Appraisal project
Air Pollution Policies
foR Assessment
of Integrated Strategies
At regional and Local scales

Grant Agreement number 308395

WP 4 Guidance on integrated air quality and health assessment systems

VITO

D4.4 Final version of the Guidance document

Reference: APPRAISAL / VITO / WP 4 / D4.4 / VERSION 1.1

Category: Coordination

Author(s): Peter Viaene and Stijn Janssen (VITO), Niko Karvesenoja (SYKE), John Douros and Lia Fragkou (AUTH), Elsa Real and Augustin Collette (Ineris), Arno Graff (UBA), Katarzyna Juda-Rezler, Zbigniew Nahorski and Marko Tainio (SRI), Nadège Blond (CNRS), Jean-Luc Ponche and Alain Clappier (UNISTRA), Giovanna Finzi, Enrico Turrini, Marialuisa Volta (UNIBS), Enrico Pisoni, Claudio Belis, Philippe Thunis(JRC), Alberto Martilli (CIEMAT), Catherine Bouland, Michèle Rasoloharimahefa(ULB), Jose Baldasano (BSC), Les White (Aeris), Jurgen Buekers, Pieter Valkering (VITO), Hanna Eskelinen (SYKE), Ana Miranda, Hélder Relvas (UAVR), Guiseppe Maffeis (Terraria),

Verification: Alberto Martillo (CIEMAT), Claudio Carnevale (UNIBS), Catherine Bouland (ULB)

Date: 08/06/2015

Status: Version 1.1

Availability: Public

SEVENTH FRAMEWORK PROGRAMME

Appraisal project

FP7-ENV CA 308395

www.appraisal-fp7.eu

Summary

An Integrated Assessment Modelling system (IAM) is a methodology to determine what set of actions has to be performed to efficiently improve air quality, reduce exposure and protect human health. In this document we provide guidance on how such an IAM can be set up.

The guidance document is structured around the different elements of an IAM as identified in WP3, in which the design of an IAM was presented based on the building blocks of the EEA DPSIR (Drivers, Pressure, State, Impact, Responses) scheme. For each of these blocks a number of recommendations is given in the form of questions and answers, taking into account that each block can be elaborated to a different level of complexity according to the available data. Some of the questions relate to important cross cutting topics that reappear for each of the blocks, such as how to take into account the different scales and their interactions and how to deal with uncertainties, while other questions are specific to single blocks.

This document is based on the first draft version of the guidance document (deliverable D4.1) which was further refined in a number of iterations and discussed during the technical meetings and updated based on the insight gained during the practical applications which are described in the deliverables D4.2 and D4.3.

Version History

Version	Status	Date	Author(s)
0.1	First Final	22/04/2015	Peter Viaene and Stijn Janssen (VITO) , Niko Karvesenoja (SYKE), John Douros and Lia Fragkou (AUTH), Elsa Real and Augustin Collette(Ineris), Arno Graff (UBA), Katarzyna Juda-Rezler, Zbigniew Nahorski and Marko Tainio (SRI), Nadège Blond (CNRS), Jean-Luc Ponche and Alain Clappier (UNISTRA), Giovanna Finzi, Enrico Turrini, Marialuisa Volta (UNIBS), Enrico Pisoni, Claudio Belis, Philippe Thunis(JRC), Alberto Martilli (CIEMAT), Catherine Bouland, Michèle Rasoloharimahefa(ULB), Jose Baldasano (BSC), Les White (Aeris), Jurgen Buekers, Pieter Valkering (VITO), Hanna Eskelinen (SYKE), Ana Miranda, Hélder Relvas (UAVR), Guiseppe Maffeis (Terraria))
1.0	Final	18/05/2015	Peter Viaene and Stijn Janssen (VITO), Nadège Blond (CNRS), Enrico Pisoni (JRC), Claudio Carnevale and Luisa Volta (UNIBS)
1.1	Final revised	1/6/2015	Peter Viaene (VITO), Alberto Martilli (CIEMAT), Claudio Carnevale (UNIBS), Catherine Bouland (ULB)

Summary of Changes

Version	Section(s)	Synopsis of Change
0.1	All	First version based on D4.1, D4.2 and D4.3
1.0	All	Review and additional references, text
1.1	All	Final thorough revision

Abbreviations

AOT40	The accumulated amount of ozone over the threshold value of 40 ppb
AQ	Air Quality
AQI	Air Quality index
AQP	Air Quality Plan
CMB	Chemical Mass Balance
CT	Circulation type
CTM	Chemical Transport Model
DALY	Disability Adjusted Life Years
DPSIR	Driver/Pressure/State/Impact/Response
EEA	European Environmental Agency
EM	Emission Measures
HIA	Health Impact Assessment
HII	Health Impact Indicator
IAM	Integrated Assessment Modelling system
NAPFUE	Nomenclature for Air Pollution of FUEls
NTM	Non-Technical Measures
NWP	Numerical Weather Prediction
PCA	Principal Component Analysis
PM	Particulate Matter
SA	Source Apportionment
SEI	Spatial Emission Inventory
SNAP	Selected Nomenclature for sources of Air Pollution
SOMO35	Sum of Ozone Means Over 35 ppb, defined as the yearly sum of the daily maximum of 8-hour running average over 35 ppb
TM	Technical Measures
YOLL	Years Of Life Lost

1 Introduction

This recommendation document, focuses on the different topics that need to be addressed to set up an Integrated Assessment Modelling (IAM) system or methodology for air quality and is the official Guidance Document of the APPRAISAL project. The objective is by no means to provide detailed technical instructions on how to set up an IAM but is to present a comprehensive set of topics that should be addressed, together with the main lines along which these can be approached and the references that can further guide the reader. The emphasis is therefore more on keeping the overview and less on providing all the details, as these can already be found in other reports and peer-reviewed literature sources.

In line with the DPSIR scheme adopted by the European Environment Agency (EEA) for describing the interactions between society and environment, the APPRAISAL project proposes a scheme to identify the different elements of an IAM. This DPSIR scheme is shortly described in the present document in terms of its building blocks (Drivers, Pressure, State, Impact, Responses) and it is detailed in the WP3 deliverable of the APPRAISAL project. Also, in line with WP3, this document does not provide a single set of recommendations but a range of solutions, so that each of the building blocks can be elaborated to a different level of detail, adapted to the data and tools that are available in practice or what is required to solve the specific air quality problem that is targeted by the IAM (‘fit for purpose’). The different topics of the document are addressed through a number of questions. Some of these questions are specific to the building block while others relate to topics that reappear in all blocks such as how the uncertainty or the interaction between scales should be tackled.

Before looking at the individual building blocks, a first general section is devoted to introduce the concept of an IAM, the DPSIR scheme and the concepts base case and scenarios.

This document is based on the first draft version of the guidance document (deliverable D4.1) which was further refined in a number of iterations and discussed during the technical meetings and updated based on the insight gained during the practical applications which are described in the deliverables D4.2 and D4.3.

2 A general overview of the Integrated Assessment Modelling system framework

2.1 What is an integrated assessment modelling system (IAM)?

Integrated assessment is an approach in which knowledge from different scientific disciplines is combined with the aim to provide insight to policy in such a way that an entire cause–effect chain of a problem can be evaluated. In the context of air quality an IAM can be considered as a methodology at the Science-Policy interface that provides the bridge between increasingly complex scientific insight into multi-pollutant/ multi-effects of air pollution and the practical policy decisions required for determining how air pollutant emissions can be **efficiently** reduced to improve air quality, protect human health and reduce ecosystem exposure. Other aspects, that could be included in an IAM but which are not the main focus of this document, are noise and the abatement of greenhouse gasses. It is important to note that integrated assessment cannot be thought of as a specific procedure at the current stage of development in air quality practice and, even less, as a unique tool. At regional and local scale levels in the EU, besides the obvious specific local conditions (meteorology and topography, emissions) that lead to air pollution problems, there is also a large difference in the detail of the available data and an even larger disparity in the decisional power of the involved agencies. Integrated assessment must thus be interpreted as an approach which links decision making, air quality dynamics and consequential health and environmental impacts in many different ways in order to suit the capability and need of a specific regional/local situation.

2.2 What can an IAM system be used for?

In general an IAM can be used to assess the effect on air quality and consequentially health or environment impacts of an emission abatement policy. If several emission abatement policies are considered the IAM can help the user decide which set of measures should be selected based on some criteria (e.g. minimisation of cost, health impact or the most socially acceptable option). While most practical IAM applications rely on expert judgement in this decision process, some IAM systems can assist the user to determine the "best" set of abatement measures through an optimisation process. In the interpretation of IAM results it is important, however, to always keep in mind the assumptions and limitations underlying the different components of the IAM.

2.3 What are the elements of an IAM?

In line with what was presented in the APPRAISAL deliverable D3.1 on the design of an IAM, we apply the DPSIR scheme to identify the different elements of an IAM. The DPSIR building blocks can be mapped onto the IAM elements as shown in Figure 1:

- driving forces: the key activities that result in pollutant emissions (e.g. traffic, residential heating);
- pressures: the pollutant emissions;
- states: the air quality i.e. the concentrations and deposition of pollutants;

- impacts: the consequences of the air quality for human exposure and health impacts and for environment;
- responses: the measures that are available to reduce the impacts.

For each of these blocks guidance is given in the next chapters of this document.

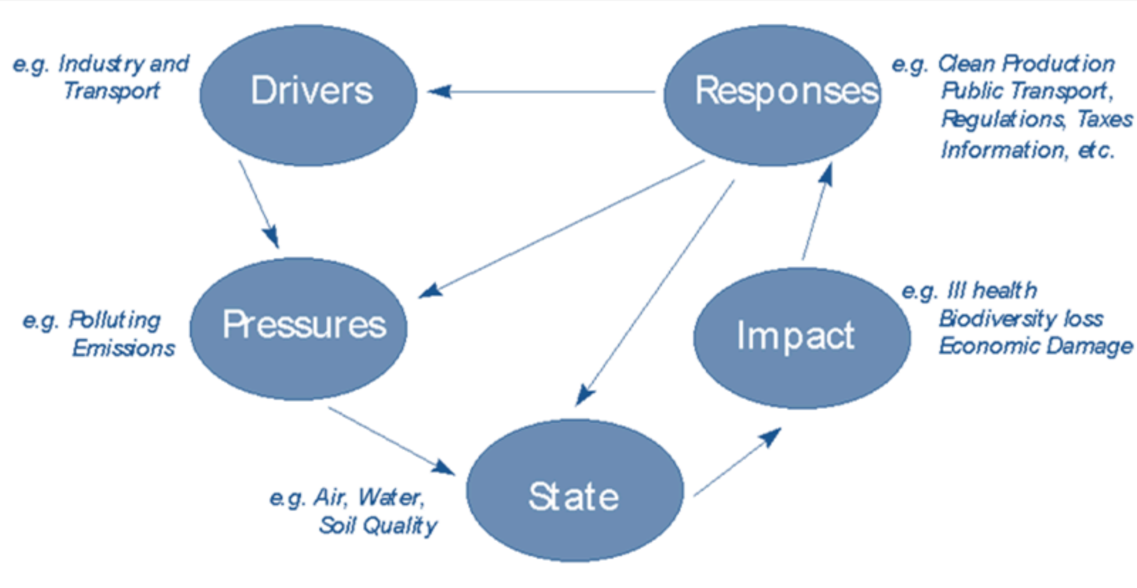


Figure 1 The DPSIR scheme applied to integrated air quality assessment modelling.

Each block can be represented to different levels of detail based on the requirements of the air quality problem the IAM targets, the policy objectives, methodological choices and last but not least the available data, tools and expertise of the staff. In this text we will distinguish three levels of complexity (1-3) where the highest level of complexity corresponds to level 3. The highest level of complexity implies a better description of the block, a better knowledge of the parameters on which to act with the objective of an improvement of the AQ and eventually the health and other impacts. In general, more complexity will go hand in hand with more detailed input data.

As blocks are interconnected and feed into each other, choice of complexity can be dictated by other blocks. For example, to produce detailed health impact results will require a detailed concentration map which, if produced using a CTM, in turn depends on a detailed, good quality emission inventory. If then a decent emission inventory is missing and impossible to acquire in the time and/or financial frame of the IAM this will also imply that it is of no use to try to produce a detailed concentration map and health impact assessment. There are however also situations where blocks of different complexity could and should be applied ('fit for purpose'). A good example might be where a single emitting source is obviously responsible for the AQ problem. In such a case source apportionment and/or an IAM accounting for all emission sectors and available technical abatement measures will have little added value while a detailed assessment of state and health impact might still be very useful to design a more efficient action to take.

2.4 What is the difference between the base case, the reference case and the scenarios?

To assess the effect of responses or abatement measures, an IAM has to assess the related change in impact. This implies a comparison between the impact with and without the measures. The base case, then, corresponds to the drivers, pressures, state and impact that correspond to a situation without the additional measures that are planned to improve the air quality and for which the IAM is used The scenario will correspond to the values of drivers, pressures, state and impact when measures are implemented. Both base case and scenario are considered at the same moment. The reference case refers to the situation in a specific (‘reference’) year, and is mainly used for the validation of (part of) the system, for example of the “state” block by comparison of results of model against results of measurements. Only in the case that the future conditions (meteorology, population, socio-economic conditions, etc.) when the abatement strategy is planned to be applied, those conditions are assumed to be the same as the reference situation, the reference case is then equal to the base case.

In general, a first challenge is to set up a reliable estimate of the activities, emissions, state and impacts for the reference year for which actual data exists. However, and even if we don’t dwell on this any further in this document, it is important to point out that it is a quite different challenge to provide a robust enough view of the future estimates related to the future policies to be implemented for a specific time horizon. To support decisions, a robust assessment of policy options could require taking into account a range of possible 'future worlds' and thus evaluate the effect of the different options (measures) on a set of possible base cases.

3 DRIVER: the activities and technologies that affect the emissions

3.1 What is the DRIVER in an IAM?

The DRIVER block combines **socio-economic activities** and their **technologies** into quantities that can be related to emissions.

How can activities be considered in an IAM?

In order to simulate future changes of the quantities that affect emissions, a model of the DRIVER block is required. This can be a model as simple as an expert judgement, or a more complex model that simulates (specific aspects of) socio-economic activities and technologies.

Changes in activities and technologies can, for example, be due to changes in population and land use, economical activities, transport needs, legislative requirements, natural renewal rate of technology stocks. This information is (dis)aggregated in a way relevant for emissions into sectors, sources and technologies.

For the reference case, emission inventories can be directly used, reflecting implicitly the underlying DRIVER block. For future base case and scenarios, however, it is important that the DRIVER block realistically translates 'the future world' of the policy horizon year (activities and technologies) into future emission. Therefore the assessment of future developments of the DRIVER block typically requires a more sophisticated framework than what would be needed for the reference situation.

In general, for the DRIVERS block implementation, the following **three-level classification** is proposed:

- **LEVEL 1**: when a top-down approach is applied, using coarse spatial and temporal allocation schemes;
- **LEVEL 2**: when a bottom-up approach with generic (i.e. national/aggregated) assumptions is applied, using more realistic spatial and temporal allocation schemes;
- **LEVEL 3**: when a bottom-up approach with specific (i.e. local/detailed) assumptions is applied, using local spatial and temporal allocation schemes.

3.2 Which activities and emission relevant technologies should be considered as driving forces for a local scale IAM?

The activity sectors which are most relevant for local (urban) scale IAM are road transport, off-road transport and machinery due to, for example, construction, residential heating and, to a lesser extent, energy production and industry. Though agriculture is typically of minor concern in a city level assessment, agriculture is a major source of ammonia emissions and is an important contributor to secondary PM formation. The same is true for activities such as forestry that result in biogenic emissions of VOC and PM. Source apportionment can help to identify those emission sources and thus activities that are most relevant to the local situation considered.

In addition to the activity rates in different sectors, technology information is important for 'emissions' assessment. Emission relevant technology parameters include vehicle engine technologies and fleet age, residential combustion technologies and emission abatement technologies used in energy and industrial plants. For future scenarios, reliable estimates on how the technology stocks evolve are essential as well as the possibilities to apply non-technical measures which affect the activities.

For the transport sector, activity estimates (traffic fleet composition and vehicle kilometres driven) are for example given in TREMOVE (www.tremove.org). Similar data set can be found for the energy sector in PRIMES (*this site*) or in the CAPR modelling system (www.capri-model.org) for the agricultural sector.

3.3 What is the recommended way to derive the activities and emission relevant technologies?

The inventory of activities and emission relevant technologies can be based on the data collected or modelled for the respective city area or region (bottom-up approach) or on statistics of a wider area (typically a country) of which the share of the respective city area or region is defined using weighting factors to disaggregate data (top-down approach). As an example of a top-down approach, legislation providing minimal efficiency requirements for technologies can be used as a proxy for relevant technology parameters.

In general, a bottom-up (= local) approach to derive activities can be considered to be more favourable as it uses, by definition, information on activity data from the respective city or region directly. However, in many cases it might be difficult to obtain reliable, representative data. Furthermore, a technology stock inventory at sub-national level is often not practical so that national level data are used (top-down approach). In case of a top-down approach, the reliability of the activity estimate depends on the representativeness of the weighting factors used. In case representative weighting factors are available for each sector, a top-down approach can produce a reliable activity estimate. To conclude: unless this is not possible the recommended way to derive activity data is to use bottom-up information.

3.4 What level of detail is required to model the driver activities?

The overall detail required for the driver model is determined by the air quality problem that is addressed by the IAM application. In addition, this detail should be in line with the level of detail in the other IAM blocks (pressure, state…).

Driver activities need to be characterized in terms of activity types, activity trends, and emission factors and can also contain spatial/temporal details. The level of detail for activity sector and technology aggregations should reflect the importance of the sector in emissions quantities and the effect of the technology on emission factors and is limited by the availability of emission factors for the detailed technologies. E.g. for the transport sector, the minimum requirement is to distinguish between the main relevant vehicle types in terms of emissions (e.g. light/heavy-duty, diesel/gasoline) and technologies (e.g. EURO-levels for cars).

The activities can be implemented with different levels of complexity, from a simple calculation using a cause-effect relation to the use of detailed traffic, housing and energy system models. City or regional level assessments can be implemented as city or regional level models (bottom-up), allocated from national level models (top-down), or as a combination of both approaches.

To assess the impacts on urban air pollution and to provide information in an appropriate format to the PRESSURE block, it is important to know not only the quantity but also the physical location and temporal variation of emission release. Therefore, in order to be able to resolve the emissions in space and time, the activities must also be allocated to certain grid and temporal patterns. Appropriate spatial and temporal resolutions in city or regional level assessments are typically 1 km (or finer) and 1 hour, respectively (e.g., King et al. 2006, Cuvelier et al. 2007, Karvosenoja 2008, Baldasano et al. 2009, Keuken et al. 2013, Guevara et al. 2013). It should be noted that 1 km spatial resolution enables only the assessment of urban background concentrations; in order to compare modelled concentrations against, e.g., air quality limit values in urban traffic locations, an even finer spatial resolution down to tens of meters is desirable.

In most sophisticated assessments, traffic, housing and energy system models with spatial information can be used. Models with dynamic spatial capabilities are also desirable to be able to assess changes in spatial patterns of activities. In case such activity models are absent, spatial gridding relies on the availability of GIS data and can be used as a proxy for different sectors. For example traffic activities can be gridded using road network and traffic volume information for each road segment. If not directly available, driving characterization and/or velocity information for the road segments can be linked to e.g. road type or speed-limit information (Baldasano et al., 2010). The temporal variation for different sectors can be based on internationally, nationally or locally defined default variations (e.g. Denier van der Gon et al. 2011) or local data obtained e.g. through questionnaires or observed data.

3.5 Which elements in the DRIVER are most uncertain?

Regarding their contribution and impact on local air quality levels (EEA, 2013) the most uncertain activities are road transport and, mostly in winter time, the residential combustion of wood and the combustion of coal/coal-substitutes as still found in some Central/Eastern European countries.

Although the transport sector is the major source of nitrogen oxides in urban areas, the compilation of reliable road transport emission inventories is still complex due to the high uncertainty levels associated with emission models and activity data. Most of the common models used (e.g. COPERT IV) have emission factors developed from vehicle emission tests in laboratories (Smit et al., 2010) which can substantially differ from real-world vehicle emissions. As shown by Kouridis et al. (2010) the uncertainty in the calculation of total emissions depends mostly on the inherent uncertainty of the emission factors rather than on the uncertainty of the data provided by the inventory compiler. Furthermore, key activity information such as fleet composition profiles based on real circulation data, which can differ substantially from statistical national fleet compositions, is still scarce and difficult to find.

On the other hand, the residential combustion sector merits further research as it has recently been found to be one of the major contributors to fine particulate and other organic air pollution in many urban areas in Europe. Residential wood/coal/coal-substitutes combustion activity information can hardly be based on sale statistics because a lot of the wood/coal/coal-substitutes fuel is from private stock and is used privately. Furthermore, household level wood heating system stock is often poorly known because such information is rarely gathered into registers. Therefore activity and combustion technology estimates have to be often based on questionnaire information about wood use amounts, frequencies and type of combustion appliances. Additionally, the questionnaire should include information about wood combustion user's practices (the ways of batching, ignition, combustion air supply, fuel quality etc.) because these parameters may have an essential impact on emission factors and would be therefore needed in the PRESSURES block assessment. (e.g. Savolahti et al., 2014, Ozgen et al., 2014).

Spatial assessment (i.e. gridding) of residential wood/coal/coal-substitutes combustion activities is important in order to assess the impacts of possible emission reduction measures and other interventions on local air quality inside the city area. Gridding might be challenging because of the lack of building registers with household level information about residential wood heating appliances. Spatial distribution of residential wood combustion activities typically differ considerably from that of many other urban emission sources (e.g. traffic) or most of the simple gridding surrogates (e.g. population density), and therefore the direct use of these surrogates results in severely incorrect spatial distribution. To consider gridding methodologies for residential wood combustion, the key question is availability of spatial (GIS) data. An optimal situation would be to have a building register with house coordinates and information about wood heating devices and their use however such data is rarely available. If there is a building register with information about main building types that are relevant for the wood use in the country (e.g. residential/other or apartment/detached/semi-detached), and an estimation of urban/rural differences in wood use, a relatively good approximation for a large area average can be achieved (Paunu et al. 2013). If a building register is missing, population data could be used. Preferably these population data should then be modified to rule out areas with limited residential combustion activities that contribute to PM e.g. areas with the highest population density might represent areas with apartment buildings where wood burning is unlikely (Paunu et al., 2013).

3.6 How should I account for the different scales in the DRIVER model?

The different activity changes, relevant for scenarios to be used in an IAM, are affected at different decision levels. Activity changes in the form of energy production, fuel switching and industrial production changes are affected largely at the international (e.g. global markets) and national (e.g. national taxation) scales. These factors, however, are typically well taken into account by respective local actors, e.g. energy production companies or other industrial actors. Thus, it is recommended to look for ways to directly use companies' strategies and knowledge for the energy and industry sector. On the other hand, population, housing and transport demand changes are affected largely at city (e.g. city taxation policies, general "attractiveness" of the city) and sub-city (e.g. traffic planning, zoning policies) scales. These local scale factors often have a strong effect on the spatial occurrence of activities and they should be taken into account in the respective activity models. For a comprehensive local level activity scenario, above-mentioned factors affecting both the local and larger scale should be addressed.

Technological changes, that are relevant for the PRESSURE block, are also affected at different scales. Many of the emission-related (e.g. EURO standards, EcoDesign and Industrial Emissions Directive) and climate-related (e.g. Renewable Energy Directive) legislation that influences technological developments are defined at EU level. National level decisions may have a big impact as well (e.g. consumption or emission based vehicle taxation). The above-mentioned technological changes must be included in activity scenarios because of their prominent effect on emissions. The technological changes that are influenced by EU or national scale legislation or policies can be adopted from national level assessments if a local level estimate is not available (e.g. transport vehicle technology stock adopted from national average information may be adequate). At city level it is possible to influence local problem spots (e.g. low emission zones, prohibition of residential wood/coal/coal-substitutes combustion) and set more general goals (e.g. city climate strategies) that influence technological developments. Obviously, technological changes adopted at the city level must be assessed at the local level.

4 PRESSURES: emissions

4.1 How is PRESSURE related to the DRIVER in an IAM?

The PRESSURE block describes the emissions as the result of the combination of activities and their technologies given by the DRIVER block and possible "end-of-pipe" abatement technologies.

4.2 How can I compute emissions for use in an IAM?

An emission inventory is a database combining emissions with a specific geographical area and time period. The emissions themselves are calculated by multiplying the activity with an emission factor, in particular considering:

- the activity of the emission sources, quantified for instance by the amount and the type of fuel burned, or the number of kilometres travelled by the vehicles, etc.;
- the amount of pollutant emitted by these sources per unit of activity linked to the technology used, i.e. the emission factors;
- additional end-of-pipe abatement technologies that reduce the overall emission.

The emission inventory may have different levels of detail depending on the availability of data and their uncertainties. Data could be given per activity sector, technology and fuel. The detail required with respect to these aspects depends on the appropriate aggregation required for calculating emissions reliably and the types of abatement measures that are taken into consideration in the RESPONSE block. For application of IAMs, information on costs and rates of application of technologies, which are needed to calculate the cost/benefit of applying this emission reduction measures, have to be included as well.

At the national level emission inventory guidelines can be found in the EMEP/EEA air pollutant emission inventory guidebook (EMEP/EEA, 2013). Emissions can be estimated through 3 different **levels of complexity** as described in Deliverable 3.1, depending on their further use (e.g. level of detail required by the abatement measures) and data availability.

- **LEVEL 1**: Emissions are estimated for the 11 macro sectors (SNAP1 classification) at a low spatial resolution (e.g. national level), using by default a top-down methodology. Uncertainties are not necessarily estimated. Due to the limited detail in the sector contributions and spatial resolution, this level does not allow for detailed emission scenarios at a local scale.
- **LEVEL 2**: A combination of bottom-up and top-down methodology is used to calculate the emissions (SNAP2 or SNAP3 classification). Emission factors and activity data representative for the study area are used when available. Uncertainties are not necessarily estimated but it is recommended to at least evaluate these qualitatively.
- **LEVEL 3:** Emissions are calculated with the finest space and time resolution required for the purpose of the IAM application, with a bottom-up approach and finest level classification at least for the significant emission sources for the area of interest. Emission factors and activity data have to correspond to the specific activities (SNAP3 classifications) and fuels of the area under study. The processes have to be detailed as well as possible, to attribute the most representative emissions. In case data are lacking, a top-down approach can be used but with the help of

complementary data to take into account regional specificities. The uncertainties must be quantified. The level 3 is the most appropriate to generate all kinds of scenarios. A necessary condition for a scenario to be considered is that the emission changes, due to the considered abatement measures, must be larger than the uncertainties for the emission values.

Other emission related inputs that are required are the specific effective height of pollutant release (e.g. Bieser et al. 2011) and the chemical speciation for NOx, NMVOC and PM2.5 that should be consistent with the applied AQ model chemical mechanism (see e.g. François et al., 2005 *or* Juda-Rezler et al., 2012 for a detailed description of emission data preparation). Biogenic emissions can be taken from the European databases (e.g. NatAir database) or can be calculated in the model as a function of meteorological parameters and vegetation cover using a state of the art model such as MEGANv2.1 (e.g. Guenther et al., 2012).

When the focus is on one or more specific areas, the actual contribution of emissions to atmospheric pollution can be tested and estimated using source apportionment techniques. Moreover, the identification of the key emission sources in the area of interest would allow a better delimitation of the problem and therefore to allocate resources to study in depth the identified more relevant sectors of activity, so that no resources are diverted to get details on emissions for which the contribution is of minor importance in that area of interest.

4.3 Can I use existing European inventories?

If no regional inventory exists to run an IAM, the emission database can be based on European inventories (e.g. EMEP inventory or TNO-MACC inventory). In case of the EMEP (European Monitoring and Evaluation Program) inventory (Vestreng et al. 2007), annual emission totals are available in gridded form at a 50 km × 50 km resolution for CO, NOx, SO_2, NH_3, non-methane volatile organic compounds (NMVOC) and primary PM fractions. The TNO-MACC inventory provides similar gridded emission over the European domain at a resolution of 7km x 7km (Kuenen et al., 2011,Kuenen et al., 2014). However all these are examples of inventories where the level of complexity is 1.

These annual total emissions should be further disaggregated in time and space (e.g. Ferreira et al., 2013). The temporal distribution can be calculated via sector-dependent emission profiles (e.g. sectorial emission profiles provided by the LOTOS-EUROS emission model, Builtjes et al. 2003). These sector-dependent emission profiles define different distributions for the month, the day of the week and the hour of the day. The low resolution emission inventory can also be further disaggregated using maps of spatial surrogates that can be related to emissions such as population density, road maps, or land use.

4.4 How should I compute emission scenarios?

Emission scenarios can be produced in several ways depending on the policies defined in the RESPONSE block:

1. by modifying the activity data of the DRIVER block. Some emission sources can be added, removed or moved to other locations, the level of activity of those sources can also be changed (increased / decreased). E.g. for traffic the number of mobile sources per unit time can be changed (including time distribution for defined periods as days, months, years).

2. by modifying the emission factors of the emission generation processes or end-of-pipe measures. This includes new technologies or technological improvement, industrial processes, changes in fuel types or characteristics, energy saving (in terms of efficiency), composition of the vehicle fleet for mobile source, etc.

The policy scenarios are limited to the Current Legislation (CLE) and Maximum Feasible Reduction (MFR) scenarios. The CLE takes into account all measures which are politically agreed and where there is a formal decision to implement these in the time horizon of the IAM. The MFR scenario includes all measures which are technically feasible within the time horizon of the IAM.

The level of detail of the scenarios is highly dependent on the level of classification of the sources and the data available for each sector. In other words, the emission scenarios may be very simple and derived from the application of an emission reduction rate directly on the gridded spatial emission inventory (SEI) or they may be the results of assumptions on the future scenarios of the activities and the emission factors. As detailed in EMEP/EEA (2013), future activity assumptions are based on a range of datasets including scenarios of industrial growth, population growth, changes in land use patterns, and transportation demand. Energy models are often based on general equilibrium theory and combine the above basic growth factors with energy price information to estimate energy demand by sector and fuel. These models can be used as a core dataset as long as the assumptions underpinning them are consistent with national economic strategies, policies and measures. Future emission factors should reflect technological advances, environmental regulations, deterioration in operating conditions and any expected changes in fuel formulations. Rates of penetration of new technologies and/or controls are important in deriving the right sector dependent emission factors for any particular scenario year. For the transportation sector, this kind of information is collected in the TREMOVE data base for 31 countries for the period 1995-2030 (www.tremove.org). TREMOVE relies on COPERT (www.emisia.com) emission factors. However, the generic emission factors reported by COPERT are under debate since real world driving conditions may results in higher emissions as e.g. reported by the ERMES group (www.ermes-group.eu) or by Fontaras et al (2014).

4.5 How should I use and combine emission inventories for different scales?

In practice, it is very difficult to manage, or even to find, a detailed and complete description of all the sources over large areas such as the scale of a continent or large countries. The large scale SEIs are therefore rather based on area than on point and line sources in comparison with small scale SEIs. The sources of large scale SEIs are more often calculated using top-down than bottom-up approaches. Consequently, the locations of the sources in large scale SEIs are not accurate and the scenarios based on such SEI on a fine resolution grid lead to an overestimation of the sources' dilution. Sources, then, need to be downscaled and redistributed using different earth surface characteristics defined at a smaller scale. For example, the emission can be redistributed according to the land use (e.g. emission release over land only and no emissions over water surfaces), the density of population (e.g. more emissions over densely populated areas like cities), the road network (e.g. road transport emissions only in cells crossed by roads), etc. (Maes et al, 2009). Instead of a simple redistribution proportional to these supplementary characteristics, which is typically done using linear regression, also more advanced approaches can be applied, e.g. using geostatistical methods, like kriging (Horabik et al. 2014, Verstraete, 2014)

When using AQ models, it often happens that an accurate detailed emission inventory is available only for the part of the grid domain on which the study has to be performed. In that case it is necessary to combine data provided from SEIs on different scales (Timmermans et al, 2013, OPERA, 2013). In this situation, the best procedure is, first, to project all the SEI outputs on the same grid and then for each cell to keep the data provided by the most accurate SEI. Even if there is a risk of inconsistency between the different SEIs because they have been produced using different methodologies (top-down or bottom-up for example) this procedure is a good compromise between consistency and accuracy. In this process it is important to take care of the sector and subsector definitions of the different SEIs. In some cases, the large scale and local scale inventories do not have the same sector classification, which requires an additional sector mapping between both data sets in order to arrive at one overall consistent emission inventory.

4.6 How can I evaluate the uncertainty of an emission inventory?

The uncertainties of an emission inventory can be evaluated in a qualitative or quantitative way. The qualitative evaluation is mainly performed by experts (IPCC, 2000; EPA, 1996), while the quantitative one is based on error propagation and Monte Carlo Methods. There is also a semi-quantitative method that can be used to evaluate the uncertainties which consist in rating the data quality. In the latter case experts attribute some numerical or alphabetical scores to emission factors and activity data to describe the uncertainties of these data. There are two main classifications that are used in these semi-quantitative methods (EPA, 1996): (1) the DARS method (Data Attribute Rating System) that attributes a score ranging between 1 and 10 (the most accurate) to each dataset (e.g. Guevara et al., 2013); (2) the AP-42 emission factor rating system that is the main reference in the USA, and is only for emission factors evaluation where the scores range from A (most accurate) to E. Both methods attribute scores which are general indications on the reliability and the robustness of the data.

The methodologies should be applied to identify the emission contributions which affect the total uncertainties the most, so that the improvement of the inventory can focus on those sources for which the impact will be largest. If the uncertainties are quantified, their knowledge will determine which emission scenarios are meaningful; that is scenarios that result in emission changes larger than the uncertainty levels.

5 STATE: pollutant concentrations and deposition

5.1 How can I determine the concentrations needed for describing the state in an IAM?

As specified in the section 2.1, an IAM can be considered as a methodology for determining how emissions should be efficiently reduced to improve air quality, reduce exposure and protect human health. This implies that we require a description of the pollutant concentrations that links concentrations to emissions so that the impact of possible changes in emission on these concentrations can be evaluated. This causal relation which is inherent to a numerical AQ model is missing when only measurements are used to described the AQ state, so that if the IAM only relies on measurements it should be made explicit how these measurements relate to the emissions. How the consequences of one or more emission reduction scenarios on air quality are evaluated and with which level of detail, is part of the decision framework described in the RESPONSE block (chapter 7).

There are several methods to determine the AQ state for an IAM, depending on its objective and on the available tools. Some of these methods involve only AQ observations and emission sources but most of them are based on numerical models. Before applying a model, the user should first verify in the documentation that the model is adapted to the objective pursued.

The different methods that can be used to evaluate the AQ state can be classified according to three different levels of complexity:

- **LEVEL 1:** The simplest way to characterize the AQ state is to use measurements taken routinely or during a measurement campaign and to interpolate these to a grid with a geo-statistic interpolation method to obtain a map of concentrations over the area of interest. Such an assessment of STATE does not require any input on emissions and thus activities. For IAM application, the difficulty is therefore to link these concentrations to emissions; that is, to estimate the contribution from identified sources to observed concentrations (source apportionment). In general, this is done through the use of more or less complex receptor modelling techniques such as superposition of different scales (Lenschow et al., 2001), Principal Component Analysis (PCA) and Chemical Mass Balance (CMB), alternatively considering a relationship between reduction in emission from a source and reduction of the contribution of this source to the measured concentration (Belis et al., 2013).
- **LEVEL 2:** is based on a characterization of the AQ state using one single deterministic model adapted to the studied spatial scale. This model should be validated over the studied area and should use emissions input data adapted to this scale. Concentrations used as boundary conditions of the model can be either extrapolated from measurements or data extracted from a larger scale model. Observed concentrations can be used to correct the model (data assimilation) at least for the reference year, often used as a starting point for IAM applications. Another input to the model are meteorological data, which can be obtained from observations or from a meteorological model. Spatial and temporal resolution of the meteorological model should be adapted to the resolution used for the AQ model.

- **LEVEL 3**: is based on a characterization of the AQ state using a chain of AQ and meteorological models, from large scale (Europe for example) to regional (country or regions) and urban/local scale (city area) and street scale. The use of a downscaling model chain allows for taking into account the interactions between the various scales, such as the transport of pollutants at a large scale or interactions between mesoscale wind flows and local dynamics. Nesting between models can be one-way or two-ways, the latter allowing local information to be passed to the larger scale model run. Sub-grid modelling approaches can also be used to combine different scales. The same model could be used for different parts of the chain, running the model itself at different resolutions; or different models could be applied at the different scales with local models such as Gaussian models using boundary conditions from a larger scale Eulerian model. For each part of the downscaling chain, emissions should be adapted to the model in term of spatial and temporal resolution. Data assimilation and meteorological data representativeness issues are similar to the ones described for Level 2.

5.2 How should I choose an air quality model for determining the state in an IAM?

IAM may be used for a number of aspects in AQ policy preparation or evaluation. Most of these aspects are related to a specific requirement of the EU AQ Directive but also other complementary decisions taken at local or national level. There where common practice in AQ assessment is to check compliance to limit levels mandated by law, one should be aware that this in itself does not guarantee that health impact is minimised. Successful AQ policy and management requires accurate and detailed information on ambient AQ levels, in order to assess the state and detect any problems that may be relevant to health impacts, but as well to assess ways to solve an exceedance of legislated limit values. Also, an AQ policy refers to a specific area. For example a local policy may focus on the impact of a stationary source. On the other hand, a national policy will have to consider the spatial and temporal distribution patterns of emissions from a number of different sources. These different assessment needs have to be taken into account when selecting an AQ model and it is therefore important to note that not all AQ modelling tools are relevant or “fit-for-purpose” to assess AQ as a response to a specific pressure (i.e. air pollution source) or at a certain scale.

Depending on the chosen scale, an AQ model of appropriate spatial coverage and resolution has to be selected. AQ assessment for policy planning concerns regulated pollutants and pollutant levels that have to be assessed by comparison to legislated limit values or health guidelines. The temporal resolution (frequency of outputs) of the model(s) used to calculate concentrations should therefore be consistent with the temporal resolution of limit values or the temporal resolution required for assessment of the health impact. For AQ assessment covering a regional or urban scale down to a resolution of a few km^2, Eulerian chemical transport models are suggested as the most appropriate tools. Most Eulerian type models have the advantage of providing the option to select either a coarser or finer spatial and temporal resolution for the AQ simulation and have the added benefit of considering scale interactions through the nesting capabilities. To accurately describe the concentration in the vicinity of specific sources e.g. the spatial variability of NO_2 near roads as required for detailed exposure assessment, micro scale modelling is necessary. In those cases a Gaussian plume model, a street canyon model or a CFD (Computational Fluid Dynamics) model are often the most appropriate tools. Other examples where a Gaussian model (Daly and Zannetti, 2007) is recommended are future emission scenarios in the case of granting an operation permit for a new industrial development, or an accidental release scenario from a specific emitting source. In general, model applications are multi-scale and have to be tackled using a combination of possibly different types of models.

Apart from the considerations of scale and resolution, the type of pollutant itself should also be taken into account. Eulerian AQ models are suitable for the simulation of transport and transformation processes of both primary and secondary pollutants, as they include comprehensive chemistry schemes. On the other hand, Gaussian, statistical and Lagrangian model types do not account for chemical processes and may not be suitable for the case of secondary pollutants such as ozone, NO_2 and PM (secondary aerosols). Limitations in the use of Gaussian models may also relate to the characteristics of the topography. If the surrounding area features a complex terrain, a Gaussian model with appropriate treatment of complex terrain has to be selected.

A comprehensive database of widely used and validated modelling tools is available at the EIONET Model Documentation System *web site*. In addition, COST728 has developed a model inventory that provides information on a significant number of mesoscale air quality and meteorological models. Detailed technical guidance on best modelling practices for assessment purposes can be found in the EEA technical report 2011/10 (Denby et al., 2011). Section 6.6 details how the quality of the model or the uncertainties of its results can be evaluated.

5.3 How can Source-Receptor models be used to determine the state?

Two approaches can be distinguished when applying an IAM. The first one, the scenario analysis approach, consists of the assessment of the impacts of an emission scenario on the air quality indexes to eventually derive health exposure, ecosystems impact, related costs. The air quality is usually estimated using deterministic 3D modelling systems, describing the linear and non-linear chemical and physical processes determining the air pollutant concentrations. An alternative is the optimization approach that implements iterative procedures to select the measures to minimize these same AQ indexes. Such procedures require to estimate the STATE, namely the AQ indexes, thousands of times once for each precursor emission scenario defined by the optimization algorithm. In this case, Chemical Transport Models (CTMs) cannot be applied directly in the optimization procedure since they require too much computational time. Therefore simplified models (so called source-receptor models) are used in the optimisation procedure to synthesize the relationship between the precursor emissions and the AQ index instead of the CTM.

In the literature, source-receptor models have been described by isolines (Shih et al., 1998) or by reduced form models such as (i) simplified photochemical models, adopting semi-empirical relations estimated on experimental data (Venkatram at al., 1994), and (ii) statistical (linear or non-linear) models, identified by processing a number of CTM simulations (Friedrich & Reis, 2000; Guariso et al., 2004; Ryoke et al. 2000; Carnevale et al., 2009) that can vary from tens (as i.e. in Carnevale et al., 2009) to thousands (as i.e. in Amann et al., 2011) depending on the type of source-receptor model used, input structure and spatial variability.

5.4 How can I use observational data with my model results?

Measurements contain valuable information which can be used together with modelling results in an IAM. Monitoring data generally estimate the true concentration levels more accurately than model results but are only representative of the measurement point itself. So, the main question which arises in integrated assessment applications is: "how can these measurement data be used most appropriately?"

One common use of observational data is obviously for evaluating the capability of the model to simulate existing air quality. Several procedures exist for this purpose (see Section 6.6). A standardised evaluation protocol organised according to the different modelling needs and characteristics with a particular focus on applications related to the Air Quality Directive is for example currently being developed in the context of FAIRMODE (Thunis et al., 2011). The results of the evaluation may be used to provide a qualitative uncertainty assessment of the final AQ indexes computed by the IAM, often limited to a discussion on the over- or under-estimation by the model.

By applying data assimilation techniques (Elbern and Schmidt, 2001; Blond and Vautard, 2004; Honoré et al., 2008), measurement data can also be used to improve the model results and provide a more accurate estimate of the concentrations for the reference case as a starting point in the integrated assessment. One can note that the set of observational data used in the data assimilation process should be adapted to the spatial resolution of the model. Indeed, observations are not always representative of what a model is able to simulate. When models are used with a relatively coarse resolution, it is recommended to select only observations representative of the background air pollution. To deal with the representativeness of any observation data, an approach consists in using statistical downscaling techniques (Colette et al., 2012; Kiesewetter et al., 2014).

Data assimilation techniques are useful on time periods over which measurements are available, or short term forecasts, but are of limited use when it comes to improving scenarios. For the specific case of scenario analyses, the possible options are:

- Propagating in the future a constant bias or quantile mapping correction
- Propagating in the future quantile mapping corrections that take into account the change in the underlying distribution (Colette et al., 2012);
- Developing a parametric model of the error, that can be applied in the future (Kiesewetter et al., 2014). This method can be classified as a Model Output Statistics technique.

Observational data can also be indirectly used to assess the air quality modelling system used to derive the source-receptor relationships. The information about sources retrieved from receptor models (i.e. derived from observations) can be compared with the output of AQ models to obtain a better quality and understanding of the whole system.

5.5 What would be an appropriate time horizon for an IAM?

AQ planning is often performed for a long time horizon. Since the air quality of a specific year may be strongly affected by the weather conditions of that year, it is important to evaluate the efficiency of the AQ plan over a longer period to filter out the inter-annual variability in air quality due to the variability of weather conditions. In general, it is preferable to not only consider a representative year as a single year with 'normal' meteorological conditions when selecting the appropriate period for an IAM.

Several techniques can be adopted:

- Use the air quality models to simulate the air quality for several years (more than a decade). The advantage of this approach is that it gives a complete representation of the variability in air quality due to the variability in weather conditions from one year to the other, and allows, in this way, to appreciate the impact of an emission reduction strategy for years characterized by different weather conditions. Moreover, it does not first require an analysis of the data. The disadvantage is that it is computationally demanding. This high cost in terms of computational time may preclude the use of a fine spatial resolution (needed for health impact studies), or the simulation of a large number of emission reduction situations (needed to design the Source-Receptor models needed for a optimization approach).
- Analyse the time series of concentrations measured over a decade or two, and combine them with an analysis of the weather patterns of each specific year compared to the climatological values. This analysis will allow eliminating anomalous years where the concentration time series are significantly different from those of the other years and these differences are not due to changes in the emissions; and allow to select a representative year that will be used to evaluate the efficiency of the AQ plan. The advantage of this approach is that it is less computationally expensive compared to the previous one, since only one full year needs to be simulated. On the other hand, it does not allow evaluating the variability in AQ due to the meteorology from one year to another. To take into account this variability, it is possible to select a few representative years within a time period (Tarrason et al., 2005).
- Perform a climatic synoptic classification over several decades of circulation types (CTs) to characterize air quality dynamics over a particular domain. Sensitivity tests to automatic classification techniques and to other factors affecting classification should be performed in order to insure quality. Then, a representative day (or group of days) for each CTs is identified using a daily score based on the sum-of-squares difference (Kirchofer score). The relevance, for air quality, of the days selected can be checked by means of back trajectory analysis with models (Valverde et al. 2014). A CTM is in the latter used to simulate those days representative of the CTs that for example historically generated more severe air quality episodes. Results can then be scaled up to a year considering climatological frequency of the different CTs. This technique requires a significant amount of analysis of the data to select the CTs and the representatives days, but it is the less demanding in terms of computational time to perform the CTM runs as only a limited number of days must be simulated. The computational time saved with this technique (compared to the two previous points), can be used either to increase the resolution, or to span a wider range of emissions reduction scenarios.

As a final comment it should be kept in mind that the choice of the period for the IAM study should take into account the specificities of the studied pollutants. This determines the scale of the problem, the resolution needed, the type of model, and the type of indicator that must be adopted to evaluate the air quality. All these aspects are relevant in the choice of one of the techniques described above.

5.6 How should I evaluate the quality and the uncertainty of the state?

When the AQ state is evaluated through measurements, the uncertainties are related to the measurements themselves, to the geo-statistical methods used to interpolate point measurements and to the representativeness of measurement sites to characterize the area under study.

The AQ models applied to simulate ambient AQ as part of IAM need to be tested and assessed in order to ensure that they meet certain quality objectives before they can be considered to be suitable for policy making. These quality objectives are described in Annex I of the AQ Directive 2008/50/EC and although in the Directive they explicitly refer to assessment of current AQ levels, there is a clear expectation that models used for future policy planning also have to be validated.

To be able to rely on model results, both model performance evaluation as well as uncertainty estimation are of imperative importance. Four types of model performance evaluation are generally recognised (Dennis et al., 2010):

- Operational model evaluation seeks an answer to the question "Are we obtaining the right results?". The method involves comparison of model results with routine monitored data and refers to both meteorology input data as well as to air quality output data of air quality models. More details can be found in the FAIRMODE WG2/SG4 report (Thunis et al., 2011).
- Diagnostic model evaluation seeks an answer to the question: "Are we obtaining the right results for the right reasons?". The diagnostic evaluation methodology is a process-oriented analysis to determine whether the individual physical and chemical processes are correctly represented in the model and investigates the way in which these physico-chemical processes can influence model results. Diagnostic model evaluation is a validation methodology that should be undertaken by model experts and model developers in the process of developing or improving an AQ model.
- Dynamic model evaluation seeks an answer to the question: "Can we capture observed changes in air quality"?. This model evaluation methodology is highly represented in the literature and is also known as the "sensitivity method". The objective is to analyse if the model responds in a correct way to changes in input data (for example emissions, or meteorology). Giving that the overall objective of an IAM is to determine the most effective change in the emissions that improves air quality, it is clear that this is the most relevant type of model evaluation for an IAM. Moreover, it is important to note the importance of the sensitivity method in the evaluation of AQ models that are used for source apportionment in the frame of policy planning, particularly when dispersion models are applied for the identification and attribution of sources.
- Probabilistic model evaluation seeks an answer to the question: "what is our confidence in model predictions?". It is performed on the basis of methods such as model inter-comparison and ensemble modelling, and attempts to capture statistical properties, including uncertainty or level of confidence in the model results, for regulatory model applications. Probabilistic model evaluation is particularly helpful for predicting the accuracy of model results when monitoring data are not available, such as in the case of future emission changes, and is

essential for future planning purposes (Hogrefe and Rao, 2001). However, it requires considerable computer and time resources and expertise in order to perform simulations with different models and is thus rarely used for policy planning purposes. Model inter-comparison is also particularly important in the case of receptor models used in source apportionment studies as these models don't have a physical basis (Favez et al., 2010).

In conclusion, as a minimum, an operational evaluation of the AQ model is required by comparison with measurements that are representative for the model scale. A number of software tools to facilitate operational model evaluation are available for use, such as the BOOT software (Chang and Hanna, 2005) and AMET (Appel et al., 2005) for dispersion models. Also in the EEA Technical report No 10/2011 (Denby et al. 2011) more information can be found on the application and evaluation of AQ models for policy purposes according to the needs of the EU Directive 2008/50/EC. The topic of operational evaluation is also being tackled in the FAIRMODE community where a model evaluation and benchmarking procedure is being developed together with a tool, the DELTA tool (Thunis et al., 2011; Thunis et al., 2012).

Evaluating the indefiniteness of a scenario calculation for future planning is more challenging than assessment of current air quality levels, as no measurements are available for comparison. Therefore, the use of dynamic (e.g., sensitivity tests) or probabilistic evaluation (e.g., errors propagation) in addition to diagnostic evaluation are in this case recommended.

6 IMPACT on health

6.1 Why do we consider health impact in an IAM?

The directive 2008/50/EC requires to draw up and implement air quality plans and short-term action plans to avoid, prevent or reduce harmful effects on human health and the environment as a whole (Chapter 1, Art.1).

In this document we limit ourselves to health impact but other impacts such as environmental and climate as well as the synergies and antagonism between these different impacts could also be considered in an IAM. An IAM is, in this sense, a framework where you can include/integrate all these impacts in a way that these can be taken into consideration by policy makers in the decisions that are taken.

Considering health impact can help the choice of non-regret measures by going beyond compliance to concentration limit values and considering the actual health impact in the assessment for comparison.

6.2 What levels of complexity can we achieve in health impact assessment?

The level of complexity that can be attained for the health impact assessment (HIA) is determined by the input data used (i.e. STATE and PRESSURE) but also by the level of detail of the other parameters used such as population distribution, level(s) of vulnerability used for the whole population or for some specific groups, and the detailed location of the population with respect to their exposure.

The STATE and PRESSURE blocks provide data that define AQ as the concentration of pollutants present in the air at a regional/local scale to which humans are exposed. **The difference between AQ concentrations and exposure to AQ requires human beings to be present and inhale the air.** Indeed, the ozone concentrations measured or modelled for the stratosphere do not account for human health, people live in the layer of air called the troposphere. In a similar way, we spend most of our time indoors (where we live, work, play), and the quality of the air indoors even if based on the outdoor air accumulates emissions from other sources, often not yet measured, modelled or even taken into account in the AQ plans.

Exposure itself then refers to the pollutant concentrations in the 50 cm diameter sphere of air around the mouth and should preferably be measured using personal exposure devices. However, in practice measured or modelled outdoor AQ data are often used as an approximation to exposure in many epidemiological studies neglecting the fact that in reality much time is spent indoor especially in the case of vulnerable groups. When using the results of such epidemiological studies for an IAM the exposure modelling approach needs to be consistent with the one used in the epidemiological exposure model.

The process between exposure and health effects implies causality, physiological mechanisms and clinical reaction called health outcomes. Pollutants interact and multiple exposures are more and more studied. The assessment of the impact on health, within an integrated assessment modelling framework, requires one to distinguish between the exposure to air pollution and the health effects of that exposure to air pollution.

Therefore, it is possible to consider three levels of complexity :

- **LEVEL 1**: A level 1 approach requires a coarse "exposure" estimate of AQ provided either by measurement or modelling (e.g. average mean annual exposure for a city), a dose-response function or concentration-response function and a simple population description. This will result in a single number to roughly indicate the 'average' exposure for the considered territory (for example a city or a country). An example could be the number of hospital emergency visits related to increased ozone levels for a city or region. Another example is the number of premature death avoided when meeting the WHO guidelines for PM2.5 for one city such as presented in the APHEKOM data, where for the Brussels Capital-Region the population life expectancy on average could increase by 7 months.
- **LEVEL 2**: Similar to level 1, but with spatial detail in the STATE description, so the variation in space of the AQ is taken into consideration. Individuals from 2 neighbourhood cities are not always exposed to the same AQ.
- **LEVEL 3**: A level 3 approach requires a detailed "exposure" estimate based on detailed, temporal and spatial, concentration and population information and will allow deriving health impact information taking into account aspects such as distance to a road, spatial distribution and vulnerable groups for instance. We specify the level of detail with respect to exposure in one hand and to health effect on the other hand. An example is to use the number of hospital emergency visits of specific age and vulnerability groups distinguishing the population according to the traffic density at the place where they live and relating this to local changes in ozone and season. Another example could be given by the results of the geo-referenced analysis of PM2.5 exposure due to road proximity in the development and exacerbation of chronic cardio-respiratory diseases such as asthma in children compared to a case using average "exposure" for the population as a whole. The use of individual dosimeters would validate the "exposure" information if available for some parameters.

It is important to remark here that a study that evaluates an abatement strategy only based on the compliance (or not) of the legislation limits on pollutant concentration, is not considered to compute the impact on health. This guidance document recommends, as necessary step of a IAM study, the evaluation of the impact on health with one of the methodologies indicated above.

6.3 Is there a preferred indicator that should be used to quantify health impact?

In general there is no 'preferred' indicator. Different pollutants have different health impacts and one pollutant can show or contribute to more than one health outcome. The choice of health indicators to support decisions has to be made to show the potential impact of policy (in)action. Different indicators have different strengths in supporting policies. The decision depends mainly on the available data for the chosen methodology and on the policy question itself. Different types of policy questions may need different kinds of state-of-the-art indicators.

Epidemiological studies provide exposure-response functions (ERFs) necessary for HIA. HIA would then provide estimates of the number of health events attributable to air pollution in the target population assuming a causal relationship between AQ and the observed health effects.

The choice of health indicator(s) used in a given assessment should anticipate what the impact assessment will be used for (WHO working group, 2000). In other terms, the choice of an indicator should consider the final goals of the HIA, the policy objectives, the potential and proven health effect and then the data availability.

The following paragraphs illustrate the previous mentioned points:

1. The burden of disease related to air quality can be expressed as mortality and morbidity impact. Indicators often used are number of premature deaths, YOLLs, DALYs, number of hospital visits and life expectancy changes related to changes in exposure conditions when computed in scenarios assessment. For that purpose, the choice of the “pollutant-exposure-health effect” indicator for HIA (e.g. DALY, number of premature death, YOLL, etc.) is in the first place driven by the available data.
2. The selection of the indicator depends on the pollutant studied, availability of data (available dose-response curve), computer resources, skills, and also on the purpose of the study. For example, cost-benefit studies usually prefer to list all mortality and morbidity outcomes to compare all health benefits with all the costs of mitigation actions. In a scenario analysis the actions or scenarios with the lowest costs for one reduction in DALYs or number of deaths, etc. can be selected and studied more in detail. This is conform to the ICUR principle in health economics (Incremental Cost-Utility Ratio). Some other HIA studies involving multiple stressors might require indicators that take into account age, or mortality and certain morbidity effects. Therefore, the choice of indicators is driven by practicality and the needs of the assessment. Therefore all of the indicators are state-of-the-art in certain conditions.
3. Air pollution has been known as a risk factor for respiratory and cardiovascular diseases but remains an important issue in public health, given its impact on long-term mortality (HEI 2009). Moreover, recent studies suggest a large range of other outcomes than cardio-respiratory diseases to be related to exposure to air pollutants, among those effects on reproduction, birth weight and duration of gestation (WHO 2013). Also annoyance which is rarely taken into consideration could be mentioned (Jacquemin et al., 2007). The choice of the indicator is not restricted to the classical “mortality” respiratory or cardio-respiratory morbidity but should consider the large range of health outcomes available from the literature.

Most epidemiological studies find a wide range of health outcomes to be consistently related to AQ (PM for instance see: Pope et al, 2004 ; Brunekreef & Fosberg 2005). Those results are supported by toxicological evidences and do not only relate to short term exposures. Study design such as time-series supported the identification of the health impact (e.g. mortality) and the importance of the time of exposure. Cohort design analyses health effects related to long-term exposure. Nevertheless, susceptibility and vulnerability are considered more and more to influence the diversity and seriousness of the health outcomes related to air pollution exposures, leading to underestimations.

In general when indicators are used in HIA, it is recommended to include an uncertainty assessment (see also 6.7).

6.4 Are there any thresholds below which there are negligible adverse health effects?

The Clean Air for Europe (CAFE) program describes pollutant thresholds at the population level to be “the concentration of the pollutant such that, at concentrations below that threshold, there is no increase in risk of adverse health effects in any of the exposed population-at risk (Hurley et al. 2005)”. The CAFE project adopted a no-threshold policy in their assessment based on the lack of scientific evidence on the existing threshold for the pollutants considered (it is mainly for PM and ozone). The World Health Organization (WHO) Air Quality Guidelines Global Update 2005 also concluded that current scientific evidence does not indicate any threshold concentration for the particulate matter, ozone and sulphur dioxide (SO_2) air pollution (WHO 2006).

Recent studies suspect adverse effects of air pollution exposure even at very low concentrations (Janssen et al 2012, WHO 2013, Pedersen et al 2013). The PM aerosol classification as carcinogenic supports also the idea that there is no safe level (IARC 2013). The latest review of the scientific literature states that “The adverse effects on health of particulate matter (PM) are especially well documented. There is no evidence of a safe level of exposure or a threshold below which no adverse health effects occur.” (WHO 2013).

At this moment, in IAM studies, it should therefore be assumed that the adverse health effects caused by PM, ozone and SO_2 do not have any threshold concentrations at individual or population level.

The implications of this 'no threshold' on policy guidance is that this can move the focus away from reducing localised peaks in hot spot areas to small reductions everywhere as this would result in an overall bigger improvement in health impact for a given emission reduction. However, this is in general however not what the policy maker wants because of the higher cost of generalised EU emission reductions. Thresholds are therefore introduced to avoid the implementation of additional measures in already highly AQ compliant areas where additional measures probably would come at a higher cost.

6.5 What are the pollutants that should be considered? Which are the most toxic ones and have the biggest impact?

In reality people are exposed to a mixture of pollutants, Among those only a few are actually measured or modelled. The review done for the REVIHAAP project concludes that PM, NOx, SO_2 and O_3 are considered responsible for the health effects seen in epidemiological studies (WHO 2013). Most epidemiological studies find a wide range of health outcomes to be consistently related to PM (Pope et al, 2004 ; Brunekreef & Fosberg 2005). Those results are supported by toxicological evidences. In the case of PM the pollutant itself is a mixture and effects on health of all individual components have not yet been assessed conclusively. The integrated science assessment published by the EPA (EPA, 2009) used evidence from both epidemiological and experimental studies to conclude that “there are many components contributing to the health effects of $PM_{2.5}$, but not sufficient evidence to differentiate those constituents (or sources) that are more closely related to specific health outcomes”. Despite the increased number of studies after 2009, the general conclusion still remains the same. Moreover, toxicity differences among the chemical components of the PM aerosol have not been quantified yet. PM_{10}, $PM_{2.5}$ and ultrafine particles have specific toxicities that interfere within the aerosol. $PM_{2.5}$ and smaller particles penetrate deeper in the lungs and are involved in other physiological mechanisms than larger particles. In HIA in IAM even if one specific parameter is taken into consideration it reflects the health effect of the whole mixture of pollutants.

If we accept that we currently can't differentiate between the toxicity of individual constituents of PM then this implies that all PM reduction strategies will have the same health impact and that there is no need to aim specifically at reducing certain PM emissions such as those for black carbon but that we instead can focus on maybe more cost effective measures for reducing secondary PM. It however also means that there is currently no health based evidence to justify subtracting the contribution from natural sources such as sea salt to PM.

6.6 What are the effects of choice of resolution?

Resolution levels can affect the robustness of the HIA. The representativeness of the calculated health impacts depends on the level of detail of the population data. Lack of spatial resolution for the population location would lead to overestimation for some and underestimation for others. The temporal resolution is of importance, decisions on short term or long term exposure should be addressed separately using the related health data.

In general, epidemiological studies that used finer spatial resolution to relate people to air pollution levels tend to report higher mortality/morbidity impacts (Tainio, 2009). By Pérez et al (2013) it was shown that a fine resolution allowed to take into consideration variations of exposure related to distance to the main road. Therefore a recommendation is to use the most detailed exposure estimate that is available in epidemiological studies when assessing the health effects of air pollution. Ideally, the "complete individual exposure to air pollution" should be used. With "complete", indoor as well as outdoor air pollution and a period of 24h/24h is meant. With "individual", monitoring the air quality at the level of the person itself, using portable-and-easy to wear monitors is meant and also taking into account the susceptibility and the context of the person. For pollutants with high spatial variability this can be based on a personal activity-based survey or modelling, associated to air pollutant measurements, or personal dosimetry.

6.7 What uncertainties should we take into account when calculating the health impact?

Assessing and quantifying the burden of illness and mortality due to air pollution exposure is subject to inherent uncertainties of the underlying scientific assumptions and epidemiological evidence as well as of the statistical estimates and input data used. It is therefore important for stakeholders and policy makers to be aware of the source of these uncertainties in order to arrive at unbiased conclusions regarding the health risks of ambient pollution levels. The main sources of uncertainty in HIA studies can be summarised as follows:

1. **Uncertainties related to the results of the epidemiological studies or to their generalization.** Identifying and assessing the health impact of exposure to a specific pollutant based on epidemiological evidence may contain intrinsic uncertainties due to interactions between different risk factors, causing so-called effect modification. Scientific results on the cause and effect relationship between air pollution associated health impacts may also not take into account the temporal scale of effects, i.e. the latency times from exposure to adverse event.

Generally speaking, uncertainty is larger in the case of long-term exposure effects, as long-term cohort studies are more complex and could be restricted due to limitations of resources (cost, time, personnel etc.) and thus scarce. Epidemiological studies are also based on health outcomes combined with a given exposure to a given pollutant, notwithstanding changes in future conditions, due to changing air quality, climate and different characteristics of the population at the study period. As scientific knowledge is changing with emerging new epidemiological evidence and results of long term cohort observations, future AQ policies need to be regularly updated.

An important issue in epidemiological studies is the generalisation of results, i.e. to support that the assessed burden of air pollution to health is the same in all populations. Therefore, it is recommended that epidemiological studies and attempts for generalisation of findings take into account the different variability factors between populations (i.e. social and education status, diet, climate, lifestyle, overall health, age etc.). The choice of the epidemiological method brings in another aspect of uncertainty. Cross-over case studies, cohorts and retrospective models do not deliver the same information, their generalisation increases the uncertainty and the potential misuse of the results.

2. **Uncertainties in estimating the impact for each health outcome.** This uncertainty is related to the health indicator that will be used in the IAM study to describe the health impact. The most commonly used indicator is premature mortality, with different variations. The mortality indicator has been criticized because the measure does not provide any information on how premature is the actual death (e.g. Brunekreef and Hoek, 2000; Rabl, 2003). This might give misleading information in cases where the stressors impact different age groups.

3. **Uncertainties in exposure assessment:** Uncertainty in the exposure assessment is due to the inherent spatial and temporal variability in the AQ but also in the population distribution. Exposure uncertainty can be reduced by selecting the appropriate horizontal grid resolution for the AQ models, in order to ensure realistic spatial distribution of concentrations. This problem can be addressed either using a combination of model types or a nesting capabilities of models. Sensitivity analysis is the recommended method to estimate uncertainties resulting from grid resolution. Recent findings in the literature may be used for selecting an appropriate spatial resolution for the AQ. According to Tainio (2009) exposure should also be estimated for different source categories, such as traffic-related emissions.

Exposure assessment has to face the uncertainty of the difference between AQ concentration and “exposure” itself. In some circumstances a validation by individual dosimeters could be useful but not always feasible.

Exposure assessment has also to face the representativeness of the chosen studied pollutants to the real exposure and its health effects. How should we handle multiple exposure, cocktail effect or a multi-pollutant situation? What is health effect of AQ when considering several pollutants at the same time and not each pollutant individually?

4. **Uncertainties related to the concentration-response functions, estimated by epidemiological models.** Exposure-response functions are often derived based on epidemiological studies e.g. effect of air pollution on mortality rates, and are identified as the main source of uncertainty in IAM. For some health effects the dose-response relation may currently not be classified as “causal” but as e.g. “suggestive to be causal”. For these effects the health impact can be calculated and it is recommended to include these calculations in a sort of sensitivity analysis (WHO, 2013). The discussion on the existence or not of the threshold in the health effects could also

lead to uncertainties. Most of the scientific literature guides decision to choose a no-threshold function even though observations have not yet concluded to the causal link of very low exposures to a variety of AQ pollutants

A number of EU funded projects focusing on HIA provide on-line valuable guidance on the estimation of uncertainty related to the calculation of health risks and on the methodologies that can be used to address this uncertainty. Useful information sources include the "Integrated Environmental Health Impact Assessment System" (*this website*) and the Aphekom on-line tool (*http://si.easp.es/aphekom*).

7 RESPONSE: choice of abatement measures

The RESPONSE block (or decision framework) deals with the choice of the abatement measures to be implemented in order to improve air quality. It is one of the key aspects that drives the integration and application of all the other (DPSI) blocks.

7.1 What are the possible levels of complexity for the decision framework?

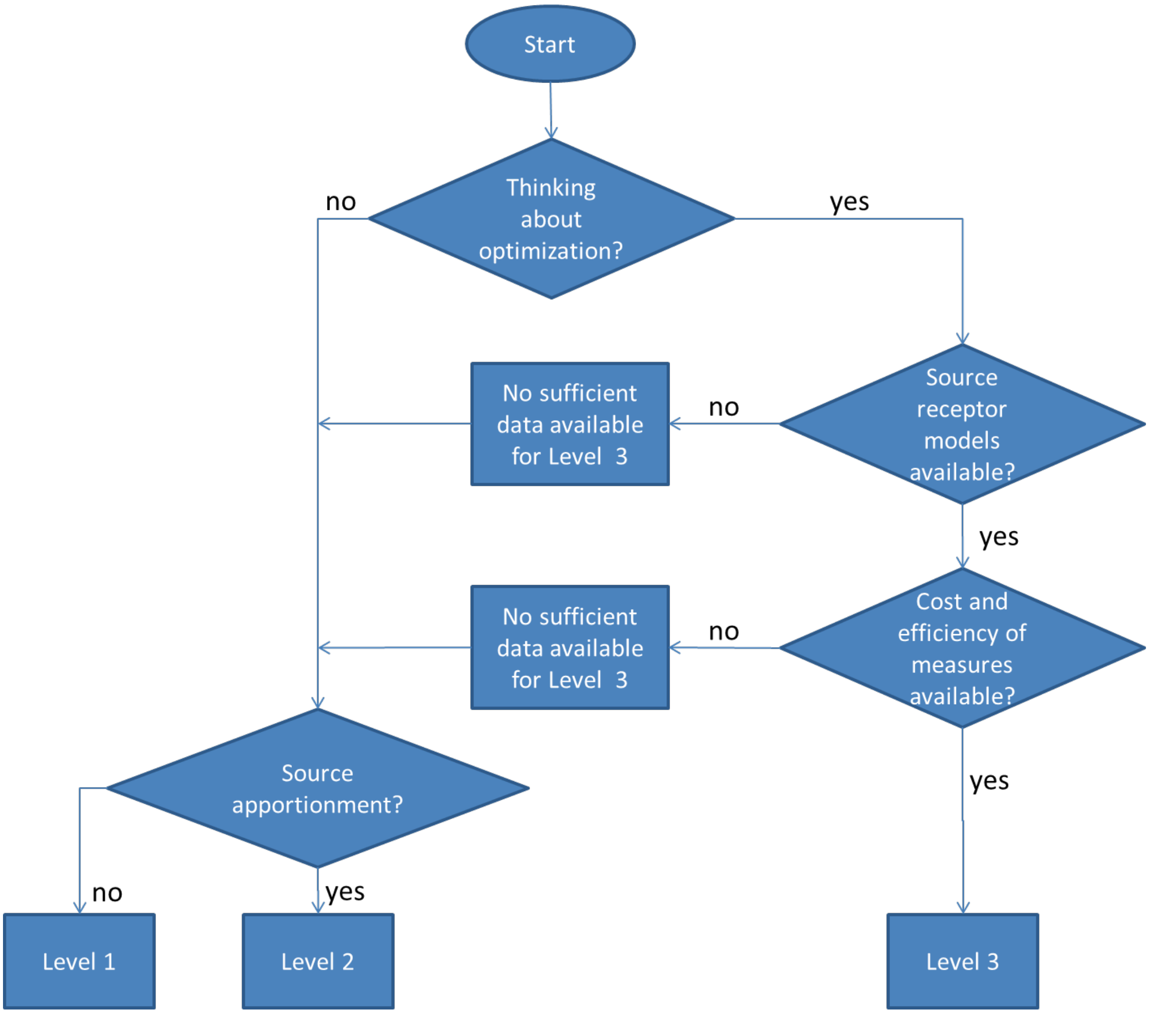
Three levels of complexity can be distinguished:

- **LEVEL 1**: Expert judgment and Scenario analysis. In this case the selection of emission abatement measures is based on expert opinion, with/without modelling support to test the consequences of a predefined emission reduction scenario on an AQI or better an health impact indicator (HII). In this context, the costs of the emission reduction actions can be evaluated as an output of the procedure if the data for such cost computation are available.
- **LEVEL 2**: Source Apportionment and Scenario analysis. In this case the main sources of emissions that are influencing an AQ index are derived through a formal approach; this then allows selection of the measures that should be applied to improve the HII(s). Again, emission reduction costs, if computed, are usually evaluated as a model output.
- **LEVEL 3**: Optimization. In this case the whole decision framework is described through a mathematical approach (Carlson et al., 2004, Amann et al., 2011), and costs are usually taken into account.

Looking at these levels of complexity from a different angle, two different pathways for the decision can be identified:

- **expert judgement/source apportionment and scenario analysis.** This is the approach that is mainly used nowadays to design “Plans and Programmes” at regional/local scale. Emission reduction measures (Policies) are selected on the basis of expert judgment or Source Apportionment and then they are tested (usually) through simulations of an air quality model. This approach does not guarantee that cost-effective measures are selected, and only allows for “ex-post evaluation” of impacts and costs.
- **optimization** This pathway suggests the most cost-effective measures for air quality improvement, by solving an optimization problem. In other words, the approach allows for the computation of the efficient set of technical (i.e. end-of-pipe) and non-technical (i.e. energy efficiency) measures/policies to be encouraged and/or introduced to reduce pollution, explicitly considering their impacts and costs.

To help in the selection of which level of complexity implement, depending on the data and knowledge available to a policy maker, the following decision tree can be used:



The graph clearly shows that the choice of the different implementation levels strongly depends on the data and knowledge available to the decision maker.

In particular, **LEVEL 1** implementation is considered in the case in which "no optimization" procedure is available, and no "source apportionment techniques" are applied. **LEVEL 2** is as **LEVEL 1** ("no optimization") but encompasses the use of source-apportionment for choosing the emission reductions scenarios to be tested. For the **LEVEL 3** implementation, more data/knowledge are required, as i.e. the possibility to perform "optimization", to develop/use "source-receptor models" and to have an "emission reduction measures database" with quantitative information (i.e., information on costs, removal efficiencies and implementation level of the different measures). As shown from the graph, if some of this information is missing, the **LEVEL 3** decision framework cannot be implemented. More details on how to implement these levels of complexity will follow in the next sections.

7.2 What do I need to implement a LEVEL1 approach? What can I derive using it?

In the **LEVEL 1** implementation, the “expert judgement” with “scenario analysis” approach is used. In this context, the decision maker is estimating the impacts of different emission reduction scenarios (usually, but not necessarily, with a CTM) applying some background/expert knowledge to choose the simulations to be performed.

For this approach the following knowledge/data are required:

- A methodology to relate emissions to concentrations (air quality model , source receptor model, …)
- The Emission reductions scenarios to be tested (optionally, this information could be integrated with a data base (DB) of measures, allowing for the computation of the effects and costs of single measure application)
- Information for impacts evaluation

The results that can be derived from this level 1 of complexity are:

- the emission reduction scenarios
- the air pollution concentrations and health impact resulting from the different scenarios.

In addition to these, and depending on the availability of data, it is also possible to derive costs associated to emission reduction scenarios, both in terms of internal (for policy implementation) and external (i.e. health related) costs. However, this additional analysis can be only performed as an ex-post evaluation.

7.3 What do I need to implement a LEVEL2 approach? What can I derive using it?

The second level of complexity is similar to the first one, with the exception that “source-apportionment techniques” are used instead of “expert judgment”.

Source-apportionment techniques can support the scenario analysis by providing the following information:

- The evaluation of transboundary pollution contribution to air quality
- The choice of the emission reductions that will have the biggest effect on local concentrations.

For this approach the following knowledge/data are required:

- An air quality model, with its required input (emissions, meteorology, boundary conditions)
- Source-apportionment techniques, to understand the transboundary contribution and design emission reduction scenarios to be analysed.

- The Emission reductions scenarios to be tested (optionally, this information could be integrated with a DB of measures, allowing for the computation of the effects and costs of single measure application)
- Information for impacts evaluation

The minimum results that can be derived from this **LEVEL 2** of complexity is the same already described in the **LEVEL 1** implementation, but in principle the LEVEL 2 results could be of a higher quality because in LEVEL 2 the choice of the RESPONSE is partially driven by the application of a formal/quantitative approach.

7.4 What do I need to implement a LEVEL3 approach? What can I derive using it?

The **LEVEL 3** of complexity deals with the selection of emission reduction measures, to improve air quality in an “optimal” way. Here the term “optimal” refers to the fact that usually the optimization takes into account

- the minimization of cost (i.e. of policy implementation)
- the use of multiple air quality indexes / impacts.

This approach is more demanding than the previous ones in terms of data/knowledge requirements. In fact it needs the following items:

- An air quality model, with its required input (emissions, meteorology, boundary conditions)
- Source-receptor models, linking emissions to concentrations in a simplified/fast way. An example on how to build simple SR model at local scale can be found in Carnevale et al., 2012b.
- a database of measures, allowing for the computation of the effects and costs of single measure application. Differently from the other level of complexity, here the use of a database of measures with quantitative information is compulsory, because otherwise the computation of costs (and of cost-effective policies) can not be implemented. Examples of such a database are the GAINS model database (Amann et al., 2011), or a regional/local database of non-technical measures.
- Information for impacts evaluation (compulsory for the evaluation of the cost of policies implementation)

The results that can be derived from this **LEVEL 3** of complexity are:

- The optimal emission reduction scenarios
- The optimal air quality fields
- The optimal costs associated to emission reduction scenarios, both in terms of internal (for policy implementation) and external (i.e. health related) costs

7.5 How can I construct measures and estimate their costs?

A first option for describing measures is through an aggregated approach (as through the CORINAIR macro sector). An example of a measure in this context is the reduction of emissions for a specific pollutant and activity sector. This is a very aggregated approach, but can provide policy makers with some insight on how to prioritize emission reduction interventions (Carnevale et al., 2012a). When working with these aggregated reductions, it is usually difficult to estimate costs; and if this is done, this is only implemented as an ex-post evaluation.

A second option is to use a database of real reduction measures, usually associated with quantitative information on costs, removal efficiency and application rates (see e.g. the GAINS database of measures, Amann et al., 2011). These measures can be classified as:

- "end-of-pipe technologies" (also called "Technical measures"), that act on emissions and are e.g. represented by large combustion plant emission filters, or diesel particle filters for cars. These measures neither modify the driving forces of emissions nor change the composition of energy systems or agricultural activities. These measures are applied to reduce emissions before being released in the atmosphere.
- "fuel change/fuel efficiency" measures. These measures are e.g. related to power plants change of fuels (e.g. from coal to gas) or process change (e.g. application of clean coal technologies). These measures do not change the driving forces (as energy demand remains unchanged) but change the structure of the primary energy supply.
- "Non-technical measures" (also called "Efficiency measures"). These are measures that reduce anthropogenic driving forces that generate pollution. Such measures can be related to behavioural changes of people (e.g. bicycle use instead of cars for personal mobility, temperature reduction in buildings) or to technologies that abate fuel consumption (e.g. high efficiency boilers or thermal insulation of buildings which reduce the overall energy demand).

To estimate costs of this second group of measures, it is necessary to:

- evaluate the quantity of "activity level that is controlled" by these measures
- multiply this "controlled activity level" with the unit cost associated to the measures themselves.

Quantitative data to implement this **second option** can e.g. be found on the GAINS website (*this website*). Additional information on non-technical measures can be also found on the OPERA project website (*this website*).

7.6 How should I account for synergies between different levels (national, regional, local) when considering abatement measures?

The effect of policies at the national or European level has to be taken into account in the assessment of the current air quality and of abatement measures, to quantify the contribution due to the different interacting scales.

Furthermore, when the effect of abatement measures has to be assessed, the decisions that regional authorities have to take are constrained by "higher level" decisions, coming from the national or EU level. In practice, this means that regional scale policies are constrained to consider the national/EU Current Legislation (CLE) as a starting point for their choices. In the effort to improve the regional air quality situation in comparison to the CLE, this "higher level" constraint cannot be disregarded or modified.

7.7 How can I take into account the uncertainty, or the lack of accuracy, in the response?

Instead of looking at an abstract uncertainty concept, for the RESPONSE block one should focus on “robust strategies”, that is strategies that do not significantly change due to the uncertainty in the system. In this context, it is important to analyse how the choice of the optimal abatement measures is influenced by the uncertainty on emissions, concentrations and impacts.

It is in fact worth underlining that, while for air quality models the accuracy can be measured by referring in one way or the other to field data (Thunis et al., 2012) for IAMs this is not possible, since an absolute “optimal” policy is not known and most often does not even exist. The traditional concept of model accuracy must thus be replaced by notions such as risk of a certain decision not resulting in the expected improvement to AQ or regret of choosing one policy instead over another.

8 Conclusions

In this guidance document we have presented how different elements of an integrated assessment modelling methodology or system (IAM) should be addressed. The guidance as presented in this document is not only based on the results of the preceding review and design work packages but also on the practical insight obtained through the evaluation of a number of existing Air Quality Plans (AQP) and two practical applications using a comprehensive IAM that were performed during the project.

As a general structure for an IAM the DPSIR (Driver/ Pressure/ State/ Impact/ Response) scheme was adopted in the design phase of the APPRAISAL project. In practice there is not a single, 'one size fits all' solution as an IAM has to take into account the available data, regional/local specificities, financial means and the actual purpose of the assessment. This is reflected in the guidance that is given for each of the DPSIR blocks by allowing for several levels of complexity to which the blocks can be elaborated. These levels can relate to the spatial and temporal resolution, to the extent to which uncertainty is accounted for and to whether different scales ranging from the European to the local scale are considered and integrated. For the choice of the abatement measures in the response block, levels were identified based on the procedure which is used to identify these measures. An overview of these different levels and their main characteristics for the different blocks is given in

Table 1.

The following aspects of the different DPSIR blocks should be carefully addressed when setting up a suitable IAM:

- The biggest task when implementing a comprehensive IAM is - as is also the case in regular air quality modelling applications – to obtain high quality input data *i.c.* information on local emissions and the cost and effectiveness of possible abatement measures. In both the AQPs and the practical examples considered much effort was put into quantifying the drivers and emissions. Regardless of this already high level of detail, the drivers and the emissions are still seen as the critical and most uncertain element in the IAM When data is lacking you can still rely on existing European inventories and databases with data on abatement measures such as EMEP and GAINS well keeping in mind the assumed validity of such data for the region of interest and the implications for the results obtained using the IAM. The availability of detailed data is also one of the major constraints for the selection of the level of complexity used for each block of the IAM.
- The most uncertain activity, and so also emission source, relevant to local air quality with respect to PM was found to be the residential combustion of wood or, in the case of Central/Eastern European countries, of coal/coal-substitutes.
- As meteorological conditions - which vary in time - significantly influence air quality, an important topic is the choice of a representative meteorology for the assessment of the efficiency of the abatement measures planned.
- In practice air quality model performance evaluation is still very limited in AQP. Adoption of the methodology developed within FAIRMODE would certainly be beneficial in this respect.
- Health Impact Assessment is clearly still a subject unfamiliar to most of those involved in developing air quality plans. An IAM that evaluates an abatement strategy only based on the compliance to legislated pollutant concentration limit values, is not considered to compute the impact on health. This guidance document recommends,

as necessary step of a IAM study, the evaluation of the impact on health with one of the methodologies outlined in chapter 6.

- For health impact assessment there is currently not enough scientific evidence to warrant the use of threshold values below which health effects are negligible.
- Many of the local abatement measures are non-technological or efficiency measures for which it is difficult to estimate the costs, particularly, because they impact many other sectors besides air quality. Further research to address this would be welcome.
- In practice, the list of options for abatement measures is restricted not only by what is technically and economically feasible but possibly even more by political and social acceptance. IAM tools should therefore be extended to allow their users to take into account the implications of political and social acceptance in an early stage of the decision process.
- If an IAM system uses source receptor relationships (artificial neural networks, linear regression, …) to relate emission changes to concentration changes, such relationships should be carefully tested to ensure that they not only correctly replicate the concentration values obtained through more complex modelling tools but also capture the concentration changes calculated by the model for which they are a surrogate.
- One should always keep in mind that more complexity is not always better. In this sense it is more advisable to adopt a simple but transparent IAM where limitations and assumptions are clear than to implement a comprehensive IAM using highly uncertain and maybe even flawed input data

Table 1 Different levels of detail for the different DPSIR blocks

<table>
<tr><th>DPSIR blocks</th><th colspan="3">Levels of complexity</th></tr>
<tr><td>Activities that produce emissions (Drivers)</td><td rowspan="2">top down information in a limited number of sectors and at a coarse resolution, detailed scenarios are not possible, uncertainty does not have to be considered</td><td rowspan="2">combines top-down with bottom-up, preferably with local activity and emission factor information, uncertainty does not have to be considered but a qualitative uncertainty estimate is recommended</td><td rowspan="2">bottom-up information at the highest possible resolution, the uncertainties for the emissions can be quantitatively calculated preferably using the Monte Carlo method</td></tr>
<tr><td>Emissions (Pressure)</td></tr>
<tr><td>Concentration (State)</td><td>based on measurements combined with a source apportionment technique to link emissions to concentrations. Validation of the source apportionment model through model intercomparison is recommended</td><td>a single air quality model adapted to the studied spatial scale. An operational model validation with observations is required</td><td>a chain of nested models adapted to the different scales ranging from the European to the local scale. An operational model validation with observations is required</td></tr>
<tr><td rowspan="2">Health impact</td><td>a simple description of “exposure” from measurements or AQ modelling and a simple description of the spatial distribution and composition of the population</td><td>a more detailed description of the concentration distribution is combined with a simple population description</td><td>detailed temporal and spatial resolution for the concentration distribution and population data with the distinction of subgroups with different vulnerability</td></tr>
<tr><td colspan="3">different sources of uncertainty should be mentioned together with results</td></tr>
<tr><td rowspan="2">Abatement measures (Responses)</td><td>a scenario based approach is used</td><td>a scenario based approach is used that is complemented with source apportionment</td><td>the selection of measures is based on an optimisation procedure</td></tr>
<tr><td colspan="3">uncertainty can be tackled by focusing on no-regret measures</td></tr>
</table>

9 References

Amann M. et al. (2011). Cost-effective control of air quality and greenhouse gases in Europe: Modelling and policy applications, Environmental Modelling & Software, 26, 1489-150.

Appel K.W, Gilliam R.C., Davis N., Zubrow A., Howard S.C. (2011) Overview of the atmospheric model evaluation tool (AMET) v1.1 for evaluating meteorological and air quality models, Environmental Modelling & Software, 26, 434-443.

Ashbaugh, L. L., Malm, W. C., Sadeh, W. Z., 1985.A Residence Time Probability Analysis of Sulfur Concentrations at Grand Canyon National Park. Atmospheric Environment - Part A General Topics 19 (8),1263-1270.

Baldasano J.M., E. López, M. Goncalves, M. T. Pay, L.P. Güereca, P. Jiménez-Guerrero and S. Gassó (2009) The HERMES high-resolution emission system: simulating anthropogenic and biogenic emissions for the Iberian Peninsula, Canary Islands and Europe. GEIA ACCENT 2009 Conference on surface emissions. University of Oslo, Oslo (Norway), October 26-28.

Baldasano J.M., M. Gonçalves, A. Soret and Pedro Jiménez-Guerrero (2010) Air Pollution Impacts of Speed Limitation Measures in Large Cities: The Need for Improving Traffic Data in a Metropolitan Area. Atmospheric Environment 44: 2997-3006 doi:10.1016/j.atmosenv.2010.05.013

Baldasano J.M., A. Soret, M. Guevara, F. Martínez, S. Gassó (2014) Integrated assessment of air pollution using observations and modelling in Santa Cruz de Tenerife (Canary Islands). Science of the Total Environment. 473-474: 576-588 dx.doi.org/10.1016/j.scitotenv.2013.12.062

Belis, C.A. , Karagulian F., Larsen B.R., Hopke P.K. (2013) Critical review and meta-analysis of ambient particulate matter source apportionment using receptor models in Europe. Atmospheric Environment 69, 94-108.

Belis C A., Larsen B. R., Amato F., El Haddad I, Favez O., Harrison R. M., Hopke P. K., Nava S., Paatero P., Prévôt A., Quass U., Vecchi R., Viana M., 2014. European Guide on with Receptor Models Air Pollution Source Apportionment. JRC Reference Reports. ISBN 978-92-79-32513-7. doi: 10.2788/9307.

Beuck, H., Quass, U., Klemm, O., Kuhlbusch, T.A.J., 2011. Assessment of sea salt and mineral dust contributions to PM10 in NW Germany using tracer models and positive matrix factorization. Atmospheric Environment 45, 5813-5821

Bieser J., Aulinger A., Matthias V., Quante M., Denier van der Gon H.A.C. 2011. Vertical emission profiles for Europe based on plume rise calculations. Environmental Pollution, 159, 2935-2946.

Borrego C, Monteiro A, Ferreira J, Miranda AI, Costa AM, Carvalho AC, Lopes M. (2008) Procedures for estimation of modelling uncertainty in air quality assessment. Environment international;34:613–20.

Borrego et al., (2012). Plans and Programmes to improve air quality over Portugal: a numerical modelling approach, Int. J. Environment and Pollution, 48, 60-68.

Briggs DJ (2008) A framework for integrated environmental health impact assessment of systemic risks, Environ Health. 2008 Nov 27;7:61.

Brunekreef B. and Hoek G. (2000). Invited commentary - Beyond the body count: Air pollution and death. American Journal of Epidemiology 151 449-51.

Brunekreef B, Forsberg B. (2005) Epidemiological evidence of effects of coarse airborne particles on health. Eur Respir J. 26: 309–318.

Builtjes PJH, van Loon M, Schaap M, Teeuwisse S and others (2003). Project on the modelling and verification of ozone reduction strategies: contribution of TNO-MEP. TNO-report MEP-R2003/166, TNO, Apeldoorn.

Carnevale C., Finzi G. , Pisoni E., Volta M. (2009) Neuro-fuzzy and neural network systems for air quality control, Atmospheric Environment. 43, 4811-482.

Carnevale C., Finzi G. , Pisoni,E , Volta M., Wagner F (2012a). Defining a nonlinear control problem to reduce particulate matter population exposure, Atmospheric Environment, 55, 410-416.

Carnevale C., Finzi G., Guariso G., Pisoni E., Volta M. (2012b). Surrogate models to compute optimal air quality planning policies at a regional scale, Environmental Modelling and Software, 34, 44-50.

Carnevale, C., Finzi, G., Pisoni, E., Volta, M., Guariso, G., Gianfreda, R., Maffeis, G., Thunis, P., White, L. Triacchini, G.(2012c). An integrated assessment tool to define effective air quality policies at regional scale, Environmental Modelling and Software, 38, 306-315.

Carlson D. et al. (2004) Large-scale convex optimization methods for air quality policy assessment. Automatica, 40, 385–395.

Chang J.C., Hanna S.R. (2004) Air quality model performance evaluation, Meteorology and Atmospheric Physics, 87, 167–196.

Cuvelier C., Thunis P., Vautard R., Amann M., Bessagnet B., Bedogni M., Berkowicz R., Brandt J., Brocheton F., Builtjes P., Carnavale C., Coppalle A., Denby B., Douros J., Graf A., Hellmuth O., Hodzic A., Honoré C., Jonson J., Kerschbaumer A., et al. (2007) CityDelta: A model intercomparison study to explore the impact of emission reductions in European cities in 2010. Atmospheric Environment 41:189-207.

Denby, B. and M. Pochmann,(2007), Basic data assimilation methods for use in urban air quality assessment. In: Proceedings of the 6th International Conference on Urban Air Quality. Limassol, Cyprus, 27-29 March 2007. Ed. by R.S. Sokhi and M. Neophytou. Hatfield, University of Hertfordshire(CD-ROM).

Denby et al. (2011) The application of models under the European Union's Air Quality Directive:A technical reference guide EEA Technical report No 10/201. ISSN 1725-2237

Denier van der Gon, H., C. Hendriks, J. Kuenen, A. Segers, A. Visschedijk (2011). Description of current temporal emission patterns and sensitivity of predicted AQ for temporal emission patterns. EU FP7 MACC deliverable report D_D-EMIS_1.3. http://www.gmes-atmosphere.eu/documents/deliverables/d-emis/MACC_TNO_del_1_3_v2.pdf

Dennis, R., et al. , (2010) A framework for evaluating of regional-scale numerical photochemical modelling systems. Environ. Fluid Mech., in press, doi:10.1007/s10652009-9163-2.

EEA (2011). Europe's environment: An Assessment of Assessments. ISBN 978-92-9213-217-0. doi:10.2800/78360.

EEA, 2013. Air quality in Europe — 2013 report. European Environment Agency. EEA Report No 9/2013. ISSN 1725-9177.

EMEP/EEA (2013). Air pollutant emission inventory guidebook. Technical guidance to prepare national emission inventories. Technical report No 12/2013. ISSN 1725-2237

EPA (1996). Evaluating the uncertainty of emission estimates, EIIP emission inventory improvement program, http://www.epa.gov/ttn/chief/eiip/techreport/volume06/vi04.pdf.

EPA (2009) Particulate Matter National Ambient Air Quality Standards : Scope and methods plan for Health risk and exposure Assessment, 2009,available on http://www.epa.gov/ttn/naaqs/standards/o3/data/2011_04_HealthREA.pdf

Favez O., El Haddad I., Plot C., Boréave A., Abidi E., Marchand N., Jaffrezo J.-L.,Besombes J.-L., Personnaz M.-B., Sciare J., Wortham H., George C. and D'Anna B. (2010) Inter-comparison of source apportionment models for the estimation of wood burning aerosols during wintertime in an Alpine city (Grenoble, France), Atmospheric Chemistry and Physics Discussions, 10, 559-613.

Ferreira J., M. Guevara, J.M. Baldasano, O. Tchepel, M. Schaap, A.I. Miranda, C. Borrego (2013) A comparative analysis of atmospheric emission inventories available for Europe. Atmospheric Environment 75: 43 – 57 dx.doi.org/10.1016/j.atmosenv.2013.03.052

Fragkou E., Douros I., Moussiopoulos N. and Belis C. A. (2012). Current Trends in the use of Models for Source Apportionment of Air Pollutants in Europe, International Journal of Environment and Pollution 50 (1-4): 363-375.

Guevara M., F. Martínez, G. Arévalo, S. Gassó, J.M. Baldasano (2013) Improved system for modelling Spanish emissions: HERMESv2.0. Atmospheric Environment 81: 209-221 doi: 10.1016/j.atmosenv.2013.08.053

Fontaras G, Franco V, Dilara P, Martini G, Manfredi U, (2014) Development and review of Euro 5 passenger car emission factors based on experimental results over various driving cycles, Science of The Total Environment 468–469, p1034–1042

François S., Grondin E., Fayet S., Ponche J.-L. (2005) The establishment of the atmospheric emission inventories of the ESCOMPTE program, Atmospheric Research 74 (2005) 5– 35

Guariso G., Pirovano G, Volta M. (2004). Multi-objective analysis of ground-level ozone concentration control, Journal of Environmental Management, 71, 25–33.

Guenther, A. B., Jiang, X., Heald, C. L., Sakulyanontvittaya, T., Duhl, T., Emmons, L. Wang, X.: The Model of Emissions of Gases and Aerosols from Nature version 2.1 (MEGAN2.1): an extended and updated framework for modeling biogenic emissions. Geoscientific Model Development 5, 1471-1492, 2012.

HEI. Health effect institute (hei) panel on the health effects of traffic-related air pollution. (2009) Traffic-related air pollution: A critical review of the literature on emissions, exposure, and health effects. Hei special report 17. Health effects institute. Boston, Mass.

Hogrefe C., Rao S.T. (2001) Demonstrating attainment of the air quality standards: integration of observations and model predictions into the probabilistic framework, J Air Waste Manag Assoc., 51(7).

Horabik J, & Nahorski Z., (2014) Improving resolution of a spatial air pollution inventory with a statistical inference approach, Climatic Change 124, p575–589

Hurley F., Hunt A., Cowie H., Holland M., Miller B., Pye A., Watkiss P. (2005) Methodology Paper (Volume 2) for Service Contract for carrying out cost-benefit analysis of air quality related issues, in particular in the clean air for Europe (CAFE) programme (http://ec.europa.eu/environment/archives/cafe/pdf/cba_methodology_vol2.pdf)

IARC (2013) Scientific Publication No. 161 Air Pollution and Cancer Editors: Kurt Straif, Aaron Cohen, and Jonathan Samet.

IPCC Task Force on National Greenhouse Gas Inventories (2000) Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories,

Jhih-Shyang Shih, a, , Armistead G. Russell, Gregory J. McRaec (1998). An optimization model for photochemical air pollution control, European Journal of Operational Research, 106, 1-14.

Jorba O., C. Pérez, F. Rocadenbosch, and J. M. Baldasano (2004) Cluster Analysis of 4-Day Back Trajectories Arriving in the Barcelona Area (Spain) from 1997 to 2002. Journal of Applied Meteorology, 43, 6: 887-901

Juda-Rezler K, Reizer M, Huszar P, Krüger BC, Zanis P, Syrakov D, Katragkou E, Trapp W, Melas D, Chervenkov H, Tegoulias I, Halenka T (2012). Modelling the effects of climate change on air quality over Central and Eastern Europe: concept, evaluation and projections. Clim Res 53(3): 179–203.

Juda-Rezler K (2010). New challenges in air quality and climate modelling. Arch Environ Prot 36(1): 3–28.

Karvosenoja N. (2008) Emission scenario model for regional air pollution. Monographs Boreal Environ. Res. 32.

Keuken et al. (2013). Modelling elemental carbon at regional, urban and traffic locations in The Netherlands. Atmospheric Environment 73:73-80.

Kim, E., Hopke, P.K., Paatero, P., Edgeton, E.S., 2003. Incorporation of Parametric Factors into Multilinear Receptor Models Studies of Atlanta Aerosol. Atmospheric Environment 37, 5009-5021

King K., Sturman J. and Passant M. (2006) NAEI UK emissions mapping methodology 2003. A report of the National Atmospheric Emission Inventory. AEAT/ENV/R/2259, March 2006.

Knol A. (2010) Health and environment: assessing the impacts, adressing the uncertainties. Utrecht University, the Netherlands.

Kouridis, Ch., Gkatzoflias, D., Kioutsioukis, I., Ntziachristos, L., Pastorello, C., Dilara, P. ,2010. Uncertainty estimates and guidance for road transport emission calculations. European Communities, DOI 10.2788/78236.

Kuenen, J., Denier van der Gon, H., Visschedijk, A., van der Brugh, H., 2011. High resolution European emission inventory for the years 2003–2007. TNO report TNO-060-UT-2011-00588.

Kuenen, J. J. P. , A. J. H. Visschedijk, M. Jozwicka, and H. A. C. Denier van der Gon, 2014, TNO-MACC_II emission inventory: a multi-year (2003–2009) consistent high-resolution European emission inventory for air quality modelling, Atmos. Chem. Phys. Discuss., 14, 5837–5869.

Lenschow, P., H.-J. Abraham, K. Kutzner, M. Lutz, J.-D. PreuB, W. Reichenbficher, Some ideas about the sources of PM10, Atmospheric Environment 35 Supplement No. 1, 23-33, 2001.

Maes J., Vliegen J., Van de Vel K., Janssen S., Deutsch F., De Ridder K., Mensink C., 2009, Spatial surrogates for the disaggregation of CORINAIR emission inventories, Atmospheric Environment 43 1246–1254

Nagl C. et al. (2005) ASSESSMENT OF PLANS AND PROGRAMMES REPORTED UNDER 1996/62/EC – FINAL REPORT, Service contract to the European Commission - DG Environment, Contract No. 070402/2005/421167/MAR/C1"

OPERA 2013, User guide of software RIAT+ http://www.operatool.eu/html/pdf/RIATplusUserGuide.pdf

Ozgen, S., S. Caserini, S. Galante, M. Giugliano, E. Angelino, A. Marongiu, F. Hugony, G. Migliavacca, C. Morreale (2014) Emission factors from small scale appliances burning wood and pellets Atmospheric Environment,94, 144-153.

Paunu, N. Karvosenoja, M. Savolahti, K. Kupiainen (2013) High quality spatial model for residential wood combustion emissions. 16th IUAPPA World Clean Air Congress, Cape Town, South Africa, 29 September - 4 October 2013. 4 pp.

Perez, L., Declercq, C., Iñiguez, C., Aguilera, I., Badaloni, C., Ballester, F., Bouland, C., Chanel, O., Cirarda, FB, Forastiere, F., Forsberg, B., Haluza, D., Hedlund, B., Cambra, K., Lacasaña, M., Moshammer, H., Otorepec, P., Rodríguez-Barranco, M., Medina, S., Künzli, N. Chronic burden of near-roadway traffic pollution in 10 European cities (APHEKOM network), Eur Respir J 2013; 42: 594–605 | DOI: 10.1183/09031936.00031112

Pisoni E., Carnevale C., Volta M. (2009). Multi-criteria analysis for PM10 planning, Atmospheric Environment, 43, 4833-4842.

Ponche J.-L. (2002). The spatialized emission inventories : a tool for the Air Quality Management French IGPB-WCRP News Letters (in french).

Pope CAI, Burnett RT, Thurston GD, et al.(2004) Cardio- vascular mortality and long-term exposure to particulate air pollution: epidemiological evidence of general pathophysiological pathways of disease. Circulation; 109: 71–77.

Rabl A. (2003). Interpretation of air pollution mortality: Number of deaths or years of life lost? Journal of the Air & Waste Management Association 53 41-50

Savolahti M., N. Karvosenoja, K. Kupiainen (2014) Ecodesign directive for residential wood combustion appliances: Impacts and emission reduction potential in Finland. The Sustainable City IX Vol. 2, WIT Transactions on Ecology and The Environment, Vol 191, pp. 1493-1504. doi:10.2495/SC141272.

Schopp W, Amann M., Cofala J., Heyes C., Klimont Z.(1999). Integrated assessment of European air pollution emission control strategies, Environmental Modelling and Software 14, 1-9.

Smit,Ntziachristos, L., Boulter, P., 2010. Validation of road vehicle and traffic emission models – A review and meta-analysis. Atmospheric Environment 44, 2943-2953.

Tarrason,L., Benedictow,A., Fagerli, H., Jonson,J.E., Klein,H., van Loon, M.,Simpson,D., Tsyro,S., Vestreng, V., Wind, P., Forster, C., Stohl, A., Amann, M., Cofala, J.,Langner,J., Andersson, C., Bergstrom, R. (2005),Transboundary Acidification, Eutrophication and Ground Level Ozone in Europe in 2003, EMEP Status Report 2005 ISSN 0806-4520.

Tainio, M. (2009). Methods and uncertainties in the assessment of the health effects of fine particulate matter (PM2.5) air pollution. National Institute for Health and Welfare (THL) and Faculty of Natural and Environmental Science at Kuopio University.

Timmermans R.M.A., H.A.C. Denier van der Gon, J.J.P. Kuenen, A.J. Segers, C. Honoré, O. Perrussel, P.J.H. Builtjes, M. Schaap, (2013) Quantification of the urban air pollution increment and its dependency on the use of down-scaled and bottom-up city emission inventories, Urban Climate, 6, p44-62

Thunis,P., E. Georgieva, S. Galmarini, (2011) A procedure for air quality models benchmarking Joint Research Centre, Ispra, Version 2, 16 February 2011: http://fairmode.ew.eea.europa.eu/models-benchmarking-sg4/wg2_sg4_benchmarking_v2.pdf.

Thunis,P. A. Pederzoli, D. Pernigotti. (2012) Performance criteria to evaluate air quality modeling applications. Atmospheric Environment, 59, 476-482.

Thunis P. E. Geogieva, A. Pederzoli (2012). A tool to evaluate air quality model performances in regulatory applications, Environmental Modeling & Software, 38, 220–230.

UNECE (2002) Progress Report Prepared by the Chairman of the Task Force on Integrated Assessment Modeling, United Nations Economic Commission for Europe, Geneva, Switzerland.

Valverde V., M.T. Pay, J.M. Baldasano (2014) Circulation Type Classification Derived on a Climatic Basis to Study Air Quality Dynamics Over The Iberian Peninsula. International Journal of Climatology doi: 10.1002/joc.4179.

Verstraete J, (2014) Solving the map overlay problem with a fuzzy approach, Climatic Change 124, p591–604.

Vestreng V, Myhre G, Fagerli H, Reis S, Tarrasón L (2007). Twenty-five years of continuous sulphur dioxide emission reduction in Europe. Atmos Chem Phys 7: 3663−3681.

Vlachokostas C. et al. (2009). Decision support system for the evaluation of urban air pollution control options: Application for particulate pollution in Thessaloniki, Greece, Science of the Total Environment, 407, 5937–5948.

Wang Y.Q., Zhang X. Y., Draxler R.R., 2009b. TrajStat: GIS-based software that uses various trajectory statistical analysis methods to identify potential sources from long-term air pollution measurement data. Environmental Modelling & Software 24, 938.

Wang, Z. S., C. -J Chien, and G. S. Tonnesen. 2009a. "Development of a Tagged Species Source Apportionment Algorithm to Characterize Three-Dimensional Transport and Transformation of Precursors and Secondary Pollutants." Journal of Geophysical Research D: Atmospheres 114 (21).

Werner S. PhD thesis (2009). Optimization of the spatialised emission inventories : estimation of uncertainties, determination of the emission factors of black carbon from the road traffic. Application to the ESCOMPTE and Nord Pas-de-Calais Region emission inventories) University of Strasbourg, Department of Chemistry (in french).

WHO (2006) Air Quality Guidelines - Global Update 2005. http://www.euro.who.int/__data/assets/pdf_file/0005/78638/E90038.pdf

WHO (2009) The choice of indicator(s) used in a given assessment should anticipate the use to which the impact assessment will be put, WHO working group.

WHO (2013) World Health Organisation. Review of evidence on health aspects of air pollution – REVIHAAP Project. Technical report. WHO Regional Office for Europe, Denmark.

ANNEX I: List of tools

		Models/methodologies
Drivers & Pressures	emission models	CAPRI, COPERT, EMEP, TREMOVE, INEMAR
State	Air quality models at regional scale	AURORA, BEL-EUROS, CAMx, CHIMERE, CMAQ, COSMO-MUSCAT, EURAD, FARM, LOTOS-EUROS, MEMO/MARS-aero, RCG, TAPM, TCAM, UAM-V WRF-Chem, WRF-MM5
	Air quality models at urban scale	ADMS, AUSTAL2000, CALINE4, CALPUF, CAR, CITYCAT, IFDM, IMMIS, LASAT, MISKAM, OFIS, OPS, PAL, PROKAS, SIRANE, SPRAY, URBAIR, VADIS
	Source-receptor models	ANNs, SR matrix
Impacts	Health assessment	ARPS, RIAT+,
Responses	Source-apportionment	CAMX, CMAQ
	Source sensitivity	CMAQ
	Scenario analysis	AURORA, BEL-EUROS, CAMx, CHIMERE, CMAQ, COSMO-MUSCAT, EURAD, FARM, LOTOS-EUROS, MEMO/MARS-aero, RCG, TAPM, TCAM, UAM-V WRF-Chem, WRF-MM5, ADMS, AUSTAL2000, CALINE4, CALPUF, CAR, CITYCAT, IFDM, IMMIS, LASAT, MISKAM, OFIS, OPS, PAL, PROKAS, SIRANE, SPRAY, URBAIR, VADIS
	Cost-effective analysis	RIAT+, GAINS
	Multi-objective analysis	RIAT+

Annex II: Useful IAM output for stakeholders

Air quality assessment:

- Maps of exceedances of limit values or information/alert thresholds. In particular:
    - number of days exceeding the daily limit value on PM_{10};
    - annual concentration of PM_{10};
    - annual concentration of NO_2;
    - number of exceedances of the hourly limit value on NO_2;
    - number of days exceeding the limit value on the 8-hour moving average on ozone concentrations;
    - number of days exceeding the information and/or alert thresholds on O_3;
    - annual concentration of $PM_{2.5}$;
    - average exposure index (AEI) on $PM_{2.5}$.
- Area and population concerned by exceedances of limits values.
- Temporal evolution of pollutant concentrations, from now till the year for which the scenario is computed, and considering different types of environment in the city (e.g. urban background, urban, traffic, industrial …).
- An Air Quality Index.

Emission fields (only used/consulted in case of exceedance of the limit value):

- For each pollutant the different relevant SNAP sectors should be taken into account.

Health impact

- Impact on mortality: loss on life expectancy (YOLL, DALY) and premature death.
- Impact on morbidity: respiratory/cardiovascular diseases in function of the age

Social impact

- Acceptability of measures

Economic impact

- Cost of implementation of measures: internal costs
- Benefits resulting from the measures: external costs.