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Summary

An Integrated Assessment Modelling system (IAM) is a methodology for determining how emissions should be reduced to 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 taking into account

that each of these blocks can be elaborated to a different level of complexity according to the available data or the purpose of the IAM. 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 a single block.

This is a first version of the document that is mostly based on the review and design work reports. In the second half of the project this guidance will be evaluated by applying it to several test cases and this practical experience will then result in an updated version of this document.

Version History

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0.2	Second draft	17/10/2013	Peter Viaene adding feedback from Niko Karvesenoja (SYKE), John Douros and Lia Fragkou (AUTH), Elsa Real (Ineris), Arno Graff (UBA), Katarzyna Juda-Rezler and Marko Tainio(SRI), Nadège Blond (CNRS), Jean-Luc Ponche (UNISTRA), Alain Clappier (UNISTRA), Alberto Martilli (CIEMAT)
0.3	Third draft	8/11/2013	Peter Viaene (VITO), Niko Karvesenoja (SYKE), John Douros and Lia Fragkou (AUTH), Claudio Carnevale (UNIBS), Enrico Pisoni (JRC), Arno Graff (UBA), Marko Tainio and Zbigniew.Nahorski (SRI), Nadège Blond (CNRS), Elsa Real (Ineris), Alberto Martilli (CIEMAT), Pieter Valkering and Jurgen Buekers (VITO)
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Summary of Changes

Version	Section(s)	Synopsis of Change
0.1	All	First version based on D1.2, D2.7 and D3.1 with structure from D3.1
0.2	All	Added feedback from partners + assigned the different topics
0.3	All	Elaborated topics for which input was given by the partners
0.4	Mainly impact on health (7) and state (6)	Elaborated impacts, restructured 6.2, shortened the topics on uncertainty
1.0	All	Final review

Abbreviations

AQ	air quality
AQI	Air quality index
CMB	chemical mass balance
CTM	chemical transport model
DALY	Disability adjusted life years
DPSIR	Driver/Pressure/State/Impact/Response
e.g.	for example (exempli gratia)
EM	emission measures
HIA	Health impact assessment
IAM	integrated assessment modelling system
i.e.	that is (idem est)
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
TM	technical measures
YOLL	years of life lost

1 Introduction

This first version of the recommendation document, focuses on the different topics that need to be addressed when setting up an Integrated Assessment Modelling (IAM) system or methodology. The topics are presented as a number of questions that relate to the building blocks of the DPSIR (Drivers, Pressure, State, Impact, Responses) scheme that was identified in WP3 as a general design framework for an IAM. In line with WP3 the 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 the purpose of the IAM. Some of these questions are specific to the building block while others relate to topics that reappear in all blocks such as how uncertainty or the interaction between scales should be tackled.

This report, which is a first version of the official Guidance Document of the APPRAISAL project, aims by no means to provide detailed technical instructions on how to set up an IAM but rather wants to present a comprehensive set of topics that should be addressed together with the main lines along which these can be approached and references that can further guide the reader. The emphasize is therefore rather on keeping the overview and less on providing all the details as these can often already be found in other reports and peer-reviewed literature sources.

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

2 A general overview of the Integrated Assessment Modelling system framework

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

IAM can be considered as a methodology for determining how emissions should be reduced to improve air quality, protect human health, reduce ecosystem exposure. Other aspects that could be included in an IAM but which are not specifically addressed in this document are noise and the abatement of greenhouse gas. 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 level in the EU, besides the obvious specific local conditions that lead to air pollution problems, there is also a large difference in the detail of the available data and even a 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 as a consequence health impacts in many different ways in order to suit the capability and need of a specific regional/local situation.

2.2 What are the elements of an IAM?

In line with what was presented in the APPRAISAL deliverable D3.1 on the design of a IAM, we apply the DPSIR scheme to identify the different elements of an IAM. The DPSIR scheme is the causal framework adopted by the European Environment Agency for describing the interactions between society and environment. 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);
- pressures : the pollutant emissions;
- states : the air quality i.e. the concentrations and deposition of pollutants;
- impacts : the implications of the air quality for health, the environment;
- responses : the measures that are available to reduce the impacts.

Notice that not in every implementation of an IAM all the building blocks are used and the level of detail of the blocks that are considered can vary. For each of the blocks guidance is given in the next chapters.

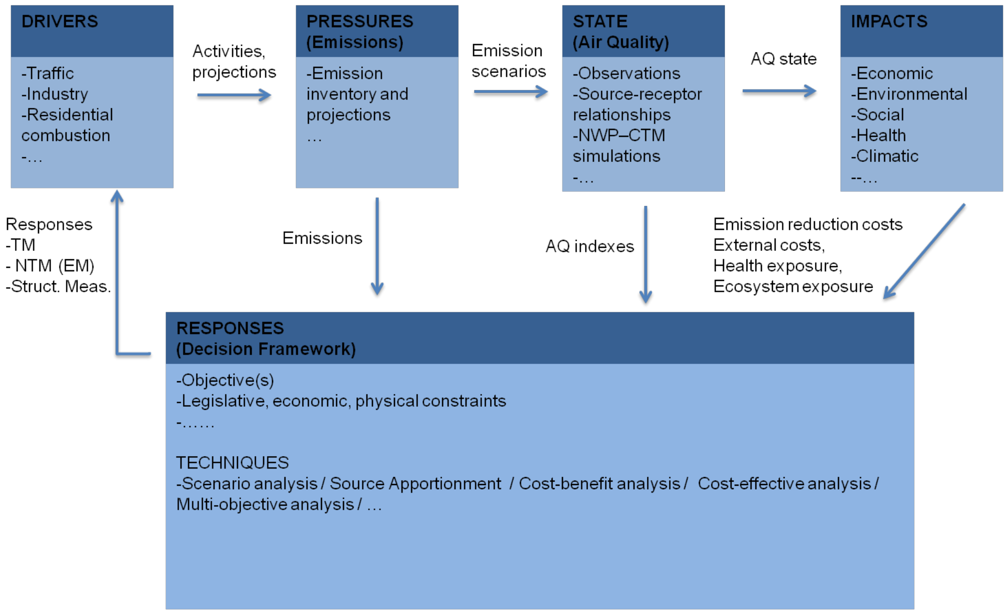

Figure 1 The DPSIR scheme applied to integrated air quality assessment modelling.

2.3 What is the base or reference case and what are projections or scenarios?

To assess the effect of responses or abatement measures an IAM will assess the change in impact due to these measures. 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 measures and the projections will correspond to the values of drivers, pressures, state and impact when measures are in place.

3 DRIVER: the activities that affect the emissions

3.1 How important is the activities model in an IAM?

The DRIVER block translates changes in activities into changes in quantities that can be related to emissions. Activity changes 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 such a way that it is classified in an emission-wise relevant way into sectors, sources and technologies. For the base-case, usually emission inventories are directly used, in some cases by-passing the DRIVERS block or activity levels can be used for base-case emissions' estimation, using associated emission factors for abatement measures. For future projections however it is particularly important that the changes in time of the input of the DRIVERS block are realistically translated into outputs (i.e. emission activities and technologies). Therefore the assessment of future developments of the DRIVERS block typically requires a more sophisticated framework than what would be needed for the base year inventory.

3.2 Which activities and emission relevant technologies should I consider as driving forces for a regional or local scale IAM?

The activity sectors which are most relevant for local scale IAM are road traffic, off-road traffic and machinery for example due to construction, residential heating and to a lesser extent energy production and industry. Agriculture is typically of minor concern in a city level assessment, but may be more important at the regional and national level, where agriculture is a major source of ammonia emissions and can be relatively important as a contributor to secondary PM formation. In addition to the activity rates in different sectors, technology information is important for emissions' assessment. Emission relevant technology parameters include traffic engine technologies and fleet age, residential combustion technologies and emission removal technologies used in energy and industrial plants. For future projections, reliable estimates on how the technology stocks evolve are essential.

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

The inventory of activities and emission relevant technologies at the local level 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 surrogates (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 approach can be considered to be more favourable as it uses, by definition, information from the respective city or region directly. However, in many cases it might be difficult to obtain reliable, representative data for certain areas. Furthermore, 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 surrogates used. In case representative weighting surrogates are available for each sector, a top-down approach can produce a reliable activity estimate.

3.4 What level of detail is required in modelling the driver activities?

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 traffic 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 using 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 of 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 (i.e. the DRIVERS block) 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. Keuken et al. 2013, Karvosenoja 2008, King et al. 2006, Cuvelier et al. 2007). 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 a finer spatial resolution down to tens of meters is desirable. Keuken et al. (2013) demonstrated a factor of 2 - 4 higher elemental carbon concentrations near traffic locations compared to background locations when modelled at 20 m and 1 km resolutions, respectively.

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 the case of absence of such activity models, spatial gridding relies on the availability of GIS data that can be used as a proxy for different sectors. E.g. 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. The temporal variation for different sectors can be based on internationally, nationally or locally defined default variations or local data (e.g. questionnaires or observed data).

3.5 Which driver activities are most uncertain?

The most uncertain activity was found to be the residential combustion of wood and, in the case of Central/Eastern European countries, combustion of coal/coal-substitutes. This sector merits further research as it has recently been found to be one of the major contributors to fine particle and other organic air pollution in many urban areas in Europe. Future research needs for the assessment of residential wood/coal/coal-substitutes combustion concern mainly:

- Activity amount assessment
- Combustion appliance and user's practice information
- Spatial assessment (i.e. gridding)

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, house-hold 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 used 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.

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 house-hold 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, apartment/detached/semi-detached), and an estimation about urban/rural differences in wood use, a relatively good approximation for a large area average can be achieved. 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.

3.6 How should I account for the different scales in the 'driver activities' model?

The different activity changes relevant for scenarios to be used in a local 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) scale. 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 use directly 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 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 at both local and larger scale should be addressed.

Technological changes, that are mainly of interest 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) legislations that influence technological developments are defined at EU level. National level decisions may have a great 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 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, prohibitions of residential wood/coal/coal-substitutes combustion) and set more general goals (city climate strategies) that influence technological developments. Naturally, the city level influenced technological changes must be assessed at local level.

Attention should also be devoted to the integration of bottom-up and top-down inventories. At the moment there are often inconsistencies between bottom-up (local/regional) and top-down (EU level) approaches and tools which prevent the implementation of a fully integrated approach connecting various governance scales. Also, while activity levels (DRIVERS) are usually available at international/national level, this is not the case at regional/local scales, where only emission inventories (PRESSURES) are compiled. This aspect can also cause inconsistencies among data provided at different levels of governance.

4 PRESSURES: emissions

4.1 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 (usually yearly-based). The emissions themselves are calculated multiplying the activity with an emission factor, in particular considering:

- the activity of the emission sources. For instance: the volume and the type of fuel burned, the number of kilometres travelled by the vehicles, etc.;
- the amount of pollutant emitted by these sources per unit of activity, i.e. the emission factors.

The emission inventory may have different levels of detail depending on the availability of the 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 and data availability:
LEVEL 1: emissions are estimated for high level sectors on a coarse grid, using per default the top-down methodology. Uncertainties are not necessarily estimated. The level 1 does not allow for detailed emission projections.
LEVEL 2: A combination of bottom-up and top-down methodology is used to calculate the emissions with the SNAP – NAPFUE classifications at level 2 or 3. Emission factors and activity data representative of the area of study are used when available. Uncertainties are not necessarily estimated.
LEVEL 3: emissions are calculated with the finest space and time resolution available, with a bottom-up approach and the SNAP-NAPFUE 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 of the area under study. The processes have to be detailed as well as possible, to attribute the most representative emissions. In case of lack of data, the top-down approach can be used but with the help of complementary data to take into account the regional specificities. The uncertainties may be quantitatively calculated preferably using the Monte Carlo method. The level 3 is the best one to allow the generation of all kinds of scenarios assuming that the emission changes due to the considered abatement measures are higher than the uncertainties for the emission values.

If no regional inventory exists to run an IAM, the emission database can be based on European inventories (e.g. EMEP inventory or TNO 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 × 50 km resolution for CO, NOx, SO2, NH3, non-methane volatile organic compounds (NMVOC) and primary PM. These annual total emissions should be further disaggregated in time and space. The temporal distribution can be calculated via sector-dependent emission profiles e.g. via sectorial emission profiles from 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. Other emission related inputs that are required are the specific effective height of pollutant release and the chemical speciation for NOx, NMVOC and PM2.5 that should be consistent with the applied AQ model chemical mechanism (see e.g. 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.

4.2 How should I compute emission scenarios?

Emission scenarios can be produced in several ways depending on the objectives of the studies:

1. by modifying the activity index or data. Some emissions sources can be added, removed or moved to other locations, the level of activity of that sources can also be changed (increased / decreased), etc. 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. 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.

Activities and technologies that affect emissions and changes to these two are quantified as part of the DRIVERS block in section 3 above. The level of detail of the scenario is highly dependent on the level of classification of the sources and the data available for each category. 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 projections of the activities and the emission factors. As detailed in EMEP/EEA (2013), future activity assumptions are based on a range of datasets including projections 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 developing the right sector dependent emission factors for any particular projection year.

4.3 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 output of the large scale SEIs is therefore rather based on area than on point and line sources in comparison with small scale SEIs. The sources of large scale SEIs are calculated using more top-down than bottom-up approaches. Consequently, the locations of the sources in large scale SEIs are not accurate and the projections of such SEI on fine resolution grid lead to an overestimation of the sources dilution. Sources then need to be downscaled an redistributed using different earth surface characteristics defined at a smaller scale. For example, the emission can be redistributed according to the land use (emissions release over the ground only and no emissions over water surfaces), the density of population (more emissions over dense population areas like cities), the road network (road transport emissions only in cells crossed by roads), etc. Apart from 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.

When using AQ models, it often happens that an accurate detailed emission inventory is available only on a part of the grid domain on which the study has to be performed. It is therefore necessary to combine data provided by different scale SEIs. 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.

4.4 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 consists in rating the data quality. Some numerical or alphabetical scores are attributed by experts to emission factors and activity data to describe the uncertainties of these data. There are two main classifications that are used in these 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; (2) the AP-42 emission factor rate system that is the main reference in the USA but 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.

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 reduced to improve air quality, reduce exposure and protect human health. This implies that we require a description of the pollutant concentrations that allows us to link 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 what level of detail, is part of the decision framework described in the RESPONSE block (paragraph 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 numerical models but others are only based on AQ observations and emission sources. 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 interpolate these to a grid with a geo-statistic interpolation method to obtain a map of concentrations over a studied area. In such case, the difficulty is as pointed out above to link concentrations to emissions, that is to say, 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 Lenschow approach, PCA and CMB or simply by analysing the emissions inventories and considering that the source contribution is directly proportional to sources emissions. Then, by considering a linear relationship between reduction in emission from a source and reduction of the contribution of this source to the measured concentration, an IAM can be applied.

LEVEL 2: is based on a characterization of the AQ state using one single 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 models, both in terms of AQ and meteorological models, from large scale (Europe for example) to regional (country or regions) and local scale (city or street level). Using a downscaling model chain allows taking into consideration 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 one 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. 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, such as 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 scale that is examined, 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. The temporal resolution of models should therefore be relevant to the temporal resolution of limit values. 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 required for detailed exposure assessment, local scale modelling is necessary. In those causes a Gaussian plume model or a street canyon model is often the most appropriate tool as this would require a very high resolution from an Eulerian model. 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 suitable for the case of secondary pollutants such as ozone or NO_2. 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).

5.3 How can I determine the Source-Receptor relationships that can be used to efficiently calculate the state?

In applications that require a large number of model evaluations, it is not feasible to use the actual numerical model due to the computational limitations. In this case you can replace the model by a source receptor relationship. Source receptor relationships are also known as surrogate models.

Unfortunately, there is no general procedure for defining the structure of such source-receptor relationships. Different structures when used within the same decision problem may lead to the same result. It must be in fact noted that, even if different set of emission reduction measures may indeed determine different air quality patterns, this may not reflect in different values of the indicators that are considered. Consider for instance the case where the indicator is defined in terms of number of exceedances of a certain threshold and the actual exceedance value is irrelevant. Thus the selection of a suitable surrogate model structure should mainly be based on the desired speed and accuracy.

The surrogate models are identified starting from the inputs and results of CTM simulations. The number and the configuration of the CTM simulations that are needed to derive the source receptor model are usually defined during the Design of Experiment phase, which can be performed in two different way:

- The CTM is linearized through matrices, performing a large number of experiments covering the full set of possible input spatial variability (Amann et al., 2011).
- The CTM is directly used to perform a limited number of simulation, selected in order to capture most of the source receptor model input information.

Different approaches can be found in literature to define the structure of a source-receptor model:

- Linear/Polynomial source-receptor models to describe the link between precursor emissions and air quality indexes are identified in Schopp et al. (1999), Guariso et al. (2004), Amann et al. (2011) to be used inside optimization procedure to select air quality control strategies.

- Artificial Neural Networks (ANNs) identified using deterministic CTM model simulations. These surrogate models (Carnevale et al., 2009) can represent the strong nonlinear relationship between emissions and concentration-based air quality indicators such as the yearly mean PM10/PM2.5 concentrations, the daily maximum 8 hour average O_3 concentration, SOMO35, and AOT40.

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

Measurements contain valuable information which can be used together with modelling results. Monitoring data generally estimate the true concentration levels more accurately than model results even if this is 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 **base-case emission scenario**. Several procedures (including model inter-comparison exercises) exist for this purpose, but with different purposes and focuses in terms of model types and/or applications. There is however enough information to provide a standardised evaluation protocol organised according to the different modelling needs and characteristics. Such a protocol is currently developed in the context of FAIRMODE with a particular focus on applications related to the Air Quality Directive.
The reference case is used as a starting point in the integrated assessment and it is therefore essential to estimate the concentrations for this base case as accurately as possible. By applying data assimilation techniques (Denby and Pochmann, 2007), measurement data can be used to improve the model results and provide a more accurate estimate of the concentrations for the base case.

Observational data can also be used in relation with **emission reduction scenarios**. Most of the model results in integrated assessment studies are dealing with future projections under certain policy options. By definition, no measurement data is available for this kind of future estimates. A solution to this problem could be to use measurement data in combination with model results for the reference case by applying data assimilation techniques and then to transfer these data assimilation corrections (or calibration factors) to the considered scenarios and to use then as relevant information in the scenario runs. However, specific and well defined methodologies on how to do so are currently not at hand.

5.5 What would be an appropriate period for an IAM?

Though emissions of atmospheric pollutants are the major cause for ambient air concentrations, meteorology is an important constraint influencing the concentration levels. The ambient air quality of a certain year is influenced by the weather conditions of this specific year. The frequency of the different prevailing weather conditions varies year by year leading to an inter-annual variation of air quality even though the amount of air emissions might be constant.

Air quality planning and modelling has to cope with these meteorological aspects since the starting point is usually to assess ambient air quality by comparing it on an annual basis with certain metrics which mostly are limit and/or target values. The extent to which a given limit value is exceeded will depend on the assessment year and determines the planning task since this year has to be the reference for air quality planning issues at least in a first approach. This might especially be true when air quality plans according to the EU air quality guidelines shall be implemented. The time horizon for the air quality plans target year may be within a few years or when aiming on mid-term or long-term strategies it might be a decade or more. To end up with a realistic perspective of which emission reduction measures will be most effective and reliable, it is important to know how the reference year can be characterised from a climatological point of view. This aspect becomes even more important as the ambient air quality exceedances of limit value become smaller. As the choice of a representative meteorology i.e. a meteorological reference year will affect the results of an IAM, it is important to make this choice properly. Below several approaches to how such a period could be chosen are presented.

A minimum requirement is to construct a concentration time series over a decade or two for various air pollutants qualitatively and to check what an intermediate year could be in this time series that could be representative of the whole time span. It should be checked in addition if there is a year that is representative with respect to long-term (climatological) meteorological conditions. Though most of the common weather situations occur within a year their statistical occurrence can differ considerably from year to year and should therefore be compared with a climatological average at least. This approach will avoid that an inappropriate extreme year in terms of meteorology is used which would obscure the expected effects of an appropriate planning.

Another approach would be to investigate meteorological parameters by correlating them with the behaviour of various air pollutants (PM_{10}, NO_2, O_3, etc.) for some years e.g. a decade. In meteorology it is standard practice to compare the meteorology for a single year with a thirty year average for climatological purposes. With respect to air quality climatology, thirty year averages are rarely available since most concentration time series for the various air pollutants are still much shorter than thirty years. This method is more elaborate than the first one and gives a deeper insight into air pollution climatology.

Both these approaches to determine a representative meteorological year are qualitative or semi-quantitative. However, they have the advantage that they are well structured and thus can be documented well and are transparent and that they can be applied with a proportionate effort. Additionally, the modelling results for the reference year could be supplemented with results obtained for a worse and a best case meteorology. The bandwidth of the results then gives an estimate on the variability that has to be expected using different meteorological years.

A straight forward and quantitative but costly way to evaluate meteorological variability and its influence on air quality is to apply CTMs for a decade or even more. The separation of meteorological variability can be achieved with three sets of model runs and a comprehensive statistical evaluation of the modelling results together with the ambient air quality measurement data:

- “Normal model runs” – simulation with individual meteorology and individual emissions for each year for the whole time span under consideration
- “Influence of emissions model runs” – simulation with a meteorological “normal” year and keeping this meteorology constant for all years while varying emissions for each individual year of the whole time span
- “Influence of weather model runs” - simulation using emissions belonging to the above mentioned meteorological “normal” year, keeping these emissions constant for all years and varying meteorology for each individual year of the whole time span.

5.6 How should I deal with uncertainty when calculating the state?

When the AQ state is evaluated through measurements only, 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 verified and 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) for which details can be found in the APPRAISAL deliverable D2.5(2013):

1. **Operational model evaluation:** 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).

2. **Diagnostic model evaluation:** Diagnostic model evaluation is a validation methodology normally undertaken by model experts and model developers in the process of developing or improving an AQ model. Therefore, this approach is mainly used and recommended for modelling applications related to research programmes and not for policy applications. 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.

3. **Dynamic model evaluation:** This model evaluation methodology is highly represented in the literature and is also known as the "sensitivity method". The underlying principle of this approach is the analysis of model responses to changes in model input data. 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.

4. **Probabilistic model evaluation** 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 it is therefore considered 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 are not founded in don't have a physical basis (Favez et al., 2010).

In conclusion, as a minimum an operational validation 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.
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 diagnostic evaluation (e.g., sensitivity tests) or probabilistic evaluation (e.g., errors propagation) are in this case recommended.

6 IMPACT on health

6.1 What corresponds to a simple and what to a more complex 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 the population distribution, the level of vulnerability used for the whole population or some specific groups and the detailed location of the population with respect to their exposure. Therefore, the assessment of the impact on health within an integrated assessment modelling framework requires one to distinguish between exposure to air pollution and the health effect of that exposure.

It is possible to distinguish three levels (see the WP3 deliverable for more details):

- **LEVEL 1**: A coarse description of exposure provided either by measurement or modelling of AQ (e.g. average mean annual exposure for a city), using 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 city. 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.
- **LEVEL 3**: A detailed temporal and spatial resolution for AQ data and population data, that will allow deriving detailed 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 effect in 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.

6.2 Which is the preferred indicator that should be used to quantify health impact?

The choice of health indicators to support decisions has to be made to show the potential policy action or inaction impact. Indicators have different strengths in supporting policies. The indicator choice depends mainly on the available data for the methodology and on the policy question itself. Different types of policy questions may need different kinds of state-of-the-art indicators. The choice of indicator(s) used in a given assessment should anticipate what the impact assessment will be used for (WHO working group, 2000).

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. 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.

The selection of the indicator depends on the stressor 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 the ICUR principle in health economics (Incremental Cost-Utility Ratio). Some other IA studies involving multiple stressors might require indicators that take into account age, or mortality and certain morbidity effects. The choice of indicators is driven by practicality and needs of the assessment. Therefore all of the indicators are state-of-the-art in certain conditions.

Air pollution has been known as a risk factor for respiratory and cardiovascular diseases but also an important issue in public health given its impact on long-term mortality (HEI 2009). 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. Otherwise, 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. Most epidemiological studies find a wide range of health outcomes to be consistently related to particulate matter (Pope et al, 2004 ; Brunekreef & Fosberg 2005). Those results are supported by toxicological evidences. 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. The review done for the REVIHAAP project concludes that PM, NOX, SO2 and O3 are considered responsible for the health effects seen in epidemiological studies (WHO 2013). These epidemiological studies provide exposure-response functions (ERFs) necessary for Health Impact Assessment (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 particulate pollution and the observed health effects.

In general when indicators are used it is recommended to include an uncertainty assessment (see also 6.6).

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

The Clean Air for Europe (CAFE) program describes thresholds at the population levels to be "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 (page 5) (Hurley et al. 2005)". The CAFE project adopted no-threshold policy in their assessment based on the lack of scientific evidence on the existing threshold for the pollutants considered in the assessment (particulate matter 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 at very low concentrations (Janssen et al 2012, WHO 2013, Pedersen et al 2013). The latest review of the scientific literature stated 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 the moment, in integrated assessment studies, it can therefore be assumed that the adverse health effects caused by PM, ozone and SO_2 air pollutants do not have any threshold concentrations in population level.

6.4 What are the PM components that I should consider? Which are the most toxic ones and have the biggest impact?

An integrated science assessment for PM was published by the EPA in 2009 to support the review of the national ambient air quality standards. The integrated science assessment 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” (EPA, 2009). Despite the increased number of studies after 2009, the general conclusion remains the same. In the technical report for the REVIHAAP project an overview is given of the current knowledge base (WHO, 2013).

6.5 What are the effects of choice of resolution, detail?

The output representativeness depends on the level of detail of population data. Also the temporal resolution is of importance, decisions on short term exposure or on long term exposure should be addressed separately using 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 Perez 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 in epidemiological studies, assessing the health effects of air pollution, the most detailed exposure estimate that is available For pollutants with high spatial variability this can be based on personal activity-based modelling or personal dosimetry.

6.6 What are the uncertainties I should 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 result in 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 restricted due to limitations of resources (cost, time, personnel etc.) and thus scarce. Epidemiological studies are also based on health outcomes measured in the past that are combined with a given exposure to a given pollutant, whereas these variables may change 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, future AQ policies need to be regularly updated

An important issue in epidemiological studies is 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.)

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 common 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. 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 through using a combination of model types or to use 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.
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. In general, epidemiological studies that have used finer spatial resolution to relate people to air pollution levels tend to report higher mortality/morbidity impacts (Tainio, 2009). Therefore, it is recommended that the most detailed available exposure estimate (e.g. for pollutants with high spatial variability this can be based on personal activity-based modelling or personal dosimetry) be used in epidemiological studies assessing the health impacts 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. 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).

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" (http://www.integrated-assessment.eu/guidebook/uncertainty_analysis) and the Aphekom on-line tool (http://si.easp.es/aphekom).

7 RESPONSE: choice of abatement measures

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 air quality index (AQI). In this context, the costs of the emission reduction actions can be evaluated as an output of the procedure.
- **LEVEL 2**: Source Apportionment and Scenario analysis. In this case the sources of emissions that are mainly influencing AQ index are derived through a formal approach; this then allows to select the measures that should be applied to improve the AQI(s). Again, emission reduction costs, if/when 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), 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 judgment/scenario analysis/source apportionment** This is the approach 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 through simulations of an air pollution model. This approach does not guarantee that cost-effective measures are selected, and only allows for “ex-post evaluation” of costs and other impacts.
- **optimization** This pathway indicates 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 (end-of-pipe) and non-technical (energy efficiency) measures/policies to be encouraged and/or introduced to reduce pollution, explicitly considering their impacts and costs.

7.2 How can source apportionment be used when determining abatement measures?

The effectiveness of any type of abatement measure strongly depends on the reliability of pollution source identification and quantification. During the set-up phase of an IAM framework, source apportionment (SA) can be used to identify the key emission sources in the area of interest. This allows for a better delimitation of the problem so that resources can be allocated to study in depth the identified more relevant sectors of activity and limit the set of measures to those that will be most effective.

SA can as such also be used together with IAM techniques like scenario analysis or optimization based approaches, such as cost-benefit, cost-effectiveness, multi-objective approaches. To that end, SA could drive the choice of the emission patterns to be tested through scenario analysis, to limit the number of simulations to be performed through a CTM. Alternatively, it could limit the degrees of freedom of cost effectiveness analysis, constraining the optimal solution to consider only a subset of the possible emission reductions previously identified applying SA.

Both AQ dispersion models as well as receptor models may be used for source apportionment purposes. Unlike dispersion models, receptor models do not rely on emissions / meteorological data and chemical transformation mechanisms to estimate the contribution of sources to ambient AQ concentrations at selected receptor points. Instead, source receptor models examine the chemical and physical characteristics of pollutants (both gases and particles) measured at source and at receptor sites to identify the presence of and to quantify source contributions. It is therefore suggested that these models are applied in complementarity to other air quality models. A number of options are available for policy makers in terms of receptor models, including the Chemical Mass Balance (CMB) and UNMIX models as well as the Positive Matrix Factorization (PMF) method. The deciding factor is generally data availability, as different receptor models require different information on source characteristics and chemical analysis of air samples taken on site. The suitability of receptor models for attribution of pollutants to their sources is recognised by experts from EU member states (Fragkou et al., 2012), as they provide, with low computational intensity, source estimations at the urban and regional scales which are independent from emission inventories and meteorological data pre-processors.

Receptor models have the disadvantage of not taking into account the physical and chemical processes that govern the transport and transformation of pollutants. Therefore, in order to explain the causal link between emissions (pressure) and concentrations (state) and understand the mechanisms of contribution of a specific emitting source to the AQ of the surrounding area, dispersion models need to be used. This is of particular importance for highly reactive pollutants such as ozone and PM. The long-distance transport of these pollutants dictates the need to account for the physical and chemical processes governing pollutant transfer, simulated by dispersion models. In the case of source apportionment for transboundary pollutants such as Saharan dust, the combination of Eulerian and back-trajectory dispersion models is favoured by the EU Commission (EC; 2011), as back trajectories constitute an ideal first screening approach for identifying the origin of transported polluted air masses. A detailed guidance on the quantification methodology for natural emissions is provided in the relevant Commission Staff Working Paper.

7.3 What are the options when measures are determined through optimization?

Different approaches (both in discrete and continuous world) are available:

- *Cost-benefit analysis*: all costs (from emission reduction technologies to efficiency measures) and benefits (improvements of health or environmental quality conditions) associated to an emission scenario are evaluated in monetary terms and an algorithm searches for solutions that maximize the difference between benefits and costs among different scenarios.

- *Cost-effectiveness analysis*: Due to the fact that quantifying benefits of non-material issues is strongly affected by subjective evaluations, the cost-effectiveness approach has been introduced. It searches for the best solutions considering non-monetizable issues (typically, health related matters) as constraints of a mathematical problem, the objective of which is simply the sum of (possibly, some) costs (Amann et al.,2011).
- *Multi-objective analysis*: it selects the efficient solutions, considering all the objectives of the problem explicitly in a vector objective function (e.g., one AQI and costs), thus determining the trade-offs and the possible conflicts among them (Guariso et al., 2004; Pisoni et al., 2009).

7.4 What types of measures can be considered?

Macro sector level emission reductions where emissions reductions to improve air quality are provided at a CORINAIR macro sector-pollutant level. This is a very aggregated approach, but can provide policy makers with some insight on how to prioritize the interventions (Carnevale et al., 2012).

Primary technical measures that reduce emissions at the source are measures such as fuel change in power plants (e.g. from coal to gas) or process change (e.g. application of clean coal technologies). These measures are applied to reduce emissions already during the combustion process. They do not change the driving forces as energy demand remains unchanged but change the structure of primary energy supply.

Examples of “end-of-pipe technologies” also called secondary technical measures, are various types of air-pollution control devices (APCD) applied to large combustion plant emissions and the adoption of diesel particle filters to cars. These measures are applied to reduce emissions before being released in the atmosphere. They neither modify the driving forces of emissions nor change the composition of energy systems or agricultural activities.

“Efficiency measures”, sometimes called "Non-technical measures", are those, 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). Localization decisions (e.g. building new industrial areas, or new highways) can also be considered as “efficiency measures”. The use of efficiency measures is now limited to scenario analysis, because it is very difficult to estimate the costs of such measures, particularly, because they impact many other sectors beside air quality. For instance, car sharing has the potential to reduce not only exhaust emissions, but also accidents and noise. How can an overall cost be associated to the benefits in such diverse sectors? It will be necessary to further investigate such actions. Also, an additional complexity is related to the use of these measures in an optimization framework which requires new formal approaches.

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

The effect of policy at the national or European level has to be accounted for both in the assessment of the current air quality and of abatement measures. In the case of the assessment of the current air quality, national and European policies should be reflected in the correct quantification of the large scale emissions and the resulting back ground pollutant concentrations through long range transport in the local area of interest. This is especially relevant in the case of secondary pollutants such as ozone and secondary PM. 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 "go beyond CLE" measures within their regional domain, these "higher level" constraints cannot be disregarded or modified.

7.6 How can I take into account the uncertainty in the response?

One should focus on robust strategies, that is policies that do not significantly change due to changes in the uncertain model elements. This requires a quantification of the sensitivity of the choice of the abatement measures to changes in the underlying emissions, concentrations and impacts that result from these measures. It is in fact worth underlining that, while for air quality models the sensitivity can be measured by referring in one way or the other to field data (Thunis et al., 2012; Carnevale et al., 2013) 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 first draft version of the guidance document we have presented how different elements of an integrated assessment modelling methodology or system (IAM) should be addressed.

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 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 where several levels to which the blocks can be elaborated have been distinguished. 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.

Some other topics that are addressed in this document concerning aspects that might be critical for a suitable 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.
- 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. The evaluation of such measures is therefore currently limited to scenario analysis.

The guidance as presented in this document is mainly based on the current results of the review and design work packages. In the following project phase this guidance will be tested through a number of practical test cases and adapted accordingly.

Table 1 - Different levels of detail for the different DPSIR blocks

<table>
<tr><td>DPSIR blocks</td><td colspan="3">Levels of complexity</td></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 projections 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</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>

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