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# Socio-economic costs of indoor air pollution: A tentative estimation for some pollutants of health interest in France

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ABSTRACT

An evaluation of the socio-economic costs of indoor air pollution can facilitate the development of appropriate public policies. For the first time in France, such an evaluation was conducted for six selected pollutants: benzene, trichloroethylene, radon, carbon monoxide, particles ($PM_{2.5}$ fraction), and environmental tobacco smoke (ETS). The health impacts of indoor exposure were either already available in published works or were calculated. For these calculations, two approaches were followed depending on the available data: the first followed the principles of quantitative health risk assessment, and the second was based on concepts and methods related to the health impact assessment. For both approaches, toxicological data and indoor concentrations related to each target pollutant were used. External costs resulting from mortality, morbidity (life quality loss) and production losses attributable to these health impacts were assessed. In addition, the monetary costs for the public were determined. Indoor pollution associated with the selected pollutants was estimated to have cost approximately €20 billion in France in 2004. Particles contributed the most to the total cost (75%), followed by radon. Premature death and the costs of the quality of life loss accounted for approximately 90% of the total cost. Despite the use of different methods and data, similar evaluations previously conducted in other countries yielded figures within the same order of magnitude.

## 1. Introduction

An evaluation of the socio-economic costs of indoor air pollution can help reveal pollutants, buildings, sources and situations that should be prioritized, thus facilitating the development of appropriate public policies. Nevertheless, extensive evaluations of indoor air pollution have rarely been conducted to date, likely because of the difficulties associated with assessing burden of disease (BOD) values for a large variety of indoor pollutants and exposure situations. In 2005, the California Air Resource Board (CARB) published an initial evaluation of the costs of indoor air pollution in California, US (CARB, 2005). Carbon monoxide (CO), volatile organic compounds (VOCs), environmental tobacco smoke (ETS), radon, mold and sick building syndrome were considered. Indoor pollution was estimated to cost California's economy more than $45 billion each year, with half of this cost attributable to ETS.

Some studies focused on specific indoor pollutants. The annual cost of dampness and mold exposure in the home was estimated to be $3.5 billion per year in the US (Mudarri and Fisk, 2007). In France, Pichery et al. (2011) estimated the annual cost of cognitive and behavioral deficiencies associated with exposure to lead in the home.

Other studies have provided economic evaluations in the context of cost-benefit analyses. Fisk and Rosenfeld (1997) estimated that the potential financial benefits of improving indoor environments exceeded costs by a factor of 18 to 47. The health and productivity benefits of complying with the American Society of Heating, Refrigerating and Air Conditioning Engineers (ASHRAE) standards were quantified by Dorgan and Dorgan (2000). Wargocki and Djukanovic (2005) compared the annual benefit from increased productivity due to a better indoor air quality, improved by the increase of the air supply rate, to the annual energy and maintenance costs of the heating, ventilation and air-conditioning system in one office building. Similarly Fisk et al. (2011) performed a cost-benefit analysis in office buildings and showed that improving indoor air quality, e.g., increasing ventilation rates and reducing mold and dampness, cost less than the return in benefits from the resulting reductions in sick building symptoms and absenteeism.

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Such changes would lead to a benefit of $20 billion annually in the US. Regarding schools, Wargocki et al. (2014) showed that improving indoor air quality through better ventilation, i.e., increased air supply, in Danish schools that do not meet the Danish Building Code requirements would lead to better learning performance and result in yearly increases of €173 million in the gross domestic product (GDP) and €37 million in public finances. Within the HEALTHVENT project, the efficiency of different strategies to reduce indoor exposure to $PM_{2.5}$, outdoor bioaerosols, VOCs, CO, radon, home dampness, and secondhand smoke in the EU-26 was assessed, and the scenarios were compared (Asikainen et al., 2016). The costs and benefits of filtration use have been assessed by several authors (Fisk et al., 2002; Bekö et al., 2008; Aldred et al., 2015).

Other studies have expressed the impacts of indoor air pollution in terms of disability-adjusted life-years (DALYs; the sum of years of life lost as a result of premature death and the years of life spent living with a disease). The DALY calculation is based on: 1) an attributable fraction of exposure or disease associated with the examined risk factor and 2) the national estimates available for the target exposure or disease. Through the European ENVIE project and its follow-up IAIAQ (Jantunen et al., 2011), the health impacts of indoor air pollution within the EU-26 were calculated. Six diseases (asthma, lung cancer, cardiovascular diseases, chronic obstructive pulmonary disease [COPD], upper and lower respiratory infections and acute intoxication) and six groups of associated indoor pollutants (particles, dampness, bioaerosols, radon, CO and VOCs) were considered. The total BOD of indoor air pollution was found to be 2 million DALYs per year within the EU-26. Two-thirds of this BOD was attributable to particles. Examining the European context from a larger scope, the World Health Organization (WHO) assessed the BOD values associated with inadequate housing (Braubach et al., 2011) (i.e., poor indoor air quality resulting from mold and dampness, radon, ETS, lead, CO, formaldehyde and the use of solid fuels for cooking or heating). Still within Europe but on a larger scale, the health impact of benzene, dioxins, secondhand smoke, formaldehyde, lead, traffic noise, ozone, particulate matter ($PM_{2.5}$), and radon represents approximately 3–7% of the annual burden of disease in Belgium, Finland, France, Germany, Italy, and the Netherlands, according to the Environmental Burden of Disease in European Countries Study (EBoDE) (Hänninen et al., 2014). $PM_{2.5}$ was the main pollutant, accounting for 68% of the estimated environmental burden of disease. Schram-Bijkerk et al. (2013) performed a similar study in the Netherlands at the request of Dutch policy makers. The targeted indoor air pollutants included dampness, CO, radon and thoron, formaldehyde and ETS. In the US, Logue et al. (2012) assessed the chronic health impacts of seventy indoor air pollutants measured in American dwellings and calculated a total of 1100 DALYs per year per 100,000 persons. In these studies, the DALYs were not converted into financial costs.

Considering the absence of any evaluation of the socio-economic impacts of indoor air pollution in France, this work aimed to provide an order of magnitude estimate based on existing indoor exposure data for the French population.

## 2. Materials and methods

### 2.1. Selected pollutants

A considerable number of pollutants, including chemical, biological, and physical pollutants, are present in indoor environments (Weschler, 2009; World Health Organization, 2010; Logue et al., 2011, 2012). The list of indoor pollutants to be considered was based on: i) the ranking of > 1000 chemical substances that may be present in indoor environments that was initially established by the national Observatory of Indoor Air Quality (OIAQ) (Kirchner, 2011); ii) an international scientific consensus on associated health effects; iii) existing accessible data on the health impacts on the French population or published dose-response relationships for health impact calculations; and iv) existing data on indoor air concentrations at the national level, e.g., data from the national OIAQ, allowing for health impact calculations to be performed when needed.

### 2.2. Health impact assessment

When the health impact of a given pollutant, i.e., diseases and deaths attributable to indoor exposure to a pollutant, had not been previously assessed in France, this impact was calculated *ad hoc*. Two approaches were used, depending on the nature of the available data. The first method followed the principles of quantitative health risk assessment based on the US National Research Council method (NRC, 1983) and was used when a toxicological reference value (TRV), i.e., an inhalation unit risk, was available. When no TRV was available but a reliable odds ratio or relative risk (RR) was identified in the literature, a second approach based on concepts and methods relating to the health impact assessment approach used in the Aphekom study (Declercq et al., 2012) was used.

For both these approaches, toxicological data and indoor concentrations related to each target pollutant were used. Toxicological data were retrieved from previous reviews and monographs by the French Agency for Food, Occupational and Environmental Health and Safety (ANSES) and from national and international agencies and institutions, while indoor concentrations were measured in dwellings at the national level by the OIAQ housing survey (Kirchner et al., 2007; Kirchner, 2011). In brief, > 30 pollutants were measured for one week (7 days) during 2003–2005 in 567 dwellings randomly selected among the 24 million main residences in France, excluding oversea residences.

In the absence of similar representative data on other indoor settings, such as schools, offices, and leisure spaces in France, the target pollutant concentrations that the French population has been exposed to indoors were assimilated into the concentrations measured in dwellings. Moreover, for this first-tier evaluation, the median indoor concentration in French dwellings was considered for target compounds. The time spent indoors by the French population was considered to constitute 90% of their lifetime (Kirchner, 2011).

In turn, the annual number of premature deaths attributable to indoor exposure to each examined pollutant was calculated when not already available. Furthermore, when not already available, the morbidity of each studied health effect, i.e., the new cases of a disease, was estimated from the mortality / morbidity ratio for this disease multiplied by the calculated (or available) mortality rate of the French population for the given disease. The mortality/morbidity ratio was obtained using data provided by the National Institute for Cancer (INCa), the French Ministry of Health and the French Institute for Public Health Surveillance (InVS).

For each disease, the difference between the average age of death and the life expectancy of the general population (80 years of age (Pison, 2005)) was needed to determine the number of life-years lost. The average age of death for each studied disease was obtained from the Center for Epidemiology on Medical Causes of Death (CepiDC; www.cepidc.inserm.fr). Similarly, the number of life-years with each studied disease was needed; INCa (2007) and World Health Organization (2004) data provided information on the survival times for each disease examined. Finally, when needed, i.e., when using the quantitative health risk assessment method, the number of people in each age category was obtained from the National Institute of Statistics and Economic Studies (INSEE).

Because exposure data, i.e., indoor air concentrations, were obtained through a survey conducted between 2003 and 2005, the reference year for this evaluation was set at 2004. As much as possible, all other collected data were from 2004. Otherwise, figures for the closest years were retrieved and considered attributable to 2004 by default.

### 2.3. Socio-economic cost evaluation

From an economic perspective, indoor air pollution is a negative externality, i.e., a consequence whereby no monetary compensation is initially planned for a transaction in which one party is affected by the intentional or unintentional behaviors of another. In this study, the socio-economic impacts of indoor air pollution are defined as the monetary value of the negative consequences of indoor air pollution, i.e., the quantity of resources lost by society as a result of pollution exposure. Because the objective was to assess the socio-economic costs to society, including public finances, private costs supported by private companies and by individuals were not assessed (Single et al., 2003).

There are two types of socio-economic costs of indoor air pollutants:

1. *External costs*, which measure the opportunity costs of allocated resources resulting from the presence of indoor air pollution;
2. *Impacts on public finances* resulting from the presence of indoor air pollutants.

The overall socio-economic costs were determined for each target pollutant and the associated health effect using Eq. (1), as follows:

$$W = \Delta CE + \Delta G \times (1 + \alpha) \quad (1)$$

with:

W = socio-economic cost variation (€)

ΔCE = external costs based on premature deaths, the loss of life quality (morbidity) and the production loss (€)

ΔG = public financing costs (€)

(1 + α) = a weighting factor demonstrating that a cost of 1 € in public finances corresponds to a (1 + α) € loss in public finances. According to Quinet et al. (2013), α = 0.2.

#### 2.3.1. Evaluation of external costs

To express the cost related to premature death and loss of quality of life, it is necessary to assign an economic value to human life. In general, the value of a statistical life (VSL) is expressed as an economic agent's willingness to pay (*WTP*) to maintain its expected utility when a risk to which it is exposed varies (Δ*risk*), as described by Eq. (2).

$$VSL = \frac{WTP}{\Delta risk} \quad (2)$$

The WTP can be estimated using the contingent valuation method (CVM). The aim of the CVM is to determine the willingness to pay or receive by individuals in terms of the capital gain or nuisance generated by a project. In France, the VSL is estimated at €3,000,000 (Quinet, 2013). A relationship described by Eq. 3 exists between the value of a statistical life year (VOLY) and the VSL.

$$VSL = \sum_{t}^{T} \frac{VOLY}{(1 + r)^t} \quad (3)$$

where T is the number of remaining years of life expected, equal to 40 years, calculated based on the average age and the life expectancy of the French population, respectively equal to 40 and 80 years; and r is the adjustment rate, equal to 2.5%. It is thus possible to calculate a VOLY equal to €115,000.

Moreover, Lebègue et al.'s (2005) report recommends using an adjustment rate of 4% per year in calculations to account for increasing costs in the future relative to the reference year. In this study, the cost of a premature death was defined as equal to the reference value adjustment over the number of life-years lost between the average age at death and life expectancy at birth, which was 80 years in 2004 (Pison, 2005). The adjusted value was then multiplied by the number of premature deaths (available or calculated) for each disease associated with indoor exposure to a studied pollutant to obtain the total cost of premature deaths for a given disease (Eq. (4)). The overall cost of mortality is the sum of the costs for each disease.

$$\text{External cost of premature deaths} = n_d \times \sum_{i} \frac{VOLY}{(1 + r)^i} \quad (4)$$

with:

$n_d$ = number of premature deaths associated with a disease generated by one pollutant

VOLY=value of a statistical life year, €115,000

r = adjustment rate, 4%

i = number of life − years lost because of the disease.

Diseases related to indoor air pollutant exposure reduce one's quality of life. The socio-economic costs in terms of the loss of quality of life were derived from the calculated morbidity value and from the average disability weight defined by the World Health Organization (2004) for each disease. Similar to the costs of mortality, quality of life costs were adjusted over the number of years of life with a given disease prior to death. Each adjusted cost was then multiplied by the number of new cases for each disease associated with indoor exposure to a studied pollutant to obtain the total cost of quality of life loss for a given disease (Eq. (5)). The overall cost of quality of life loss is the sum of costs for each disease.

$$\text{External cost of quality of life loss} = n_c \times \sum_{j} \frac{\delta \times VOLY}{(1 + r)^j} \quad (5)$$

with:

$n_c$ = number of new cases associated with a disease generated by one pollutant

δ = loss of quality of life, i. e. ,average disability weight defined by the WHO for the disease

VOLY=value of a statistical life year, €115,000

r = adjustment rate, 4%

j = number of years of life with a given disease prior to death.

Finally, to estimate production loss, two situations were identified. First, in the case of cancer, INCa (2007) methods and data were used to generate estimates. INCa used the 'discounted income stream' approach to assess the production value that an individual would have generated if he/she had not died of cancer. This method assumes that a person of a given age and sex would survive to the age of life expectancy if he/she had not died of cancer. For each year that he/she would have been alive, his/her production in both the commercial sphere and the non-market sector (e.g., domestic work) is calculated. The production is weighted by the probability that the individual has a job and is adjusted over the years. The production in the commercial sphere is calculated only until the age of retirement. The annual production loss for a given cancer is the sum of the individual production loss per age category and sex multiplied by the number of men and women who die yearly in each age group. For a given cancer, the total loss of production per year is divided by the incidence to obtain a cost per individual. For our study, this cost was multiplied by the number of deaths per year associated with the indoor exposure to a given compound.

For deaths due to the other diseases, in the absence of published data, assumptions were made to infer the production losses associated with a given disease based on the production loss associated with another known pathology that affects the same target organ. Thus, it was assumed that the production losses per year caused by cardiovascular diseases and COPD are one-half and one-quarter of the losses due to lung cancer, respectively. Regarding production loss costs due to deaths from CO poisoning, the discounted income stream was calculated considering that a person loses an average of 47 years of production valued at €35,000 per year.

#### 2.3.2. Impacts on public finances

The payment for treatments represents expenditures associated with the medical care of sick people. These costs were estimated using INCa (2007) data in the case of cancer. For the other diseases, data were derived from the literature on health economics (Spieler et al., 2004; Ministère de la Santé et des Solidarités, 2005) or from an analysis of the national hospital database (PMSI).

Expenditures related to research on indoor air pollution were calculated using an approximate estimate of the number of full-time equivalents allocated to this topic in the country during the 2004 reference year ($n = 108$) and based on the average annual cost of a full-time equivalent position equal to €99,000, which includes salaries with charges (€90,000) and associated expenses (10% of the salary).

Finally, public finance savings resulting from the non-payment of full or partial retirement pensions as a result of premature deaths were considered. The average annual retirement pension value in 2004 (€15,000) was adjusted by 4% over the number of years of retirement lost. Twenty-one percent of the workforce in France comprises public officials. This figure is considered to calculate savings for public finances. Unpaid pensions by the private sector do not affect public finances. The number of lost retirement years was calculated for each disease based on the respective average age at death, a retirement age of 60 years and a life expectancy of 80 years (Eq. (6)). The overall cost of unpaid pensions is the sum of costs for each disease.

$$\text{Unpaid public pensions} = n_d \times \beta \times \sum_k \frac{P}{(1+r)^k} \quad (6)$$

with:

$n_d$ = number of premature deaths associated with a disease generated by one pollutant

$\beta$ = portion of retirement pensions from the public sector, 21%

P = annual retirement pension value, €15,000

r = adjustment rate, 4%

k = number of retirement years lost because of the disease.

## 3. Results

### 3.1. Selected pollutants

Six indoor pollutants were considered either because their health impacts at the national scale were available (namely for radon, CO and ETS) or because their health impacts could be calculated because both the indoor concentrations in the French housing stock and the dose-response/relative risk data were available (namely, for benzene, trichloroethylene and particulate matter with an aerodynamic diameter of $< 2.5$ µm [$PM_{2.5}$]). While particulate matter with an aerodynamic diameter of $< 10$ µm ($PM_{10}$) may also have been considered because it overlaps with $PM_{2.5}$, only one of the two indicators was selected. Thus, $PM_{10}$ is not addressed in this study.

### 3.2. Health impact assessment

#### 3.2.1. Radon

The most reliable and extensive data on radon's health impact in France is for the year 1999 (Catelinois et al., 2006, 2007). To assess the lung cancer mortality attributable to indoor radon exposure, Catelinois et al. (2007) used the mean indoor concentration of radon in French dwellings, the total number of lung cancer deaths and several exposure-response relationships. The residential radon concentrations were estimated using 12,261 measurements carried out in 1980 as part of a nationwide survey; the national arithmetic mean adjusted for the season, type of housing and population density was 63 Bq $m^3$. Six exposure-response relationships were selected: four linear relationships with modifying factors and two relationships without modifying factors. The model derived from the North-American joint analysis and the model from the European analysis were used. The tobacco-radon interaction was taken into account. The authors used a probabilistic method to quantify the uncertainties around the risk factors (Catelinois et al., 2007). The annual number of deaths from lung cancer attributable to domestic radon exposure in France ranged from 1234 (90% confidence interval: 593–2156) to 2913 (2763–3221), depending on the exposure-response relationship used. The arithmetic mean of these two values was calculated to obtain the number of annual premature deaths used in our work: 2074.

In 2005 in France, 30,651 new cases of lung cancer caused 26,624 deaths. The morbidity/mortality ratio for lung cancer caused by radon is considered to be the same as the all-cause morbidity/mortality ratio for lung cancer in the general population. Under this assumption, the incidence of lung cancer attributable to radon is 2388 cases (30,651/26,624 × 2074 = 2388). The deaths occurred at a mean age of 69 years, corresponding to a loss of 11 years of life. The average survival time is 1.5 years (INCa, 2007).

#### 3.2.2. Carbon monoxide

The incidence and mortality data for CO from accidental exposure, i.e., excluding voluntary poisoning and exposure related to fire, between 2000 and 2004 were derived from the national monitoring system for CO poisoning, which compiles death certificates processed by the CepiDC. Most cases were related to housing. Over three-quarters of domestic accidental poisonings were related to a connected boiler installation-type stove/heater or water heater. Other poisonings were related to the use of a non-connected device, such as a brazier/barbecue, a generator or a mobile heating source. Improper use of these devices in an enclosed area, such as the garage, basement or living room of a house, was the primary cause of poisoning in connection with a brazier/barbecue or a generator. A few cases pertained to poisonings in public buildings or at work as a result of CO sources similar to those encountered in homes (boilers, water heaters, etc.). In addition, there were cases of CO poisoning in specific types of buildings such as places of worship, ice skating rinks, go-kart tracks or parking garages. For 2004, 98 annual premature deaths (corresponding to the annual average for 2000–2004) were considered (InVS, 2013).

#### 3.2.3. Environmental tobacco smoke

Only one report – published in 2006 under the aegis of the European Cancer Society, Cancer Research UK, the European Health Network and INCa entitled “Lifting the smokescreen: 10 reasons for a smoke-free Europe” – provided estimates of the number of deaths associated with ETS for France in 2002 (The Smoke Free Partnership, 2006). This report is commonly used in France by the scientific community and authorities and was used to estimate the impacts of ETS in France for our study. The deaths considered were those related to lung cancer, ischemic heart disease (mainly myocardial infarction), cerebrovascular disease (mainly stroke), and chronic lower respiratory tract diseases (mainly COPD). Premature deaths related to smokers' exposure to their own secondhand smoke are obviously excluded. The number of premature deaths caused by exposure to passive ETS in France was estimated as 152 lung cancer cases, 510 myocardial infarctions (ischemic heart disease), 392 strokes and 60 chronic non-neoplastic pulmonary disease cases (mainly COPD). In total, 1114 premature deaths were considered. The estimates proposed for 2002 were extrapolated to 2004.

Similar to the method used to calculate radon-related cancer cases, the number of yearly cases of lung cancer attributable to ETS indoors was calculated as follows: 30,651 / 26,624 × 152 = 175.

In 2004, 510 individuals died of myocardial infarction related to indoor ETS. Based on the CepiDC data and following De Peretti et al. (2012a), the ICD10 nomenclature identifies myocardial infarction with the codes I21 (acute myocardial infarction), I22 (recurrent myocardial

infarction) and I23 (certain current complications following acute myocardial infarction). The number of deaths related to these codes was 22,268, 64 and 0, respectively, for a total of 22,332 deaths for these three codes for 2004. The same year, there were 58,270 hospitalizations for myocardial infarction in France (De Peretti et al. 2012a). Thus, the incidence/death ratio for myocardial infarction is 58,270/22,332 = 2.61. Therefore, the incidence of myocardial infarction related to passive ETS exposure indoors was estimated as 1331 cases (510 deaths × 2.61).

Three hundred ninety-two individuals died from a stroke related to exposure to passive ETS indoors. According to De Peretti et al. (2012b) and the CepiDC data, 29,414 stroke deaths, and 88,515 hospitalizations for stroke were recorded in 2004. Applying the ratio incidence/deaths (88,515/29,414 = 3.01), 1180 cases (392 deaths × 3.01) of stroke caused by exposure to indoor ETS were estimated in 2004 in France.

Additionally, 60 individuals died from chronic bronchitis in 2004. According to a report by the Ministry of Health (Ministère de la Santé et des Solidarités, 2005), COPD causes 16,000 deaths in France each year, and 40,000 new patients are admitted every year for long-term treatment. The ratio of incidence/deaths is estimated at 2.5. Based on 60 COPD deaths in 2004, 150 new cases (60 deaths × 2.5) were diagnosed with COPD through exposure to passive ETS indoors.

The average age of death from myocardial infarction (ischemic heart disease) was 77 years in the general population between 2004 and 2006 according to CepiDC, leading to an average of 3 life-years lost. The average age of death is 80 years for strokes, resulting in a number of years of life lost equal to zero. COPD deprives individuals of a year of life because death from COPD occurs at an average age of 79 years old (Godard, 2007). The survival time after COPD diagnosis is estimated as 12 years on average (Antó et al., 2001; Chapman et al., 2006; Dal Negro et al., 2007).

#### 3.2.4. Trichloroethylene

The excess collective risk (ECR) resulting from indoor trichloroethylene exposure among the French population was calculated based on the principles of quantitative health risk assessment and using Eq. (7).

$$\mathrm{ECR} = \sum \left( \text{inhalation unit risk}_{\text{adjusted}} \times \text{median indoor air concentration} \times F \times \text{number of exposed individuals}_{\text{age-class}} \times \frac{\text{median age}_{\text{class}}}{80} \right) \quad (7)$$

The inhalation unit risk proposed by the US EPA for kidney cancer was used (U.S. Environmental Protection Agency, 2011). Kidney cancer is the sole form of cancer that the International Agency for Research on Cancer estimates to have a causal relationship to trichloroethylene exposure (IARC, 2012). The value is $1.10^{-6}$ per ($\mu g\ m^{-3}$). Furthermore, the US EPA addresses population classes that may have increased susceptibility. The data on trichloroethylene are limited. Nevertheless, in agreement with the guidelines on the susceptibility of young children to carcinogens (U.S. Environmental Protection Agency, 2005) and based on the genotoxic mode of action of this substance, additional uncertainty factors (age-dependent adjustment factors) were applied in accordance with US EPA guidelines.

The median indoor air concentration in French dwellings was 1.0 $\mu g\ m^{-3}$ (Kirchner et al., 2007). This value was considered to be the same for all other indoor environments examined (i.e., concentrations inhaled over the entire period spent indoors).

The time adjustment factor F is the average time spent indoors for the general population, i.e., 90%.

The number of individuals in France reported by the INSEE census for 2004 was used to estimate the number of premature deaths associated with kidney cancer resulting from trichloroethylene indoor exposure. For each age class, an adjustment is introduced based on the average number of years of exposure to trichloroethylene divided by the average life expectancy, assumed to equal 80 years. The detailed calculation is provided in Table S1 in the supporting information. In 2004, 15 kidney cancer cases were caused by exposure to trichloroethylene in indoor air.

INCa (2007) indicates 3052 annual deaths from cancer of the urinary tract, including kidney cancer, in France. The associated annual morbidity is 8293 cases. Based on the ratio of incidence/deaths, 15 kidney cancer deaths correspond to a yearly incidence of 41 cases. For kidney cancer, individuals lose an average of 15 years of life, i.e., the average age of death is 65 years (INCa, 2007). A 1.5-year survival time is considered (INCa, 2007).

#### 3.2.5. Benzene

The excess collective risk associated with benzene exposure was also determined using Eq. (7). The inhalation unit risk proposed by the World Health Organization (2000) for acute leukemia was used: $6.10^{-6}$ per ($\mu g\ m^{-3}$). The median indoor air concentration in French dwellings was 2.1 $\mu g\ m^{-3}$ (Kirchner et al., 2007). This value was considered to be the same for all other indoor environments. The time adjustment factor F is the average time spent indoors for the general population, i.e., 90%. The number of individuals in France reported by the INSEE census for 2004 was used to estimate the number of premature deaths associated with acute leukemia resulting from indoor benzene exposure. In France, according to the 2004 INSEE census, 25% of the population is under 20 years, 27% is between 20 and 39 years, 27% is between 40 and 59 years, 13% is between 60 and 74 years, and 8% is over 75 years. For each age class, an adjustment is introduced based on the average number of years of indoor exposure to benzene divided by the average life expectancy, assumed to equal to 80 years.

The detailed calculation is provided in Table S2: in 2004, 342 acute leukemia cases were caused by exposure to benzene in indoor air.

Based on the ratio of incidence/deaths from acute leukemia in 2005 in France (3082/2733), an incidence of 385 cases attributable to indoor benzene exposure was calculated. The average number of years of life lost is 15 for cancer of the lymph tissue (INCa, 2007). Because of the lack of specific data for leukemia, this figure of 15 years of life lost on average was used. A 15-year survival time was considered (INCa, 2007).

#### 3.2.6. Particulate matter: $PM_{2.5}$ fraction

$PM_{2.5}$ health impacts were determined based on the relative risks reported in epidemiological studies relative to ambient air pollution and based on the approach employed in the European Aphekom study (Declercq et al., 2012). The major effects are associated with long-term exposure, and thus were the effects that were considered.

Several assumptions were made. First, time-series studies were used to explore the exposure-risk relationships established for the ambient air between particles, defined as indicators of urban air pollution and measured at urban-background monitoring stations, and health indicators (morbidity or mortality). Thus, it was assumed that the effects associated with urban particulate matter are analogous with those of indoor air particles. This assumption is based on the fact that a large proportion of indoor particles comes from outdoors, particularly in non-smoking buildings (Lai et al., 2006; Meng et al., 2009; Chen and Zhao, 2011).

Second, the median indoor air concentration in French dwellings was measured in non-smoker dwellings because the impacts of ETS, which includes particles emitted through smoking, were treated separately. Although the nature, composition and size distribution of particles in urban areas differ from those in rural areas because of the different emission sources, the risk-exposure relationship was applied to the entire population using a conservative approach. However, 61 million people, i.e., 95% of the French population, live within the influence of cities, according to INSEE.

Health impacts were calculated for the population aged 30 years and over and using the same RR for all age groups (Declercq et al., 2012). The calculations were limited to the adults because the RRs used

were derived from data collected by the American Cancer Society as part of the Cancer Prevention Study II, an ongoing prospective mortality study of approximately 1.2 million adults (Pope et al., 2002, 2004). Consequently, the RRs are relevant and applicable to the population aged 30 years and over. More precisely, the number of deaths attributable to particle exposure indoors ($\Delta y$) was calculated using Eq. 8, as follows:

$$\Delta y = F \times y_0 \times (1 - e^{-\beta \Delta x}) \quad (8)$$

with:

$F$ = the average time that the population spends in indoor environments, i.e., 90%

$y_0$ = the annual number of deaths depending on the cause extracted from the CepiDC death census. In France in 2004, the annual number of deaths, excluding violent ones, was equal to 470,422. The following figures were used for the annual number of deaths of the population aged 30 years and over: 464,364 (total mortality, excluding violent deaths, ICD10 codes: A00-R99), 149,892 (deaths from cardiovascular causes, ICD10 codes: I00 to I99) and 27,028 (deaths from lung cancer, ICD10 codes: C34).

$\Delta x$ = the difference between the median concentration and the 5th percentile of the $PM_{2.5}$ concentration distribution in non-smoker dwellings. This quantile was used to represent a background indoor concentration of $PM_{2.5}$ without specific anthropogenic outdoor and indoor sources, considering that particles are also emitted by natural processes. These concentrations were equal to 15.2 and 8.4 $\mu g\, m^{-3}$, respectively. Therefore, $\Delta x$ is equal to 6.8 $\mu g\, m^{-3}$.

$\beta$ = the coefficient of the exposure-response function. In practice, $\beta = \log(RR_{10}) / 10$, wherein $RR_{10}$ is the relative risk for a 10-$\mu g\, m^{-3}$ increase in the concentration of the pollutant considered. The mean RRs for a 10-$\mu g\, m^{-3}$ increase in the $PM_{2.5}$ concentration from Pope et al. (2002, 2004) were selected; these included a death from all causes RR = 1.06 [1.02–1.11], a death from cardiovascular causes RR = 1.12 [1.08–1.15] and a death from lung cancer RR = 1.14 [1.04–1.23].

The results for particle-related mortality are presented in Table S3: in 2004, 16,236 individuals aged 30 years and older died as a result of exposure to particles in indoor environments. A proportion of the 4156 residual deaths ('deaths from all causes') remained unexplained. However, this proportion was needed to calculate the costs and, in turn, to determine the associated disease. Respiratory diseases were assumed to be the cause of these deaths; COPDs were considered to represent this group of diseases. Thus, three diseases were assumed to be associated with indoor exposure to particulate matter and were considered in the calculations: lung cancer, cardiovascular diseases and COPD.

As for radon, it is possible to estimate that the annual lung cancer incidence caused by exposure to particles indoors was 2388 cases. Regarding the cardiovascular morbidity, the incidence of cardiovascular diseases was assumed to be equal to the number of deaths. Regarding COPD, as mentioned for ETS, the ratio of incidence/deaths is 2.5. Therefore, for the 4156 COPD deaths in 2004, there were 10,390 new patients diagnosed with COPD caused by exposure to particles in indoor air.

The average age of death from lung cancer is 69 years. Regarding cardiovascular diseases, the assumptions are the same as those for ETS, i.e., an average age of death of 77 years. The average age of death from COPD is 79 years, as reported for ETS.

Table S4 provides the calculated morbidity and mortality in 2004 related to exposure to particles in indoor air.

### 3.3. Socio-economic cost evaluation

The number of deaths in 2004 and the number of life-years lost used to calculate the external cost of mortality for each disease are presented in Table 1. For the calculation of the external cost of morbidity, the number of new cases of each disease in 2004 and the number of life-years with the disease are reported in Table 1; the respective average disability weights are in Table S5. The production loss per capita and per disease used to calculate the external cost of production loss is presented in Table S5. Regarding the impact on public finances, the cost of medical treatment per case for each disease is reported in Table S5. Finally, the number of lost retirement years for each disease used to calculate the unpaid pensions is reported in Table 1.

The detailed and overall socio-economic costs are presented in Table 2. Indoor pollution was estimated to cost approximately €20 billion in France in 2004.

The main factors contributing to costs include intangible costs related to premature deaths, quality of life losses and production losses. These external costs represent nearly all of the total costs, and the impacts on public finances represent < 1% of the total cost. Overall, mortality and morbidity costs have the same effect (46% of the external costs). However, depending on the pollutant and its associated health outcomes, the mortality costs are higher (CO, trichloroethylene, radon), lower (ETS) or approximately equal (benzene, $PM_{2.5}$) to the morbidity costs.

Particles contribute the most to the total cost (approximately €14 billion). Moreover, radon, a natural soil pollutant found in several French regions, also has a significant health cost (€2.7 billion). The pollutants' relative contributions to costs are ranked as follows: particles > radon > ETS > benzene > carbon monoxide > trichloroethylene.

## 4. Discussion

This socio-economic evaluation is the first one performed for the French population. This evaluation is based on various assumptions and methodological choices, which are discussed below. These limitations are described to identify future methods that may improve assessments of health costs related to indoor air pollution exposure.

### 4.1. Selected pollutants

The pollutants were selected based on the availability of the basic data needed for the health impact assessment: the number of annual deaths, indoor air concentrations, dose-response relationships, and fractions attributable to diseases. The exclusion of certain pollutants commonly found in indoor air that have well-known health effects may raise questions. For example, the health ranking conducted to identify priority pollutants in indoor air in dwellings (Alméras, 2010; Logue et al., 2011) highlights formaldehyde and acrolein, which do not appear in this study because of a lack of published dose-response relationships on effects with a dose-threshold mode of action. More generally, pollutants with a dose-threshold mode of action were excluded because of a lack of related publications and valid dose-response relationships over the threshold needed to quantify health impacts. Thus, the studied health effects mainly included genotoxic carcinogenic effects with a non-threshold mode of action. Our selection of pollutants was also constrained by a lack of representative measurements for indoor environments in France (e.g., in the case of asbestos and mold). Dose-response relationships are available for ozone and nitrogen dioxide, and these pollutants were highlighted by Logue et al. (2012). However, they were not measured in French dwellings and could not be considered in this study.

### 4.2. Health impact assessment

The health impact estimation method used was based on quantitative health risk assessments or health impact calculation principles, depending on the available data. Other approaches exist, such as those that assess the BOD developed by the WHO. The WHO recommends the following two methods for calculating BOD values: a health event-based

**Table 1**
Estimate of the health impacts associated with exposure to each of the six target indoor air pollutants, France, 2004.

| Pollutant | Associated health effect | Number of years with the disease | Average age at death | Number of years of life lost[a] | Number of years of pension lost[b] | Morbidity incidence | Number of deaths |
|---|---|---|---|---|---|---|---|
| Benzene | Leukemia | 15 | 65 | 15 | 15 | 385 | 342 |
| Trichloroethylene | Kidney cancer | 1.5 | 65 | 15 | 15 | 41 | 15 |
| Radon | Lung cancer | 1.5 | 69 | 11 | 11 | 2388 | 2074 |
| Carbon monoxide | Asphyxia | 0 | 33 | 47 | 20 | – | 98 |
| Particles | Lung cancer | 1.5 | 69 | 11 | 11 | 2388 | 2074 |
| | Cardiovascular | 13 | 77 | 3 | 3 | 10,006 | 10,006 |
| | COPD[c] | 12 | 79 | 1 | 1 | 10,390 | 4156 |
| Environmental tobacco smoke | Lung cancer | 1.5 | 69 | 11 | 11 | 175 | 152 |
| | Infarction | 13 | 77 | 3 | 3 | 1331 | 510 |
| | Stroke | 11 | 80 | 0 | 0 | 1180 | 392 |
| | COPD[c] | 12 | 79 | 1 | 1 | 150 | 60 |

[a] Considering the life expectancy of the general population equal to 80 year old (Pison, 2005).
[b] Considering the age of retirement equal to 60 year old.
[c] Chronic obstructive pulmonary disease.

**Table 2**
Costs (millions €) of indoor air pollution exposure for the target pollutants, France, 2004.

| | Benzene | Trichloroethylene | Radon | CO | Particles | ETS | Total |
|---|---|---|---|---|---|---|---|
| External costs | | | | | | | |
| Premature death | − 453 | − 19.6 | − 2089 | − 237 | − 5760 | − 322 | − 8880 |
| Quality of life loss | − 383 | − 6.7 | − 309 | 0 | − 7350 | − 837 | − 8886 |
| Lost productivity | − 38 | − 1.5 | − 282 | − 72 | − 1102 | − 85 | − 1580 |
| *Total external costs* | − 874 | − 27.8 | − 2680 | − 309 | − 14,212 | − 1244 | − 19,347 |
| Public finances | | | | | | | |
| Health | − 18 | − 2.9 | − 61 | − 3 | − 236 | − 37 | − 358 |
| Research | * | * | * | * | * | * | − 11 |
| Unpaid pensions | 10.7 | 0.45 | 49 | 4 | 136.5 | 88 | 289 |
| *Raw costs for public finances* | − 7.3 | − 2.45 | − 12 | 0.9 | − 99.5 | − 29 | − 80 |
| *Total costs for public finances* | − 8.8 | − 2.9 | − 14.4 | 1.1 | − 119.4 | − 35 | − 96 |
| Total costs | − 883 | − 30.7 | − 2694 | − 308 | − 14,331 | − 1182 | − 19,443 |

* : overall evaluation for all the selected pollutants; CO: carbon monoxide; ETS: environmental tobacco smoke.

approach (calculations are based on the incidence and/or prevalence of health events) and the exposure-based approach (calculations are based on population exposure levels). These approaches estimate health impacts using the DALY method for different health outcomes (De Hollander, 2004). In addition to issues associated with the relevance of these data to the French population (Inserm, 2011), converting these data into monetary terms allows the calculation of the external costs of the studied diseases, i.e., one of the two dimensions covered under the socio-economic cost approach.

Regarding health effects, we studied individual substance effects and thus did not consider potential co-exposure effects, including addition, antagonism, potentiation, and synergies. Moreover, the incidence of a disease and its related deaths could be the result of co-exposure to multiple pollutants, e.g., lung cancer could be related to concomitant exposure to radon, particles and ETS. When pollutants are considered individually and then their respective impacts are summed, the incidences of a disease and the related deaths may be counted several times because of co-exposure.

One health effect or partial effect was considered for each pollutant because no published dose-response relationships are available for associated effects. Furthermore, with the exception of CO effects, only the effects associated with long-term exposure were considered, and the effects associated with short-term exposure were not considered. This focus ultimately resulted in an underestimation of the health costs resulting from indoor air pollution.

The mechanism of action could result in cumulative exposure at levels that differ between indoor and outdoor environments. For this evaluation, it was assumed that the development of a disease related solely to exposure in indoor environments was proportional to the time spent in these environments.

Regarding exposure levels, within the framework of this first-tiered approach, calculations were based on the median concentrations in indoor air without using the entire distribution of concentrations because this was considered the best estimate of the situations encountered when extreme situations were excluded. In addition, because of a lack of available data at the national level, the air concentrations measured in dwellings (bedroom or living room) were equated to the air concentrations found in all indoor environments. This hypothesis may have a minor impact; benzene, trichloroethylene and $PM_{2.5}$ indoor air concentrations in French schools ($n$ = 51 classrooms) and office buildings ($n$ = 36 office rooms) showed the same orders of magnitude as the medians used in the present study (Canha et al., 2016; Mandin et al., 2017). Moreover, an assumption of continuous exposure to each pollutant was made for the same concentration level during the years prior to the target year of 2004, although indoor air pollution evolution patterns are known (Weschler, 2009). However, at present, this is a common hypothesis within the framework of risk assessment.

Regarding our health impact calculations, the calculation of the ratio of morbidity to mortality also presents limitations because of missing information, e.g., for cancer subtypes, such as acute non-lymphocytic leukemia and acute myeloid leukemia. Moreover, this ratio does not consider age variations or the nature of each pollutant, which can affect the duration of cancer symptoms. Finally, there are uncertainties regarding the average age at death and regarding life expectancy following the development of the studied diseases because the figures do not specifically pertain to diseases that are strictly related to exposure to the studied pollutants.

### 4.3. Specific issues for certain pollutants

Certain authors have discussed the exposure-risk relationship between benzene exposure and the onset of leukemia (Finkelstein, 2000; Richardson, 2008; Vlaanderen et al., 2011). In this study, the assumption of linear extrapolation to the origin used by the WHO was adopted.

Regarding trichloroethylene, the development of renal cell carcinomas was considered. Other types of cancer should also be considered; for instance, the International Agency for Research on Cancer concluded that there is limited evidence in support of an association between exposure to trichloroethylene and the development of liver cancer and non-Hodgkins lymphoma (IARC, 2012).

Regarding particulate matter, effects associated with indoor air particles were considered to be analogous with well-established effects of urban particulate matter based on changes in PM concentrations measured at urban-background monitoring stations. Moreover, an average estimate was used, whereas the confidence interval [5590; 28,630] for the number of deaths from 'all causes' other than violent deaths (> 30 years) related to particulate matter emphasizes the ambiguities in the results. A proportion of the deaths related to 'all causes' remains unexplained and is assumed to be related to respiratory diseases. To determine the incidence level and the average age at death, these respiratory deaths were equated to COPD. Furthermore, the incidence of cardiovascular diseases was considered to be equal to the number of deaths to facilitate our calculations. Lastly, only adults > 30 years of age were considered. Nevertheless, considering the number of all-causes deaths in the French population under 30 years in 2004 (6058 versus 464,364 for adults > 30), this hypothesis likely had a low impact on the results.

Regarding ETS, the most recent data from the 2006 "Lifting the smokescreen" report by Hill in 2011 were used. Estimates were made for 2002. Because of a ban on smoking in workplaces and public buildings that has been in effect since February 2007, the 1114 estimated deaths per year (including 107 work-related deaths) led to an overestimation relative to the present situation. In parallel, the number of deaths was limited to four risks (infarction, stroke, lung cancer, and chronic respiratory diseases) and strictly applies to the adult population.

### 4.4. Economic impact assessment

From an economic perspective, several limitations can be discussed. First, we attempted to estimate the costs of indoor air pollution to society. Thus, private investments made to improve indoor air quality levels or the quality of life of patients were not included in our estimations.

The reference value of a statistical life-year is based on a figure of €115,000, as proposed by Quinet et al. (2013). This estimate is derived from a calculation in which the exploratory approach was stressed by the authors and which is not specific to indoor air pollution.

In addition, adjustments to the cost of a life-year were based on a single rate of 4% (Lebègue et al., 2005), which was independent of the nature of the cost to be estimated. Quinet et al. (2013) emphasized the relevance of using differentiated adjustment rates depending on the time horizon considered or the nature of the morbid consequences of a disease.

Morbidity cost calculations differ from mortality cost calculations in their weighting of the quality of life loss. According to the WHO, a disability weight is a weight factor that reflects the severity of a disease on a scale from 0 (perfect health) to 1 (equivalent to death) and that is used to calculate DALY values. Several tools that calculate this weight across various disciplines exist, such as econometrics and psychometrics. Such tools are essentially based on questionnaires that are administered to patients. These weights can be developed for various diseases (in a general approach) or can be specific to a given disease. Moreover, these factors are based on assessments that use scores that occasionally require a subjective evaluation. These factors are not specific to France and are likely to vary among health care systems. Another source of uncertainty related to morbidity arises from assumptions that were adopted because of a lack of available data. As an illustration, the monetary value of elements comprising the costs of treatment resulting from CO poisoning was estimated by the authors. In future studies, it would be useful to estimate this cost more precisely.

The loss of production related to cancer was determined using INCa (2007) method and data. It was assumed that the exposure profile and individual responses to a cancer diagnosis were homogeneous. However, the exposure events that induce these cancers can exhibit significant geographic fluctuations. For instance, radon is present in certain areas, such as Brittany (West Coast), Franche-Comté (East), Centre, and Corsica. The socio-economic levels in these regions vary; therefore, the average production loss per individual is also likely to vary. Moreover, the production losses caused by diseases other than cancer were hypothesized. Finally, production losses may also be related to productivity losses on the job resulting from deteriorated health. This point was not considered because the production losses considered were limited to losses related to absenteeism.

Lastly, the marginal cost of public finances α was set at 0.2. This estimate was validated by the Quinet report and based on the work of Beaud (2008). Using a general equilibrium model, the author provides an analytical expression of α and estimates the overall α. Estimates may vary depending on the elasticities of the labor supply and the demand for goods. Maurice and Roquigny (2013) reviewed the α calculated for 14 European countries, which ranged from 1.07 in Spain to 1.32 in Belgium. Considering the low contribution of public expenses to the overall cost of indoor air pollution, the impact of α remains low.

### 4.5. Sensitivity analysis

To illustrate the range of the socio-economic cost related to the burden of disease induced by exposure to indoor air pollution, a probabilistic approach focusing on particles has been developed; the results are reported in the supporting information. Particles were selected because they represent the largest proportion of the calculated total socio-economic cost, i.e., approximately 75%; consequently, they may widely influence the results. The sensitivity analysis highlighted a large interval directly correlated with the variability and uncertainties related to the input parameters and the selected assumptions. The value of €14.3 billion reported here corresponds to the 35th percentile of the distribution, with a mean and a 95th percentile of €15.1 and €18.3 billion, respectively.

### 4.6. Comparisons with other evaluations

The results of similar studies are reported in Table 3. The results are not strictly comparable across studies because of the use of different methods, assumptions, selected pollutants, health outcomes, reference years, populations, and cost perimeters. In particular, the selected pollutants differ from one study to another and never represent all of the indoor air pollutants.

Nevertheless, common findings can be derived from these different studies, such as the following: i) the important impacts of indoor air pollution on health and, consequently the socio-economic costs to the community, regardless of the study perimeter; ii) the significant health burden attributable to particles (Jantunen et al., 2011; Hänninen et al., 2014) and corollary to outdoor air pollution, which contributes to a large portion of indoor particulate matter; and iii) the lack of information available for complete and accurate evaluations of health outcomes, dose-response relationships, and exposure levels.

Moreover, a rough comparison of common pollutants can be performed across studies. DALYs can be calculated from our study by adding the number of deaths multiplied by the number of years of life

**Table 3**
Review of published studies that report costs of indoor air pollutant exposure.

| Reference | Target pollutants | Health endpoints | Calculation method |
|---|---|---|---|
| Jantunen et al. (2011) | Bioaerosols, carbon monoxide, radon, VOCs, combustion products | Allergies and asthma<br>Lung cancer<br>COPD<br>Respiratory infections<br>Cardiovascular morbidity and mortality<br>Odor perception and irritations (sick building syndrome) | DALY method: attributable fraction of diseases associated to the indoor air pollutant exposure |
| Braubach et al. (2011) | Radon<br>Environmental tobacco smoke<br>Lead<br>Carbon monoxide<br>Formaldehyde<br>Dampness and mold | Lung cancer<br>Respiratory infections, asthma, cardiovascular diseases and lung cancer<br>Cardiovascular diseases in adults<br>IQ loss in children<br>Poisoning<br>Respiratory infections in children<br>Asthma in children | DALY method: attributable fraction of diseases associated to the indoor air pollutant exposure |
| Logue et al. (2012) | $PM_{2.5}$<br>Carbon monoxide<br>$NO_2$<br>$O_3$<br>$SO_2$ | Mortality all causes, chronic bronchitis, and non-fatal stroke<br>Hospital admissions for: asthma, lung disease, dysrhythmias, and heart failure<br>Hospital admissions for: respiratory issues, congestive heart failure, ischemic heart disease, and respiratory illness<br>Hospital admissions for asthma, lung disease, respiratory infection, and dysrhythmias<br>Hospital admissions | Intake–incidence–DALY method that uses epidemiology-based concentration–response functions to quantify disease incidence rates |
| Schram-Bijkerk et al. (2013) | Dampness<br>CO<br>Radon/thoron<br>Formaldehyde<br>Environmental tobacco smoke | Upper respiratory tract symptoms<br>Lower respiratory tract symptoms in children<br>Asthma incidence<br>Hospital admissions<br>Lung cancer<br>Asthma in children<br>Lower respiratory tract symptoms in children<br>Asthma incidence<br>Ischemic heart disease<br>Sudden infant death syndrome<br>Otitis media in children<br>Lung cancer | DALY method: attributable fraction of diseases associated to the indoor air pollutant exposure |
| Hänninen et al. (2014) | Benzene<br>Environmental tobacco smoke<br>Formaldehyde<br>Lead<br>$O_3$<br>$PM_{2.5}$<br>Radon | Leukemia<br>Trachea, bronchus, and lung cancers in non-smoker adults<br>Ischemic heart disease in non-smoker adults<br>Asthma in non-smoker adults and children<br>Lower respiratory infections in infants<br>Otitis media in toddlers<br>Asthma aggravation in children<br>IQ loss in children<br>Mild mental retardation in children<br>Hypertensive diseases<br>Increased blood pressure<br>Total mortality (non-violent) in adults<br>Minor restricted activity days in workers<br>Cough days in children<br>Lower respiratory symptom in children (excluding cough)<br>Cardiopulmonary disease in adults<br>Lung cancer in adults<br>Chronic bronchitis in adults<br>Restricted activity days in workers<br>Lung cancer | DALY method: attributable fraction of diseases associated to the indoor air pollutant exposure |

lost to the number of sick people multiplied by the number of years with disease. Considering a French population of 62 million in 2004, such a calculation leads to DALYs equal to 176 for benzene, 426 for radon, 5084 for $PM_{2.5}$, and 574 for ETS per 1000,000 people. For France, Hänninen et al. (2004) obtained the following DALY per 1000,000 people: 3.4 for benzene, 1146 for radon, 4572 for $PM_{2.5}$, and 550 for ETS. In the Netherlands, Schram-Bijkerk et al. (2013) calculated DALYs per million people equal to 500 for radon and between 313 and 2188 for ETS. At the European level, Jantunen et al. (2011) calculated DALYs of 346 for radon and 2851 for particles per million people (total figures reported for 500 million Europeans). Despite the limitations, the orders of magnitude across studies are rather consistent.

## 5. Conclusion

The objective of this study was to determine the availability of data to assess the socio-economic costs of indoor air pollution in France and to determine the magnitude of these costs based on these data. According to the method used and the assumptions made, the costs to the community that are attributable to indoor exposure to six air pollutants amount to approximately €20 billion per year in France. The

costs associated with particles represent 75% of the overall cost. Despite the limitations, the results of the present study are consistent with the orders of magnitude reported in previous national or European evaluations.

Such a study implies many assumptions and choices at each step of the process. To refine the evaluation, knowledge of dose-response relationships and additional data on population indoor exposures are needed. The probabilistic approach and sensitivity analysis, which were roughly performed for particles, should be systematized. A refined evaluation in the future could also serve as a baseline for cost-benefit analyses of possible strategies for reducing indoor air exposure levels, with the objective of identifying the most efficient strategy. These results can nevertheless be used to support public health policies; they confirm that it is imperative for researchers and policymakers to continue addressing indoor air quality issues.

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.envint.2017.03.025.

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