

Air pollution:
from epidemiological studies
to regulation policies

Study realized by **Agnès LEFRANC**

Project coordinator, IAURIF – Observatoire Régional de Santé Ile-de-France

Senior fellow, Johns Hopkins University – Institute for Policy Studies

a.lefranc@ors-idf.org

Tel: +33.1.44.42.64.79

Institut d'Aménagement et d'Urbanisme de la Région Ile-de-France (IAURIF)

15, rue Falguière

75015 Paris

France

General director: Hervé GAY

Observatoire Régional de Santé Ile-de-France (ORS)

21-23, rue Miollis

75732 Paris cedex 15

France

Director: Ruth FERRY

University Johns Hopkins – Institute for Policy Studies (IPS)

3400 N. Charles Street

Baltimore, MD 21218

United States

Director: Sandra NEWMAN

This study was realized in the context of a three-month fellowship, between September and December 2003 at the Institute for Policy Studies of Johns Hopkins University (Baltimore) as a senior fellow. A one-month business trip between mid-October and mid-November was made in order to meet experts involved in the evaluation of health effects of air pollution and in the development of air pollution regulations.

Thanks to:

all the staff's members of the Institute for Policy Studies, especially

Sandra Newman, Director

Marscha Schachtel, senior research fellow

Laura Vernon-Russell, assistant

Thanks to all the persons who accepted to receive me (see Appendix)

Table of contents

Acronyms and abbreviations	7
Introduction.....	9
I. Studying the health effects of air pollution.....	15
Environmental epidemiology studies	17
Time-series studies.....	19
Cohort studies	28
Causality.....	30
II. Air pollution controls in France and in the United States.....	33
Monitoring ambient air quality and informing citizens	36
National ambient air quality standards.....	39
Implementation plans	45
Air pollution regulations as risk management policies	47

III. Estimating the public health benefits of proposed air pollution regulations.....	49
The framework of the Health Impact Assessment	51
The principles of air pollution HIA.....	52
Uncertainties in air pollution HIA.....	54
From HIA to cost-benefits analyses.....	61
Comparing risks	65
Conclusions.....	67
Science and environmental policy	69
Necessary decisions	70
Communicating epidemiology and HIA results to public and decision makers	71
References.....	75
Appendix.....	85

Acronyms and abbreviations

APHEA: Air Pollution and Health, a European Analysis

AQS: Air Quality System

BACT: Best Available Control Technology

CASAC: Clean Air Science Advisory Comity

CAA: Clean Air Act

EPA: Environmental Protection Agency

GAM: Generalized Additive Models

HEI: Health Effects Institute

HIA: Health Impact Assessment

LAURE: Loi sur l’Air et l’Utilisation Rationnelle de l’Energie (Law on Air and Rational Use of Energy)

NAAQS: National Ambient Air Quality Standards

NMMAAPS : National Morbidity, Mortality and Air Pollution Study

NRC : National Research Council

PDU: Plan de Déplacements Urbains (Urban Mobility and Transportation Plan)

PPA: Plan de Protection de l'Atmosphère (Air quality Protection Plan)

PRQA: Plan Régional pour la Qualité de l'Air (Regional Plan for Air Quality)

RR: Relative Risk

SIP: State Implementation Plan

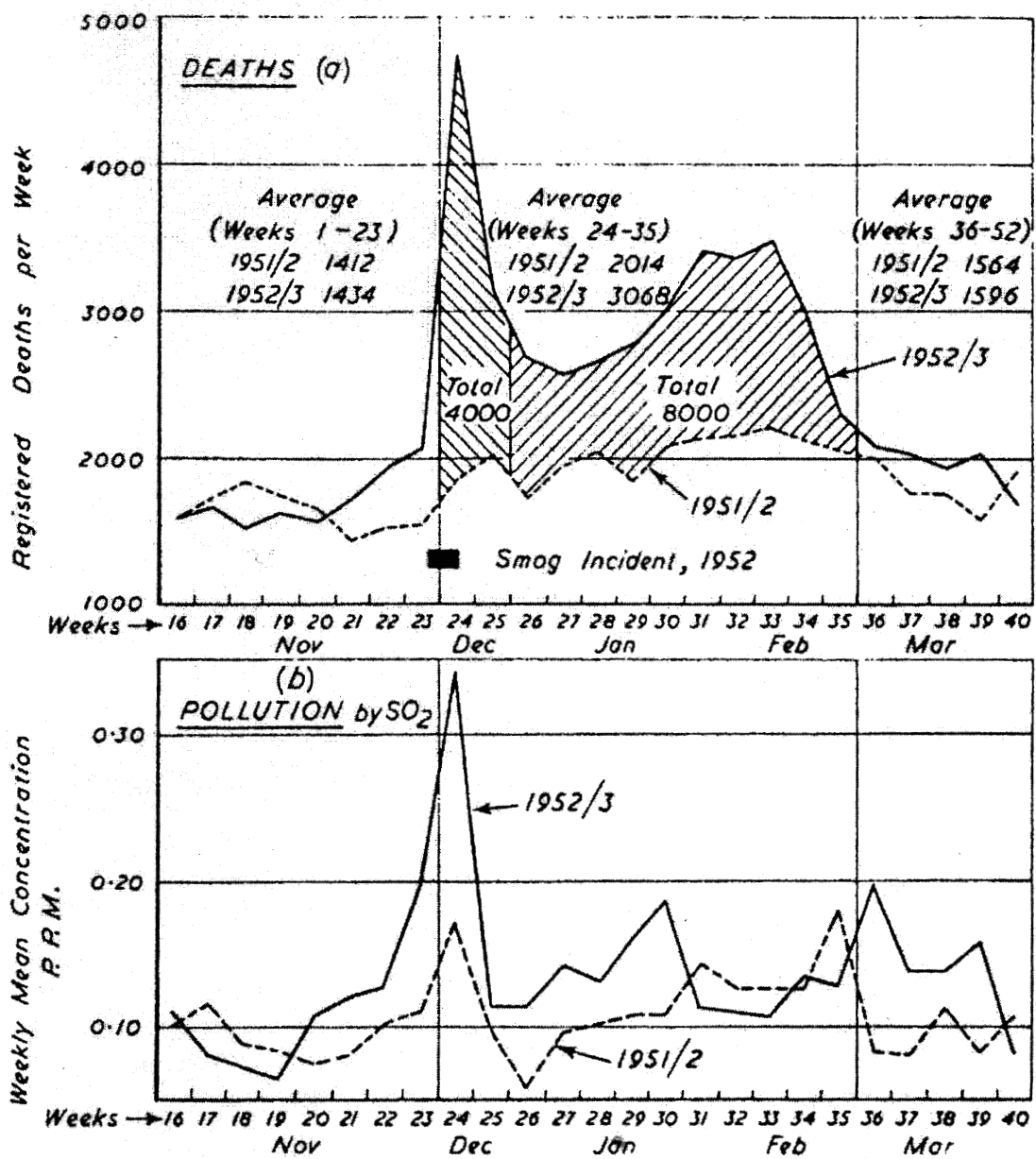
TCP: Transport Control Plan

VSL: Value of a Statistical Life

WTP: Willingness to pay

Introduction

Figure 1: Daily mean pollution concentration and daily number of deaths during the London fog episode of 1952 (from Wilkins 1954)



Environmental health is defined by the World Health Organization as comprising “those aspects of human health and disease that are determined by factors in the environment. It also refers to the theory and practice of assessing and controlling factors in the environment that can potentially affect health”¹. Among these environmental factors, air quality is very important as it represents a source of exposure which is very difficult to escape.

The atmosphere is a mixture of various gases. Human activities and some natural processes generate some compounds that are rejected in the atmosphere. Multiple definitions can be proposed for **air pollution**. Some authors rely on the human origin of the substances to define them as “pollutants”, whereas some others clearly take into account the potential harmful effects of the substances to define air pollution. Using this second criterion, Thad Godish (2003) proposes the following definition: “Air becomes polluted when it is changed by the introduction of gas- or particulate-phase substances or energy forms (heat, noise, radioactivity) so that locally, regionally, or globally altered atmosphere poses harm to humans, biological systems, materials, or the atmosphere itself”. This very general definition allows taking into account in a single definition very different problems such as the effects on humans, plants, animals and buildings of gaseous and particulate air pollutants, the “greenhouse” effect of carbon dioxide or methane, the harmful effect of CFC on the ozone layer, etc.

In the following parts of this report, “air pollution” will be restricted to the gaseous and particulate air pollutants.

The **health effects of air pollution** have been suspected for a very long time, especially after dramatic air pollution episodes such as the ones encountered in the Meuse Vallée (Belgium, 1930), in the Donora Valley (U.S.A., 1948) and in London (UK, 1952, see Figure 1). During each of these exceptional air pollution episodes, excess deaths were recorded. But both decision makers and pollution emitters have not always been convinced of the relationship between the presence of pollutants in the air and these health effects (for more information, see Davis 2002).

One of the first attempts to study scientifically the link between air pollution and health with the help of statistical methods was conducted by two econometrists, Lave and Seskin. They published a book in 1977, after more than 10 years of work, including peer reviews and reanalysis in order to

¹ See http://www.euro.who.int/eprise/main/WHO/Progs/HEP/20030612_1

answer the criticisms emitted by various people (Davis 2002). This book contains an entire section (section IV) discussing the **policy implications** of the links observed between air pollution and health. Originating from the economy field, the two authors used a **cost-benefits analysis** of air pollution abatement measures as a base for their discussion. This kind of approach is still in use today.

The recent advances in environmental epidemiology have given tools that allow quantifying more and more precisely the intensity of the relationships between pollutants levels in the air and both short- and long-term health effects in humans. But even with these very powerful methods, **uncertainties** remain at various levels. Klapp (1992) distinguishes four types of uncertainties affecting risks that may delegitimize regulatory decisions based on them:

- The “extrapolation” uncertainty, that occurs when scientists disagree over whether findings concerning risks in one species can be extrapolated to another (typically from animals to humans),
- The “data” uncertainty, that occurs when scientists disagree about what data, types of sample, or number of studies to use to analyze a risk,
- The “model” uncertainty occurs when scientists disagree over which parameters should be included in models of risk,
- The “parameter” uncertainty occurs when scientists disagree over how to estimate the same parameter within a model.

Risk evaluations concerning the health effects of air pollution can be affected by any of these kinds of uncertainties.

These uncertainties are however more and more precisely quantified and can hence be taken into account during the process of decision making.

In order to study how results from epidemiological studies are used to analyze the public health implications of proposed air pollution regulations, and therefore helps decision makers to design policies, this report is going to present successively:

- The main **epidemiological methods** presently used for the assessment of short- and long-term health effects of air pollution.
- The **public policies** used to control air pollution in France and in the United-States.
- The **methods used to assess the public-health benefits of air pollution regulations**, and how the uncertainties inherent in these methods may affect the regulatory decisions.

The aim of this report is not to provide extensive information about the health effects of the pollutants themselves, this information will hence not be directly reported in the different parts.

I. Studying the health effects of air pollution

The recent advances in environmental epidemiology and biostatistics provide powerful tools to study the links between air pollutants levels and health outcomes. Most of these analyses give always the same result: there is a significant link between air pollution and health. But the nature of the data and of the methods used makes it impossible for the moment to get rid of certain uncertainties.

Studying the effects of air pollution can be done through various approaches. One is the experimental study of the effects of controlled amounts of pollutant on animals' health. This is an approach commonly used in toxicology, and it usually gives very interesting results. But when this approach is used, the results by themselves can not be used in order to take a public-health regulation decision. In this case, the interpolation incertitude would be too important: one could easily object that, as animals and humans biology is at least slightly different, one compound having a harmful effect in animals could be innocuous for humans (and reciprocally).

Experimental studies in humans can not be done in most cases for evident ethic reasons. In order to evaluate the risks associated with exposure to air pollutants, one hence has to use, among others, the results of **environmental epidemiology studies**. The general aim of these studies is to analyze relationships between environmental exposures and human diseases.

Environmental epidemiology studies

The general framework for these analyses is to apply statistical methods to data concerning the health status of a population (e.g. mortality or morbidity data), its exposure to potentially harmful components and potential confounding factors. For example, in the case of air pollution, health data could be the number of deaths registered in a city, the exposure data could be the levels of pollutants monitored in the air, and confounding factors could be factors linked to both exposure and health outcomes, such as temperature and humidity...

The results from these analyses usually provide useful information on possible causal relationships between air pollutants exposure and health effects. They do not provide by themselves a guarantee that a causal relationship exists between these two variables, but when pooled with other results (including results from toxicology studies) and compared to certain viewpoints, they can give a good indication that such causal relationships exist (Bradford Hill 1994, Weed 2000, Parascandola and Weed 2001, Weed 2002).

Concerning the health effects of air pollution, many different study designs can be used:

- **Cross-sectional designs:** these designs assess the relationship between pollutants exposures and health across a population. Typically, in such a study, death rates of people living in highly polluted areas are compared with the ones of people living in low pollution areas. This kind of study can be found in the primer work of Lave and Seskin (1977), and has been used many times since.
- **Longitudinal designs:** in these designs, exposure and health status is evaluated in the same population over a period of time. The relationship between these two variables is then evaluated through the use of appropriate statistical methods. This kind of method has also been used in the work of Lave and Seskin (1977), but since then, many refinements have been introduced in the methodologies used.

Analyses concerning the links between air pollution and health can also be classified into two categories, depending on the kind of effects they are studying:

- **Short-term studies** concentrate on the short-term effects of air pollution. “Short term” usually corresponds to a time lag of less than two months between the exposure to air pollution and the health effect. Mainly two designs can be used to study the short term health effects of air pollution:
 - **Panel studies** concern a small group of volunteers in which symptoms (e.g. cardiovascular or respiratory symptoms), and/or measures of physiological functions (e.g. cardiac function or lung function) are recorded during a short period of time for every person on a daily or weekly basis, and then studied in relation with air pollution exposure.
 - **Ecological time-series studies** concern an entire population in which the number of health events (e.g. deaths or hospital admissions), aggregated at the population level, is recorded during a long period of time. The day-to-day variations in the number of health events are then studied in relation with day-to-day variation in air pollution.

- **Long-term studies**, in which association between long-term exposure to air pollution and life shortening or morbidity is studied. Usually this is done through **cohort studies**, in which a group of persons is followed during years. The following process involves obtaining information concerning both the health status and exposure to air pollution of each person. These data are then analyzed in order to see if air pollution exposure has long term effects on mortality or morbidity risks.

The results obtained from each of these kinds of study have of course different signification (Künzli *et al.* 2001a).

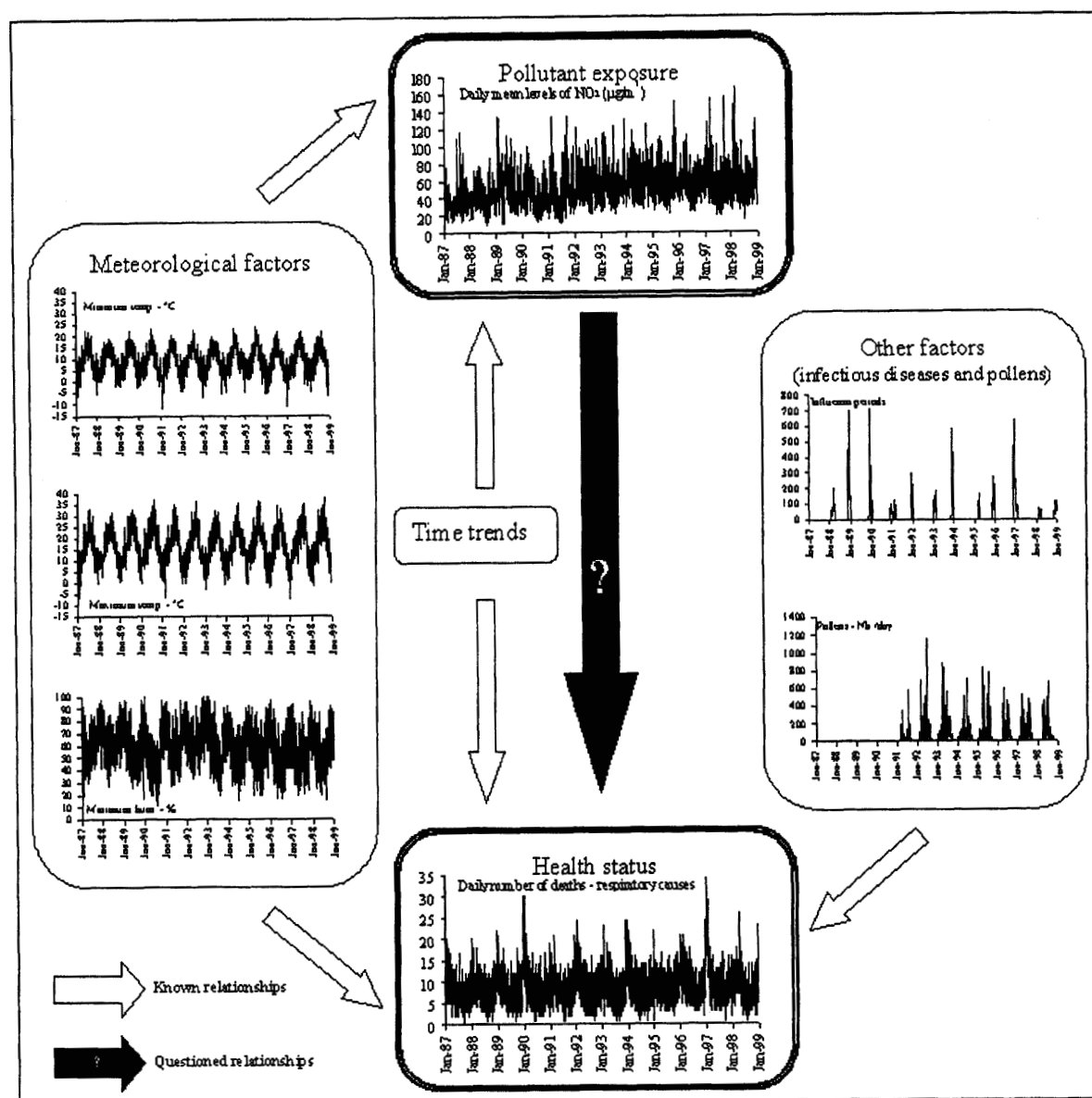
In this report, ecological time-series analysis and to a less extend cohort studies are going to be presented with more details, as they represent a large proportion of the studies available today.

Time-series studies

Studies of the short-term effects of air pollution can be conducted at a large scale, using measures of exposure supposed to represent the mean exposure across the population, and seeking for a correlation between this mean daily exposure and the daily number of deaths or hospital admissions in the same population. This kind of **ecological time-series studies** has been widely used since the beginning of the 90's.

The general principle of these studies is to analyze the relation between day-to-day variation in air pollutants concentrations and day-to-day variations in number of health events (deaths, hospital admissions, etc.), taking into account confounding factors such as the meteorological parameters, time trends, infectious diseases and pollination periods (see Figure 2 on page 20).

Figure 2: General framework for a time-series analysis (data from the ERPURS study - Paris metropolitan area - ORS)



Exposure to air pollutants

A first source of uncertainty in these analyses comes from the variable used as the exposure variable: usually, data concerning the actual exposure of the population are not available. Hence, the exposure is generally assessed under the two following assumptions:

- It is possible to define a geographical area in which population's exposure to air pollutants can be considered as homogeneous.
- In this area, background levels of air pollution can be considered as reliable indicators of individuals' air pollution exposure.

Hence for time-series analysis, exposure is usually assessed by averaging the daily values recorded by background air quality monitoring stations situated within the study-area. This study-area is designed in order to ensure that background air pollution levels do not differ drastically within it.

Of course, these two assumptions are questionable. Exposure is assessed using outdoor measures, whereas the time spent inside buildings can represent up to 90% and varies among individuals. However, studies (Janssen *et al.* 1998, Rojas-Bracho *et al.* 2000, Sarnat *et al.* 2000, Harrison *et al.* 2003) have shown that, at least for fine particulate matter, outdoor background levels are reliable indicators of mean personal exposure. What is more, if some differences exist between individual and measured exposures, they may only induce an underestimation of the effects of air pollution on health (Linacker *et al.* 2000).

The representativeness of the daily average of values recorded by the background monitoring stations also depends on the number and the repartition of these stations within the study area. Usually, the more stations available, the more representative the estimated exposure will be.

Health variables

Health variables must concern the population living in the exact same area used for the evaluation of exposure to air pollution.

Usually, **mortality** data are easily available, reliable and exhaustive, at least in countries such as France, or the U.S, where data collection and coding of deaths causes are done through a well defined and quality insured process. However, even in these conditions, misclassification of death causes may happen, especially between cardio-vascular and respiratory causes (NRC 2002).

When one wants to study **morbidity**, obtaining relevant daily data can be more complicated. Hospital admissions data are sometimes available, but they are often non exhaustive (they may concern only a subset of the hospitals, or a subset of patients), and may present some reliability problems. Some other indicators are also sometimes studied as morbidity indicators: sales of drugs, doctor's house calls (Medina *et al.* 1997)...

Confounding factors and other factors sources of bias

Many factors may act as **confounding factors** if they are correlated with both the pollution exposure and the health variable. If they are not taken into account in the analysis, they may induce a **bias**: the effects observed and attributed to air pollution may in fact be due to these factors.

These factors may include individual sensitivity, age, existing disease, gender, race, socioeconomic status, tobacco smoking, lifestyle, occupation (Godish 2003). But the incidence of all these factors within the population studied is not changing from one day to another, and if there are some long term changes, they are taken into account in the model through the use of time-trends (see Figure 2 on page 20). Hence, in an ecological time-series analysis, these factors do not act as confounding factors.

On the other hand, meteorological factors are of course correlated with both the pollutant exposure and the health variable, and vary from day to day. Hence, they are to be taken into account during the statistical analysis (see Figure 2 on page 20).

Some other factors may also induce some bias in the results of the analysis by interacting with the exposure variable or the health variable. This is the case, for example, of infectious diseases, such as influenza, and pollination periods, which are both known to have a very important effect on

respiratory health outcomes. When data concerning these factors are available, they are hence to be taken into account in the analysis (Braga *et al.* 2000).

Another issue somewhat related to confounding factors concerns the correlations between the levels of the various pollutants. Usually, mono-pollutant models are constructed, and the level of the other pollutants are not taken into account. This represents a problem, because this kind of model does not allow adjusting for simultaneous exposure to a mixture of pollutants that may interact in their health effects. Multi-pollutants models have sometimes been used, but the high correlation between pollutants levels leads to results difficult to interpret.

Statistical method

The method currently used usually involves Generalized Additive Models (GAM, Hastie and Tibishirani 1990). This method has been used for nation-wide studies such as the European APHEA2 study (Atkinson *et al.* 2001, Katsouyanni *et al.* 2001, Samoli *et al.* 2001, Aga *et al.* 2003, Sunyer *et al.* 2003), or the U.S. NMMAPS study (Samet *et al.* 2000a, 2000b). This method has also been used for many local analyses, such as the one made in Paris (Campagna *et al.* 2003).

Briefly, in this method a model linking the number of daily health events to the daily levels of air pollution is built. This model also takes into account the time-trends and all the other confounding factors. Usually, a Poisson link function is used, because the health events have a low incidence, and the total population is large.

The originality of GAM models when compared to other models such as Generalized Linear Models is to allow modeling of non-linear effects by using non-parametric smoothing functions. These non-parametric smoothing functions can be locally weighted smoothing functions called *loess* that allow a very flexible modeling of various trends and effects. When these non-parametric smoothers are used, the estimation of both coefficients of the variables introduced in the model and standard-errors involves an iterating process, which stops when a convergence criteria is reached.

Recently, two potential biases incurred when using this method have been found. One, reported by Dominici *et al.* (2002) is related to the default setting of the convergence criteria in the software

commonly used for these analyses (S-Plus) and may lead to incorrect estimations of the coefficients. The other one, reported by Ramsay *et al.* (2003) is related to the method used in S-Plus for the estimation of the standard error of the estimated coefficients when *loess* non-parametric smoothers are involved. This bias may induce an underestimation of the value of this standard error.

Much has been written about these two problems and their consequences for the results of epidemiological studies using this method (Katsouyanni *et al.* 2002, Samet *et al.* 2003, Lumley and Sheppard 2003). Finally, a complete reanalysis of NMMAPS and of 37 other time-series studies, taking into account these two problems, was published by the Health Effects Institute in May 2003 (HEI 2003). These re-analyses show that the importance of the biases due to the lack of stringency of the convergence criteria vary greatly across studies, but even in the most affected studies, this does not drastically change the conclusions that can be drawn from the study. The bias affecting the standard error was also found in the re-analyses, but its impact on statistical significance of the results was minor (HEI 2003).

Anyway, in order to get rid of these uncertainties, models used for the analysis of time-series now frequently use semi-parametric functions, such as *p-splines* (HEI 2003) instead of non-parametric *loess* functions.

Some authors (Lumley and Sheppard 2003) have also underlined that, apart from there two problems, GAM modeling implies some potentially more important flaws. As an example, control of seasonal effects is a very important point in GAM modeling, as these effects are much larger than air pollution effects. This is usually done in GAM modeling by using some smooth functions of time. The problem is then to choose the appropriate degree of smoothing for these functions. Today, there is no objective way of selecting the degree of smoothing. Hence, selection of the “appropriate” model usually involves both taking into account some subjective assumptions, and screening multiple analyses. This may probably be a source of bias and of increase of the type I error due to multiple testing... As written in the HEI statement (HEI 2003), as there is presently no methods to get rid of these uncertainties, “demonstration of sensitivity or lack of it, to a range of sensible smoothing choices seems a reasonable approach”.

Relative Risks evaluations provided by time-series analysis

Time-series analyses have proved to be very powerful tools to detect small associations between air pollution and health effects. Reviews of these results can be found in various publications, such as the recent review articles by Brunekreef and Holgate (2002) or Vedal (2002), or for each pollutant, in the criteria documents published by the U.S. Environmental Protection Agency².

Usually, time series analyses provide a **Relative Risk** (RR) associated with air pollution, which is significantly larger than 1 if there is a significant acute adverse effect of air pollution on the studied health outcome. Usually, a 95% confidence interval, derived from the estimated standard error, and corresponding hence to the random sampling error of the RR estimate is also given for this relative risk.

Time-series analyses and the shape of the exposure-response relationship

Time-series analyses also allow exploring the **shape of the exposure-response relationship** between ambient air pollutant concentrations and health outcomes (Daniels *et al.* 2000, Arden Pope 2000). The shape of the exposure-response relationship is an important issue, as it conditions the risk associated to each concentration of pollutant.

The methodology used in GAM, by allowing flexible modeling of this dose-response relationship, when combined with complementary analyses, allows an efficient exploration of the shape of this curve. However, the knowledge on this shape obtained from time-series analyses can be less reliable for very low and high concentrations of air pollutants, as usually fewer data points are available for the extreme ends of the concentrations range.

The knowledge of the lower part of the exposure-response curve is especially of interest, as this shape will determine if there is a threshold under which the pollutant does not have any harmful effect. During the last years, this question received a lot of attention, especially in the case of

² All criteria documents are available on line <http://www.epa.gov/ttn/oarpg/t1cd.html>

particulate matter, and statistical methods were developed for this purpose (Smith *et al.* 2000, Schwartz and Zanobetti 2000, Daniels *et al.* 2000).

Mortality displacement and time-series analyses

One concern about the results of time-series analyses is the signification of the excess death observed following high levels of air pollution. Do these excess deaths correspond to a mortality displacement, *i.e.* were the people dying on these days about to die anyway? This hypothesis is called “**harvesting**”, and would totally change the interpretation and the implications of the results of time-series analyzes if it was true.

According to studies conducted on this subject (Kelsall *et al.* 1999, Zeger *et al.* 1999, Schwartz 2000a and b, 2001, Dominici *et al.* 2003a, 2003b, Zanobetti 2003), the harvesting hypothesis does not seem to be true. The methodology used in some of these studies has been discussed (Smith 2003), but the fact that different methodologies lead to the same result, *i.e.* the lack of significant harvesting effect, indicates that harvesting effect does not account for most of the effects observed in time-series analysis.

The excess deaths recorded in time-series analyzes following high air pollution days hence seems to correspond to deaths advanced by months to years.

The time-lag between air pollution exposure and health effects

The earlier studies using time-series analyzes concentrated on the association between daily deaths and air pollutant concentrations recorded on the same or one to two days before. Usually a few lags were tested and the one giving the “best” result was chosen. This method was criticized, as it involved multiple testing and may tend to overestimate the effect of air pollution.

Today, flexible **distributed-lag models** (Zanobetti *et al.* 2000) allow taking into account lags up to 40 days, in a single model. The results of the analyzes using this method (Zanobetti *et al.* 2002,

2003) have shown that the cumulative effect of air pollution over this longer period was usually higher than the one obtained with “classical” fixed lags models. Moreover, these results have also contributed to show that air pollution effects observed in time-series studies are not due short-term mortality displacement (see above).

Time series and meta-analyses

Recently, **meta-analyses** have been conducted on time-series analyses results. These meta-analyses allow combining RR obtained in various places in order to obtain RR estimation for an entire country or continent. This has been done for the U.S. in the NMMAPS study (HEI 2003), for Europe in the APHEIS study (Katsouyanni *et al.* 1997), and for France in the PSAS-9 study (Cassadou *et al.* 2002).

Meta-analyses offer the interest of combining a large amount of data, and hence provide a powerful source of information to estimate the RR associated to air pollution (Dominici and Burnett 2003). What is more, between-study variability can be explored in order to determine which factors can influence the relationship between air pollution and health (Levy *et al.* 2000).

Time series analyses and particulate matter air pollution

One example of a potential source of variation between studies is specific to particulate matter air pollution, and deserves to be underlined here.

The term “particulate matter” can in fact cover a wide variety of both solid- and liquid- phase substances that vary in size and density (Godish 2003). Some of them are directly emitted by pollution sources (primary particles), whereas some others are produced as a result of chemical reactions involving gaseous pollutants and water vapor.

The common methods for quantitatively assessing the concentration of particulate matter in the ambient air discriminate only on the size of the particles. For example, the total amount of particles

less than 10 μ m wide can be measured (this measure is called PM₁₀), or the total amount of particles less than 2.5 μ m (called PM_{2.5}). Concerning the health effects of particles, it is known that smaller particles are able to penetrate deeper into the respiratory tract of humans, and can then have more important health effects (McClellan 2002). But this assessment does not take into account the fact that for a given size, particles composition, and hence chemical properties may certainly vary.

Time-series analyses concentrating on particulate matter health effects have non-surprisingly found some variability in the strength of the association between particulate matter concentrations and mortality or morbidity among locations (Levy *et al.* 2000, Samet *et al.* 2000a and b, Katsouyanni *et al.* 2001, HEI 2003). Among other factors, such as differences in the gravimetric distribution of the particles (PM₁₀/PM_{2.5} ratio, Levy *et al.* 2000) and effect modification due to interactions with other pollutants or climate (Levy *et al.* 2000, Katsouyanni *et al.* 2001), the composition of the particles mix is often thought has having an important role in this heterogeneity (Samet and Pope 2003, Dominici and Burnett 2003).

These uncertainties concerning particulate matter effects apply to time-series studies, but the exact same questions can also be asked concerning the long-term cohort studies, most of which actually concentrate on the effects of particulate matter.

Cohort studies

Cohort-studies are designed to study the long-term effects of air pollution. In order to do so, these studies need to be pursued during a long period of time, especially if they are studying mortality effects.

As in time-series analysis, the aim is to study the relationship between air pollution levels and a health outcome, often mortality. But in cohort studies, it is not the death itself that is considered as the health variable, but the time to death (Künzli *et al.* 2001a).

Usually, in a cohort study, a group a volunteers is followed during years, and the relationship between their life expectancy and their exposure to air pollution is studied. The two main studies of this kind were published during the 90's (Dockery *et al.* 1993, Pope *et al.* 1995). The first one

involved 8,111 adult subjects in northeast and midwest of the U.S. that were followed during 14 to 16 years. In the second one, 552,138 adult subjects from 154 U.S. cities were followed from 1982 to 1989.

Both studies found a significant link between levels of exposure to air pollution (particulate matter and sulfur dioxide) and total and increased risks of death for cardio-pulmonary causes. But soon, some critics concerning the methodology used in these studies were raised, especially when the results of these studies were used by the U.S. EPA to set the new Air Quality Standards for particulate matter in 1997.

These studies were hence subjected by the HEI to a reanalysis, published in 2000 (HEI 2000). This reanalysis concentrated on various factors that could have introduced flaws in the original results:

- **Quality of the original data**
- **Quality of the original analysis:** using the same statistical methods, the data were reanalyzed.
- **Alternative risk models:** the original analyses were conducted using a model that assumed that the relative increase in the death rate associated to pollutants concentrations was constant throughout the period of the follow-up. Models allowing for more flexibility were used to reanalyze the data.
- **Identification of sensitive subgroups:** stratified analyses were conducted during the reanalysis, in order to determine if some population subgroups were responsible for most of the effect observed in the original analyzes.
- **Occupational exposures:** the confounding effect of occupational exposure was taken into account in the original analyzes, but during the reanalysis, more indicators of environmental exposure were used in the analyses.
- **Flexible exposure-response models:** the original analyses constrained the shape of the exposure-response curve. Flexible exposure-response models were hence considered during the reanalysis.

- **Time dependent covariates:** in the original analyses, smoking habits and weight were introduced in the model as constant over time for each individual. Reanalysis were conducted introducing some time-dependency for these variables. Reanalysis also included the introduction of time-dependency for the average level of air pollutants in each city.
- **Population mobility:** in the original studies, population mobility was not taken into account.
- **Ecologic covariates:** for the reanalysis, various ecological covariates (demographic factors, socioeconomic factors, availability of health services, climate, physical environment and gaseous pollutants) were introduced in the model, in order to see if they could possibly act as confounding or modifying factors.

This complete set of reanalysis showed that the results obtained were very similar to the ones obtained in the original analysis, apart from the fact that education may have a modifying effect. The interpretation of this finding is still rather unclear and one may suggest that in this case education acts in fact as a “proxy” for another determinant that has a modifying effect (possibly socioeconomic status, which is known to be correlated with health status).

Today, these results are now widely accepted, and cohort studies have been conducted more recently (Hoeck *et al.* 2002) and are currently conducted, with more and more sophisticated designs (Tager 2003) in order to study the effects of air pollution on a wide range of health outcomes, from death to bronchitic symptoms (McConnell *et al.* 2003) or lung function (Gauderman *et al.* 2002).

Causality

All the results obtained from observational studies such as time-series or cohort studies raise an interrogation concerning the signification of the observed associations.

Usually, several guidelines are used to assess whether an association between a risk factor and a health effect can be interpreted as a cause-effect relationship.

Among the many and differing frameworks proposed to assess the probability of a cause-effect relationship when an association is observed, the “Bradford-Hill viewpoints” are among the most widely used (Bradford-Hill 1994):

- **Temporal sequence** of the associations (exposure precedes effect).
- **Consistency** of the findings in different studies (using different methodologies).
- **Size of the effect.**
- **Monotonic exposure-response relationship**
- **Coherence** of the study results
- **A plausible biological mechanism**
- **Specificity** of outcome
- **Analogy** with similar exposures
- **Evidence of change following an intervention.**

These viewpoints are not criteria *stricto sensu*, as none is sufficient, and only one is necessary (the temporality) to establish causality. The use of these viewpoints has been discussed (Weed 2000, Parascandola and Weed 2001, Weed 2002), but usually there is a consensus toward their use.

Anyway, concerning air pollution, most of these viewpoints are reached. Time-series studies show that there is an exposure-response relationship and a consistency of effects (Bates 1992). More generally, there is a specificity of the outcome (cardiovascular and respiratory health outcomes are more affected). There is an analogy between air pollution exposure and environmental tobacco smoke exposure. Recent studies have shown that interventions were associated with a significant change in population health status (Pope 1989, Hedley *et al.* 1996, Heinrich *et al.* 2000, Friedman *et al.* 2001, Clancy *et al.* 2002). However, concerning particles, the toxicology and biology studies have not been able to determine the particle characteristics or biological mechanisms by which particles may affect human health (McClellan 2002).

Even before the very sophisticated and convincing studies published during the last ten years, a lot of works demonstrated the adverse effects of air pollution on health. These results represented the context in which a lot of early air pollution regulations were developed in both U.S. and France.

II. Air pollution controls in France and in the United States

The framework for air pollution regulations is established in the U.S. by the Clean Air Act (CAA), and in France by the « Loi sur l’Air et l’Utilisation Rationnelle de l’Energie » (LAURE).

Both texts make explicit references to the principle of public health preservation.

In both countries, air pollution regulations involve both the national and local authorities.

Table 1: Key laws controlling air pollution in the U.S. and in France

Year	United-States		France	
	Title	Content	Title	Content
1955	Air Pollution Control Act	Launches the first federal efforts to study and remedy the problem of air pollution, produced mainly by coal-fired industrial plants and vehicle exhaust		
1961			Loi sur l'air	Authorizes the government to issue standards for pollutants emitted by cars, plants and buildings. Authorizes the government to define geographical zones where the air pollutants emissions can be tightly regulated.
1963	Clean Air Act	Funds research and enables federal, state, and local government to issue regulations to curb harmful emissions		
1965	Motor Vehicle Air Pollution Control Act	Authorizes the Department of Health, Education and Welfare to set standards for hydrocarbons and carbon monoxide emitted by new cars.		
1967	Air Quality Act	Authorizes the federal government to identify metropolitan air quality regions and to issue and enforce federal pollution standards if the state fails to do so.		
1970	Clean Air Act Amendment	Authorizes the government to set air quality standards for six major air pollutants and requires the states to devise plans to meet them.		
1976			Loi relative aux installations classées	Authorizes the government to set standards for the emissions of plants, factories and commercial buildings. Authorization required before setting up for these installations.
1977	Clean Air Act Amendment	Relaxation of previous auto emission requirements, prevention of significant deterioration for nonattainment areas		

1980	Acide Precipitation Act	Development of a long-term research plan		
1990	Clean Air Act Amendment	Strengthen the EPA's enforcement powers and provides greater flexibility for meeting air quality standards		
1996			Loi sur l'air et l'utilisation rationnelle de l'énergie	<p>Defines the principles for monitoring of air quality and its effects on health. Results must be available for the citizens.</p> <p>Requires that national air quality objectives are set according to those set by the European community and WHO.</p> <p>Requires that each region prepares its PRQA, which is a framework for a regional air quality policy, aiming at reaching the national air quality standards.</p> <p>Requires that each metropolitan area of more than 250,000 inhabitants prepares a PPA, which defines precisely the actions that can be taken to reach the air quality objectives.</p> <p>Requires that the PDU take into account both the needs for efficient transportations and the protection of environment and public health.</p>
2002	Clear Skies Initiatives	Amends Clean Air Acts and cuts power plant emissions of sulfur dioxide, nitrogen oxide and mercury by 70% by letting individual companies trade pollution credits.		

Sources:

CQResearch Volume 7, No. 9 "New Air Quality Standards", 1997

CQResearch Volume 12, No. 37 "Bush and the Environment", 2003

APHEIS Health Impact Assessment of Air Pollution in 26 European Cities. Second year report, 2000-2001. Institut de Veille Sanitaire, Saint-Maurice, September 2002

Both U.S. and France have a whole framework for the control and monitoring of air pollution. In both countries, this framework involves both local (state, or “region”) and national institutions. Concerning France, the European Community also intervenes in the domain of air pollution regulations.

In both countries, air pollution regulation frameworks deal with three main points: monitoring of air quality, setting of ambient air quality standards, and design of implementation plans to reach these standards, including emission regulations.

Monitoring ambient air quality and informing citizens

In both countries, the technical issues requirements for air quality monitoring are uniform within the country.

In the U.S., according to the section 319³ of the Clean Air Act (see Table 1), the Environmental Protection Agency is in charge of “promulgating regulations establishing an air quality monitoring system throughout the United States which

- Utilizes **uniform air quality monitoring criteria and methodology** and measures such air quality according to a uniform air quality index,
- Provides for air quality monitoring stations in **major urban areas and other appropriate areas** throughout the United States to provide monitoring such as will supplement (but not duplicate) air quality monitoring carried out by the States required under any applicable implementation plan,
- Provides for **daily analysis and reporting of air quality** based upon such uniform air quality index, and

³ See <http://www.epa.gov/air/caa/caa319.txt>

- Provides for **recordkeeping** with respect to such monitoring data and for periodic analysis and reporting to the general public by the Administrator with respect to air quality based upon such data.”

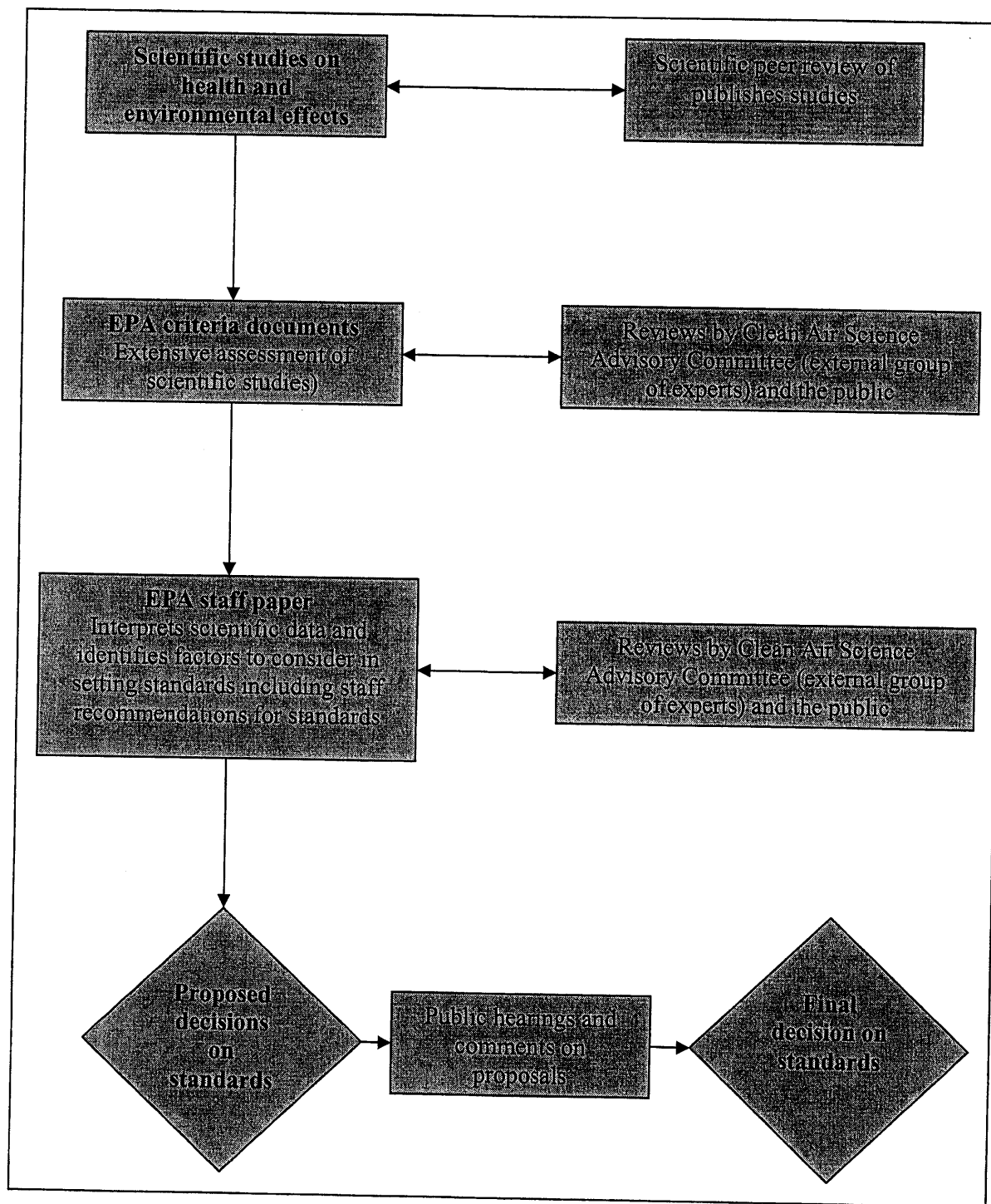
Practically, there is a national monitoring network (**National Air Monitoring Stations, NAMS**), but monitoring networks are also developed and operated by the states (**States and Local Air Monitoring Stations, SLAMS**), which are funded by EPA to do so. For the SLAMS, there are some nationally defined design and quality assurance requirements that must be met. These networks must monitor highest pollutant concentrations, representative concentrations in the areas of high population density, the impact of major emission sources and regional background concentrations. All the data are centralized and kept in a national database, the Air Quality System database (AQS), operated by the EPA.

In France, air quality monitoring is operated in each “region” by an organization. State representatives, local governments in the “region”, representatives from the different business contributing to the emissions of pollutants, approved organizations for the protection of the environment and qualified key figures are members of this organization. Together, these different entities decide on the general terms for the air monitoring policy, according to the standards issued by the French government and the European Union. These standards, as in the U.S. include quality assurance requirements, and the necessity of monitoring background, regional and highest concentrations of air pollutants.

Apart from monitoring air pollution, and according to the terms of the “Loi sur l’air et l’utilisation rationnelle de l’énergie” (1996, see Table 1), this organization is also entitled to forecast air pollution episodes, assess the impact of emission reduction measures and give the results of these measurements to local governments and general public

In both countries, the results derived from the air quality monitoring networks are compared with the national ambient air quality standards, in order to see where these standards are reached, and where they are overshot.

Figure 3: Review process for NAAQS (source H. Richmond, EPA)



National ambient air quality standards

The U.S. National Ambient Air Quality Standards

In the U.S., air quality standards are now set at the national scale. This was not the case before the 1970 amendments to the Clean Air Act (CAA).

Since these amendments, the Environmental Protection Agency is in charge of setting **National Ambient Air Quality Standards** (NAAQS). The EPA administrator is required to prescribe national primary and secondary ambient air quality for the six criteria pollutants (particulate matter, sulfur dioxide, carbon monoxide, nitrogen dioxide, lead and ozone). These criteria pollutants are ubiquitous, have multiple sources and “may reasonably be anticipated to endanger public health or welfare” (section 108 of the CAA⁴).

According to the section 109 of the CAA⁵, primary standards “shall be ambient air quality standards the attainment and maintenance of which in the judgment of the Administrator [of the EPA], based on such criteria and allowing an adequate margin of safety, are requisite to protect the **public health**”. Secondary standards “shall specify a level of air quality the attainment and maintenance of which in the judgment of the Administrator, based on such criteria, is requisite to protect the **public welfare** from any known or anticipated adverse effects associated with the presence of such air pollutant in the ambient air”. Both definitions make explicit reference to public health, whereas they do not take into consideration economic costs.

According to these definitions, NAAQS for each pollutant are set through a review process that includes scientific expertise and peer review (see figure 1).

First of all, an EPA (office of air quality planning and standards) workgroup manage the review of scientific work available. This includes work concerning:

- Background information on **physical and chemical properties** of the pollutant; **sources and emissions**; **atmospheric transport**; **transformation and fate** of the pollutant;

⁴ See <http://www.epa.gov/air/caa/caa108.txt>

⁵ See <http://www.epa.gov/air/caa/caa109.txt>

methods for the collection and measurement of the pollutant; and ambient air concentrations.

- **Environmental effects** of the pollutant on vegetation and ecosystems, impacts on man-made materials and visibility, and relationships to global climate change processes,
- **Exposure** of the general population to this pollutant, and factors affecting this exposure,
- **Health effects** of the pollutants, including the nature of these health effects, judgments about adversity and severity and exposure-response relationships. For this purpose, toxicology, controlled human exposure and epidemiology studies are considered.

The workgroup produces a document synthesizing this review, which is called “criteria document”⁶. This document is then reviewed by an external group of experts: the Clean Air Science Advisory Comity (CASAC), and may be modified and reviewed several times.

When a stable version of the “criteria document” is reached, a “staff paper” is written. Using the information contained in the “criteria document”, the “staff papers” synthesizes the scientific data and identifies factors to consider in setting standards. The “staff papers” includes:

- **Air quality characterization**, including sources, measurement methods, trends and spatial patterns, relationships between human exposure and ambient and central monitor measurements.
- **Characterization of health effects**, including the nature of these effects, the sensitive subgroups of the population.
- **Characterization of health risks**, including risks estimates for the current air quality and just meeting current and alternative standards.

⁶ For an example, see 2003. AIR QUALITY CRITERIA FOR PARTICULATE MATTER (FOURTH EXTERNAL REVIEW DRAFT). USEPA EPA/600/P-99/002aD and bD.. 01 Jun 2003. U.S. Environmental Protection Agency, Office of Research and Development, National Center For Environmental Assessment, Research Triangle Park Office, Research Triangle Park, NC, .
<http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=58003>

- **Characterization of welfare effects**, including effects on materials, vegetation, ecosystem and climate change.
- **Conclusions and recommendations on NAAQS.**

The “staff document” goes through the review process, and then the EPA can propose air quality standards. After this proposition, the EPA holds hearings to solicit public comments before issuing a final decision on standards.

The ambient air quality standards in France

In France, the air quality objectives mentioned in the 1996 “Loi sur l’air et l’utilisation rationnelle de l’énergie” are in fact set at the European scale. The European directive setting the framework of European air quality policies (Framework Directive 96/62/EC on ambient air quality assessment and management⁷, 1996) indicates the following general principles for setting the European air quality standards of 6 criteria pollutants (sulfur dioxide, nitrogen dioxide, fine particulate matter, suspended particulate matter, lead, and ozone):

- Objectives for ambient air quality should be designed to “avoid, prevent or reduce harmful effects on **human health and the environment** as a whole”.
- “**limit value**” shall mean a level fixed on the basis of scientific knowledge, with the aim of avoiding, preventing or reducing harmful effects on human health and/or the environment as a whole, to be attained within a given period and not to be exceeded once attained;
- “**target value**” shall mean a level fixed with the aim of avoiding more long-term harmful effects on human health and/or the environment as a whole, to be attained where possible over a given period;

⁷ See
http://europa.eu.int/smartapi/cgi/sga_doc?smartapi!celexapi!prod!CELEXnumdoc&lg=en&numdoc=31996L0062

- “**alert threshold**” shall mean a level beyond which there is a risk to human health from brief exposure and at which immediate steps shall be taken by the Member States as laid down in this Directive;
- “**margin of tolerance**” shall mean the percentage of the limit value by which this value may be exceeded subject to the conditions laid down in this Directive;
- “When setting the limit value and, as appropriate, alert threshold, the following factors may, by way of example, be taken into account:
 - degree of **exposure** of sectors of the population, and in particular sensitive sub-groups,
 - **climatic conditions**,
 - **sensitivity of flora and fauna** and their habitats,
 - **historic heritage** exposed to pollutants,
 - **economic and technical feasibility**,
 - **long-range transmission of pollutants**, of which secondary pollutants, including ozone.”

One of the differences between this European directive and the CAA is that economic and technical feasibilities are clearly to be taken into account in the process of setting numerical values for the air quality objectives in Europe.

This framework directive was followed by “daughter directives” that set the numerical limit or target values for each of the identified pollutants. Besides setting air quality limit and alert thresholds, the objectives of the daughter directives are to ensure that standards methods and quality insurances are used across Europe.

As in the U.S. system, the development of daughter legislations is being supported by **expert working groups**. They prepare their “position papers” that the Commission uses as a basis to draft legislation. The working groups consist of technical experts from the Commission, Member States, industry and environmental non governmental organizations and are supported as appropriate by the European Environment Agency, the World Health Organisation, the United Nations Economic

Commission for Europe and consultants involved in cost-benefit analysis studies, among others. A Steering Group comprising representatives from these stakeholders and supporting institutions is guiding the work.

The “position papers”⁸ contain information on:

- **Pollutant description**, including sources, current ambient concentrations, trends in emission...
- **Risk assessment**, including exposure of the general population to the pollutant and factors affecting this exposure, health effects of the pollutant, sensitive subgroups, and environmental effects.
- **Measurement**, including the measurements methods and monitoring network design.
- **Costs implications.**
- **Recommendations for limit values, alert threshold and air quality objectives.**

A “daughter directive” is then issued, taking into account the information contained in the position paper. Member states are then supposed to issue a national law in order to conform to this directive. In France, the European daughter directive 1999/30/CE concerning the standards ambient values for sulphur dioxide, nitrogen dioxide, fine particulate matter and lead is applied through a “décret” issued in 2002⁹.

⁸ For an example see the position paper on PM10 (1997)
http://europa.eu.int/comm/environment/air/pdf/pp_pm.pdf

⁹ Décret n° 2002-213 du 15 février 2002 portant transposition des directives 1999/30/CE du Conseil du 22 avril 1999 et 2000/69/CE du Parlement européen et du Conseil du 16 novembre 2000 et modifiant le décret n° 98-360 du 6 mai 1998 relatif à la surveillance de la qualité de l'air et de ses effets sur la santé et sur l'environnement, aux objectifs de qualité de l'air, aux seuils d'alerte et aux valeurs limites.
<http://www.legifrance.gouv.fr/WAspad/Visu?cid=297134&indice=1&table=JORF&ligneDeb=1#>

Table 2: Major air pollution abatement strategies (from Künzli, 2002)

Sector	Measure
General policies	Application of the “polluter pay principle” Energy/fuel pricing Ecotaxes Prioritize public transport systems Urban planning Favor renewable energies to replace fossil fuels
Stationary sources	Emission control in industries, power plants, incineration Regulation of solvent use Fuel production and distribution Coal, coke, oil reformulation for power plants, industrial boilers, etc. Modification of combustion processes Closed circuit in dry cleaning
Mobile sources	Catalyst and particle filters for heavy duty traffic, cars, trains, motorcycles, off-road engines Taxing kerosene for aircrafts Low- or no-emission engine Traffic regimens (car share, rush-hour toll, etc.) Fuel consumption dependent taxes
Products	Solvent replacement in paints, adhesives, etc.

Implementation plans

Outdoor air pollution abatement strategies consist into reducing the emissions.

The national plans for reducing emissions

In both countries, national plans and regulations are designed to reduce emissions. They mainly consist into:

- **Emissions limitations for mobiles sources** (cars, new and “in use”, etc.).
- **Emissions limitations for stationary sources** (power plants, industries and incineration).

These limitations can be reached by use of alternative fuels, by use of control technologies, or by renovating combustion installations.

Usually, the laws concerning these emissions limitations give a threshold that should not be exceeded. If this threshold is overshoot, then penalties are incurred.

Specificity in the U.S. emission reduction program concerns the **emission trading scheme for sulfur dioxide** (Ortolano, 1997). It was introduced by the 1990 Clean Air Acts Amendments (see Table 1 page 34). Emissions of sulfur dioxide can be carried over long distances, and then deposit (“acid rains”), causing damages to the ecosystems. The 1990 Clean Air Acts amendments encourage cost-effective reduction of sulfur dioxide emissions: coal-fired power plants included in the acid rain control program were allowed to buy and sell sulfur dioxide emissions allowances. Each utility received a certain number of emissions allowances (1 emission allowance = right to emit one ton of sulfur dioxide during one year), according to its baseline fuel consumption. If at the end of a year a utility has emitted more sulfur dioxide than its allowances can cover, it faces a penalty and allowances from the following years are used to cover the difference. Each year, there is an EPA auction, where allowances can be bought from the EPA “special allowance reserve”, or from private parties that have not used all the allowances they were initially given. Organizations

can also negotiate private trades. This system of tradable pollution permits, with other measures inciting to limit emissions led to a 54% decrease of average ambient sulfur dioxide concentrations between 1983 and 2002.

The extension of the use of tradable emission allowances to some other pollutants (nitrogen oxides) is currently discussed in the U.S.

The local plans for compliance with the national air quality standards

In both countries, **local governments are responsible for reaching the air quality standards** in their area.

The local governments, by the mean of State Implementation Plans (SIP) and Transport Control Plans (TCP) in the U.S., and “Plan Régional pour la Qualité de l’Air” (PRQA), “Plan pour la Protection de l’Atmosphère” (PPA) and “Plan de Déplacements Urbain” (PDU) in France, have the authority to decide on how to achieve locally the national air quality standards.

However, local governments are not entirely free to make decisions, because in the U.S. for example, all these plans are subject to federal approval. Of course, local regulations must be compliant with the national ones.

These plans contain:

- **Emission inventories.**
- **Comparison of the monitored air quality in the area with the national air quality standards.**
- **Regulations and incentive** in order to:
 - Reduce the number of vehicles miles traveled (car pooling, flex time, priority lanes for buses, parking regulations, developing and improving public transit systems).

- Enhance vehicle emissions inspection and maintenance.
- Apply national emission reduction measures.
- **Projected emission inventories** (with and without the regulations measures proposed).
- **Projected air quality** (with and without the regulations measures proposed).

These plans are usually written by working groups that involves local governments' representatives, government's representatives, industry representatives, non governmental organizations representatives, public health, epidemiology and toxicology experts...

Air pollution regulations as risk management policies

Air pollution clearly represents a risk to human health, and in both countries, very similar approaches have been chosen to manage this risk.

According to Wilson and Crouch (2001), various criteria may be used to decide how to manage risks:

- **Zero risk:** any action which involves any risk should be rejected. This is clearly not possible for air pollution, as there does not seem to be a threshold under which no harmful effects on health could be recorded, and getting totally rid of air pollution seems absolutely improbable.
- **As low as reasonably achievable:** risk should be made as low as reasonably achievable through the regulation process. This criterion requires a decision rule to specify what is "reasonable".
- **Best available control technology (BACT):** the best available technology should be used to reduce the risk. Here, a definition of "best available technology" is needed. In air pollution regulations, it usually means a technology commercially available at a reasonable cost. Explicit reference is made to this criterion in the Clean Air Act, where it is stated that

in regions compliant with the NAAQS, the BACT must be used. The European law concerning the setting of ambient air quality standards also makes a reference to “technological feasibility”, which can be related to the BACT.

- **Cost-benefits analysis:** valuation of benefits and costs of the regulation should be studied in order to take a decision. This criterion is used as a way of designing ambient air quality standards in Europe, where costs are explicitly to be taken into account when setting these standards. In the U.S., the balance between costs and benefits is implicitly taken into account when a regulation is issued, and may also affect the timing for compliance with standards. Furthermore, the federal law requires a Regulatory Impact Analysis to be performed if the consequences of a regulation would cost more than 500 millions dollars.

Setting a standard for an air pollutant suggests that under this standard, the great majority of the population will not suffer from adverse effects. But many recent studies (HEI 2000, 2003, Krewski *et al.* 2000, Hoeck *et al.* 2002) show that significant effects, both acute and long-term, occur at current ambient concentrations, and at concentrations lower than the various standards and guidelines. According to these studies, it seems that no threshold of effect can be defined at the population level. This represents a challenge for the design of regulations, especially the one concerning ambient air quality standards. In this case, forgetting about technological feasibility and costs, one may wonder how to set the numerical value of the risk that would be “acceptable”.

Air quality standards are often considered as playing a central role in regulating air pollution. However they are not the only useful approach for this purpose (Maynard 2003, Raizenne 2003) and epidemiological results underline the need for a whole set of policies directed toward reduction of air pollutants ambient concentrations.

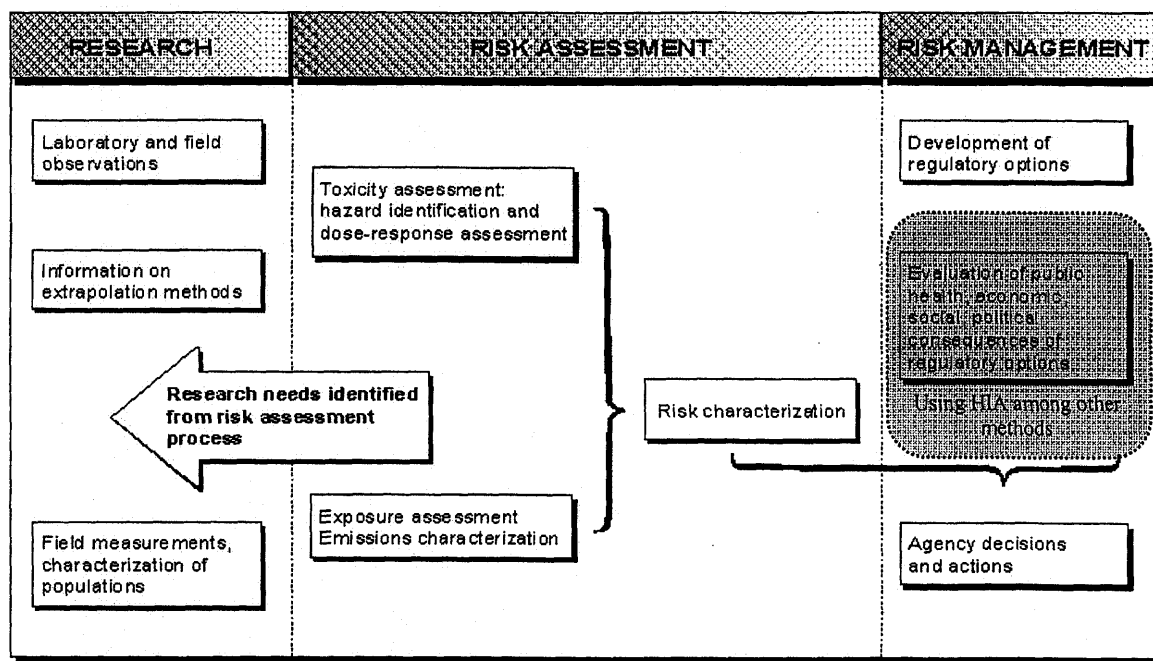
During the whole process of designing air pollution regulations, results from epidemiology and toxicology studies are clearly taken into account (Morrone and Lohner 2002). Knowledge of the exposure-response curves can help the regulators to devise rational policies for dealing with pollutants, by giving an appreciation of the benefits likely to accrue by a reduction in levels of these pollutants.

III. Estimating the public health benefits of proposed air pollution regulations

The U.S. Office of Technology Assessment provided a list of factors that influence regulatory decisions: legislative and political factors, social factors, economic factors, technical feasibility, risk assessment and research.

Results from environmental epidemiology study can be used for risk assessment and cost-benefits analysis, especially through the Health Impact Assessment methodology.

Figure 4: Position of the HIA in the risk assessment and management process (source: EPA)



NAS/NRC risk assessment/management paradigm.

Figure 5: The determinants of determinants (adapted from Joffe and Mindell 2002)

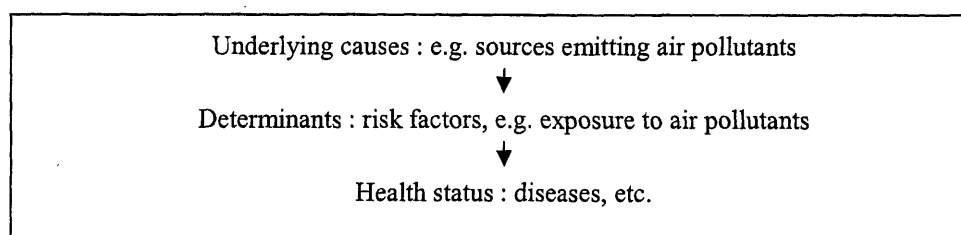
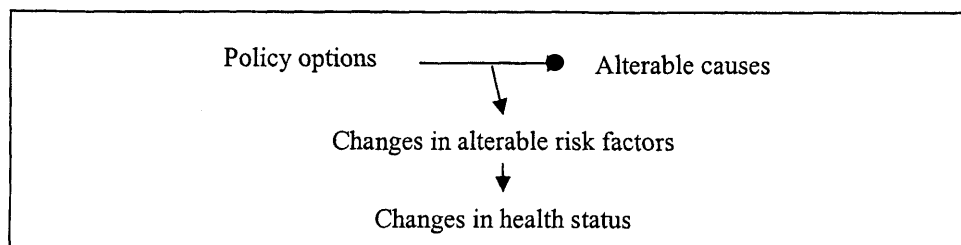


Figure 6: Altering the determinants of determinants (from Joffe and Mindell 2002)



The framework of the Health Impact Assessment

Results from epidemiological studies concerning air pollution are of course used to inform the general public and the decision makers of the qualitative links between the air quality and many health outcomes. Today, a majority of people are aware of this relationship, even if this has not always been the case (Davis 2002).

The most recent advances in environmental epidemiology now allow giving more quantitative estimations of the effects of air pollution on health. More precisely the Health Impact Assessment (HIA) method provides a framework for the quantitative analysis of the consequences of decreases or increases of air pollutants concentrations. This method can hence be used during the process of assessing and managing risks (see Figure 4). Together with risk characterization, the HIA is used to help decision-makers to make choices concerning regulations.

Generally, the HIA is defined (WHO 1999) as “a combination of procedures, methods and tools by which a policy, program or project may be judged as to its potential effects on the health of a population and the distribution of those effects within a population”. As underlined by Joffe and Mindell (2002), one of the main differences between HIA and epidemiology is that in HIA, “risk factors, exposures or determinants are not just taken as given, but are considered in the context of their own underlying causes” (see Figure 5), and policy options are considered as potentially acting on some of these underlying causes (see Figure 6).

Many different kinds of health impact assessments can be performed, depending on their aim. Concerning air pollution, HIAs can be conducted in order to analyze the health effects of attaining targets values, during the process of settings NAAQS for example, or to analyze the health benefits of emission regulations. In the case of NAAQS, estimations of the health impacts of reaching policy targets for exposure levels are conducted, without necessarily specifying the policy options used to achieve them.

The principles of air pollution HIA

The principle of these HIAs is to use exposure-response relationships in order to forecast the health effects of proposed ambient air quality standards, or other air pollution regulations. Usually the approach used for the HIA can be decomposed into three points (Martuzzi *et al.* 2003):

- Definition of the **relevant health endpoints**: the health endpoints should be known to be associated with the pollutant studied.
- Identification of an **exposure-response relationship**, of the associated relative risk and its confidence interval.
- Estimation of the proportion of the health events observed in the study population which are attributable to pollutant's concentration, or **attributable cases**.

The choices made at each of these three steps can be questioned.

General principles

The **relevant health endpoints** are usually chosen taking into account both the known biological effects of the pollutants studied and the availability of exposure-response relationships. Data concerning the frequency of these health events in the population studied must also be available.

The **exposure-response relationships** are usually obtained from the epidemiological literature. When more than one study is available for one pollutant and one health effect, their results can be pooled in order to obtain a "combined estimate". This method attributes a different "weight" to the various studies according to the precision of their estimates. A single relative-risk can hence be obtained for each health endpoint and each pollutant. Usually, the magnitude of the effects is very low, and hence the relationship can be taken as linear (Martuzzi *et al.* 2003). It is then possible to infer an associated increase per unit change in exposure level from the relative risk.

The **rate of the health event** in the population must also be known. It is hence necessary to have data concerning both the demography of the population (population size, total or age-stratified, etc.), and the epidemiology of the health outcome (incidence, prevalence, etc.)

The **change in air pollution concentrations** influences the outcome of the HIA. If the purpose is to compare various policy options, then one HIA may be conducted for each of these options. What is more, target values or values expected with different policies may be compared to the current measured values, but they can also be compared to forecasted values “without regulations”.

Conducting a HIA also requires the **time period** of the assessment to be defined: are the attributable cases going to be considered over a one-year or a ten-year period? If a long period of time is to be considered, demographic projections may be necessary to estimate the size of the exposed population for remote years.

The estimation of the proportion (or absolute number, also known as “**attributable cases**”) of health events observed in the study population which are attributable to the pollutant levels can be calculated with the following formula:

$$E = dRR \times B \times C \times P$$

where E is the number of attributable cases, dRR is the associated increase of RR per unit change in exposure level, B is the rate of the health event observed in the population, C is the change in air pollution concentrations and P is the size of the population exposed to C . This equation supposes that the exposure-response relationship is linear.

What is more, the HIA process assumes that there is a **cause-effect relationship** between exposure to air pollution and health endpoints.

Uncertainties in air pollution HIA

The HIA method has been widely used for policies evaluation in both U.S. and France. After some critics were raised concerning this method, a complete review of HIA uncertainties was conducted in the U.S. (NRC 2002). Here are presented a few major sources of uncertainty in HIAs.

The random sampling error and the uncertainty inherent to the epidemiological studies

The 95% confidence interval surrounding the RR estimate obtained from epidemiological studies represents only the **random sampling error**. Hence the 95% confidence interval for the attributable cases derived from the confidence interval of the RR does not take into account all the potential sources of uncertainty coming from the RR estimate.

Possible **confounding effects** in the epidemiological studies may add some uncertainties in the results of these studies, and of course, the uncertainty concerning the causality of the observed relationship remains as important as in the epidemiological studies. As there are no clear criteria that would ensure that an observed relationship is causal, this point remains one of the most discussed concerning HIA (Künzli 2002).

Extrapolation of risks

Time series analyses of short-time effects of air pollution are not available in every location, and very few cohort studies are available. It is then quite always necessary to extrapolate results across locations or time.

This may represent a source of uncertainty for the following reasons:

- Usually, RRs link ambient concentrations of air pollutants to health effects. But the **relation between ambient concentrations and individuals' exposure** may differ

according to ways of life, climate or other factors. Hence, the RRs obtained in one location may not be applied directly in another location.

- **Population susceptibility** may differ from one location to another, according to their demographic and socio-economic characteristics, and this may influence the issue of the HIA (Levy *et al.* 2002)
- The **baseline incidence rate** of some diseases may differ greatly between populations, and this may affect the relevance of spatial extrapolation (NRC 2002, Rössli *et al.* 2003).
- If **pollutants concentrations are really different** between the original location and the location where the HIA is conducted, extrapolating the slope of the exposure-response relationship may not be relevant. In this case, it can be worth trying to find at least one epidemiological study that could give some information about the shape of the exposure-response relationship in this domain of concentrations, in order to see if the extrapolation seems relevant.
- Concerning particulate matter, the effects may depend on the **composition of the particulate mix**. It has been shown that this composition can differ across locations. Hence using the results obtained in one location to assess the health impact of particulate matter in another location may give incorrect results.

The necessary extrapolation of RRs across locations and time may hence be a source of uncertainty in HIA results, even if it does not seem to be the most important one (Künzli 2002).

Meta-analysis vs. local RRs?

The shape of the exposure-response relationship can represent a source of uncertainty. Each epidemiological study gives an exposure-response relationship that is supposed to represent the “best estimate” of the true relationship for the population studied.

When generalizing the exposure-response function to another population, another location or another time, uncertainty increases (see “extrapolation of risks”). Using results from **meta-analyses** might be a way to limit this source of uncertainty, as these results represent an “average” among different populations and locations, which may hence be less sensitive to this kind of extrapolation.

However, considering that air pollution composition and baseline incidence rates of health outcomes may differ greatly from one location to another, if **local** estimates of exposure-response relationships are available, it may be preferable with regard to the uncertainties to use them than to use meta-analysis estimates.

Much has been written about the choice of epidemiological studies used for the HIA, and it seems that it is not possible to define criteria that could always justify the inclusion or exclusion of some epidemiological studies for the purpose of any HIA (NRC 2002). Anyway, for a particular HIA, some criteria may be defined (see for example Ostro and Chestnut 1998).

Short- vs. long term risks estimates?

The effects of air pollution are usually sorted into two categories: short-term effects, analyzed with time-series analyses for example, and long-term effects, analyzed with cohort studies for example. Both kinds of effects can be associated with a corresponding relative risk. But when doing an HIA, which of these two kinds of relative risks should be used?

This question has been discussed by many authors (McMichael *et al.* 1998, McMichael *et al.* 1999, Quénel *et al.* 1999, Ostro and Chestnut 1999, Künzli *et al.* 2001a and b, Martuzzi 2001, Burnett *et al.* 2003). Künzli *et al.* (2001a) underline the following differences between time-series and cohort studies:

- In time-series studies, the relation between the temporal variability of exposure and the temporal variability of health outcomes is studied. Exposure duration taken into account in these studies varies from 1 day to two months. The RR obtained from a time-series analyses hence does not take into account the deaths corresponding to people that developed frailty

following chronic exposure to air pollution and whose death occurred in a timing unrelated to daily variations of air pollution levels.

- In cohort studies, short term effects of air pollution are impossible to disentangle from long-term effects, but both are taken into account. Even if the short-term effects of air pollution can not be quantified in a cohort analysis, most of these deaths are taken into account in the global RR associated to air pollution. In a cohort study, most of the possible effects of air pollution are hence taken into account. Furthermore, the apparition of frailty, and its association with air pollution exposure can also be studied. Cohort studies also give information on the number of **years of life lost** due to air pollution effects.

According to these descriptions, in order to take into account most of the effects of air pollution and to be able to calculate a number of years of life lost, Künzli *et al.* (2001a and b) concluded that cohort-estimated RRs should be used for HIA. More recently, Burnett *et al.* (2003) showed that it was also possible to calculate a number of years of life lost using time-series-estimated RRs under some particular hypothesis.

The estimation of future concentrations of pollutants, future populations' sizes and future baseline incidence rates

When the aim of the HIA is to compare the health benefits expected in the future with and without a regulation, it is necessary to use some **projections of the air pollutants levels** under these two hypotheses. Usually, very complex models are used for this purpose, and of course, the results of these models are surrounded by a certain uncertainty.

It can also be necessary to have projected values of population sizes and baseline incidence rates. Of course, all these data are usually obtained through modeling, and they hence carry an intrinsic uncertainty that is transmitted to the results of the HIA.

Table 3: Key sources of uncertainties in the Tier 2 benefits analysis (from NRC 2002)

1. Uncertainties associated with Concentration-Response (C-R) functions

- The value of ozone- or particulate matter-coefficient in each C-R function
- Application of a single C-R function to pollutant changes and populations in all locations
- Similarity of future year C-R relationships to current C-R relationships
- Correct functional form of each C-R relationship
- Extrapolation of the C-R relationship beyond the range of ozone or PM concentrations observed in the study

2. Uncertainties associated with ozone and PM concentrations

- Estimating future-year baseline and hourly ozone and daily PM concentrations
- Estimating the change in ozone and PM resulting from the control policy

3. Uncertainties associated with PM mortality risk

- No scientific literature supports a direct biological effect for observed epidemiological evidence
- Direct causal agents within the complex mixture of PM responsible for reported health effects have not been identified
- The extent to which adverse health effects are associated with low levels of exposure that occur many times in the year vs. peak exposures
- Possible confounding in the epidemiological studies of PM_{2.5} effects with other factors (such as other air pollutants, weather, indoor and outdoor air)
- The extent to which effects reported in long term studies are associated with historically higher concentrations of PM rather than concentrations occurring during the period of the study
- Reliability of the limited ambient PM_{2.5} monitoring data in reflecting actual PM_{2.5} exposures

4. Uncertainties associated with possible lagged effects

- What portion of the PM-related long-term exposure mortality associated with changes in annual PM levels would occur in subsequent years

5. Uncertainties associated with baseline incidence rates

- Some baseline incidence rates are not location-specific and might not accurately represent the location-specific rates of interest
- Current baseline incidence rates might not approximate baseline incidence rates in the year 2030
- Projected population and current demographics - used to derive incidences - might not approximate future-year populations and demographics

The time lag between pollution reduction and health benefits

When studying long-term effects of air pollution, it is very difficult to assess the time lag that would take place between a change in air pollutants concentration and a change in the studied health outcomes. This represents a problem in the context of a HIA, because usually, the HIA is conducted for a precise year, separated from the regulation application by a precise duration. It is hence very difficult to decide whether all health benefits would be expected to be present immediately after the decrease of air pollution, or if these benefits would appear progressively, and if so, how progressively?

As an illustration, a list of all sources of uncertainties listed by the EPA for its “Tier 2 analysis” (a HIA to assess the health benefits of a regulation concerning PM and ozone) is presented in Table 3.

Estimating the uncertainties

It is hence clear that the 95% confidence interval around the central estimate derived from the confidence interval of the RR does not take into account most of the uncertainty of the HIA estimate. Some of these uncertainties can be quantified, and hence reported in the uncertainty surrounding the HIA. Reporting some of these uncertainties can be done using some **sensitivity analyses** that “examine the sensitivity of estimated benefits results to less plausible alternatives to the assumptions used in the primary analysis” (EPA). Practically, this means doing a HIA using an alternative value for the parameter whose uncertainty is to be taken into account. With this method, usually only a single source of uncertainty can be examined at a time.

Some more complicated methods that allow taking into account multiple sources of uncertainty at a time exist. For example (Ostro and Chestnut 1998), for each parameter introduced in the HIA calculation, central, low and high estimates can be selected. When the final health benefits are assessed, each of these estimates (high, low and central) is assigned a probability weight and this is used to compute a probability distribution for the total health benefits. A simpler method would have been to calculate a low estimate of the total health benefits by using the low estimates of each parameter, and a high estimate of the total benefits by using the high estimates of each parameter.

However, with this method, the uncertainty would be highly overestimated, as it is very unlikely that all the low (or high) estimates would be correct.

The difficulties in this method arise from the selection of the low and high estimates for each of the parameter, and the choice of the probability weighting. Concerning the **low and high estimates**, one can choose to use:

- the lowest and the highest values available for this parameter from previous studies,
- the most likely range of variation for this parameter (hence not taking into account studies that does not seem relevant),
- when a single study is available, high and low estimates can be generated using the standard errors provided in the study.

Concerning the **probability weighting**, it is ultimately a decision of the scientist, guided by the data available on the subject. This parameter is hence highly dependant from the expert opinion and choice. Hence, different experts may use different probability weighting, and hence generate different estimates for the final estimated health benefits. The NRC (2002) recommended the use of **expert judgment** for the purpose of uncertainty analyses in HIA. However, for parameters distributions which are only or mainly based on expert judgment, the NRC recommended to conduct some sensitivity analysis, confronting opinions from various experts.

Appropriate estimation of the uncertainty is fundamental. Under estimating the uncertainty may lead decision makers to be too confident in the central estimations provided by the HIAs, hence biasing their decision-making. Furthermore, if the uncertainty analysis shows that even taking into account uncertainties, the same regulation decision should be taken, this is a very informative result for decision makers. As a conclusion, the NRC committee (2002) recommended that more sources of uncertainty should be included in the HIAs, and that all these sources of uncertainty should be documented when the results of the HIA are given to the decision makers.

From HIA to cost-benefits analyses

When a quantified health benefit is obtained from a HIA, it is possible to attribute an economic value to these cases, and to compare it with the cost of the proposed air pollution regulation.

Valuation of health benefits

One problem is then the **valuation of life and health**. Ideally, this valuation should include both tangible costs, such as medical costs and lost of income, and costs more difficult to assess objectively such as effects on the well-being. Usually, the **Willingness to Pay** (WTP), corresponding to what individuals would be willing to pay to reduce their risk of illness and death, is used. Concerning death risks, the measure used is the Value of a Statistical Life, i.e. the WTP to save a statistical life which is equal to the sum across different individuals of WTPs for risk reduction that together equal one statistical life. Of course, the VSL is subject to many discussions and cautions (Wilson and Crouch 2001). Other approaches involve health-based measures such as Quality Adjusted Life Years (QALYs) or Disability Adjusted Life Years (DALYs) that can be related to the number of years of life lost, and given a price. But these last health-based measures are less relevant for economic valuation and are hence less used for cost-benefits analysis (Levy 2003).

The WTP can be difficult to estimate for some of the health outcome associated with air pollution, for which there are no estimates of what people would be ready to pay to avoid effects on their well-being, or to reduce a risk. In these cases, the WTP evaluated for the health outcome usually only takes into account the medical costs and the lost of incomes.

According to economical theories, the WTP for a particular health outcome should increase with income, whereas the VSL should decrease with age. Hence, one may wonder if separate health-benefits analyses should be done for each socio-economic and age strata of the population. This raises an important **equity** issue, and usually, no socio-economic stratification is used, but benefits in terms of avoided deaths are presented by age stratum.

When an international point of view is applied to these issues, other problems arise: the valuation of life, the cost of medical care may differ drastically between countries, as do political, institutional and technological frameworks, health and welfare systems. Hence, studies should give a precise description of the uncertainties incurred when transferring valuation functions from one country to another (Bell *et al.* 2002), and the equity issue could also be discussed in this case.

Another difficulty concerning cost-benefits analysis concerns the **range of health benefits** that should be explored: if mortality and hospital admissions are health outcomes for which both data and RRs are available, this is not the case for other endpoints (Krupnick and Morgenstern 2002).

The use of economic valuation may hence add some uncertainties to the ones preexisting in the epidemiologic and HIA results. For this reason, it seems important to give an indication of the uncertainty surrounding cost-benefits results (Krupnick and Morgenstern 2002), and to increase the communication and the collaboration between public health researchers, epidemiologists and economists in order to improve the methodology (Bell *et al.* 2002).

Valuation of regulation-associated costs

Concerning the **evaluation of the costs associated with the air pollution regulations**, valuation of technological costs may seem more straightforward than valuation of health effects, but when forecasted values of technological costs are needed, which is often the case when a proposed regulation is assessed, it can be difficult to take into account all the implications of the regulation.

Many a posteriori reanalysis, that compared forecasted and real costs associated to some air pollution regulations showed that important discrepancies could exist between these two values.

Apart from the technical difficulties enumerated above, one may also have more ethical considerations concerning the use of cost-benefits analysis for policy evaluation: for example, other kinds of analysis integrating equity issues may be more relevant for this purpose

The most important and general cost-benefits analysis realized were the ones conducted by EPA in 1997 and 1999 to comply with the section 812 of the 1990 amendment to the Clean Air Act¹⁰. Both analyses considered the almost entire range of health and environment endpoints for the past (1997 study) and the future (1999 study) years. Another example of a recent large cost-benefits analysis of regulations, including air pollution regulations is the one realized by the Office of Management and Budget (2003)¹¹ that showed that among the many regulatory policies existing in the U.S., the ones concerning air pollution clearly implied some economic benefits. This information was largely relayed in the media¹², showing that this methodology, even it is intrinsically associated with very large uncertainties and hypothesis questionings presents a great interest for both decision makers and general public.

¹⁰ Section 812 of the Clean Air Act requires the EPA to conduce periodic, scientifically reviewed studies on the effects of the Clean Air Act on the “public health, economy and the environment of the United States”, see <http://www.epa.gov/oar/caa/caaa.txt>

¹¹ Informing Regulatory Decisions: 2003 Report to Congress on the Costs and Benefits of Federal Regulations and Unfunded Mandates on State, Local, and Tribal Entities. Office of Management and Budget, Office of Information and Regulatory Affairs. Available online at http://www.whitehouse.gov/omb/inforeg/2003_cost-ben_final_rpt.pdf

¹² See for example The Washington Post, 12/27/2002, A Section, Pg.A1 “Study finds net gain from pollution rules; OMB overturns past findings on benefits” or The Baltimore Sun, 09/28/2003, A section, Pg.9A “Environmental rules’ benefits said to exceed costs”

Table 4: Major criteria in judging the public health relevance of environmental exposure (from Künzli 2002)

Domain of judgement	Criterion
Exposure	Probability of exposure Intensity of exposure Frequency of exposure Life period of exposure Number of people exposed Degree of choice (voluntary exposure) Benefit of the source that causes exposure
Health effect	Type of health effect Degree and intensity of effect Size of effect (Relative risk) Specificity of effect Acute vs. chronic effect Frequency of health outcome among nonexposed Reversibility of effect Acceptance of effect Cost of effects
Abatement / prevention	Number of susceptible among exposed Feasibility of abatement strategies Costs of abatement Benefits of abatement Specificity of the abatement strategy Reversibility of health problems Time of benefit of abatement Acceptance of abatement strategy Level of abatement (individual behaviour vs. structural)

Comparing risks

Today, there is a large concern for various environmental problems: not only air pollution, but also water, soil and food pollution are subjects on which the public asks policy makers to take decisions.

Of course, there is a limited amount of economic resources available for all these issues. One may hence wonder what would be the better way to allocate these resources, in order to obtain the greater public health benefit available (Morrone and Lohner 2002, Maynard *et al.* 2003).

Künzli (2002) gives a list of criteria that can be used to assess the public health relevance of environmental exposures (see Table 4). Concerning air pollution itself, these criteria may be used to define the most relevant regulation strategy. For example, if a pollutant has a low toxicity, but is present in large amounts in the air and is easy to control, should a regulation affecting the emissions of this pollutant be preferred to another one affecting the emissions of a pollutant with a high toxicity, but present in very low concentration and difficult to control? The answer to this question is not always obvious and furthermore, pollutants may interact between them in their health effect (Maynard 2003).

In such cases, or when very different risk factors are to be compared, formal comparative risk assessment may hence be used. For example, comparative risks assessment have been conducted by the WHO (Ezzati *et al.* 2002), using the methodology used for the Global Burden of Diseases¹³. The results obtained with these methods are interesting, but it is important to remind that for all studied risk factors, uncertainties comparable, or even larger than for air pollution are present in these calculations (Powles and Day 2002).

¹³ See <http://www3.who.int/whosis/menu.cfm?path=evidence,burden>

Conclusions

Epidemiology and Health Impact Assessment methods can provide useful information for the design of regulation policies. But all the results obtained with these methods are surrounded by a great amount of uncertainty.

How should these uncertainties been taken into account during the process of policy-making?

Science and environmental policy

The articulation between science and policy has always been very important in the environmental health domain. This link is so important that many opponents to environmental policies argue that these policies are based on “junk science” (Morrone and Lohner 2002, Davis 2002) in order to criticize the policies themselves.

A recent example of the very tight link existing between policies and the underlying epidemiological results comes from the software glitch discovered in S-Plus and affecting the results of the GAM models (Dominici *et al.* 2002). Articles were written in general newspapers about this apparently very technical issue¹⁴. This may seem surprising, but in fact most of the articles concentrated on the signification of this methodological issue affecting RRs estimation for environmental health policies. Of course, opponents to air pollution regulation policies were very prompt to (mis)use these results in order to criticize these policies.

This is a recent example of a discussion that has lasted for the past 20 years, and that has certainly not reached its end, because policy making in the field of environmental health and science are definitively intricate. Morrone and Lohner (2002) distinguish three stages in the making of an environmental policy:

- **Identification of a problem:** this stage usually involves public and politicians who become concerned about a specific situation. Scientists then help to frame the problem, advise the public and politicians of existing studies that address the problem, and conduct research to evaluate human and environmental effects (HIAs for example). This should give policymakers an understanding of the magnitude of the problem
- **Policy making** can then begin. It usually involves administrative agencies and governmental organization that develop environmental regulations and policies. Scientists may be consulted to give advice on the technical feasibility of some regulations, or the costs of the regulations...

¹⁴ See for example The New York Times, 06/05/2002, Section A, page 23 “Data revised on soot in air and deaths”, or The Ottawa Citizen, 06/29/2002, page A7 “Studies on smog deaths flawed: health risk might be overstated in Canada by 25 to 100 per cent, research journal says”

- The last stage is **evaluation**, where science plays an important role in evaluating and monitoring the effects of the policies.

Hence, science plays a role during each stage of environmental policy making, and as underlined by Morrone and Lohner (2002), “to further complicate the use of science in environmental decision making, stakeholders who have special interests can manipulate, and sometimes distort, scientific information to affect outcomes”.

It’s in this context that in the U.S., a special institute, the “Health Effect Institute” (HEI) has been created in 1980 “to provide high-quality, impartial, and relevant science on the health effects of pollutants from motor vehicles and from other sources in the environment”. In order to make sure that no conflicts of interest interfere with this aim, this institute is funded jointly by the EPA and by the industry. The HEI can then fund some studies, and ensure a high-quality scientific reviewing process on some particular topics (see HEI 2000 and HEI 2003 for examples).

Anyway, in a domain such as air pollution, where there are so many sources of uncertainties, it is of course very easy for stakeholders to use these uncertainties as opportunities to criticize and try to refuse air pollution regulations (Klapp 1992, Davis 2002). On the other hand, policy makers can feel uncomfortable with decision making because there are so many uncertainties and unknown things about the effects of air pollution on health.

Necessary decisions

“All scientific work is incomplete-whether it be observational or experimental. All scientific work is liable to be upset or modified by advancing knowledge. That does not confer upon us a freedom to ignore the knowledge we already have, or to postpone the action that it appears to demand at a given time” (Bradford Hill 1994).

This advice can of course be applied to the air pollution issue, and is actually given under slightly different forms in recent publications (Künzli 2002, Levy 2003, Martuzzi *et al.* 2003).

Today, the amount of results showing that air pollutants, even at very low concentrations, have a significant harmful health effect can not be ignored. Even if each of these studies taken individually does not prove that there is a strong causal link between air pollution and health endpoints, the reunion of all of them gives a significant scientific base to say that such a causal relationship exists.

What is more, scientific studies show that some subpopulations are more at risk than others when facing similar levels of air pollution. This represents an opportunity for targeted interventions and prevention actions (for example, giving recommendations to asthmatics to consider increasing levels of medication when pollution level is high) that may induce relatively important public health benefits (Maynard *et al.* 2003).

However, both the uncertainties and the complex concepts involved in the epidemiological analyses of air pollution health effects can blur the understanding that decision makers have of these results.

Communicating epidemiology and HIA results to public and decision makers

Policies intended to reduce the levels of air pollution usually imply some very important changes in the way people live, travel and work. Hence these policies do not have any chance of success if they do not have a strong public support. It is then very important to have a good communication between scientists, policy makers and the public (Maynard and Cohen 2003).

Medical doctors have been often identified as a good relay between scientists and general public. Medical doctors are usually trusted by their patients, they can transmit results concerning the health effects of air pollution and give useful prevention recommendations. It can hence be fruitful to direct specific communication actions toward medical doctors in order to have them then transmitting this information to the general public (such an action is currently done in Paris metropolitan area).

Table 5: Items to be reported in the summary of a benefits analysis of an air pollution control regulation (from NRC 2002)

Planning the analysis	Emissions and air quality	Health benefits
Describe each regulatory option: <ul style="list-style-type: none"> - Geographic scope - Timing - Parties affected 	Summarize emissions at the national level by sector and without the regulation <ul style="list-style-type: none"> - Compare baseline emissions to historical trends 	List health outcomes evaluated and describe each Indicate time path of avoided cases for each outcome
Describe the boundaries of the analysis: <ul style="list-style-type: none"> - Time period of benefits analysis - Intervals at which benefits are calculated - Pollutants evaluated - Degree of compliance with regulation 	Summarize ambient air quality by region and at the national level with and without regulation <ul style="list-style-type: none"> - Report as population weighted averages - Compare baseline air quality to historical trends 	For quantified outcomes at each time period for which results are presented, the following information should be presented: <ul style="list-style-type: none"> - Size of exposed population - Baseline number of cases (per 100,000) - Coefficient of concentration response function
Describe the regulatory baseline: <ul style="list-style-type: none"> - Conditions without regulations, including other regulations in place and assumptions about the economy and population 	<ul style="list-style-type: none"> - Present pollution changes associated with the regulation in absolute and percentage terms 	<ul style="list-style-type: none"> - Number of avoided cases
Highlight any assumptions that have a substantial impact on the results of the analysis		For avoided mortality and chronic morbidity, information should be presented by age at onset and remaining life expectancy

More generally, results from epidemiological studies and HIA should be given in the **clearest way available**, but without omitting the **hypothesis** on which are based these estimates, and the **uncertainties** that surround the results (NRC 2002, Maynard *et al.* 2003). Giving a clear view of the results of an epidemiological study to public or decision makers can be very difficult, because they are usually not very familiar with the scientific concepts and vocabulary, especially the ones concerning uncertainties, or the meaning of an HIA. Epidemiologists should hence do their best in order to ensure that the way they are communicating the results of their studies can not be misunderstood.

Concerning the results of the HIAs, taking into account the very large uncertainties inherent in this method may induce giving an order of magnitude of the number of attributable cases rather than the exact value of the central estimate for this number. When an exact central estimate is given, only this number tends to be remembered. As an alternative, Maynard *et al.* (2003) also propose to give a “distribution of plausible risks” in order to give information on the uncertainties.

As an illustration, concerning the benefits analyses of air pollution control regulations, the National Research Council (2002) gave the recommendations summarized in Table 5 for the elements to be reported.

It hence seems very important to improve and develop:

- The methodologies used in epidemiological studies, in order to get accurate risks evaluations and to identify the sources of uncertainty. Concerning the source of risk variation, it is important to distinguish between differences in exposure and susceptibility, that induce some variability, and that can not be reduced with additional studies and other “true” sources of uncertainty.
- The methods that allow to take into account and to quantify all the sources of uncertainty during a HIA.

All these aspects concern the communication of scientific results from epidemiological or HIA studies. One may wonder if the scientists should also give **policy recommendations**. Much has been written about that (see Begier and Samet 2002 for a review of bibliographic references), and

both points of view seem defensible. One argument against the release of policy recommendations in scientific papers is that policy statements could jeopardize scientific objectivity and trivialize complicated policy issues. From an opposite point of view, policy statements could help to clarify policy implications of scientific results, and replace them in a public health context.

It seems clear that scientists, when reporting results from scientific studies should not discuss details of policy recommendations, as usually these details are very complicated and can not be derived from the results of a single scientific study. However, it seems totally relevant in a scientific paper to give some “general” policy implications. Giving a review of policy options available (see for example Künzli *et al.* 2003) provides useful information about the public health and regulation context of the scientific results.

References

Aga E, Samoli E, Touloumi G, Anderson HR, Cadum E, Forsberg B, Goodman P, Goren A, Kotesovec F, Kriz B, Macarol-Hiti M, Medina S, Paldy A, Schindler C, Sunyer J, Tittanen P, Wojtyniak B, Zmirou D, Schwartz J, Katsouyanni K (2003) Short-term effects of ambient particles on mortality in the elderly: results from 28 cities in the APHEA2 project. *European Respiratory Journal* S40:28s-33s.

Arden Pope III C (2000) Particulate matter-mortality exposure-response relations and threshold. *American Journal of Epidemiology*, 152(5): 407-412

Arden Pope III C, Thun MJ, Namboodiri MM, Dockery DW, Evans JS, Speizer FE, Heath CW (1995) Particulate air pollution as a predictor of mortality in a prospective study of U.S. adults. *American Journal of respiratory and Critical Care Medicine*, 151: 669-674

Atkinson RW, Anderson HR, Sunyer J, Ayres J, Baccini M, Vonk JM, Boumghar A, Forastiere F, Forsberg B, Touloumi G, Schwartz J, Katsouyanni K (2001) Acute effects of particulate air pollution on respiratory admissions: results from APHEA 2 project. *Air Pollution and Health: a European Approach. Am J Respir Crit Care Med.*, 164(10 Pt 1):1860-6.

Bates DV (1992) Health indices of the adverse effects of air pollution: the question of coherence. *Environmental Research*, 59(2):336-349

Begier EM, Samet JM (2002) Studies of particulate air pollution and mortality: when do authors comment on the policy implications? *Epidemiology*, 13(6): 743-744

Bell ML, Davis D, Cifuentes L, Cohen A, Gouveia N, Grant L, Green C, Johnson T, Rogat J, Spengler J, Thurston G (2002) International expert workshop on the analysis of the economic and public health impacts of air pollution: workshop summary. *Environmental Health Perspectives*, 110(11): 1163-1168

Bradford Hill A (1994) The environment and disease: association and causation” in Report to the workers’ compensation board on lung cancer in the hardrock mining industry, report 12 of Canada Industrial Disease Standard Panel (Toronto, Ontario: IDSP), appendix, available at <http://canoshweb.org/odp/html/rpt12>

Brunekreef B and Holgate ST (2002) Air pollution and health. *The Lancet*, 360: 1233-1242

Burnett RT, Dewanji A, Dominici F, Godberg MS, Cohen A, Krewski D (2003) On the relationship between time-series studies, dynamic population studies and estimating loss of life due to short-term exposure to environmental risks. *Environmental Health Perspectives*, 11(9): 1170-1174

Campagna D, Lefranc A, Nunes-Odasso C, Ferry R (2003) ERPURS, Evaluation des Risques de la Pollution Urbaine sur la Santé, 1987-2000. Observatoire Régionale de Santé d’Ile-de-France, Paris, 201p.

Cassadou S and the PSAS-9 team (2002) Programme surveillance air et santé 9 villes, phase II. Institut National de Veille Sanitaire, Saint Maurice, France, 184p.

Clancy L, Goodman P, Sinclair H, Dockery DW (2002) Effect of air-pollution control on death rates in Dublin, Ireland: an intervention study. *Lancet*, 360(9341):1210-1214

Daniels MJ, Dominici F, Samet JL, Zeger SL (2000) Estimating particulate matter-mortality dose-response curves and threshold levels: an analysis of daily time-series for the 20 largest U.S. cities. *American Journal of Epidemiology*, 152(5): 397-406

Davis D (2002) When smoke ran like water: tales of environmental deception and the battle against pollution. Basic Books, 316p.

Dockery DW, Arden Pope C, Xiping Xu, Spengler JD, Ware JD, Fay ME, Ferris BG, Speizer FE (1993) An association between mortality and air pollution in six U.S. cities. *New England Journal of Medicine*, 329: 1753-1759

Dominici F, McDermott A, Zeger SL, Samet JM (2002) On the use of Generalized Additive Models in Time-series studies of air pollution and health. *American Journal of Epidemiology*, 156(3):193-203

Dominici F and Burnett RT (2003) Risk models for particulate air pollution. *Journal of Toxicology and Environmental Health, Part A*, 66: 1883-1889

Ezzati M, Lopez AD, Rodgers A, vander horn S, Murray CJL and the comparative assessment collaborating group (2002) Selected major risk factors and global and regional burden of disease. *Lancet*, 360: 1347-1360.

Friedman MS, Powell KE, Hutwagner L, Graham LM, Teague WG (2001) Impact of changes in transportation and commuting behaviors during the 1996 Summer Olympic Games in Atlanta on air quality and childhood asthma. *JAMA*. 285(7):897-905

Gauderman WJ, Gilliland GF, Vora H, Avol E, Stram D, McConnell R, Thomas D, Lurmann F, Margolis HG, Rappaport EB, Berhane K, Peters JM (2002) Association between air pollution and lung function growth in Southern California children. *American Journal of Respiratory and Critical Care Medicine*, 166: 76-84

Godish T (2003) Air quality, 4th edition. Lewis Publishers, 460p.

Harrison RM, Kinnersley RP, Lawrence RG, Ayres JG, Mark D (2003) Personal exposure assessment in the epidemiology of air pollutants. *Occupational and Environmental Medicine*, 60: 458-460

Hastie T, Tibishirani R (1990) Generalized Additive models. Chapman and Hall, London.

Hedley AJ, Wong CM, Thach TQ, Ma S, Lam TH, Anderson HR (2002) Cardiorespiratory and all-cause mortality after restrictions on sulphur content of fuel in Hong Kong: an intervention study. *Lancet*. 360(9346):1646-52.

HEI (2000) Special report. Reanalysis of the Harvard Six Cities Study and the American Cancer Society study of particulate air pollution and mortality. Cambridge, MA: Health Effects Institute, 293p.

HEI (2003) Special report: Revised analysis of time-series studies of air pollution and health. Cambridge, MA: Health Effects Institute, 271p.

Heinrich J, Hoelscher B, Wichmann HE (2000) Decline of ambient air pollution and respiratory symptoms in children. *American Journal of Respiratory and Critical Care Medicine*, 161(6):1930-1936

Hoeck G, Brunekreef B, Goldbohm S, Fisher P and van der Brandt PA (2002) Association between mortality and indicators of traffic related air pollution in the Netherlands: a cohort study. *The Lancet* <http://image.thelancet.com/extras/01art7366web.pdf>

- Janssen NA, Hoeck G, Brunekreef B, Harssema H, Mensink I, Zuidhof A (1998) Personal sampling of particles in adults: relation among personal, indoor and outdoor air concentrations. *American Journal of Epidemiology*, 147(6): 537-547
- Joffe M, Mindell J (2002) A framework for the evidence base to support Health Impact Assessment. *Journal of Epidemiology and Community Health*, 56: 132-138
- Katsouyanni K, Touloumi G, Spix C *et al.* (1997) Short term effect of ambient of ambient sulphur dioxide and particulate matter on mortality in 12 European cities: results from time series data from the APHEA project. *British Medical Journal*, 314: 1658-1663
- Katsouyanni K, Touloumi G, Samoli E *et al.* (2001) Confounding and effect modification in the short-term effects of ambient particles on total mortality: results from 29 European cities within the APHEA2 project. *Epidemiology*, 12(5):521-31.
- Katsouyanni K, Touloumi G, Samoli E *et al.* (2002) Different convergence parameters applied to the S-PLUS GAM function. *Epidemiology*, 13(6): 744-745
- Kelsall J, Zeger SL, Samet JM (1999) Frequency domain log-linear models; air pollution and mortality. *Journal of the Royal Statistical Society series C Applied Statistics* 48: 331-344
- Klapp MG (1992) *Bargaining with uncertainty: decision-making in public health, technological safety, and environmental quality*. Auburn House (New-York), 149p.
- Krupnick A and Morgenstern R (2002) The future of benefit-cost analyses of the Clean Air Act. *Annual Reviews of Public Health*, 23: 427-448
- Künzli N (2002) The public health relevance of air pollution abatement. *European Respiratory Journal* 20:189-209.
- Künzli N, Medina S, Kaiser R, Quénel Ph, Horak Jr. F, Studnicka M (2001a) Assessment of deaths attributable to air pollution: should we use risk estimates based on time series or on cohort studies. *American Journal of Epidemiology*, 153(11):1050-1055.

- Künzli N, Kaiser R, Medina S (2001b) The first three authors reply : Re" Assessment of deaths attributable to air pollution: should we use risk estimates based on time series or on cohort studies". American Journal of Epidemiology, 154(10):974-975.
- Künzli N, McConnell R, Bates D, Bastain T, Hricko A, Lurmann F, Avol E, Gilliland F, Peters J (2003) Breathless in Los Angeles: the exhausting search for clean air. American Journal of Public Health, 93(9): 1494-1499
- Lave LB and Seskin EP (1977) Air pollution and human health. Published for Resources for the Future by the Johns Hopkins University Press (Baltimore), 368 p.
- Levy JI (2003) Issues and uncertainties in estimating the health benefits of air pollution control. Journal of Toxicology and Environmental Health, Part A, 66: 1865-1871
- Levy JI, Hammitt JK, Spengler JD (2000) Estimating the mortality impact of particulate matter: what can be learned from between study variability ? Environmental Health Perspectives, 108(2): 109-117
- Levy JI, Greco SL, Spengler JD (2002) The importance of population susceptibility for air pollution risk assessment: a case study of power plants near Washington, DC. Environmental Health Perspectives, 110(12): 1253-1260
- Linacker CH, Chauthan AJ, Inskip HM *et al.* (2000) Personal exposure monitoring of particulate matter, nitrogen dioxide, and carbon monoxide, including susceptible groups. Occupational and Environmental Medicine, 59:671-679
- Lumley T, Sheppard L (2003) Time series analysis of air pollution and health: straining at gnats and swallowing camels. Epidemiology, 14(1): 13-14
- Martuzzi M (2001) Re: "Assessment of deaths attributable to air pollution: should we use risk estimates based on time series or on cohort studies?" American Journal of Epidemiology, 154(10): 974-975
- Martuzzi M, Kryzanowski M, Bertollini R (2003) Health impact assessment of air pollution: providing further evidence for public health action. European Respiratory Journal, S40: 86s-91s
- Maynard R (2003) Scientific information needs for regulatory decision making. Journal of Toxicology and environmental health, Part A 66:1499-1501.

Maynard R and Cohen AJ (2003) Public health significance of research results. *Journal of Toxicology and environmental health, Part A* 66:1877-1878.

Maynard R, Krewski D, Samet J, Brook JR, Granville G, Craig L (2003) Health and air quality: directions for policy –relevant research. *Journal of Toxicology and Environmental Health, Part A*, 66: 1891-1903

McClellan RO (2002) Setting ambient air quality standards for particulate matter. *Toxicology*, 181-182: 329-347

McConnell R, Berhane K, Gilliland F, Molitor J, Thomas D, Lurmann F, Avol E, Gauderman WJ, Peters JM (2003) Prospective study of air pollution and bronchitic symptoms in children with asthma. *American Journal of Respiratory and Critical Care Medicine*, 168: 790-797

McMichael A, Anderson HR, Brunekreef B, Cohen AJ (1998) Inappropriate use of daily mortality analyses to estimate longer-term mortality effects of air pollution. *International Journal of Epidemiology*, 27: 450-453

McMichael A, Anderson HR, Brunekreef B, Cohen AJ (1999) Authors' response (Re: "Inappropriate use of daily mortality analyses to estimate longer-term mortality effects of air pollution"). *International Journal of Epidemiology*, 28: 991-992

Medina S, Le Tertre A, Quénel Ph, le Moullec Y, Lameloise Ph, Guzzo J-C, Festy B, Ferry R, Dab W (1997) Air Pollution and Doctors' House Calls: Results from the ERPURS System for Monitoring the Effects of Air Pollution on Public Health in Greater Paris, France, 1991–1995. *Environmental Research*, 75: 73-84

Morrone M, Lohner TW (2002) Sound science, Junk policy: Environmental health science and the decision making process. Auburn House, Westport, Connecticut. 194p.

NRC (2002) Estimating the public health benefits of proposed air pollution regulations. National Research Council, Committee on estimating the health-risk-reduction benefits of proposed air pollution regulations, National Research Council, Board on environmental studies and toxicology. Washington, DC: National Academy Press 170p.

Ortolano L (1997) Environmental regulations and impact assessment. New York, NY: Wiley. 604p.

Ostro B and Chestnut L (1998) Assessing the health benefits of reducing particulate matter air pollution in the United State. *Environmental Research*, section A, 76: 94-106

- Ostro BD, Chestnut LG (1999) Inappropriate use of daily mortality analyses: a response. *International Journal of Epidemiology*, 28: 990-992
- Parascandola M and Weed DL (2001) Causation in epidemiology. *Journal of Epidemiology and Community Health*, 55: 905-912.
- Powles J and Day N (2002) Interpreting the global burden of diseases. *Lancet*, 360: 1342-1343
- Pope CA III (1989) Respiratory disease associated with community air pollution and a steel mill, Utah Valley. *American Journal of Public Health*, 79(5):623-628
- Pope CA III, Thun MJ, Namboodiri MM, Dockery DW, Evans JS, Speizer FE and Heath CW (1995) Particulate air pollution as a predictor of mortality in a prospective study of U.S. adults. *American Journal of Respiratory and Critical Care Medicine*. 151: 669-674
- Quénel Ph, Zmirou D, Dab W (1999) Premature deaths and long-term mortality effects of air pollution (Letter). *International Journal of Epidemiology*, 27: 362
- Raisenne M (2003) Science and regulation – U.S. and Canadian overview. *Journal of Toxicology and Environmental Health, Part A*, 66: 1503-1506
- Ramsay T, Burnett R, Krewski D (2003) The effect of concurvity in generalized additive models linking mortality to ambient air pollution. *Epidemiology*, 14: 18-23
- Rojas-Bracho L, Suh HH, Koutrakis P (2000) Relationships among personal, indoor and outdoor fine and coarse particle concentrations for individuals with COPD. *Journal of Exposure Analysis and Environmental Epidemiology*, 10(3): 294-306
- Röösli M, Künzli N, Schindler C, Braun-Fahrlander C (2003) Which effect measure should be used for impact assessment in a new population context? *Human and Ecological Risk Assessment*, 9(3): 709-719
- Samet JM, Dominici F, Zeger SL, Schwartz J, Dockery DW (2000a) the National Morbidity, Mortality, and Air pollution Study, Part I: Methods and methodologic issues. Research report 94. Health Effects Institute, Cambridge MA.

Samet JM, Zeger SL, Dominici F, Curriero F, Coursac I, Dockery DW, Schwartz J, Zanobetti A (2000b) the National Morbidity, Mortality, and Air pollution Study, Part II: Morbidity and mortality from air pollution in the United States. Research report 94. Health Effects Institute, Cambridge MA.

Samet JM, Pope III CA (2003) Epidemiologic research needs for particulate air pollution. *Journal of Toxicology and Environmental Health Part A*, 66: 1873-1876.

Samet JM, Dominici F, McDermott A, Zeger SL (2003) New problems for an old design: time series analysis of air pollution and health. *Epidemiology*, 14(1): 11-12

Samoli E, Schwartz J, Wojtyniak B, Touloumi G, Spix C, Balducci F, Medina S, Rossi G, Sunyer J, Bacharova L, Anderson HR, Katsouyanni K (2001) Investigating regional differences in short-term effects of air pollution on daily mortality in the APHEA project: a sensitivity analysis for controlling long-term trends and seasonality. *Environ Health Perspect.*, 109(4):349-53.

Sarnat JM, Koutrakis P, Suh HH (2000) Assessing the relationship between personal air particulate and gaseous exposure of senior citizens living in Baltimore, MD. *Journal of Air and Waste Management Association*, 50(7): 1184-1198

Schwartz J (2000a) The distributed lag between air pollution and daily deaths. *Epidemiology*, 11: 320-326

Schwartz J (2000b) Harvesting and long term exposure effects in the relation between air pollution and mortality. *American Journal of Epidemiology*, 151(5): 440-448

Schwartz J and Zanobetti A (2000) Is there a threshold in the relation between particulate air pollution and daily deaths. A meta smoothing approach. *Epidemiology*, 11(6): 666-672

Schwartz J (2001) Is there harvesting in the association of airborne particles with daily deaths and hospital admissions? *Epidemiology*. 12(1):55-61

Smith RL, Kim Y, Fuentes M (2000) Threshold dependence of mortality effects for fine and coarse particles in Phoenix, Arizona. *Journal of Air and Waste Management Association*, 50: 1367-1379

Smith RL (2003) Timescale-dependent Mortality Effects of Air Pollution. *American Journal of Epidemiology*, 157(12): 1066-1070

- Sunyer J, Ballester F, Tertre AL, Atkinson R, Ayres JG, Forastiere F, Forsberg B, Vonk JM, Bisanti L, Tenias JM, Medina S, Schwartz J, Katsouyanni K (2003) The association of daily sulfur dioxide air pollution levels with hospital admissions for cardiovascular diseases in Europe (The Aphea-II study). *Eur Heart J*, 24(8):752-60
- Tager IB (2003) Commentary on the design of cohort studies. *Journal of Toxicology and Environmental Health Part A*, 66: 1731-1734
- Vedal S (2002) Update on the health effects of outdoor air pollution. *Clinics in Chest Medicine*, 23(4): 763-775
- Weed DL (2000) Epidemiologic evidence and causal inference. *Hematology/oncology Clinics of North America*, 14(4): 797-807
- Weed DL (2002) Environmental epidemiology. Basics and proof of cause-effect. *Toxicology* 181-182: 399-403
- Wilkins ET (1954). Air pollution and the London fog of December, 1952. *J R Sanitary Inst* 74(1):1-21
- Wilso R and Crouch EA (2001) Risk-benefit analysis. Harvard University Press, 370 p.
- WHO (1999) Health Impact Assessment. Main concepts and suggested approaches. Gothenburg Consensus Paper, Copenhagen WHO regional office for Europe.
- Zanobetti A, Wand MP, Schwartz J, Ryan LM (2000) Generalized additive distributed lag models: quantifying mortality displacement. *Biostatistics*, 1(3): 279-292
- Zanobetti A, Schwartz J, Samoli E, Gryparis A, Touloumi G, Atkinson R, Le Tertre A, Bobros J, Celko M, Goren A, Forsberg B, Michelozzi P, Rabczenko D, Aranguez Ruiz E, Katsouyanni K (2002) The temporal pattern of mortality responses to air pollution: a multicity assessment of mortality displacement. *Epidemiology*, 13(1):87-93
- Zanobetti A, Schwartz J, Samoli E, Gryparis A, Touloumi G, Peacock J, Anderson RH, Le Tertre A, Bobros J, Celko M, Goren A, Forsberg B, Michelozzi P, Rabczenko D, Hoyos SP, Wichmann HE, Katsouyanni K (2003) The temporal pattern of respiratory and heart disease mortality in response to air pollution. *Environmental Health Perspectives*, 111(9): 1188-1193

Zeger SL, Dominici F, Samet J (1999) Harvesting-resistant estimates of air pollution effects on mortality. *Epidemiology*, 10(2): 171-175

Appendix

The author is grateful to all the persons that took some of their time to meet her.

By chronological order:

**Environmental Protection Agency (EPA),
Office of Air and Radiation**

Carl MAZZA
Science Advisor

Harvey RICHMOND
Office of Air Quality Planning and Standards
Air Quality Strategies and Standards Division
Health and Ecosystems Effects Group

Cesar CORDERO
Intern

**Harvard School of Public Health,
Exposure, Epidemiology and Risk program**

Christine ROGERS
Senior research scientist
Environmental Science Engineering Program

Douglas DOCKERY
Professor of Environmental Epidemiology,
Associate Professor of Medicine (Epidemiology), Harvard Medical School

Jonathan LEVY
Assistant professor of Environmental Health and Risk Assessment
Departments of Environmental Health and Health Policy and Management

Marie S. O'NEILL
Research Fellow
Environmental Epidemiology Program

Antonella ZANOBBETTI
Research Associate
Environmental Epidemiology Program

Stephen FERGUSON
Engineering supervisor
Environmental Science Engineering Program

Mike WOLFSON
Research associate
Environmental Science Engineering Program

Deborah BENNETT
Assistant Professor of Environmental Health and Risk Assessment
Departments of Environmental Health and Health Policy and Management

Joel SCHWARTZ

Associate Professor of Environmental Epidemiology

Associate Professor of Medicine, Harvard Medical School

**California Environmental Protection Agency (Cal-EPA),
Office of Environmental Health Hazard Assessment (OEHHA),
Air Toxicology and Epidemiology Section**

Bart OSTRO

Chief

Air Pollution Epidemiology Unit

Andrew G. SALMON

Chief

Air Toxicology and Risk Assessment Unit

Robert J. BLAISDELL

Chief

Exposure Modelling Unit

**University of Southern California (USC, Los Angeles),
Keck School of Medicine,
Division of Environmental Health**

Nino KUENZLI,

Associate Professor

**Columbia university,
Mailman school of public health,
Department of Environmental Health Sciences**

Joyce E. ROSENTHAL
Project director

Kim M. KNOWLTON
DrPh Candidate

Patrick L. KINNEY
Associate Professor