9433333	0.00058013
-2.19421461	48.1161303	0.00171566
-0.79134298	47.5835702	0.00424057

Fig.9. Example of a pollutant file (data shown are for France)

Region files

PATHWAYS assumes that receptor data are supplied in region files. Each region file consists of the coordinates of the region outline and of receptor data (e.g., population, tons/year of wheat, etc.) The receptor data are assumed uniform within this region. *Region files* contain the demographic, agricultural and physical data for a given geographic region. The first two lines of the region file are used to identify where the region is located. The list of receptors in this region starts on line 3 and continues for as many receptors as desired. Each line with a receptor should start with the receptor *keyword*, a text string up to 20 characters which is used to uniquely identify the receptor. The receptor keyword cannot contain any spaces. The keyword is followed by an amount (e.g., the number of people living in the region or the amount of corn produced each year) and the units (e.g., persons, kg, head). The rest of the line is available for any comments you wish to add.

```
FRANCE
Country of France
Cereals 54998300000 kg NUTS 1990
Wheat 33312500000 kg NUTS 1990
Barley 10019900000 kg NUTS 1990
.
.
Population 57206200 persons NUTS 1992 (all ages)
Children 11411900 persons NUTS 1991 (ages 0 through 14)
Adults 34528600 persons NUTS 1991 (ages 15 through 59)
Seniors 10952700 persons NUTS 1991 (greater than age 60)
start
1.623105 50.37521
1.623105 50.37521
.
.
1.585658 50.382202
1.623105 50.37521
end
start
9.399103 41.861397
9.407001 41.828789
.
.
9.397402 41.873486
9.399103 41.861397
end
```

Fig.10. Example of a region file

The physical boundary of the region consists of two columns of numbers, the longitude and latitude, which describe a closed polygon. There can be as many coordinates as needed.

The list of coordinates must start with the word “start” on a separate line and must end with the word “end” on a separate line. You can include as many sets of coordinates as desired provided that each set describes a closed polygon and begins and ends with the words “start” and “end”. This is useful if you have a region which consists of islands or separate shapes. An example of a region file is given in Fig.10.

An interesting alternative for the representation of cities is to use point data, superimposed on region files with uniform distribution.

EQUATION.LST

The dose-response functions are stored in the file EQUATION.LST. This file is an ASCII file and can be edited by the user directory in DOS, although it is strongly suggested that all changes to this file be made through the Edit Defaults feature in PATHWAYS. The first two lines in this file are for comments, and each successive line contains information about a single dose/response function. Up to 100 dose/response functions can be included in this file. The data, per line, are

1. text description of the function, up to 40 characters (can include spaces)
2. text description of the pollutant (must match a pollutant name from the POLLUTAN.LST file)
3. the coefficients A, B, C, D and E used in the function $\Delta Y = (\Delta C \;\; A + B)^{C} \;\; D + E$
   where ΔY is the change of the impact, ΔC is the change of the ambient pollutant concentration, and A, B, C, D and E are user-selected constants (in future versions more general equations will be permitted).
4. text description of the receptor (i.e., the receptor keyword)
5. text description of the cost category (must match a category from the PRICES.LST file)

An example of the EQUATION.LST file is given in Fig.11.

```
** Equation list
** format is Description (40 chars), pollutant, A, B, C, D, E, ReceptorName, PriceCategory
Phlegm                                NOx              1.00000 0.0 1.00000 0.0039 0.0 POPULATION Cough
Sore Throat                           NOx              1.00000 0.0 1.00000 0.017 0.0 POPULATION Cough
Eye Irritation                        NOx              1.00000 0.0 1.00000 0.013 0.0 POPULATION EyeIrritation
Death                                 SulphurDioxide   0.0100 0.0 1.00000 0.00100 0.0 POPULATION HumanLife
```

Fig.11. Example of the file EQUATION.LST

8.2. Interpolation of Data

The interpolation of data may appear a minor detail, but it is of ubiquitous importance for all the calculations of the impact pathway analysis. The need for interpolation arises from the different nature of different data types. There are

- point data (data for background concentrations of pollutants),
- line data (flow data for segments of rivers),
- area data (for receptors, e.g. population, crops or forests, typically given in terms of administrative regions such as towns, counties or départements),
- data calculated on regular grids (e.g. output of atmospheric dispersion models).

The EUROGRID [1990] data base, which is included as an option in PATHWAYS, is based on a regular longitude-latitude grid but has been derived from data for administrative regions. Examples of region files are shown in Fig.12. The NUTS data of EUROSTAT [1992] are also used, in particular for the outlines of the regions.

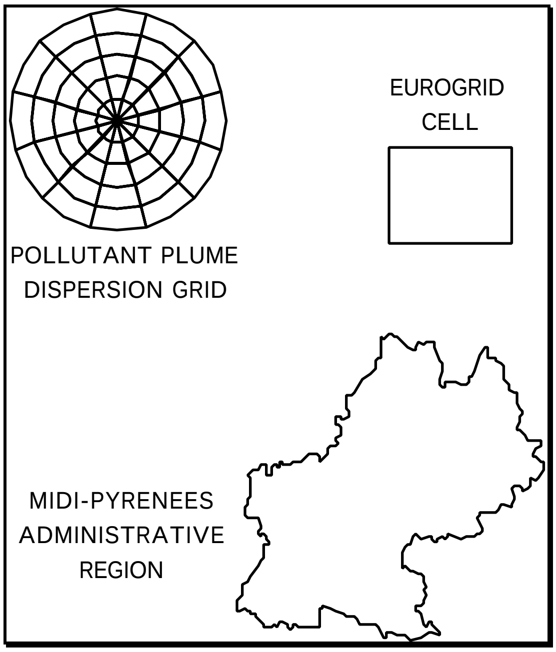

Fig.12. Examples of regions shapes
(all on same scale where EUROGRID block represents 10,000 square kilometers).

As a typical example of a calculation to be performed, consider the analysis of the effect of a pollutant on human health. The population data are given for an arbitrarily-shaped administrative region. The incremental pollution data are in a radial format (provided by an atmospheric dispersion model) while the background concentrations are given in terms of yet another spatial system, extrapolated from point data. How do you combine the various grid systems?

One would like to minimize both interpolation errors and computation time. Note that the accuracy is limited by the available data. In particular, without additional information the resolution of area data is no better than uniform density within the respective region.

Therefore we have chosen the following solution for PATHWAYS. First the background pollution data are assigned as average values to a suitably chosen grid of regions with sufficient resolution (in view of the insensitivity of most dose-response functions to background concentrations this is usually not a point of concern). Then one finds the intersection "cells" of all overlapping grids and regions, and performs the calculations for each resulting cell. In this way we guarantee that each cell has uniform data across its surface. The result for the cell is the product of area, receptor density and dose-response function. Finally one sums over all such cells.

Fig.13 illustrates this process. The region in question is shown at the bottom of the chart, along with the location of the pollutant source. The grid system in the middle represents the background concentration data (here a rectangular grid is assumed for the sake of illustration). The top grid represents the incremental concentrations. A sample intersection cell is shown as the small dark region.

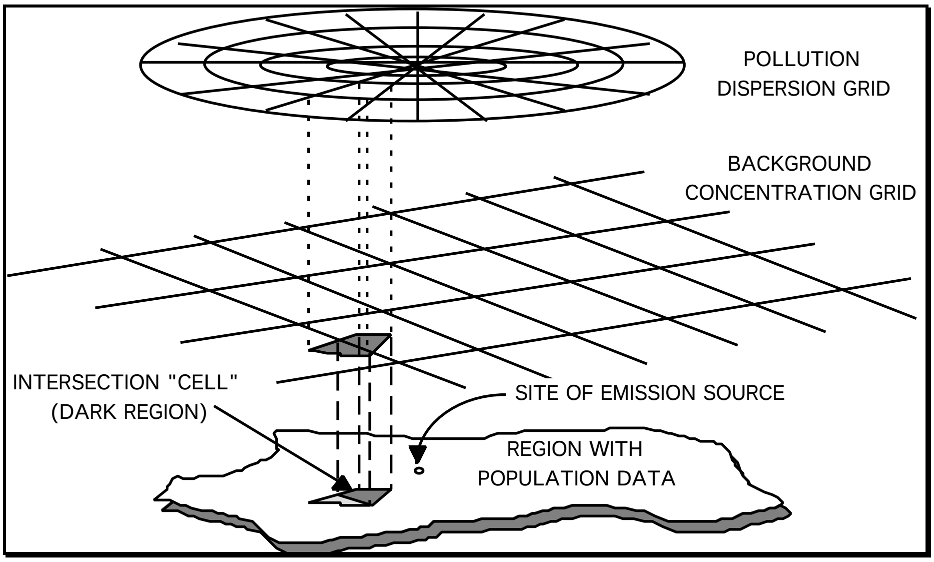

Fig.13. Example of overlapping grid systems

8.3. Simplification of Shapes

For complicated shapes, calculating the intersections can take a long time. Furthermore, the extremely detailed precision of many region files, while useful for creating interesting graphical output, is often not needed for the intersection files. For this reason, PATHWAYS incorporates a smoothing procedure into the routine which calculates the intersection areas, reducing the number of vertices of many regions by a significant amount while still retaining enough data to properly account for the shape of the region.

The process by which each shape is "smoothed" involves performing a running average of a sequence of coordinates. A series of "new" points are created which replace the detailed data of the original sequence. A new point is created each time the distance between either coordinate of the last new point created and the corresponding coordinate in the current point of the running average is greater than 2.5% of the dimension of the bounding box in that direction. Thus, the displacement of the location of a receptor will not vary more than 1.25% from its original location based on the overall size of the region.

The Fig.14 shows the results of smoothing the outline of Luxembourg. The original shape from the NUTS database, shown on the left, contains 113 vertices; after smoothing it is approximated by 71 vertices, shown on the right. The more irregular an outline, the greater the potential for simplification, as indicated by the following examples:

Cyprus 513 vertices → 108 vertices (79% reduction)
United Kingdom 3286 vertices → 252 vertices (92% reduction)
Norway 8595 vertices → 166 vertices (98% reduction).

While the smoothing of the regions will decrease the amount of computations required, it can also lead to possible slight overlap of adjacent regions. This can be seen in Fig.15 which shows the smoothed versions of Norway, Sweden and Finland. For instance, there is a section between Norway and Sweden which overlaps at around 14°E x 64.5°N. Since PATHWAYS takes each region into account individually, the only effect is a very slight geographic displacement of a small fraction of the receptors. There is no change in the number of receptors. Therefore the affect on the calculation of impacts is entirely negligible as has been verified by Curtiss and Rabl [1995b].

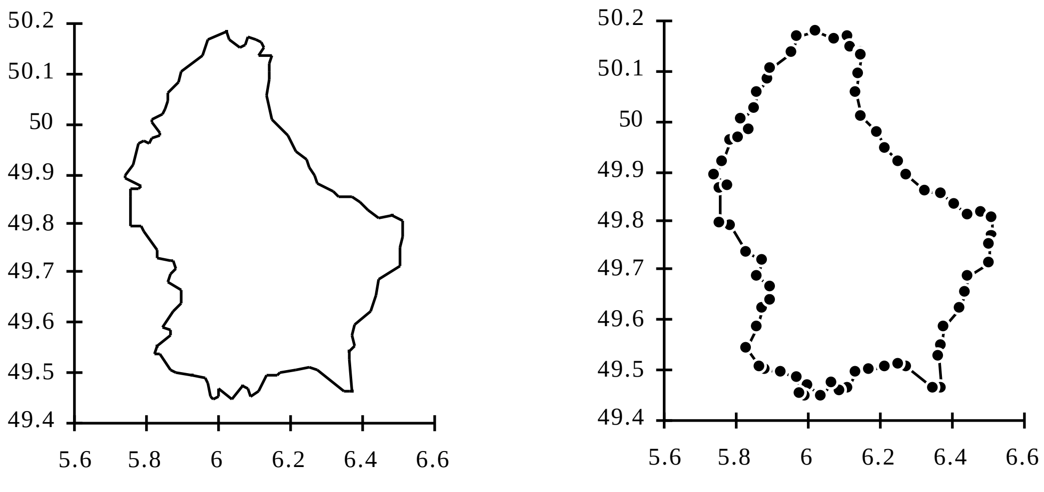

Fig.14. Simplification of the outline of Luxembourg:
original outline (left) 113 vertices → 71 vertices (right), a 37% reduction.

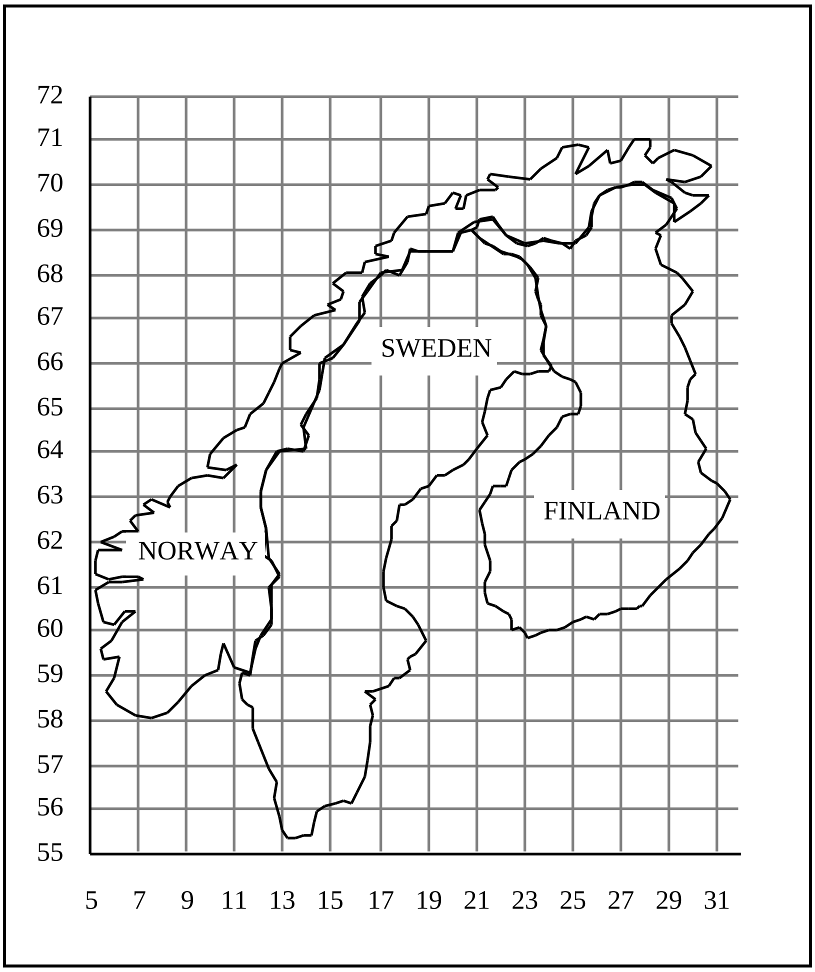

Fig.15. Overlapping of smoothed regions

9. Example: Impacts of Power Plants

In recent years many studies have been carried out to estimate the environmental costs of energy use. A summary of several such studies has been presented by Koomey [1990]. Other examples are Hohmeyer [1988] and Ottinger et al. [1991]. Comparing different studies, one finds that different cost estimates can differ by several orders of magnitude. That there could be substantial differences should not be surprising after the discussion of site dependence in Section 5 and of uncertainties in Section 7 above.

In fact the differences can be even larger because of differences in assumptions or in methodology (that may or may not have been stated clearly by the authors). For instance, when stating an impact per kWh of electricity produced by coal fired power plants, one should note the assumed technology; different flue gas treatment can alter the emissions by an order of magnitude. When numbers are stated for total costs, one needs to consider which impacts have been included (e.g. health impacts, global warming, impacts on ecosystems, land use, etc.) and how large a geographic region has been covered. And the authors may chose among a wide range of possible assumptions, for example with regard to the value of human life; needless to say, the resulting cost estimates can diverge widely.

In order to give a firmer and more consistent basis to such studies, the Commission of the European Communities, in a cooperative program with the US Department of Energy, is developing an accounting framework for identifying and quantifying the external costs associated with fuel cycles [EC 1995, ORNL/RFF 1994]. This program is based on the methodology of impact pathways, applied on the basis of a common set of assumptions that are clearly stated. The authors have participated in the implementation of this program in France [Curtiss et al 1995]. Analogous studies have been carried out in Canada by Ontario Hydro [1993] and for the State of New York by Rowe et al [1995].

In Table 1 we present some results from Curtiss et al [1995], calculated for the impacts and costs due to NO_X, SO_X, and particulates, from a coal fired power plant near Nantes on the Atlantic coast of France. The dispersion has been calculated with a gaussian plume for the local range and the EMEP data for the long range [Barrett 1992, Eliassen and Saltbones 1983]. For the greenhouse gases CO_2 and CH_4 we have not performed a separate calculation because their life times in the atmosphere are decades to centuries, long enough for almost uniform mixing (scaling the global warming costs of EC[1995] to the coal plant near Nantes we find 1.9 ¢/kWh).

We want to emphasize several points about this summary of these results:
1. The results are site dependent (see Fig.7) and technology dependent.
2. Apart from global warming, the dominant impact appears to be mortality; the value for acute mortality is a lower bound. The mortality impacts could be almost a factor of ten larger if the dose-response function for chronic mortality [Dockery et al 1993] is correct.
3. We have not been able to analyze the impacts of coal mining, because the coal is imported from several countries overseas.
4. We have not yet analyzed the impacts of solid wastes, liquid wastes, acid rain, land use, or accidents.
5. Even though ozone can have severe impacts on health and on agricultural crops, in the present case the formation of ozone is negligible because the plant runs only in winter.
6. At the present state of scientific knowledge there is some uncertainty whether the damage categories of health and buildings are due to particulates or to SO_2 or some combination; we have found it most plausible to assign the damage to particulates.
7. For sulfate aerosols, formed from SO_2 as secondary pollutant we have assumed the same acute mortality dose-response function as for particulates. We have not considered health impacts of nitrate aerosols.
8. There is considerable scientific uncertainty, especially about the dose-response functions.

Table 1. Overview of damages in Europe for **coal** fuel cycle:
intermediate load power plant near Nantes, France, output 2.1×10^{9} Kwh/yr.

Emissions according to regulations ("best available technology").

Average retail price of electricity in France = 8.1 ¢//kWh [EdF 1994].

Burdens	**Impact category**	**Physical impact**	**¢/kWh**
Greenhouse gases 1085 g.CO_2 equiv/kWh, includ. upstream emiss.	global warming (sea level rise, crop failures, etc.)	5.9×10^{-16} °K/kWh after 10yr 3.4×10^{-16} °K/kWh after 100yr	**1.9** [EC 1995b]
Particulates 0.17 g/kWh	mortality, acute	3.2×10^{-10} deaths/kWh	**0.10**
	mortality, chronic	2.6×10^{-9} deaths/kWh	**0.8**
	morbidity	respiratory diseases	**0.02**
	buildings	maintenance cost increase by 10^{-13}%/kWh	**0.003**
	historical monuments		**0.0003**
SO_2 1.36 g/kWh	mortality	we assume that damage in these categories is due to particulates rather than SO_2	
	morbidity		
	building+hist.mon.		
	crops	fertilizer or damage	**- to + 0.001**
Aerosols from SO_2	mortality, acute	1.1×10^{-9} deaths/kWh	**0.4**
NO 2.22 g/kWh	crops	fertilizer	mostly benefit
Aerosols from NO	mortality +morbidity	not yet quantified	
O_3 (from NO + VOC)	mortality	O_3 formation negligible, because fossil plants in France run only in winter	
	morbidity		
	crops		

In view of the weight of the epidemiological evidence, the numbers for acute mortality due to particulate emissions are quite firm; they can be considered lower bounds for the true mortality. The latter is higher because of aerosols, and because the dose-response function for chronic mortality [Dockery et al 1993] could be almost an order of magnitude higher than the one for acute mortality. Chronic effects are notoriously difficult to measure, and there is much uncertainty. Of course, the mortality costs scale with the assumed reference value of statistical life. The numbers in Table 1 are based on the value of 2.6 MECU ($ 3.4 million) that has been recommended for the EC studies [EC 1995a].

The morbidity costs, even if simply added together, amount to less than a quarter of the acute mortality costs at $ 3.4 million per life. For that reason we have not worried very much about double counting, i.e. the possibility of overlap in the definition of health impacts.

Even though damage to buildings is often thought to be a major item, we find that soiling of utilitarian buildings is at least an order of magnitude below mortality. Note, however, that there is significant uncertainty, and that we have not included damage to other materials such as galvanized steel (which has been found appreciable in EC [1995b]). Our damage estimate for historical buildings and monuments is even more uncertain, but in any case much smaller than that for utilitarian buildings.

Damage to crops is small because in France the fossil fuel power plants run only in winter when ozone formation is negligible. Of the direct pollutants, particulates have no significant effect, NO is a fertilizer, and for SO_2 the background concentration in France is such that the dose-response function is near the transition between damage and fertilizer.

That the non-greenhouse damage of the fossil plants is small compared to the price of electricity is thanks to regulations that impose stringent pollution controls. Without such regulations the emissions of the classical pollutants would be an order of magnitude higher. Studies such as the ones presented here can serve to determine optimal levels of pollution control. For example, a rough estimate suggests that the current regulations are justified on economic grounds because the cost of current pollution control measures adds less than twenty percent to the cost of electric power.

The field is evolving, and the reader who wants the most reliable or the most complete information is advised to consult also the other studies of this type [Ontario Hydro 1993, ORNL 1994, EC 1995, Rowe et al 1995].

Acknowledgment

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