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Ranking indoor air health problems using health impact assessment

final report

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LIST OF ABBREVIATIONS AND ACRONYMS

AIRMEX	European Indoor Air Monitoring and Exposure Assessment Project
AIVC	Air Infiltration and Ventilation Centre
APHEA	Air Pollution and Health: a European Approach
APHEIS	Air Pollution and Health, an European Information System
CAFE	Clean Air for Europe
CBA	Cost Benefit Analysis
CEN	European Committee for Standardization
ECA	European Collaborative Action
ECRHS	European Community Respiratory Health Survey
EHAP	the Commission’s Environment and Health Action Plan
EL	Exposure Limit
ENHIS	European Environment and Health Information System
ENVIE	Co-ordination action on indoor air quality and health effects
ERF	Exposure Response Function
ETS	Environmental Tobacco Smoke
EWGIA	European Working Group on Indoor Air
EXPOLIS	Air pollution exposure distributions of adult urban populations in Europe
EXTERNE	Externalities of Energy
GC-FID/MS	Gas Chromatography- Flame Ionization Detection/ Mass Spectrometry
GC-LC	Gas Chromatography- Liquid Chromatography
HIA	Health Impact Assessment
IA(Q)	Indoor Air (Quality)
IARC	International Agency for Research on Cancer
INDEX	Critical Appraisal of the Setting and Implementation of Indoor Exposure Limits in the EU
ISAAC	International Study of Asthma and Allergies in Childhood
ISO	International Organization for Standardization
LOAEL	Lowest Observed Adverse Effect Level
NEHAP	National Environmental Health Action Plan
NOAEL	No Observed Adverse Effect Level
PEOPLE	Population Exposure to Air Pollutants in Europe
PINCHE	Policy Interpretation Network for Children's Health and Environment
PM	Particulate Matter
RA	Risk Assessment
SCHER	Scientific Committee on Health and Environmental Risks
SIDS	Sudden Infant Death Syndrome
THADE	Towards Healthy Air in Dwellings in Europe
US-EPA	US Environmental Protection Agency
VOCs	Volatile Organic Compounds
WHO	World Health Organization

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0 EXECUTIVE SUMMARY

The Commission's Environment and Health Action Plan 2004-2010 (EHAP)[1] contains a specific action on indoor air. With this action the Commission wants to prepare policies to tackle exposure to environmental tobacco smoke (ETS), and to develop networks and guidelines on other factors affecting indoor air quality by using research and exchange of best practice. This report aims at including the existing information from scientific research, from Member States and from stakeholders to establish the priority indoor air pollutants in private and public spaces, to assess the risks and health impacts associated with these priority pollutants, and to evaluate the current Member States' policies and monitoring systems for indoor air quality. Recommendations to fill the gaps in knowledge and information, to reduce the uncertainties, and recommendations for policy interventions at EU level have been elaborated in order to enable a European science based policy on indoor air. This study summarizes the available evidence from literature, from European research projects (e.g. INDEX, THADE, EXPOLIS and studies from the *European Collaborative Action on "Urban air, indoor environment and human exposure"* (ECA) including some ongoing studies like AIRMEX and ENVIE, …). Member States' policies and expertise on this matter was gathered and discussed during a workshop involving over 40 participants from EU Member states, the Commission and the European Working Group on Indoor Air (EWGIA). A good EU geographical coverage was obtained, also including New Member States.

Key messages are:

1. There is a consensus on a cross-section of priority pollutants: ETS, formaldehyde, CO, particles ($PM_{2.5}$ and PM_{10}), NO_2, benzene, naphthalene, moulds and mites, dampness/moisture, CO_2 (measure for ventilation) and radon.
2. Participation and consensus is the road to follow, for which the EWGIA offers a good platform.
3. The development of European guideline values or limit values for this set of pollutants should be considered.
4. A harmonized monitoring approach should be developed for chemical pollution and ventilation (CO_2) in schools. Member States should be encouraged to monitor microbial contamination in hospitals, and care centres for the elderly. More knowledge on the acute exposure in different transport systems needs to be developed before implementing more elaborated monitoring schemes.
5. Member States should be consulted on how to tackle indoor moulds and dampness in existing private residences.
6. An information and communication campaign should be developed aiming to reduce children's exposure to ETS in private residences. The momentum and success story from the smoking ban should be used to tackle this hidden problem.
7. The basic tools and instruments should be harmonised at EU level: emission testing protocols, labelling schemes, monitoring strategies according to existing ISO, CEN standards and following ECA proposals.

[1] COM(2004)416 final adopted by the Commission in June 2004.

8. The different directives related to indoor air quality should be integrated in a common framework, supported by guidelines or limit values to ensure that the risks due to total exposure indoors are minimal.
9. European research including indoor air in epidemiological studies and in European human biomonitoring studies should be initiated, using harmonised protocols for exposure assessment, exposure modelling and analysis.

Priority pollutants

Taking into account (1) the conclusions of several EU-funded studies and actions in the domain of indoor air quality and human health (INDEX, THADE, SCHER opinion, EWGIA, WHO indoor air working group), and (2) the opinion of workshop participants, there is a general consensus on the priority pollutants: ***ETS, formaldehyde, CO, particles (PM2.5 and PM10), NO_2, benzene, naphthalene, moulds and mites, dampness/moisture, CO_2 (measure for ventilation) and radon***. This prioritization relies on the existing body of evidence, more than on a formal health impact assessment. A formal health impact assessment is however crucial to include the health impacts of indoor air pollution and the benefits of indoor air quality interventions in a cost-benefit assessment, as part of an impact assessment of policies.

Health impact assessment

Previous health impact assessments have been carried out for ETS, radon and dampness/moulds. These studies revealed a significant health burden for these factors: it was estimated that 72 000 people in the EU 25 die each year due to the exposure to ETS at home. In 2006, 21 000 lung cancer deaths occurred in the EU 25 due to radon and 13 % of childhood asthma in developed countries could be attributable to moulds and dampness. For other (chemical) indoor pollutants, a health impact assessment and a meta-analysis of exposure response functions (ERF) across various epidemiologic studies is lacking. Using available epidemiological studies, in combination with existing exposure information and health data in a health impact assessment confirms the set of pollutants. For carcinogenic substances ETS is dominant in comparison to other pollutants, followed by radon. Care should be taken with the fact that radon increases the mortality risk from lung cancer in smokers. From our assessment the carcinogenic impact from formaldehyde seems more important than that of benzene. The incompleteness of ERF and the incomparability of endpoints and methods in current epidemiological studies (often from outside the EU) hampers the ranking of indoor priorities, especially for morbidity effects. To overcome these constraints, there is a need to extend the database of ERF by EU-wide epidemiological assessments. For such assessments, it is strongly advised to apply harmonized protocols to select pollutants, indoor environments, to assess the exposure and to analyze the health outcomes. In addition, research into the dose-effect relationships of emerging or new pollutants should be encouraged. A formal health impact assessment of particles is not possible at this moment.

Risk assessment in public spaces

Standardized methods and instruments to perform risk assessment are available, but are not always applied in exposure and risk assessment studies. A priority list of public spaces is developed combining the general results from the INDEX risk assessment (which includes more than only public spaces), the input from the workshop, and information on those public spaces where sensitive groups spent time and the level of exposure combined with the duration of the exposure. ***Schools, hospitals, care centres for the elderly and to a lesser extent public transport are put forward as the most important public spaces contributing to personal exposure, and thus risk.*** Chemical pollution and poor ventilation is considered a priority topic for schools where children spent a lot of time. Hospitals and care centres are important with respect to microbial exposures and transport systems due to the high peak exposures that might occur.

Indoor air policies

Member States consider a healthy indoor environment as a very important topic. Implementation of European legislation, the National Environmental Health Action Plan (NEHAP) and specific national problems and priorities have encouraged policy interventions on indoor air in Member States. Generally, in most Member States voluntary measures are used for private spaces, and mandatory for public spaces. When mandatory measures are used for private places, they generally apply to new buildings and products. Legislation on indoor air quality is not a stand alone issue: it is regarded as part of a broader picture, i.e. in connection with legislation on health, housing, spatial planning, energy and sustainability, and in connection with communication and participation of stakeholders. The indoor air aspects of different laws and regulations could be embedded as building blocks in an indoor air framework, with EU indoor air guideline or limit values as a cornerstone. Legislation however is only useful when implemented, when enforced, e.g. when accompanied by an operating monitoring scheme. It is proposed to focus on the implementation of existing policies and legislation first, and to target some clear priorities EU-wide. This needs to be done while finding a balance between harmonisation and an equal approach across the EU, and the Member States' freedom of implementation.

The current ***strategy on ETS*** in the EU is very successful, but there is still a variation in degree of implementation between Member States. It seems that the stricter the rules, the more satisfying results in health improvement and in public approval. The success of the EU strategy to reduce ETS in indoor spaces could serve as an example for other IAQ problems and illustrate that acceptance of measures interfering with the individual's life is possible.

There is no scientific or methodological problem related to ***monitoring.*** There are numerous examples of monitoring in several EU countries, ranging from complaint-based interventions (e.g. Green ambulances in Belgium) to standardized measurement campaigns in schools in France or survey based monitoring schemes like in Germany. It is recommended however to continue working on a standardized approach to serve European wide policies, to enable comparisons and non-biased impact assessments.
It is generally recommended to tackle public spaces first, because it's easier to implement measures in public spaces. Nevertheless this is only an argument of feasibility and not of health importance. An assessment of the importance of exposures to certain pollutants in public spaces versus private homes is non-existing. This might create a mismatch between what's being done and what needs to be done. Especially children's exposure to ETS in indoor environments is of concern.

Finally it is necessary to communicate the fact that the majority of indoor air problems requires a "***do it yourself" solution***, with individuals understanding the risk, managing the risk and reducing the risk. Indoor air quality is too comprehensive to tackle by any legal construction. Policies should enable this type of risk management by defining what is ‘risky’, and by setting product standards, by information campaigns, and by stimulating innovation.

1 INTRODUCTION

1.1 Background and scope

The Commission's Environment and Health Action Plan 2004-2010[2] (EHAP) contains a specific action on indoor air. With this action the Commission wants to prepare policies to tackle exposure to environmental tobacco smoke (ETS), and to develop networks and guidelines on other factors affecting indoor air quality by using research and exchange of best practice. To this purpose an expert working Group has been set up to exchange information on best practices and to assist the Commission in the development of European programmes to reduce emissions from and lower exposures to a number of priority pollutants. A green paper has been published to discuss the policy options to reduce exposure to ETS[3]. Research is ongoing to examine the source - exposure - health impact chain of pollutants in the indoor environment.

1.2 Objectives

This report aims at including the existing information from science, from Member States and from stakeholders to inform the Commission on:

(a) the health impacts arising from the various contamination issues, including uncertainties, and to make recommendations for filling any information gaps;

(b) the key indoor air pollutants in homes and key public spaces across the EU, with an indication of the potential for intervention;

and, based on the Member States' current practice, to inform the Commission on:

(c1) the risks associated with the exposure to indoor air pollutants in public spaces;

(c2) the existing surveillance monitoring schemes of public spaces and private homes;

(c3) the implementation of exposure limits.

1.3 Approach and methodology

This study is based on information and insights gathered by (1) a literature review and (2) an expert workshop to provide the means to collate relevant information across Europe.

[2] COM(2004)416 final

[3] COM(2007) 27 final (Green Paper Towards a Europe free from tobacco smoke: policy options at EU level) adopted by the Commission in January 2007.

1.3.1 Literature review

The literature review focused on European studies such as INDEX[4], THADE[5], EXPOLIS[6],... and studies from the *European Collaborative Action on "Urban air, indoor environment and human exposure" (ECA)*, including some ongoing studies like AIRMEX[7] and ENVIE[8]. The INDEX study has been used predominantly, as it consists of the best available information on exposure and risks associated with several priority pollutants. The EXPOLIS data has been proven useful also to derive estimates of the health impact associated with these priority pollutants, since it reports distributions of exposure, and has used standardised protocols to collect data on a European scale. In addition, the peer-reviewed scientific literature was consulted using the search engines Web of Science, PUBMED and screening of air pollution related journals; and some 'grey literature' sources on the internet to find local or national information on e.g. monitoring campaigns.

1.3.2 Expert workshop

Information on the Member States' policies and expertise was collected, through a workshop in Brussels (29-30/03/2007) involving over 40 participants from Member States, scientists, European Commission delegates and members from the Expert Working Group on Indoor Air (EWGIA). Attention was given to geographical coverage, by including countries from each of the following regions: Southern Europe (Italy and Portugal), Northern Europe (Finland and Sweden), Western Europe (Germany, Belgium, the Netherlands), Eastern Europe (Bulgaria) and Central Europe (Hungary, Poland) (Figure 1). We aimed at including 4 New Member States. Representatives from Bulgaria, Poland and Hungary attended the workshop. Unfortunately, the Slovakian delegate was unable to attend the workshop, but provided input by means of completing written questionnaires. Conclusions from the workshop cannot be seen as a European-wide consensus between countries, the Commission and stakeholders, nor as an EWGIA consensus, because some countries or members were not present. Background information is given in Annex A.

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[4] the INDEX project. 2005 Critical Appraisal of the Setting and Implementation of Indoor Exposure Limits in the EU. EC, JRC. Kotzias et al.

[5] THADE: Towards Healthy Air in Dwellings in Europe, available at http://www.efanet.org/activities/documents/THADEReport.pdf

[6] EXPOLIS: Air pollution exposure distributions of adult urban populations in Europe, more info at http://www.ktl.fi/expolis/

[7] AIRMEX: European Indoor Air Monitoring and Exposure Assessment Project

[8] ENVIE, 6th FP. Co-ordination action on indoor air quality and health effects (http://indoorairenvie.cstb.fr)

Figure 1: Geographical coverage of EU Member States in the workshop.

1.4 Structure of the report

This study consists of two parts:

- Part 1 deals with the risk assessment, Member States' policies, recommendations on surveillance, monitoring and policy interventions (Chapter 2 + 3 + 4) with an emphasis on public spaces.
- In part 2, a health impact assessment (HIA) of indoor air pollution (Chapter 5) is developed, an analysis of knowledge and information gaps carried out, followed by uncertainties of a HIA (Chapter 6).

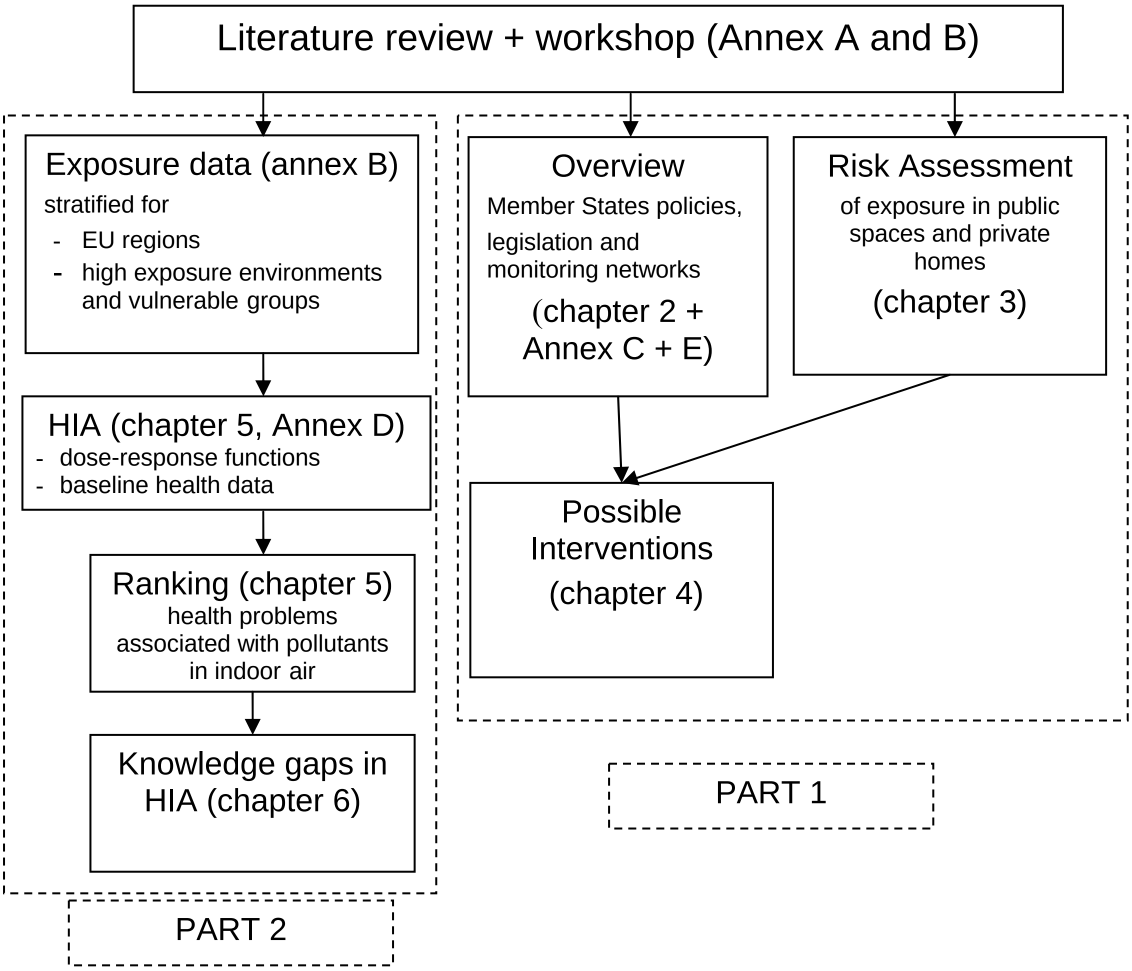

Figure 2: General structure of this report.

2 OVERVIEW OF MEMBER STATES' POLICIES, LEGISLATION AND MONITORING NETWORKS

2.1 Incentives for and key features of Member States' indoor air policy

Countries like Germany, Finland, Poland have a long tradition on indoor air quality policy making. Evidently European Directives play an important role:

- The Commission Recommendation on the protection of the public against indoor exposure to radon[9] and Commission Recommendation on the protection of the public against exposure to radon in drinking water supplies[10].

- Green Paper on Tobacco Smoke[11] and the World Health Organisation Framework Convention on Tobacco Control (WHO FCTC). The current strategy on ETS in the EU is fruitful. There is still a variation in the degree of implementation of this measure between the different EU Member States, but it seems that the stricter the rules, the more satisfying the results in health improvement and in public approval. The success of the EU strategy to reduce ETS in indoor spaces could serve as an example for other IAQ problems and illustrates the acceptance of measures interfering with the individual's life. Next to the smoking ban, Member States use financial stimuli (taxes) to restrict smoking in general.

- The energy performance of the buildings directive[12] and the gas- and heating appliances directives[13]. One of the more successful legislative actions seems to be ventilation standards. Member States have included minimum ventilation rates in their national building codes that apply to private and public spaces. But minimum ventilation rates have caused concern in e.g. Sweden and Finland, where the improved insulation has had a negative impact on effectiveness of radon policy measures.

- The Construction Products Directive[14] is prominent as an incentive in various Member States, e.g. in Germany where the AgBB[15] protocol is developed and adopted by the government and it is the first mandatory emission label scheme (for flooring products). Emission standards for building products exist in different forms. An overview has been made available by ECA in 2005 (ECA Report N°24, see also ANNEX C). The incentive for most of these labels was essential requirement N°3 "Hygiene, Health and the Environment" of the Construction Products Directive. Most labelling schemes have been developed by private organisations (scientific and industrial), except for France.

- CPD Essential Requirement N°3 also stipulates that dampness should be controlled. However, contrary to policies on radon and ETS, dampness and moulds are considered a persistent problem in old buildings in many Member States.

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9 Commission Recommendation 90/143/EURATOM of 21 February 1990

10 Commission Recommendation 2001/928/EURATOM of 20 December 2001

11 COM(2007)27

12 Directive 2002/91/EC of the European Parliament and of the Council of 16 December 2002 on the energy performance of buildings

13 1990/396/EEC and 1992/42/EEC

14 89/106/EEC

15 AgBB. 2004. Health-related Evaluation Procedure for Volatile Organic Compounds Emissions (VOC and SVOC) from Building Products.

Further, indoor air policy is in many cases driven by national problems (e.g. NO_2 problem in Hungary and NO_2, SO_2 in Bulgaria, moulds and dampness in Finland and Slovakia, formaldehyde from building materials). Indoor air quality is one of the priorities in the National Environmental Health Action Plan (NEHAP) in Belgium, The Netherlands, Sweden, Slovakia, Italy and Bulgaria. Generally, in most Member States voluntary measures are used for private spaces, and mandatory for public spaces. When mandatory measures are used for private places, they generally apply to new buildings and products. Most consulted east European Member States prefer mandatory measures, while in north European Member States there is a long tradition of voluntary measures. Voluntary actions that were successful are material emission agreements and information campaigns, and especially the anti smoking campaigns and laws.

2.2 Indoor air quality guidelines in Member States

Every Member State has limit values for workplace environments, but only some Member States have guideline values for private and public places. Portugal has implemented limit values for public spaces. Indoor air guidelines for public spaces such as schools have been established for example in Austria, Germany and Norway. Indoor air quality guidelines for private spaces have been established for instance in Finland, Belgium (Flanders Region) Germany, France (CO and Formaldehyde guidelines have been published 3rd of September 2007), Norway, Poland and the United Kingdom (Table 1). Values vary between countries. Limit values for private spaces are very rare. A draft Ministerial Order containing health limit values for indoor air pollutants was prepared in Hungary, but is not in force.

Table 1: Guidelines values for indoor chemical pollutants in private spaces in European Member States

		Formaldehyde	**CO**	**NO_2**	**Naphthalene**	**Toluene**	**Styrene**	**NH_3**	**Monoteroene (a-pinene)**
		µg/m³	**mg/m³**	**µg/m³**	**µg/m³**	**µg/m³**	**µg/m³**	**µg/m³**	**µg/m³**
Belgium (Flanders)[1]	GL	10	5.7	135		260			
	IV	100 (30-min)	30	200 (1-h)					
Finland[2]	S1	30	2					30	
	S2	50	3					30	
	S3	100	8					40	
Germany[3,4,5]	GVII		15 (8-h)	60 (1-w)	20	3000 (1-w)	300 (1-w)		2000 (1-w)
	GVII		60 (30 -min)						
	GVI		1.5 (8-h)		2	300 (1-w)	30 (1-w)		200 (1-w)
	GVI		6 (30 -min)						
Norway[6]		100 (30 –min)	10 (8-h)	100 (1-h)					
			25 (1-h)						
Poland[7]	Cat B	100	6		150	250	30	300	
	Cat A	50	3		100	200	20	300	
UK[8]		100 (30 –min)	100 (15 -min)	300 (1-h)					
			60 (30 -min)	40 (1-y)					

		Formaldehyde	CO	NO_2	Naphthalene	Toluene	Styrene	NH_3	Monoteroene (a-pinene)
		µg/m³	mg/m³	µg/m³	µg/m³	µg/m³	µg/m³	µg/m³	µg/m³
			30 (1-h)						
			10 (8-h)						
Abbreviations for averaging time (-min) = -minute; (-h) = -hour; (-w)=-week and (-y) = -year									
1	Flemish Indoor Decree (BS: 19/10/2004); GL: Guideline Value; IV, intervention Value								
2	Target values for indoor air quality and climate; S1 = very good indoor air climate (Individual Indoor Climate), S2 = good indoor air climate, S3 = satisfactory indoor air climate. Values given in the table are maximum values for S1, S2 and S3. Source: Finnish classification of indoor climate. Finnish Society of Indoor Air Quality and Climate (FiSIAQ), 2000 (in English).								
3	Guidelines values (GV) for indoor air pollutants; GV II is a health-related value based on current toxicological and epidemiological knowledge. If the concentration corresponding to GV II is reached or exceeded immediate action must be taken because permanent stay in a room at this concentration level is likely to represent a threat to health especially for sensitive people. GV I is the concentration level at which a substance, taken individually, does not give rise to adverse health effects even at life-long exposure. An exceedance of GV I is linked with an exposure beyond normal which is undesirable from a hygienic viewpoint. GV I and GV II are given as 1-week average, except carbon monoxide, which was given as 8-hour (8-h) and 30-minute (30-min) average. Source: Seifert B. et al. (1999). Guidelines values for indoor air pollutants, Proceedings of Indoor Air '99, Edinburgh, vol 1: 499-504.								
4	Sagunski H, Heger W (2004). Richtwerte fur die Innenraumluft: Naphthalin. BundesgesundheitsblGesundheitsforsch – Gesundheitsschutz. 47:705-712 (in German).								
5	Sagunski H, Heinzow B (2003). Richtwerte fur die Innenraumluft: Bicyclische Terpene (Leitsubstanz Pinen). Bundesgesundheitsbl – Gesundheitsforsch – Gesundheitsschutz. 46:346-352 (in German).								
6	Becher (1999). Recommended Guidelines for Indoor Air Quality, Proceedings of Indoor Air '99, Edinburgh, Bol 1:171-176								
7	Category A – exposure up to 24 h per day; Category B – exposure limited to 8-10 h per day								
8	COMEAP (2004) Guidance on the effects on Health of Indoor Air Pollutants. Committee on the medical Effects of Air Pollutants (COMEAP). December 2004								
9	WHO (2000). Air Quality Guidelines for Europe. WHO Regional Publications, European Series, N° 91, Regional Office for Europe, Copenhagen								

2.3 Communication and collaboration with the public

Information guides can be very efficient measures to raise public awareness. Persons that are well informed can better protect themselves against harmful influence of certain substances present indoors. They can also influence the market by choosing low emitting products or avoiding purchase of products containing toxic substances. Germany, for instance, has a quite extensive set of guides to increase the awareness of the general public. These documents can be consulted online on the UBA[16] website. Guides like this are also published by other Member States such as Austria, France, Sweden, the United Kingdom. Educating the public and building trust has been very successful in Finland.

2.4 Indoor air monitoring and control programmes

Pursuing indoor air policies should be accompanied by efficient controlling and monitoring to test if policies are successful in complying the aims of good indoor air quality, to alert if a sanitation plan is mandatory, or to steer new policies if aims are not achieved. Currently, none of the EU directives prescribes explicitly a monitoring- and control programme for indoor air quality and no pan-European systematic indoor air monitoring system is currently installed. Indoor air monitoring studies in the EU have been performed in the framework of scientific research programmes (for an overview of European research projects up to 2005, see Annex E). Assessment protocols have been described, e.g. in the UK and by ECA, but have not been implemented on a countrywide and permanent basis.

[16] http://www.umweltbundesamt.de/gesundheit-e/irk.htm

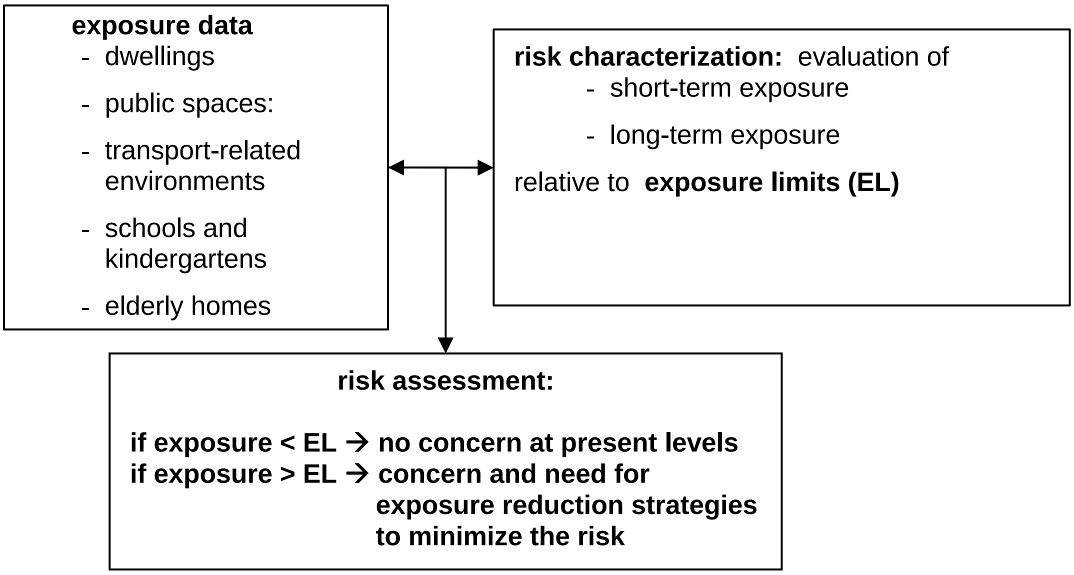
Where national authorities still have the freedom to intervene and to measure in public spaces, consent of the inhabitants is needed for private spaces, But country-wide coverage is difficult, given the high number of places and measurements needed. Consequently, systematic monitoring and control programmes are lacking in most Member States. In some Member States, monitoring studies occur on a project basis, most of them through European projects. In Germany, the German Environmental Survey (GerES) started in 1985 and is currently at its 4th edition and focuses on private spaces. In France, the French Indoor Air Quality Observatory[17] focuses on private spaces and dwellings. In some Member States, monitoring data are available through services where the general public can call upon in case of indoor air quality problems ("Green Ambulances"). It should be noted however that such complaints-based data are biased and do not represent the general IAQ. In Italy, the necessary facilities for a control programme are being set up, and in Sweden a national register is being set up to monitor the implementation of the Energy Directive. In the Netherlands, they will have participating projects in human biomonitoring studies.
Evaluation of the policy measures taken is not a general practice in most Member States. From the Member States present at the workshop, only Italy and The Netherlands has evaluation in practice. In Hungary, the smoking ban is constantly evaluated and in Sweden measures that are related to environmental objectives are continuously evaluated. In Sweden, the effective ventilation rates have been measured in 1999, and it was revealed that in many cases, the actual ventilation was not in compliance with the original building design, which shows the importance of control programmes.
In Poland, the follow-up of registered complaints can be taken as an evaluation of policy. Monitoring systems (especially for private dwellings) are however often based on services for dwellings with complaints, and thus do not depict the overall status of the IAQ (biased dataset).

[17] Observatoire de la qualité de l'air intérieur

3 RISK ASSESSMENT OF EXPOSURES IN PRIVATE HOMES AND PUBLIC SPACES

3.1 Risk assessment: general concept

Risk assessment (RA) and health impact assessment (HIA) (see chapter 5) are related to each other, and both use indoor exposure data as input. Basically, risk assessment is limited to evaluating exposure against thresholds (Exposure Limits (EL) based on the No Observed Adverse Effect Level (NOAEL) and Assessment Factors) - *without evaluating the **impact** if thresholds are exceeded* - whereas health impact assessment (chapter 5) goes beyond this point and attempts to calculate the health impact of exposure, in terms of attributable cases/diseases.



The INDEX study has been used predominantly, as it consists of the best available information on exposure and risks associated with indoor air pollutants in general. It provides insight in the selection of pollutants to develop further risk assessment of public spaces in particular.

3.2 Selection of stressors to conduct risk assessment

Following the recommendation of the Scientific Committee on Health and Environmental Risks (SCHER)[18], the risk assessment of pollutants in indoor environments is performed according to the principles used in the EU for risk assessment of chemicals as this is an evidence based approach. However, this substance by substance approach is an enormous task given the over 900 chemicals, particles and biological materials that may be present in indoor air. A first selection of substances is mandatory. Taking into account (1) the conclusions of several (EU initiated) studies and actions in the domain of indoor air quality

[18] SCHER opinion http://ec.europa.eu/health/ph_risk/committees/04_scher/docs/scher_o_048.pdf

and human health (INDEX, THADE, SCHER opinion, EU EWGIA[19], WHO indoor air working group[20]), and (2) the opinion of VITO workshop participants (workshop "Indoor Air Health Priorities" , Brussels, 29-30 March 2007), substances and factors which deserve high priority are ***ETS, formaldehyde, CO, particles (PM2.5 and PM10), NO_2, benzene, naphthalene, moulds and mites, dampness/moisture, CO_2 (measure for ventilation) and radon.***

3.3 Risk assessment: availability of methods and exposure limits of indoor stressors

Risk assessment is only feasible if tools are there:
1) methods and instruments to measure exposure, and
2) threshold values below which the risk is acceptable.

3.3.1 Availability of standard methods for sampling and analysis.

The sampling protocols for indoor air pollutants depend on the objectives of measurements. In general, active, pumped sampling – with normal sampling time of 30 min to 1 hour – is used for the investigation of peak or worst-case concentration, while the diffuse samplers provide a measure of the mean concentration over periods of days or weeks.
A review of strategies and protocols for indoor air monitoring of pollutants is published by Crump[21]. While for some indoor stressors like formaldehyde, VOCs, NO_2 and moulds ISO standardized methods exist, this is not the case for CO and PM, although for these pollutants general methods also exist.

- For formaldehyde, the standard method protocol is described in the ISO 16000-4 protocol 'indoor air, part 4: determination of formaldehyde in indoor air quality by the diffuse methods.' This technique involves capturing formaldehyde by the reagent DNPH on a filter, followed by extracted and analysis with GC-LC or GC-FID. The ISO 16000-3 protocol provides guidance on the active sampling method for formaldehyde and other carbonyl compounds.
- For CO, the applied method depends largely on the duration time of monitoring. For short-time monitoring, non-dispersive infrared analysers are applied. Equipment for long-time monitoring (e.g. 2 weeks) is less complex: colorimetric reaction of CO diffusion tubes.
- For PM determination, two measuring types exist: gravimetrical measurements and continuous measurements e.g. by optical laser aerosol spectrometers. The first method involves collection of PM on a filter by an active pumping system, followed by micro-balance weighting of the residue on the filters. The optical methods allow a higher time resolution compared to the gravimetric method, and simultaneous determination of the different size fractions. Gravimetrical measurements are currently the reference methods.

[19] presentation on Consultative Forum on Environment & Health , 30 November 2006, Brussels
[20] WHO working group http://www.euro.who.int/Document/AIQ/IAQ_mtgrep_Bonn_Oct06.pdf
[21] Crump. 2001. Review: Strategies and Protocols for Indoor Air Monitoring of Pollutants. Indoor and Built Environment, 10: 125 -131.

- The widely applied method for indoor air NO_2 sampling is a diffusion tube method (Atkins, 1978), followed by colorimetrical analysis. In addition, there is the ISO/DIS 16000-15 protocol 'indoor air- part 15: sampling strategies for nitrogen dioxide'
- CO_2/ventilation. CO_2 monitors with infrared sensor. For ventilation, the ISO/DIS 16000-8 protocol 'indoor air- part 8: determination of local mean ages of air in buildings for characterizing ventilation conditions' provides guidance.
- Benzene and naphthalene: passive samplers with activated charcoal, followed by desorption with carbon disulphide and subsequent analysis with GC-LC or GC-FID/MS. The standard method is described in the ISO 16017-1, 2 procedure 'indoor air – parts 1,2: Sampling and analysis of volatile organic compounds by sorbent tube/thermal desorption/capillary gas chromatography. – part1: pumped sampling, - part 2: diffusive sampling.
- Moulds: visual inspection (non-quantitative) techniques of mould growth on walls and ceiling is very often used as a measure for moulds. A quantitative approach is described in the DIS/ISO 160000-15,16,17 protocols 'indoor air, parts 15,16,17: detection and enumeration of moulds:-sampling by filtration, - culture base method, - sampling by impaction.

In addition to these standard methods applied in scientific research – often with high tech apparatus – commercial multi-parameter (CO_2, CO, humidity, temperature) indoor air quality monitors are available on the market.

As a rough estimate of costs for these sampling and analyzing these stressors, €50 per stressor is a realistic estimate. It should be taken into account that this price might be influenced by sample size, and by the number of stressors that can be measured simultaneously.

In conclusion, standard sampling and analysis methods are available for most of the indoor air pollutants, and there are no major sampling and analytical constraints hampering the assessment of indoor air stressors. The use of these ISO protocols should be encouraged.

3.3.2 Availability of threshold values of selected indoor air stressors

For *formaldehyde, CO, NO_2, benzene and naphthalene*, indoor air threshold values exist (e.g. the thresholds proposed in the INDEX reports, and are useful in RA (see further in Table 3). These thresholds originate from toxicological and epidemiological evidence. The basis for these thresholds, and assumptions behind them (e.g. assessment factors) however may differ between pollutants. For indoor PM, dampness/moulds and radon, no clear-cut thresholds values are in place.

Particles
No NOAEL/LOAELs or unit risk factors exist for PM, hampering the performance of a risk assessment. In addition, Fromme et al. (2006)[22] concluded that for a reliable risk assessment it is also essential to characterize the chemical and, particularly, toxicological properties of

[22] Fromme et al., 2006. Indoor air concentrations of PM (PM2.5 and PM10) in German schools. WIT transactions on Ecology and the Environment. Vol 86, Air Pollution XIV 393.

indoor PM samples. Schneider et al. (2003)[23] reached a similar conclusion in the EUROPART project. Based on an extensive literature survey on associations between exposure to particles and health effects, that there is inadequate scientific evidence to establish limit values or guidelines for indoor airborne particulate matter based on mass or number concentrations.

Dampness and moulds
The mechanisms behind the adverse health effect of moulds and dampness are still unclear. In a recent report, the European Agency for Safety and Health at Work (2007)[24] concluded that there is a lack of epidemiological and clinical data to establish exposure-disease and dose-response relationships. Therefore, health-based exposure limits cannot yet be proposed.

Radon
Radon is a known human carcinogen (group A according to IARC). No safe levels of exposure can be determined. The risk estimates obtained in the studies conducted among miners in a Swedish study corresponds to an unit cancer risk of 3-6 10^{-5} Bq/m³. No guideline value for radon is recommended[25] according to WHO.

3.3.3 Exposure data availability

A summary of the data availability on indoor concentrations of the priority pollutants and trends/highlights in data is given in Table 2. This brief summary is based on an extensive literature survey and on data collection via workshop participants. A full description of data and references is given in Annex B.
The exposure survey is limited to the priority pollutants formaldehyde, CO, NO_2, benzene, naphthalene and CO_2/dampness. Health impact assessments for radon and dampness/moulds (and ETS) have been previously reported (see chapter 5), and included an exposure assessment. In addition, for radon, we refer to the inventory of radon in dwellings reported by the WHO.

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[23] Schneider et al., 2003. EUROPART. Airborne particles in the indoor environment. A European interdisciplinary review of scientific evidence on associations between exposure to particles in buildings and health effects. Indoor Air, 13: 38-48.
[24] European Agency for Safety and Health at Work, 2007. European Risk Observatory Report: Expert forecast on Emerging Biological Risks related to Occupational Safety and Health.
[25] WHO. Air quality guidelines for Europe. Second edition. WHO Regional Publications, European Series, No 91. Chapter 8..3: radon p 209-217.
[26] MACBETH: Monitoring of atmospheric concentrations of benzene in European Towns and Homes. (http://www.fsm.it/padova/homepage.html)

Table 2: literature review of availability of exposure data in dwellings, public spaces, transport, schools and elderly homes

substance	widely investigated? [b]	data available for indoor environment?					statistical distrib. of data?	EU stratification?
		dwelling	public spaces[a]	transport[a]	schools[a]	elderly homes[a]		
formaldehyde	YES[b]++	YES (new > old)	YES [a]=/>(libraries)	YES [a]=/ > (heat in car)	YES [a]=/>	NO	MED/MEAN/MAX	S>N*
carbon monoxide	YES+	YES	YES [a]/>(bar/resto)	YES [a]/ > (bus, tram, metro, car)	YES [a]/=>	NO	entire distribution	S>N**
PM2.5/PM10	YES++	YES (main source: smoking)	YES [a]/>>(bar/resto)	YES [a]/> to >>(metro)	YES [a]/>	NO	entire distribution	S>N*
NO_2	YES++	YES (main source: gas appliance)	[a]=/>>(ice arena)	YES [a]=/ >	YES [a]=	NO	MED/MEAN/MAX	no clear trend
CO_2 as measure for ventilation	NO	YES	NO	NO	YES	NO	MED/MEAN/MAX	not enough data
benzene	YES++	YES	NO	NO	YES	NO	entire distribution	S>N**
naphthalene	NO	YES	NO	NO	NO	NO	entire distribution	no clear trend (outlier: Athens)

++ more than 10 studies; + fewer studies, however these studies had a wide coverage
[a] concentration ranges compared to dwellings
[b] 'YES' means data availability for that combination of substance and environment
>higher concentrations than in dwellings
* weak evidence, (not based on EU-wide study) or based on EU-wide study but smaller differences
** strong evidence, based on EU wide study

For most of the priority pollutants, information on indoor air concentrations in various micro-environments is available. However, it should be kept in mind that different measurement techniques/periods are applied in different studies, and most studies are limited to a short period of time and place, not necessarily representative for the EU, nor for a EU region. The most useful information in this context is found in the pan European studies like EXPOLIS, MACBETH[26], PEOPLE[27] and AIRMEX. In these studies, the same methods and study-setup is applied across different EU cities.

3.3.3.1 Formaldehyde

For formaldehyde, quite a lot of studies have been published reporting indoor formaldehyde concentrations, especially in dwellings. A point of attention is the trend of higher indoor formaldehyde concentrations in new homes compared to older homes, about a factor of 2. Indoor formaldehyde concentrations appear to be slightly higher in public spaces, transport and schools than in dwellings. Especially cars parked in the heat might lead to very high formaldehyde concentrations. The comparison of the various studies pointed to a slight trend of higher formaldehyde indoor concentrations in warmer compared to colder regions in the EU.

[27] PEOPLE: Population Exposure to Air Pollutants in Europe

3.3.3.2 Carbon monoxide

Carbon monoxide (CO) concentrations in dwellings and personal exposure have been investigated in a few large scale studies in the EU. Most of the studies focus on the influence of smoking, since this is a major contributor to CO exposure. CO concentrations in transport modes (bus, tram, metro, car), were (slightly) above CO concentrations in private dwellings and personal exposure of non-smokers. CO concentrations in public spaces depend on the implementation of the smoking ban. Indoor concentrations and CO exposures were typically lower in northern Europe than in central Europe, and again lower than in Southern Europe (based on the cities in the EXPOLIS study).

3.3.3.3 Particulate matter (PM2.5 and PM10)

Inventories of PM indoor concentrations in dwellings in the EU have been made in the framework of e.g. the THADE study, and others. Analogously to CO, smoking is a major contributor to indoor PM levels. Indoor PM levels in public spaces where smoking is allowed were much higher than common residential PM concentrations. People are also exposed to higher PM levels in transport compared to indoor residences. Concern has risen about very high PM levels in undergrounds (see below). The EXPOLIS study indicated a trend of higher indoor PM levels in southern EU countries compared to northern EU countries.

3.3.3.4 NO_2

Indoor concentrations vary depending on the presence of special indoor sources of NO_2. Elevated indoor NO_2 concentrations were typically related to gas cooking, gas heating and incense burning. Concentrations in homes without NO_2 sources are typically lower than outdoor concentrations and in those cases indoor levels are driven by outdoor sources. Across different studies, no clear EU geographical distribution of NO_2 indoor levels could be identified. Information on NO_2 in common public indoor spaces is lacking. Very high NO_2 concentrations have been reported in indoor ice arenas where propane and gasoline driven ice resurfacers cause NO_2 emissions. Average NO_2 during transport was higher (about 2-fold) than during non-transport activities. The studies on NO_2 in schools do not point to higher concentrations compared to residential environments.

3.3.3.5 CO_2/ventilation

The Air Infiltration and Ventilation Centre (AIVC, www.aivc.org) is specialized in design of ventilation. An inventory of the population at risk of living or working in poorly ventilated dwellings or offices is however not available. We found only very limited published information on CO_2 concentrations in indoor micro-environments. Most of the available data are for schools. CO_2 concentrations in classrooms are slightly higher in winter compared to summer.

3.3.3.6 Benzene

Mean indoor concentrations are typically higher than the respective outdoor levels all over Europe. In northern European cities, benzene indoor concentrations in dwellings appeared to be lower than in southern European cities (EXPOLIS, MACBETH, PEOPLE and AIRMEX). The benzene indoor concentrations were lower in public spaces where a smoking ban was installed compared to public spaces where smoking was allowed. One study (Schupp et al., 2006)[28] mentions that benzene exposure inside cars is problematic. In general, benzene indoor concentrations in schools and kindergartens were in the same range as in residential spaces.

3.3.3.7 Naphthalene

Only limited information on naphthalene indoor concentrations could be retrieved from the literature. The most important study is the EXPOLIS study. Indoor (dwelling) concentrations and personal exposures are usually low in Europe, except in Athens (> 20 fold above other concentrations in Basel, Helsinki, Oxford and Prague). Edwards et al. (2005)[29] attributed high naphthalene concentrations in Athens to, in the following order: to 1) time actively smoking, 2) presence of attached garage, 3) home located in the downtown area (~emissions from automobiles) and 4) time using gas stove.
No data on naphthalene in public spaces, transport and schools in the EU could be found.

3.3.3.8 Radon

The arithmetic mean of average indoor radon varies from 7 Bq/m³ (Cyprus) to 120 Bq/m³ (Finland and Estonia). A population weighted EU average radon concentration is probably close to 50 Bq/m³.[30]

3.4 Risk assessment of IAQ in private dwellings and public spaces

3.4.1 Risk assessment of IAQ in private dwellings

After reviewing the available evidence, the INDEX study is considered the single-most important source of information, being a well conducted RA, including all the state-of-the-art knowledge. The risk assessment and characterization of the INDEX study is summarized in Table 3.

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[28] Schupp et al., 2006. Benzene and its methyl-derivates: derivation of maximum exposure levels in automobiles. Toxicology letters, 160: 93-104.
[29] Edwards et al., 2005. Personal exposures to VOC in the upper end of the distribution - relationships to indoor, outdoor and workplace concentrations. Atmospheric Environment, 39: 2299-2307.
[30] McLaughlin and Bochicchio. 2007. In Focus: Radon and lung cancer. In: Proceedings of the first Envie Conference on indoor air quality and health for EU policy. p 161-171.

Table 3: risk assessment of indoor pollution in the EU (source: the INDEX report)

substance	threshold level	threshold value	unit	frequency of exceeding[$]
formaldehyde	exposure limit (EL)	1	µg/m³	100%
	NOAEL	30	µg/m³	20-65 %
carbon monoxide (1-h personal exposure indoors)	acceptable level*	35	mg/m³	<5 - 10 %
	desirable*	15	mg/m³	5-30 %
carbon monoxide (48-h personal exposure)	acceptable level*	15	mg/m³	< 5 %
	desirable level*	6	mg/m³	< 5%
particulate matter	-			**
NO_2	WHO-recommended value 1-yr average	40	µg/m³	10-75 %
	German GV II 1-week average	60	µg/m³	<5 - 45 %
benzene	EL for non cancer effects	60	µg/m³	< 5%
	NOAEL for non cancer effects	600	µg/m³	< 5 - 5 %
	current EU limit value (1 yr av.)	10	µg/m³	<5 - 50%
	EU limit value to be met at 01/01/10	5	µg/m³	<5 -75 %
naphthalene	EL	10		< 5 (all except Athens - 80 % (Athens)
	LOAEL	10000	µg/m³	< 5 %

[$] within the indoor concentration databases in INDEX for dwellings and personal exposure (e.g. EXPOLIS studies)
* acceptable: based on a COH_b level of 2 %; desirable: based on a COH_b level of 1 %
** RA of indoor PM is not feasible at this moment (lack of threshold value). INDEX did not include PM

The risk assessment for carcinogenic effects are summarized in the HIA section (see below), because for carcinogenic effects, no threshold values are in place.

3.4.1.1 Formaldehyde

Almost the entire population in the EU is exposed at levels which are higher than the exposure limit of 1 µg/m³. Even omitting the assessment factor of 30, it shows that at least 20 % (e.g. French National Survey Study) to 65 % (Helsinki EXPOLIS study) of the population is exposed to levels above the NOAEL (30 µg/m³) of formaldehyde.

3.4.1.2 Carbon monoxide

Personal exposure outcomes averaged over 1-hour were considered of moderate concern (5-30 % above the desirable concentration of 15 mg CO/m³) even for the most susceptible subpopulations. Nevertheless, there are uncertainties on the models used do derive the thresholds, especially for individuals exposed to low CO concentrations and its applicability to sensitive subpopulations. It was suggested that about 10% of the general non-smoking population experience CO levels which could be hazardous for individuals with heart diseases.

3.4.1.3 NO_2

The INDEX approach of risk assessment and characterization of NO_2 levels in the EU was by referencing the indoor concentration against the WHO recommended 1-year average value of 40 µg/m³, and by referencing against the German GV II 1-week average value of 60 µg/m³. The authors of the INDEX project concluded that a *remarkable proportion of the European houses, and thus the population, is exposed to NO_2 levels higher than the current guideline values protecting from respiratory effects in children*. Up to 25 % of the investigated residences NO_2 levels exceeded the German indoor related value of 60 µg/m³.
The authors of INDEX concluded that safe levels in homes (<40 µg/m³) are not likely to be achieved everywhere (e.g. in areas with intensive automotive traffic) given that ventilation alone already may introduce outdoor air containing such concentrations.

3.4.1.4 Benzene

The EL (and NOAEL) for non-cancer effects is exceeded in less than 5 % cases of the investigated indoor spaces in INDEX.
Although related to the carcinogenic effects of benzene instead of non-cancer effect, the evaluation of prevailing benzene indoor concentrations against EU limit values (current and to be met at 01/01/10) points out that less than < 5 % (Basel, Helsinki) to more than 50 % (Milan) exceedances of the current EU limit were recorded in the EXPOLIS study.

3.4.1.5 Naphthalene

The LOAEL of 10 mg/m³ for nasal effects in mice is converted to an EL of 10 µg/m³ taking into account an assessment factor 1000, as a combination of a factor 10 for LOAEL to NOAEL, a factor of 10 interspecies variability and a factor 10 of intraspecies variability.
In Basel, Helsinki, Oxford and Prague, in less than 5 % of the investigated indoor environments this EL is exceeded, whereas the EL is exceeded in 5 % of the cases in Milan and in 80 % of the cases in Athens (EXPOLIS study).

3.4.1.6 Radon, moulds/dampness, ETS and PM

No formal RA is possible for these indoor air stressors given the lack of exposure limits, because of lack of scientific evidence to derive such threshold, or because there is no safe threshold.
Further INDEX-like research activities on these stressors are needed to establish RA of these indoor stressors.

3.5 Risk assessment in public spaces, transport and environments of sensitive groups

Although people spend most of their time indoors at home, and hence exposure in the home environment dominates total personal exposure, some other micro-environments deserve also attention, namely
(1) micro-environments such as public spaces and transport where exposure levels can be significantly elevated compared to the home environment, and

(2) micro-environments in which sensitive persons spend a major part of their time (schools and elderly homes). Workshop participants also added hospitals and other medical facilities, in view of the exposure to microbial contaminants.

This section summarizes the main studies which have been published on RA of the selected chemical indoor pollutants in these environments, and tackles the environments which were judged as priority spaces among workshop members.

3.5.1 Transport

3.5.1.1 Transport in general

There is no unique trend in exposure and risk across different transport modes: while PM exposure is 3-8 higher in the underground system compared to above transport, the opposite trend was found for NO_2: 3 –fold lower NO_2 exposure in trams and undergrounds compared to transport by car or motorcycle. Thus, when comparing the risks to IAQ related to different transport modes, the pro's and con's of each transport mode should be weighted against each another. In the EXPOLIS study, air pollutant concentrations in various transport environments (bus, tram, metro, car, taxi, cyclist) have been investigated across a few EU cities. Differences in indoor CO concentrations between these different motorized transport compartments were minor compared to differences between the cities. For example, in Athens, indoor concentrations in bus-tram (4.4 mg CO/m³) were very close to concentrations in cars and taxis in Athens (4.2 mg CO/m³), whereas corresponding concentrations were much lower in Helsinki (bus-tram: 0.7 mg CO/m³; car-taxi: 1.2 mg CO/m³). These concentrations are below the threshold value of 6 mg/m³ (for long-term exposure). Adams et al. (2001)[31] also investigated $PM_{2.5}$ exposure in transport micro-environments in London. Cyclists had the lowest exposure levels, bus and car were slightly higher, while exposure levels on the London Underground rail system were 3 to 8 times higher than the surface transport modes. Piechock-Minguy et al. (2006)[32] revealed that during journeys by train, exposure to NO_2 was 20-50 µg/m³, by tramway or underground 33-68 µg/m³, by bicycle 69-96 µg/m³, and by car or motorcycle 97-125 µg/m³. These values exceed the thresholds of 40 and 60 µg/m³ (Table 3), indicating a potential health risk at these exposures, but averaging times were different.

3.5.1.2 Transport: formaldehyde and benzene in cars

Indoor car formaldehyde concentrations were higher in cars in heavy traffic circumstances (27 µg/m³) compared to parked cars (14 µg/m³) or cars in fluid traffic (17 µg/m³). Albeit, under normal circumstances, the concentrations in cars were not above typical concentrations for dwellings under normal thermal conditions. However, formaldehyde concentrations inside cars increase drastically with increased temperature. Under normal thermal conditions (23 °C) inside car concentrations of 48 µg/m³ have been reported by Schupp et al. (2005)[33] while at 65 °C, the inside car concentration can be as high as 1470 µg

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[31] Adams et al., 2001. Fine particles (PM2.5) personal exposure levels in transport microenvironments, London, UK. The Science of the Total Environment 279 2001 29- 44.

[32] Piechock-Minguy et al., 2006. A case study of personal exposure to nitrogen dioxide using a new high sensitive diffusive sampler. Science of the Total Environment 366 (2006) 55– 64

[33] Schupp et al., 2005. Maximum exposure levels for xylene, formaldehyde and acetaldehyde in cars. Toxicology 206 (3): 461-470.

formaldehyde/m³. This is 10-fold above the acceptable exposure levels inside cars as proposed by Schupp et al. (2005), based on a toxicological analysis. This shows that cars parked in the sun, indoor formaldehyde concentrations can be of concern and a reduction may be necessary (Schupp et al., 2005). The study of Schupp et al. (2005) is a first indication for this problem, but this phenomena needs being further explored (more cars, realistic instead of experimental temperature data). Schupp et al. (2006)[34] recently made a literature review of benzene concentrations in automobiles. Among the various studies, they found a range between 13 - 560 µg benzene/m³. The higher end of this range exceeds the maximum exposure levels for chronic exposure (called 'ELIA': 83 µg/m³), not for short term exposure (called 'STELIA': 16 mg/m³), proposed by Schupp et al. (2006). The authors concluded that benzene exposure inside cars is problematic, especially because benzene is a genotoxic carcinogen that probably acts by non-threshold mechanisms. Interestingly, the conclusion of a comparable exercise for toluene, xylene and trimethyl benzene was that exposure inside cars to the latter components are unlikely to pose a risk to the health of drivers.

3.5.1.3 Transport: particulate matter in metro and underground systems

The difficulties in performing a RA of indoor PM as mentioned above are still in place here, but nevertheless, a comparison of PM concentrations in transport (compared to other indoor environments) can indicate to some extent the relative risk of PM exposure during transport (and here commuting by metro in particular).

Various studies point to increased risk to elevated PM exposure and hence adverse health effects upon commuting by metro. For most of the investigated underground systems in the EU (i.e. London, Prague, Stockholm), very high (>100 µg PM2.5/m³) were recorded, except for the Helsinki subway. The personal PM2.5 exposure of office workers commuting by underground in London was 36.8 µg PM2.5/m³, i.e. 1.5-fold higher than personal exposure of office workers commuting to the office by another transport mode than the underground (Pfeifer et al., 1999)[35]. This difference between persons travelling by the underground tube (238.7 µg/m³) was even more pronounced (8 times higher levels than other modes' mean journey exposure) in the study of Adams et al. (2001)[36], also performed in London.

But composition (and thus toxicity and health effects) of PM in the underground system is very different compared to that of above ground transport systems. The former consists mainly of iron oxide particles, released through wear of steel and brakes, while the latter are combustion generated particles. This specific composition of underground PM is certainly an aspect to study more in detail, and to take forward in risk assessment of commuting by the underground.

[34] Schupp et al., 2006. Benzene and its methyl-derivatives: Derivation of maximum exposure levels in automobiles. Toxicology letters 160 (2): 93-104

[35] Pfeifer et al., 1999. Personal exposures to airborne metals in London taxi drivers and office workers in 1995 and 1996. The Science of the Total Environment 235: 253-260

[36] Adams et al., 2001. Fine particles (PM2.5) personal exposure levels in transport microenvironments, London, UK. The Science of the Total Environment 279: 29- 44

3.5.2 Schools

IAQ in schools should be recognized as a priority topic for public health (Carrer et al., 2002) [37]. The importance of good IAQ in schools is underlined by the following figures: there are more than 70 million students in the European Union, representing about 18 % of the population. In most countries, children attend schools five or six days a week, for over 800 hours a year. In addition, the teaching profession constitutes 3 % of the total working population in the EU. In the framework of a EU-funded project, Carrer et al. (2002) assessed the status of the indoor air quality in European Schools based on a literature review. The main conclusion of this study were:

- *Poor ventilation rate, air exchanges and airflows inside schools causing increased CO_2 levels in classrooms are common throughout Europe;*
- *The concentrations of PM are often higher in schools than in adult work environment. This is related to the material brought in on children's shoes, the use of chalk to write on the blackboard and the higher indoor physical activity of children causing resuspension of dust (Roorda-Knape et al., 1998)[38].*
- *VOCs and formaldehyde are emitted from the ceilings of schools and from furniture;*
- *No studies about the importance of passive smoking at schools were identified;*
- *School is an exposure risk environment to cat and dog allergic children, and this is related to the children who have a cat or dog at home and who transport allergens on their hairs, clothes, shoes and bags to schools.*

In conclusion, the study of Carrer et al. (2002) points out that the IAQ in school environment is in many cases probably lower than at home. However, there is a lack of representative IAQ audits in European Schools to capture the magnitude of this problem.

3.5.3 Hospitals and elderly homes

Hardly any data could be found on IAQ in hospitals or elderly homes. Indoor climates in nursing departments are typically characterized by high room temperatures and low relative air humidity (Smedbold et al., 2002)[39]. The major problem of IAQ in hospitals is primarily one of a microbial kind (see workshop discussion), rather than chemical pollution.

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[37] Carrer et al., 2002. The EFA project: Indoor Air Quality in European schools. Proceedings Indoor Air 2002, 794 -799.

[38] Roorda-Knape et al., 1998. Air pollution from traffic in city districts near major motorways. Atmospheric Environment, 32: 1921 -1930

[39] Smedbold et al., 2002. Relationships between indoor environments and nasal inflammation in nursing personnel. Archives of environmental health, 57 (2): 155-161.

4 RECOMMENDATIONS ON MONITORING STRATEGIES AND POLICY INTERVENTIONS

Based on risk assessment, expertise from the workshop members, and from experience of Member States, the following recommendations on monitoring strategies and policy interventions can be made:

4.1 Selection of public spaces and monitoring systems

4.1.1 Selection of public spaces for surveillance and monitoring

There is a general consensus supported by scientific data that **schools, public transport and health care places** (hospitals and elderly homes) are the public spaces that should come on top of the list of public spaces. Common driver hereto is the presence of *sensitive groups*, *elevated exposures*, and *duration of time of stay* in these environments.

4.1.2 Type of monitoring systems

Continuous monitoring is the best option to capture detailed trends in IAQ, however, for a selected set of pollutants (formaldehyde, CO, NO_2, benzene, PM, moulds,..), it is unlikely that continuous monitoring is cost-efficient. Instead, *periodic monitoring* (e.g. yearly or every 2-3-5 year) is probably the best cost-efficient solution. One-off studies are less suitable since these are too restricted in time, and do not allow to evaluate the effectiveness of the policy. Hereto, a *pan European design* should be developed and used for monitoring studies in all Member States, to allow inter Member State analysis. This is a task involving efforts of both the European Commission and the Member States.

Monitoring campaigns should focus on concentrations of *priority pollutants*, which is feasible. Passive sampling strategies (except for PM) are the most convenient option (3.3.1). In addition, attention should go to developing exposure data, especially for the less studied compounds (including particle composition) according to standardized methods and harmonised approaches across Europe. Here too examples exist, like EXPOLIS, but also other initiatives from the ECA (STRATEX…).

Monitoring of concentrations should focus on *public spaces*, where periodic measurements are feasible. Monitoring studies for private spaces are advisable, but problematic to conduct systematically. The solution could be to start with vulnerable groups, such as children or elderly. Another option is to work with complaint-based services, such as green ambulances, that can be consulted on a voluntary basis by the general public when needed.

A first indication of the costs associated with risk assessment of public spaces can be made by extrapolating the expenditures of research projects that have monitoring as the main focus (AIRMEX,… others), and including the cost of a risk assessment (e.g. INDEX, or costs to the EC for work in the scientific committee on environment and health risk) to cover all countries and pollutants.

4.2 Policy interventions at EU level

4.2.1 Development of indoor air quality guidelines or limit values

Inspection and intervention in the strict sense fall under the authority of the Member States and not of the Commission. The development of indoor air exposure guidelines or limits can be a task for the Commission. Currently, Member States have no European incentives for inspection. A common set of guidelines for indoor air quality is a key action to enable the integration of specific EU directives with the common objective to improve indoor air quality by achieving these guideline values. The WHO indoor air quality guidelines that are currently under development could serve as a basis for EU indoor air guidelines.
The Commission could then consider the development of indoor air exposure limits, in consensus with MS and the EWGIA.
Vulnerable groups should be taken into account while developing IAQ guidelines.
Meanwhile, monitoring data could be used as a reference for inspection and intervention.

4.2.2 Integration of indoor air legislation

Legislation in indoor air quality is not a stand alone issue: it is regarded as part of a broader picture, i.e. in connection with legislation on health, housing, spatial planning, energy and sustainability (also in connection with communication and participation). The indoor air aspects of these laws and regulations could be embedded as building blocks in an indoor air legislative framework, with EU indoor air guidelines or limit values as a cornerstone. Integrating the various aspects of indoor air regulated in the various 'sub'legislations in such a framework would be helpful to evaluate if the sum of the exposures is below the indoor air exposure limits or guideline values. Development of such framework on indoor air quality in a green paper would facilitate the debate.
Legislation is only useful when implemented, when enforced, e.g. when accompanied by an operating monitoring schema. It might be useful to consider an approach in which a priority is to focus on the implementation of existing policies and legislation, and to target some clear priorities EU-wide. This needs to be done while finding a balance between harmonisation and an equal approach across the EU, and the Member States' freedom of implementation, to take cultural differences into account. It needs to be considered to which extent the subsidiary principle[40] applies in the field of indoor air quality.

4.2.3 Harmonization of testing procedures and monitoring methods

Harmonisation of emission testing procedures at EU level are welcomed. The introduction of harmonisation on monitoring requirements (pollutants, analytical techniques, measuring locations (schools were frequently mentioned), periods and frequencies) will improve knowledge on IAQ in Europe. The facilities are available: the European Collaborative Action can serve as knowledge centre, within CEN pre-normative work can be done. There was a general preference for voluntary and harmonised labelling schemes among the

[40] Under the principle of subsidiary, in areas which do not fall within its exclusive competence the Union shall act only if and insofar as the objectives of the intended action cannot be sufficiently achieved by the Member States, either at central level or at regional and local level, but can rather, by reason of the scale or effects of the proposed action, be better achieved at Union level.

workshop participants. Still, voluntary agreements with industry are preferred but should be EU-wide using the same testing methods, to be in agreement with the EU common market principles.

4.2.4 Communication and collaboration between the European Commission, EWGIA and Member States

Participation and consensus is the road to follow. A platform such as the EWGIA where different Member States can discuss existing and new actions amongst each other offers a very useful and informal opportunity to exchange ideas. There is great interest in the topic and there is a strong need for a forum like the EWGIA to have meetings on a regular basis.
In addition, disclosure of exposure data for Member States should be stimulated. Data are certainly there, but are very often not publicly available, published in Member States' language. It would be useful for science and risk managers to have this information stored in a large database. The Commission could establish such a forum and stimulate Member States to enter their information (key words of studies, contact persons,…) in such a database or system.

4.2.5 Communication and collaboration with the public

On top of mandatory or voluntary emission reduction schemes, raising public awareness through information campaigns, and disseminations of "Codes of Good Practice" is essential. Whereas legislation can influence IAQ in public spaces, this is far less the case for private indoor dwellings, in which people spend the major part of their time. Education, consciousness raising, communication and participation of the public is of key importance to improve indoor (and especially private indoor) air quality in Europe.

Major policy gaps rely in the question how to intervene in private homes. Especially ETS in indoor environments of young children is of concern. This topic is somewhat on the hidden agenda because of the gap in knowledge how to tackle this problem. Conducting IAQ policy should always bear in mind the severity of the problem: one should strive to find policy actions improving 'serious problems' rather than focussing on easily implementable actions that only resolve a 'marginal' problem.

4.2.6 Stimulating research in order to fill scientific knowledge gaps

In general, there is a need for more and more comparable exposure and risk data. Specifically, there is a need to develop:

- methodologies to assess risk from and define measures against combined exposures, toxicology and synergies. The knowledge on chemical reactions between pollutants, or with indoor surfaces, the mixed composition and properties of (mixed) house dust is lacking at the moment. The further elaboration of integrated exposure assessment (human biomonitoring, source attribution, impact pathway) is necessary to evaluate health effects of poor IAQ and to evaluate the policy. Tools and methodologies are there, but the interpretation (e.g. for human biomonitoring) towards health relevance, inhaled or ingested doses and main responsible sources of pollution is still difficult. To assess the risk from combined exposure, it is also important to know the lifestyle and housing conditions of the general public.

- develop risk based evidence for particles and for some new or less studied compounds (pesticides, phthalates, flame retardants…). The process of reviewing and assessing compound-specific exposures and risks that started with INDEX should be continued.

5 OVERVIEW OF RELATIVE SCALES OF HEALTH IMPACT BY A HEALTH IMPACT ASSESSMENT

There is a general consensus on priority pollutants (formaldehyde, NO_2, CO, benzene, particles, dampness and moulds, ventilation,…). This consensus is among different Member States (based on established body of evidence, see chapter 2), and is in line with the risk assessment exercises as performed in INDEX. However, this prioritization is not based on a formal health impact assessment. In this chapter, a formal health impact assessment on these priority pollutants is tested, and gaps are identified. Gaps and uncertainty, plans to rectify these gaps and improve HIA are further discussed in chapter 6.

5.1 Health impact assessment (HIA): definition, data requirements and methodology

HIA involves the quantification of the expected burden of disease due to an environmental exposure (e.g. indoor air pollution) in a specific population. Whereas risk assessment is a standard approach in environmental pollution, Health Impact Assessment (HIA) is relatively new. Within the on-going 6th Framework Programme project HEIMTSA (health and environment integrated methodology and toolbox for scenario assessment), HIA/CBA (Cost Benefit Analysis) methods are being developed for environmental related problems in the EU, including indoor air. Ideas and methodologies have been well-established for ambient air, in research projects like ExternE[41] and in policy studies like the CAFE-CBA[42].
A general description of HIA is: "*a combination of procedures, methods and tools by which a policy, programme or project may be judged as to its potential effects on the health of a population, and the distribution of those effects within the population*" (WHO, 1999)[43].

5.2 HIA of indoor air pollutants: state of the art and feasibility

5.2.1 HIA of indoor air pollutants : review of previous assessments

Up to now, only a few complete EU-wide health impact assessment studies of the selected indoor air pollutants have been published. This is in contrast to the more advanced status of HIA for outdoor air pollution. For outdoor air pollution (with main focus on PM and ozone), HIA has been elaborated for example in the ExternE, CAFE and APHEIS[44] studies. We can think of 3 reasons why HIA on indoor air is less elaborated than HIA of outdoor air:

- There is in general a longer tradition of the field of ambient AQ compared to IAQ;
- There is a substantial body of epidemiological evidence for health effects of ambient air, supported by harmonized methods (like the APHEA study), and EU-wide studies;

[41] ExternE. Externalities of Energy. Externalities of Energy, Vol. 7, Methodology 1998 Update. European Commission, DG XII, Science, Research and Development. Chapter 8: health effects of PM10, SO_2, NO_X, O_3 and CO.
[42] CAFE: the Clean Air for Europe Programme
[43] WHO Regional Office for Europe. Gothenburg Consensus Paper. Health impact assessment: main concepts and suggested approach. 1999.
[44] APHEIS: Air Pollution and Health; an European Information System

- There is a strong policy trigger, with long-term cohort studies making ambient PM a high priority. No such chronic effects studies exist for IAQ.

Two studies of HIA on IAQ were developed in the framework of the ENHIS[45] programme,
1) a HIA for children exposed to Environmental Tobacco Smoke (ETS), and
2) a HIA for children living in homes with mould and dampness.
In addition, the HIA of radon exposure in the EU was assessed by McLaughlin et al. (2007) in the ENVIE project.

5.2.1.1 ENHIS case study 1: HIA for ETS

The most recent estimates indicate that more than 72 000 people in the 25 EU countries die each year due to exposure to ETS at home. WHO has estimated that 9–13% of all cancer cases can be attributed to exposure to ETS in a non-smoking population where 50% are exposed to ETS. If it is assumed that 35% of mothers smoke in the home, then 15–26% of lower respiratory illness in infants can be attributed to exposure to ETS. Applying these estimates to the population of the European Region suggests that 3000 to 4500 cases of cancer in adults and 300 000 and 550 000 episodes of lower respiratory illness in infants are attributable to ETS each year. For the evaluation of the impact of ETS on sudden infant death syndrome cases (SIDS), exposure-response functions developed in the meta-analysis of Anderson et al. (1997)[46] were applied using prevalence of maternal smoking as a proxy for exposure to ETS, and in the ENHIS HIA it was calculated that around 25 % of all SIDS cases could be attributable to exposure to ETS in the home.

5.2.1.2 ENHIS case study 2: HIA for moulds and dampness

The WHO estimated that 13 % of childhood asthma in the developed countries could be attributable to dampness.[47] However, in the framework of the ENHIS case study of HIA for moulds and dampness, the authors concluded that collection of comparable data on moulds and dampness is not available for the majority of EU countries. Instead of performing a EU-wide HIA, a Czech case study was performed.
The dampness/mould attributable cases of diagnosed asthma in the Czech Republic amounted to 4 % (range: 1 – 9 %), attributable cases of wheeze and night cough were respectively 7 % (3-13 %) and 3 % (1-5 %). This corresponds to about 200 attributable cases asthma, 330 cases of wheeze per 100 000 children and 140 cases night cough per 100 000 children in the age of 7-15 years.

It should be remarked that this ENHIS HIA is based on a exposure-response function from one single Czech study and not based on a meta-analysis. The outcome of the HIA would be different if other exposure-response functions (ERF) for home dampness and moulds would be used, e.g. ERF of Pirhoren et al.(1996)[48], Smedje et al. (2001)[49] or Venn et al. (2003)[50].

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[45] ENHIS: European Environment and Health Information System.
available at http://enhiscms.rivm.nl/object_class/enhis_casestudies.html
[46] Anderson et al., 1997. Passive smoking and sudden infant death syndrome: review of the epidemiological evidence. Thorax, 50: 1003-1009.
[47] WHO second technical meeting on quantifying disease from inadequate housing. Copenhagen, WHO Regional Office for Europe, 2007.
[48] Pirhoren et al., 1996. Home dampness, moulds and their influence on respiratory infections and symptoms in adults in Finland. Eur. Respir. J, 9, 2618-2622.

5.2.1.3 Radon

McLaughlin and Bochicchio (2007)[51] collected indoor radon data for each of the EU 25 Member States. Combining this information with the exposure response functions from residential radon epidemiological studies, McLaughlin and Bochicchio (2007) estimated that in 2006 in the EU 25 about 21 000 lung cancer deaths were due to radon exposure. An important footnote is that the majority of these estimated radon related lung cancer deaths occurred in active smokers exposed to radon.

For indoor pollutants other than ETS and mould/dampness, and radon we did not find published indoor HIA assessments. Hereto, a preliminary HIA will be performed in the next paragraph for the list of target indoor air pollutants. The classical methods described in the literature will be used. Based on the outcome, the feasibility of preliminary ranking of impacts of different pollutants will be investigated.

5.2.2 HIA of indoor air pollutants: feasibility for indoor air priority pollutants

5.2.2.1 indoor air concentration and exposure data availability

A summary of the data availability on indoor air concentration of the priority pollutants and trends/highlights in data is given in Table 2 (see chapter 3). It was decided to base the HIA on selected EU-wide datasets, namely the EXPOLIS datasets, eventually appended with national large scale studies with information on distribution (e.g. France) (also used in the INDEX report), instead of performing the HIA on a variety of (scattered) exposure studies. Two main advantages justify this use: the EXPOLIS study provides information of *distribution* of exposure data (in contrast to most of the other studies providing only mean/median and/or min/max exposure data), and allows comparison between different EU regions. In addition, the risk assessment of private dwellings (mainly based on the INDEX results) is also mainly based on the EXPOLIS dataset. This should enable drawing parallels between outcomes of the RA and the HIA.

5.2.2.2 Identification of appropriate health outcomes

The next step in HIA is the identification of appropriate health outcomes on the basis of epidemiological evidence and the availability of necessary data. This step is straightforward and supported by a wealth of toxicological and epidemiological data for most of the indoor

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[49] Smedje and Norbäck, 2001. Incidence of asthma diagnosis and self-reported allergy in relation to the school environment – a four-year follow-up study in schoolchildren. Int. J. Tuberc. Lung Dis., 5(11):1059-1066.
[50] Venn et al., 2003. Effects of volatile organic compounds, damp, and other environmental exposures in the home on wheezing illness in children. Thorax, 58: 955-960.
[51] McLaughlin and Bochicchio. 2007. In Focus: Radon and lung cancer. In: Proceedings of the first Envie Conference on indoor air quality and health for EU policy. p 161-171.

air pollutants under investigation. Main health outcomes affected by indoor air pollutants are respiratory symptoms (formaldehyde, PM, NO_2, ventilation, naphthalene), nasal and eye irritation (formaldehyde), cardiovascular diseases (CO, PM), cardiopulmonary disorders (PM), hematological and neurological effects (benzene, naphthalene) and carcinogenic effects (formaldehyde, benzene, naphthalene, PM). A detailed case by case identification of appropriate health outcomes is given in Annex D.

5.2.2.3 Exposure – response functions (ERF)

The ERF is the key contribution of epidemiology to HIA, and enables the calculation of a health impact beyond the risk assessment of exposure compared to toxicological limits. It provides the health risk associated with the hazard. The ERF is reported as a slope of a regression line or as a relative risk for a given change in exposure.

For outdoor key pollutants ($PM_{2.5}$ and O_3), such ERFs are based on epidemiological evidence and are widely accepted. For example, based on extensive review of the literature, WHO Air Quality Guidelines (WHO 2005) concludes for $PM_{2.5}$ increases the mortality risk by 6 (2-11) % for an increase of 10 $\mu g/m^3$.
Such exposure-response relationships for indoor air pollutants are far less elaborated or validated, and if, less widely accepted than for key outdoor air pollutants. To develop a useful exposure – response function, for each health outcome, a systematic literature review has to be performed to identify all eligible studies. With help of meta-analytical statistical techniques, a pooled exposure-response relationship from all selected publications has to be determined for each outcome separately. Such meta-analysis has, to our knowledge, not yet been elaborated for any of the target indoor air pollutants of this study (except for ETS by Anderson et al., 1997, see above). For some pollutants, review papers on health outcomes associated with indoor air pollutants exist (e.g. formaldehyde, benzene,…)[52], though, these review papers do not reach the step of statistical meta-analysis resulting in one pooled exposure – response relationship for each endpoint.

In this study, the availability of ERF functions, and the usefulness of these ERFs for indoor air HIA is investigated. A brief overview of ERFs of the indoor air pollutants is given in Table 4; more details are given in Annex D. While for most of the pollutants indoor-specific ERF studies were found in the literature, this was not the case for CO and PM.

[52] Mendell, 2007. Indoor residential chemical emissions as risk factors for respiratory and allergic effects in children: a review. Indoor Air. doi.10.1111/j.1600-0668.2007.00478x

Table 4: overview of exposure-response functions described in the literature (selected results; for more details: see Annex D)

substance	indoor/ ambient	receptor	region of the study	health effect	study (full references: see Annex D)
FORMALDEHYDE					
non-carcinogenic effects					
	indoor	children	Australia	diagnosed asthma	Garrett et al., 1999
		children	Australia	atopy - skin prick test	Garrett et al., 1999
	indoor	children	Australia	diagnosed asthma	Rumchev et al., 2002
	indoor	children	U.K.	asthma - self reported wheezing	Venn et al., 2003
	indoor	children	Sweden	diagnosed asthma	Smedje and Norbäck, 2001
carcinogenic effects				nasopharagyngeal cancer	IARC/US EPA
CO					
non-carcinogenic effects					
	ambient	heart patients	EU	cardiac diseases (hospital readmission)	von Klot et al. 2005
	ambient	newborns	Canada	effects on newborns	Liu et al., 2003
	ambient	children	Taiwan	asthma -prevalence of allergic rhinitis	Hwang et al., 2006
	ambient	all ages	Italy	asthma, respiratory symptoms, acute respiratory infections, COPD	Fusco et al., 2001
PM					
non-carcinogenic effects					
	ambient	all ages	EU	total mortality	APHEA2
	ambient	> 65 years	EU	COPD -hospital admission	APHEA2
	ambient	children		postneonatal respiratory mortality, lung function parameters,…	PINCHE
NO_2					
non-carcinogenic effects					
	indoor	children	Sweden	asthma	Emenius et al. ,2003
	indoor	children	U.K.	asthma - self-reported wheezing	Venn et al., 2003
	indoor	children	U.S.	respiratory symptoms	Li et al., 2006
	indoor	children	Australia	respiratory symptoms (self-reported prevalence during one year)	Garrett et al., 1998
	indoor	children	U.S.	lower respiratory tract symptoms	van Strien et al., 2004
CO_2/ventilation					
non-carcinogenic effects					
	indoor	children	Korea	asthma - increased risk for wheezing attacks	Kim et al., 2002
	indoor	children	Sweden and Estonia	asthma	Frisk et al., 2002
	indoor	adults	Sweden	nocturnal breathlessness	Norback et al., 1995
BENZENE					
non-carcinogenic effects					
	indoor	adults	U.S.	asthma - physician-diagnosed	Arif et al., 2007
	indoor	children		asthma- prevalence	Rumchev et al.,2004
	ambient or breath air	asthmatic children		asthma - degree of severness	Delfino et al.,2003
carcinogenic effects				cancer (leukemia)	IARC/US EPA
NAPHTALENE					
carcinogenic effects				cancer	California Office of Env. Health Hazard Assessment

5.2.2.4 Health impact assessment calculations

Existing exposure data in several countries and the ERF from literature have been used to assess the potential health impact per pollutant (see Annex D). Existing methodologies (e.g. the ExternE methodology or the methods used in the CAFE-CBA) have been applied. The resulting health impact is off course subjected to uncertainties, due to extrapolation beyond the original study and due to transferring the ERF to other populations. Nevertheless, the impact assessment results illustrate the magnitude of health effects due to these pollutants in Europe.

Formaldehyde

The HIA for indoor air formaldehyde concentrations in the EU points out that the increase in **asthma** for children due to formaldehyde in indoor air – given the INDEX formaldehyde data - is rather low, namely ranging from 0 % (95 % C.I.: 0- 3%; Rumchev ERF) - 1.9% (95 % C.I.: 0-6 %; Smedje ERF) in France to 3.5 % (95 % C.I.: 0-12 %; Smedje ERF) - 2.9% (95 % C.I.: 0.5 - 8%; Rumchev ERF) in Helsinki. This increase in health outcome is often called the 'attributable fraction'.

In contrast, the formaldehyde attributable fraction for **atopy** (prevalence) is substantial: from 17 % (95 % C.I.: 0-33 %) in France to 32 % (95 % C.I.: 0-61 %) in Helsinki. Among the atopic children, wheezing frequency is increased by 13 % (95 % C.I.: 2-23 %) in France to 24 % (95 % C.I.: 4-42 %) in Helsinki under prevailing formaldehyde concentrations.

Applying the carcinogenicity unit risk factor of $1.3\ 10^{-5}$ (US-EPA) to the prevailing indoor formaldehyde concentrations, figures out that 4 formaldehyde attributable cancer cases per year occur per 1 million inhabitants in France, and 5, 8 and 6 per 1 million inhabitants respectively for the U.K., Finland (Helsinki) and Sweden.

Carbon monoxide

The health impact assessment of carbon monoxide is performed in a similar way as for formaldehyde, except on one major point: given the lack of an indoor ERF, the outdoor ERFs were used. This puts a larger uncertainty on results, and care should be taken when interpreting these results. Mainly, the use of outdoor ERF for indoor ERF might lead to double counting of indoor and outdoor effects, since impact assessment, or ERF, of ambient pollutants, indirectly takes into account that people spend time indoors.

The adverse health effect of CO on newborns (birth weight, preterm birth , intrauterine growth retardation) is small (<1 %) at typically prevailing CO indoor concentrations in Helsinki, France and Milan. We estimated that the CO attributable fraction of allergic rhinitis prevalence among children is 3.3 % (95 % C.I.: 2.6-4.6 %) in France and 16 % (95 % C.I.: 13-22 %) in Milan. For adults, indoor CO exposure influences the hospital admission rate for respiratory symptoms by 0.7 % (95 % C.I.: 0.3-1.2 %) in France to 3.6 % (95 % C.I.: 1.7-5.6 %) in Milan. Among all investigated effects for adults, indoor CO has probably the largest influence on asthma (hospital admission for asthma for adults, i.e. 1.5 % in France with 95 % C.I.: 0.2 -2.8 %; and 7 % in Milan with 95 % C.I.: 1.2 -13 %.

However, the impact of accidental, acute CO poisoning, which is the largest risk of indoor CO exposure, can simply not be calculated by an ERF function because of lack of exposure data for these events. Instead, an additional HIA for CO is based on recorded statistics of CO poisoning.

The U.K. Health and Safety Executive tried to make an inventory of CO incidents in the EU during the past decades[53]. They noticed that not in all Member States statistics on CO incidents were collected (e.g. Italy, Belgium), and if collected and published (in many cases not public available), there was a variety in reporting formats and seriousness of reported as 'CO incidents', making the overall picture incomplete and difficult. Nevertheless, citing few selected data from that report, namely yearly 150-200 serious CO intoxication incidents and 25-30 acute CO caused deaths per year in the U.K., demonstrates that the seriousness of the acute CO poisoning (in the EU).

PM

At this stage, HIA of indoor PM is not feasible given the lack of indoor-specific ERF. Substitution by outdoor ERF is not justified because of different composition of outdoor PM compared to indoor PM. Using the ERF from ambient air would also lead to double counting with outdoor impact assessments. Since personal exposure to PM is correlated with ambient PM (because of infiltration of PM indoors), an impact assessment with ambient ERF essentially misses out on the PM fraction that is produced indoors. At the same time an impact assessment of ambient PM implicitly takes into account that people spent time indoors.

NO_2

The result of the HIA for NO_2 depends, even for the same endpoint (e.g. cough), strongly on the study used for deriving ERF. Applying the ERF of Belanger et al. (2006) [54]. shows 0.7 % (95 % C.I.: 0 - 4.2 %) increase in cough prevalence in Helsinki to 2.5 % increase (95 % C.I.: 0 -15 %) in the Po Delta, while the NO_2 attributable fraction is more than factor 10 higher when using the ERF of Garrett et al. (1998)[55]: 16 % (95 % C.I.: 0 - 30 %) in Helsinki to 54 % (0 - 103 %) in the Po Delta.
This shows that there is no consensus on magnitude of impacts of NO_2 on health. This is in analogy with the review paper of Basu et al. (1999)[56] who concluded that there is inconsistent evidence of adverse effects of NO_2 exposure. Comparing the endpoints within one study of Garrett et al. (1999), shows that effects of NO_2 are more pronounced on cough prevalence, shortness of breath, wheezing prevalence, and less on asthma attacks and chest tightness prevalence.

CO_2

A health impact assessment of CO_2 should be considered as a reflection of the impact of *overall* indoor air quality than of a HIA of CO_2 in se. No EU-wide CO_2 indoor monitoring data are available, and instead, a HIA for 2 cities, i.e. Örebro (Sweden) and Tallinn (Estonia)[57] results in the observation that wheeze attacks in children are influenced by

[53] Reducing carbon monoxide incidents. Contract Research Report 386/2001. Study prepared by Advantica Technologies Limited. 2001, Norwich, U.K.
[54] Belanger et al., 2006. Association of Indoor Nitrogen Dioxide Exposure with Respiratory Symptoms in Children with Asthma. Am J Respir Crit Care Med Vol 173. pp 297–303.
[55] Garrett et al., 1998. Respiratory symptoms in children and indoor exposure to nitrogen dioxide and gas stove. Am. J. Respir. Crit. Care Med., 158: 891-895.
[56] Basu et al., 1999 A review of the epidemiological evidence on health effects of nitrogen dioxide exposure from gas stoves. J. Environ. Med., 1: 173-187
[57] Frisk et al., 2002. Are there any differences in the indoor environment of asthmatic and non-asthmatic persons? A case-control study performed in Sweden and Estonia. Proceedings of Indoor Air 2002;1:97-102.

elevated CO_2 concentrations, caused by bad ventilation, which accompanies elevated indoor pollutants. However, attributable cases are not significantly different from zero. Further, exact numbers of attributable cases should be interpreted with care since the ERF is based on a Korean study, based on lower CO_2 indoor concentrations than in Örebro and Tallinn as reported by Frisk et al. (2002). Extrapolation of ERF beyond the concentration window of the ERF study might cause over-or underestimates.

Benzene
Although the major health concern of benzene is related to its carcinogenic effect, indoor benzene exposure is also associated with effects on asthma.
Comparing with other IAQ factors (e.g. formaldehyde), benzene has a larger impact on asthma prevalence, ranging from 6 % (95 % C.I.: 5-8 %) in Helsinki to 41 % (95 % C.I.: 32-49 %) in Milan. In contrast to most of the other impacts of the HIA is statistically significant different from zero.
The number of cancer cases per year per million persons associated with benzene vary from 0.1 – 0.3 (Helsinki) to 0.5 – 1.7 (Milan).

Naphthalene
The HIA of naphthalene is hampered by a lack of consensus on the carcinogenicity of naphthalene. Whereas the IARC classifies naphtalene in group 2B: possibly carcinogenic to humans, the US-EPA did not derived an inhalation unit risk estimate for naphthalene because of the weakness of the evidence that naphthalene may be carcinogenic in humans.[58] Alternatively, one could apply the unit risk factor handled by the California office of environmental health hazard assessment. However, results based on this risk factor (3.4 x 10^{-5} $(\mu g/m^3)^{-1}$) should be interpreted with caution, especially when ranking with carcinogenicity of other substances. Comparing the results for naphthalene with attributable cancer cases caused by benzene, suggests a similar number of attributable cancer cases (except for Athens). However, the evidence for the carcinogenic effects of benzene is stronger than for naphthalene.
It is puzzling that the concentration ranges of naphthalene, and hence the attributable numbers of cancer cases in Athens is one or two orders of magnitude above that of other investigated EU cities included here.

Table 5: Health impact assessment of indoor formaldehyde, CO, PM, NO_2, CO_2, benzene and naphthalene in the EU. Ranges illustrate the difference in exposure distribution; the 95 % C.I. indicates variation in the ERF. Background rates and prevalences are taken from the original studies (more details: see Annex D).

health outcome	study	receptor	from… (95 % C.I.)	to … (95 % C.I.)
FORMALDEHYDE				
non-carcinogenic effects: increase in adverse health effects in region x				
diagnosed asthma (prevalence)	Rumchev	children	0% (0-3 %)	2.9% (0.5 - 8%)
self-reported wheezing (prevalence during past year)	Venn	children	0.7% (0-11 %)	1.3% (0 -19%)
more frequent nocturnal wheezing prevalence (among atopic children)	Venn	atopic children	13% (2 -23 %)	24% (4-42 %)
atopy prevalence (skin prick tests)	Garrett	children	17% (0-33 %)	32 % (0-61 %)
diagnosed asthma incidence (over 4 years period)	Smedje	children	1.9% (0-6 %)	3.5% (0-12 %)

[58] http://www.epa.gov/iris/subst/0436.htm

carcinogenic effects (number of cancer cases per year per 1 million persons)			4	8
CO				
non-carcinogenic effects: increase in adverse health effects in region x				
hospital readmission for cardiac diseases (prevalence)	von Klot	heart patients	0.6% (0.-1.1 %)	2.9% (0.2 -5 %)
low birth weight (prevalence)	Liu	newborns	0% (0-0.1 %)	0% (0-0.3 %)
preterm birth (prevalence)	Liu	newborns	0.1% (0 -0.3 %)	0.7% (0.1 - 1.3 %)
Intrauterine growth retardation (prevalence)	Liu	newborns	0.2% (0 - 0.3 %)	0.9% (0.1 - 1.4 %)
allergic rhinitis (prevalence)	Hwang	children	3.3% (2.6 – 4.6%)	16% (13 -22 %)
hospital admission for asthma	Fusco	all ages	1.5% (0.2-2.8%)	7.1% (1.2 -13 %)
hospital admission for respiratory symptoms	Fusco	all ages	0.7% (0.3 -1.2 %)	3.6% (1.7 - 5.6 %)
hospital admission for acute respiratory infections	Fusco	all ages	0.6% (0 -1.2 %)	2.8% (0-5.7 %)
hospital admission for COPD	Fusco	all ages	1.2% (0-1.2 %)	5.6% (0-5.7 %)
PM				
not feasible - no adequate indoor PM ERF; outdoor PM ERF is inapplicable due to different composition, and would cause double-counting of outdoor effect				
NO_2				
non-carcinogenic effects: increase in adverse health effects in region x				
recurrent wheezing (prevalence)	Emenius	children	2.3% (0-44 %)	8% (0-154 %)
wheezing (prevalence)	Venn	children	0% (0- 1.5 %)	0% (0-5 %)
cough (prevalence)	Garrett	children	16% (0-30 %)	54 % (-103 %)
shortness of breath (prevalence)	Garrett	children	8% (0-20 %)	28% (0-70 %)
wheeze prevalence	Garrett	children	5% (0-15 %)	16% (0-53 %)
asthma attack prevalence	Garrett	children	1.8% (0 -13 %)	6.4% (0-45 %)
chest tightness prevalence	Garrett	children	2% (0-10 %)	8.2% (0-36 %)
wheeze prevalence	Belanger	asthmatic children	0% (0-3.3 %)	0% (0-11%)
persistent cough	Belanger	asthmatic children	0.7% (0-4.2 %)	2.5% (0-15 %)
shortness of breath (prevalence)	Belanger	asthmatic children	0% (0-2.2 %)	0% (0-7.5 %)
chest tightness prevalence	Belanger	asthmatic children	0.6% (0-3.3 %)	2.2% (0-11.5 %)
CO_2/VENTILATION				
non-carcinogenic effects: increase in adverse health effects in region x				
wheezing attacks (prevalence)	Kim	children	42% (0 -125 %)	45% (0-134 %)
BENZENE				
non-carcinogenic effects: increase in adverse health effects in region x				
severity of asthma	Delfino	asthmatic children	17% (4-28 %)	113% (26 -185 %)
asthma (prevalence)	Rumchev	children	6% (5-8 %)	41% (32 -49 %)
carcinogenic effects (number of cancer cases per year per 1 million persons)			0.1- 0.3	0.5- 1.7
NAPHTALENE				
carcinogenic effects (number of cancer cases per year per 1 million persons)			0.3	31**

* the regions/cities are France, Helsinki, Milan, Athens, Prague, Basel, U.K., Athens, Oxford and the Po Delta. Not all cities are included for all pollutants, ranging from 3 regions/cities for carbon monoxide to 6 regions/cities for benzene and naphthalene.

** for one city, namely Athens, health impact of naphthalene exposure is an order of magnitude above all other investigated cities and regions in the EU.

5.3 Interpretation and provisional ranking of indoor air pollutants by HIA

The following constraints apply to the health impact assessment:

- An Exposure Response Function shows a statistical relationship between the exposure and health outcome. They do however not show a causal relationship. Thus, additional (experimental) toxicological evidence is needed to support the causal relationship between exposure and health outcome. The impacts considered here are supported by other (toxicological and medical) evidence.
- The HIA at this stage does not weigh the severity (e.g. in terms of costs) of the various diseases, making it difficult to make allow a real ranking.

However the following conclusions can be made:

1. For fatal cancers ETS is dominant in comparison to other pollutants, followed by the assessment of radon deaths. Care should be taken with the fact that radon increases the mortality risk from lung cancer in smokers. From our assessment the carcinogenic impact from formaldehyde seems more important than that of benzene. Naphthalene cancer cases are smaller in most cases, but results for Athens where exposure is very high should be taken into account when prioritising pollutants.
2. Morbidity effects are difficult to compare. More health endpoints can be quantified for CO and for NO_2, but increases in effects are small to moderate. One study for NO_2 and cough in children results in a high prevalence increase. Formaldehyde has a high to moderate effect on sensitive children. The results for CO_2/poor ventilation are difficult to compare because of the very large uncertainties involved.

Benzene and formaldehyde were also evaluated as the two pollutants with the highest cancers risk of organic hazardous air pollutants at prevailing concentrations in the U.S. (Loh et al., 2007)[59]. We did not make an explicit and separate HIA of dampness and moulds. Though, the moulds/dampness HIA performed by ENHIS demonstrates that moulds and dampness exposure lead to a significant increase in night cough, wheeze and asthma. The study of Venn et al. (2003) identified dampness and moulds, and formaldehyde as the only factors causing an increase in wheezing illness in children, while other environmental exposures such as NO_2 and TVOC did not influence wheezing illness. Using the same study, where different exposures are assessed (such as the U.K. study of Venn et al., 2003) helps ranking indoor air priorities (under the assumption that the exposures in that study are representative for other regions). This underlines that dampness and moulds should be seen as a top indoor air priority.

In conclusion, the HIA for indoor air is feasible but incomplete for ranking effects/pollutants. In chapter 6, ideas to rectify these information gaps are elaborated.

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[59] Loh et al., 2007. Ranking cancer risks of organic hazardous air pollutants in the United States. Environmental Health Perspectives, 115: 1160 – 1168.

5.4 Uncertainty associated with Health Impact Assessment

Several factors contribute to uncertainty on the calculated numbers of health impact assessments. Attributing exact numbers to the uncertainty is difficult at the moment, though, we believe that the following list gives the uncertainty factors (listed following decreasing degree of importance):

1. Uncertainty on the ***exposure-response function***.
   There is a rather wide statistical uncertainty on attributable fraction (expressed by the 95 % confidence intervals in Table 5). Such wide uncertainty is however not unusual given the kind of the studies (namely smaller panel studies).
2. Uncertainty on ***country-specific exposure data*** (e.g. naphthalene)
   The HIA is based on a limited number of exposure data. This puts uncertainty on representativeness of these datasets for other EU regions. Whereas for some pollutants, the variability of exposure across the EU is rather limited (e.g. formaldehyde), this is not the case for e.g. naphthalene. For such pollutants, the geographical coverage of exposure data should be improved to reduce uncertainty.
3. There is a ***lack of adequate indoor PM ERF***. From knowledge for outdoor air pollution related HIA, we know that PM is top cause of adverse health effects related to outdoor pollutant. Without the knowledge of indoor PM ERF, it is difficult to rank the health impact of indoor PM.
4. Uncertainty ***on background prevalence of disease.***
   Uncertainty also increases when country specific health background data is missing. However like in policies for ambient air the use of average background rates across (regions of) Europe might be reasonable for EU-wide assessment (cfr. The CAFE CBA), when the set of ERF is large enough. Within the total impact of a certain pollutant uncertainty of one function or of the underlying health data will be less problematic, and the assessment might still provide robust answers.
5. Uncertainty on ***transferability of ERF.***
   Some of the ERFs are based on studies performed outside the EU. For example, the studies of Rumchev et al. (2002, 2004), Garrett et al. (1998, 1999) rely on Australian data. The transferability of these ERFs to the EU might be criticized due to differences in building and ventilation characteristics between Australia and the EU, and thus puts uncertainty on transferability of ERF.
6. Uncertainty on ***comparability across pollutants and studies.***
   Comparability across pollutants is hampered by the fact that still limited health endpoints are assessed. In addition, the study design of one study is not comparable to that of another study investigating health effects of the same pollutant.

6 GAPS IN THE INFORMATION BASE FOR HEALTH IMPACT ASSESSMENT AND PLANS TO RECFIFY THEM

6.1 Why do we need a health impact assessment of indoor air pollutants?

Developing a health impact assessment for the selected list of priority pollutants is crucial to include the health impacts of air pollution and the benefits of indoor air quality interventions in a cost benefit assessment, as part of an impact assessment of policies. It is also useful to enable comparison between pollutants and to assist in the selection of new and emerging pollutants for inclusion in the indoor policies.
Health impact assessment is a rather new discipline in the field of indoor air quality. Thus, development of HIA methods and HIA studies should be encouraged. Ongoing research projects like HEIMTSA may provide a strong basis for this.

6.2 What are the knowledge and information gaps and plans to rectify them ?

6.2.1 Identification of health endpoints

Gap: for the set of priority pollutants there are sufficient studies indicating the potential health problems related to these pollutants (Overview annex D+ chapter 5). For emerging pollutants and issues of indoor air quality knowledge is still limited to a few studies indicating a potential hazard without assessing the risk.
Plan to rectify: continue research efforts, specify that each research proposal on indoor air should include provisions to assess the dose-effect relationship of some emerging or new pollutant.

6.2.2 Selection of exposure-response functions

Gap: due to the complexity of an adequate epidemiological design to assess indoor air as a potential cause for health effects, literature results are limited to smaller panel studies and case-control studies. Thus, the information is incomplete, resulting in a potential bias when comparing different pollutants. To extend the database on ERF EU-wide epidemiological assessments are needed, applying a harmonized protocol to select pollutants, indoor environments, to assess the exposure and to analyze the health outcomes.
Plan to rectify: consider setting up an epidemiological research programme for 4 years or more (comparable to APHEA1&2 for ambient air, that constituted of 8 years of EU-wide research), and to combine the assessment of outdoor air pollution with indoor air pollution. This would enable development of a balanced health impact assessment on a European scale, like it has been done for ambient air, using e.g. results from the APHEA studies in health impact assessments for CAFE. It is suggested that a process of review and expert working group is set up to develop a set of ERFs of indoor air pollutants based on a meta-analysis of several studies. This approach has been followed for ambient air (e.g. WHO expert working groups,...).
Gap: exposure assessment (defining the measure of exposure, and estimating the prevalence of exposure) is crucial. It is the main (logistic) challenge here, requiring new techniques to perform large scale studies.

Plan to rectify: consider investing in large scale exposure assessment studies, as specific task within the epidemiological research, in combination with the human biomonitoring network under development. Human biomonitoring can be used as validation of exposure measurements and for the assessment of dose and effect. Human biomonitoring has proven its feasibility for a limited set of indoor air pollutants, including cotinine as indicator for ETS. When a EU-wide and harmonised approach for human biomonitoring is implemented, this system can be used to the benefit of indoor air quality assessments. Further research is however required to establish validated biomarkers of exposure and effect for the complete set of priority indoor air pollutants. This can gradually be included in such a European human biomonitoring system. In addition, further elaboration of integrated exposure assessment (human biomonitoring, source attribution, impact pathway) is necessary to evaluate health effects of poor IAQ and to evaluate the policy.

6.2.3 Adequate exposure-response function of indoor PM

Gap: information on particles is limited to qualitative studies or to studies drawing a parallel with ambient PM. This information is however insufficient to derive ERF of indoor exposure to particles, or specifically to indoor sources of particles.
Plan to rectify: establish a relation with the presence of (indoor) sources like wood burning, consumer product uses,… Source-effect relationships and not only exposure-response relationships are useful for policy makers to implement measures. The ENVIE project is taking this into account.

6.2.4 Population baseline frequencies for the health outcomes

Although exposure-response relationships may be derived from the international literature, preferably after a careful meta-analysis of the available evidence, the baseline frequency of disease should be gathered for the target population. In the assessment of (minor) respiratory effects these health data are either lacking or inconsistent across Europe. If such data are unavailable or inadequate, health frequency data from other populations may be used, taking into account the limitations in use. In such cases, the potential limitations of such substitutions should be considered and thoroughly discussed in the health impact assessment. There is no formal assessment of the existing health data in relation to its use in health impact assessments in general and for indoor air in particular. Data exist for example on respiratory outcomes in the ECRHS study or the ISAAC study, that is indirectly linked with (indoor) air pollution, and that might be used to account for (socio-economic) differences in impact in different European countries.

6.3 Actors to rectify these gaps

Responsible parties in all these four aspects are:
a) the European Commission (Directorates General Environment, Health and Consumer Protection and Research) in initiating actions to fill these gaps;
b) the scientific community to execute the research and
c) the Member States to enable and support indoor air quality research in each country (e.g. through the NEHAP).