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II. GUIDANCE ON MAPPING CONCENTRATIONS LEVELS AND DEPOSITION LEVELS

Updated by Anne Christine Le Gall, Chairwoman of the Task Force on Modelling and Mapping, from initial text edited by D. Fowler and R. Smith, Mapping Manual 2004

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(This refers to Chapter 1. This title is useful here in order to have the benefice of Word automatic section numbering. It will be removed in the final layout.)

II.2   INFORMATION RELATIVE TO MODELLING AND MAPPING CONCENTRATION AND DEPOSITION

II.2.1   GENERAL REMARKS AND OBJECTIVES

The purpose of this chapter is to provide general information to the participating countries on the generation and use of concentration level and deposition maps for a range of pollutants. These maps may be used for different purposes, such as assessment of air quality for human health and assessment of ecosystems via critical loads exceedances. Depositions are then compared with critical level/load maps. In this chapter, main principles of mapping concentrations levels and deposition are described and discussed. A range of different techniques for the provision of maps of concentration and deposition, depending on the resources and ambition of the country are described. Modelling procedures are outlined and the reader is referred to specialist publications for further measurement and modelling approaches. Note that within the LRTAP Convention, modelling and mapping of pollutant concentrations in air and their deposition is EMEP mandate (see section II.2). The reader is therefore referred to EMEP documents and website for further details.

There are three main objectives for mapping of concentrations and depositions over a territory within the LRTAP Convention.

The first aim is to construct exceedance maps relative to critical levels and loads. At large scale (covering several countries), transfer coefficients may be calculated. They allocate critical loads exceedances in each grid cell of a transport chemistry model to emissions in all European countries This approach is particularly well suited to assess policy scenarios in integrated assessment modelling. It provides thus scientific results for a) implementation of, and compliance with, existing LRTAP Convention protocols and b) their review and extension.

The second aim is to map concentrations and depositions which can be used for effects assessments in specific ecosystems. Such data are needed with a much better spatial and temporal resolutions than required for integrated assessment modelling. National Focal Centers should aim at a sufficient spatial resolution for the assessment process, making use of national models and measurement networks. Large scale transport chemistry models provide background, long-range transported air components which can be used as boundary conditions for such national models.

The third aim is to gain information on deposition at site level. This is particularly important for "monitoring" ICPs such as ICP Forests and ICP Integrated monitoring. Then concentration measurements, coupled with wet, dry and throughfall deposition measurements are used for deriving process parametrisations (mainly micrometeorological measurements) and for independent model validation (mainly throughfall measurements; see Chapter II.3.1, II.3.2 and II.3.10). This chapter does not detail field and laboratory methods for concentrations and deposition measurements. The interested reader is refered to specific ICPs´ Manuals and to the EMEP publications on monitoring methodology (ICP Integrated Monitoring[1]; ICP Forests[2]; ICP Waters[3]; ICP Materials[4]; EMEP/CCC[5]).

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[1] http://www.syke.fi/en-US/Research__Development/Ecosystem_services_and_biological_diversity/Monitoring/Integrated_Monitoring/Manual_for_Integrated_Monitoring

[2] http://icp-forests.net/page/icp-forests-manual

[3] http://www.icp-waters.no/

[4] http://www.corr-institute.se/ICP-Materials/web/page.aspx

[5] http://www.nilu.no/projects/ccc/

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II.2.2 UN-ECE EMEP MODEL: A LRTAP CHEMISTRY-TRANSPORT MODEL

The EMEP model is the reference chemistry-transport model for the LRTAP Convention and European air pollution policy assessments. The model covers all of Europe. Its geographic resolution has increased over the years (initially it was 150 x 150 km²). Since 1999, EMEP model was run on a 50 x 50 km² grid (1.0° long x 0.5° lat) although local air pollution emission and chemistry may now be simulated on a 7 x 7 km² grid (0.125° long x 0.0625° lat). Its vertical resolution extends from ground level to the tropopause (100 hPa or arout 16 000 m). It is divided in 20 layers, the lowest having a thickness of about 90m.

The model has changed extensively over the last ten years: with flexible processing of chemical schemes, meteorological inputs (from European Centre for Medium-Range Weather Forecasts[6]) and with nesting capability. The model is used to simulate photo-oxidants and both inorganic and organic aerosols. It takes into account land use (source CCE/SEI for Europe, Global Land Cover 2000[7] elsewhere, or the CCE harmonised land cover map), at a resolution of about 5 km.

The EMEP model is now available as public domain code, along with all required input data for model runs for one year. It is extensively documented in EMEP status reports, EMEP publications and in the scientific literature (EMEP, 2013; Simpson et al., 2012 and see also http://emep.int/emep_publications.html).

Since 2003, the methodology has been revised for the simulation of dry deposition for particles, the emissions of hydrocarbons from vegetation, NO emissions from soils, co-deposition of SO_2 and NH_3, the calculation of mixing heights, or the introduction of pH response during sulphate formation. Smaller changes in the equations or parameters values have also been carried out.

The EMEP model is the fruit of 30 years of discussions and intense scientific work carried out within the LRTAP Convention, and in particular within the MSC West and East Centres of the European Monitoring and Evaluation Programme (EMEP programme, http://emep.int). It is continuously evolving. A recent update of the model characteristics has been given by Simpson et al., 2012. Initial work is described in proceedings from several UN-ECE Workshops dealing with this subject, most notably the 1992 "Workshop on Deposition" in Göteborg, Sweden (Lövblad et al. 1993), the 1993 "Workshop on the Accuracy of Measurements" with WMO sponsored sessions on "Determining the Representativeness of Measured Parameters in a Given Grid Square as Compared to Model Calculations" in Passau, Germany (Berg and Schaug 1994), and in Erisman and Draaijers (1995), Sutton et al. (1998), Slanina (1996), Fowler et al. (1995a, 2001a) and ICP Forests Manual (UN-ECE 1999). The model characteristics and performances have also been discussed within the task Force on measurements and modelling (cf. http://www.nilu.no/projects/ccc/tfmm/index.html) and as part of European Research Programmes (such as EC4MACS, EURODELTA…). Supplementary information can be found in other workshop proceedings and in scientific journals.

Many other national chemistry-transport models are also available. A large number of inter-comparisons in recent years have discussed their strengths and weakenesses (Colette et al., 2011; Colette et al., 2012; Cuvelier et al., 2013; Cuvelier et al., 2007; Fiore et al., 2009; Huijnen et al., 2010; Jonson et al., 2010; Langner et al., 2012; van Loon et al., 2007). In terms of performance, the EMEP model has ranked well in these studies, with consistently good performances for different pollutants (ozone, PM, etc.). In terms of complexity, the EMEP model is fairly similar to other regional-scale European chemistry-transport models, such as MATCH (Robertson et al., 1999), CHIMERE (Bessagnet et al., 2004) or DEHM (Christensen, 1997; Frohn et al., 2001). All of these models have some flexibility with regard to chemical schemes and have zooming-capabilities.

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[6] http://www.ecmwf.int/

[7] http://bioval.jrc.ec.europa.eu/products/glc2000/glc2000.php

Within the LRTAP convention, the EMEP values are regarded as default data allowing the assessment process to be completed everywhere. In the following sections, when references to chemistry-transport models are references to the EMEP model.

Further description of the EMEP grid is given in chapter VIII.

EMEP model has been developed together with the EMEP monitoring network. This is managed by the Chemical Coordinating Centre[8] (CCC), responsible for collating contributing parties data, as well as developing monitoring methods and standards.

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[8] http://www.nilu.no/projects/ccc/index.html

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II.2.3 MAPPED ITEMS IN LONG RANGE CHEMISTRY-TRANSPORT MODELS RELEVANT TO CRITICAL LOADS

In order to fulfil their role, long range chemistry-transport models simulate reactions between a large number of natural and anthropogenic substances and take into account many processes. Here, in the context of the calculation of critical levels/loads and their exceedances, the following items are mapped:

For critical level exceedance maps:

- ozone flux (PODy for vegetation, SOMO35 for human health) and ozone concentration (AOT40 values),
- sulphur dioxide concentration,
- nitrogen dioxide concentration,
- ammonia concentration.

For critical load exceedance maps:

- oxidized sulphur (SO_x) deposition (total and non-sea-salt),
- oxidized nitrogen (NO_x) deposition,
- reduced nitrogen (NH_y) deposition,
- total nitrogen deposition,
- base cation and chloride deposition (total and non-sea-salt),
- total potential acid deposition.

Heavy metal deposition:

- total deposition of mercury, lead and cadmium,
- other priority heavy metals as data become available and policy needs are expressed (such as copper, nickel, zinc, arsenic, chromium and selenium).

Black carbon is also included amongst the species for which concentrations and deposition are calculated. Although its impact was initially mainly calculated for human health, this pollutant also intervene in material soiling and to an extent that now remains to be defined in ecosystem functioning.

As input into deposition and critical load computations:

- precipitation amount and other meteorological parameters,
- wet, dry, cloudwater/fog and aerosol deposition.
- In the context of biodiversity and climate changes, meteorological parameters such as temperature, light availability, soil wetness become relevant.

II.2.4 PROCESSES RELEVANT TO AIRBORNE POLLUTANT CONCENTRATIONS AND DEPOSITION

The behaviour and the fate of each chemical species depend on its physico-chemical properties. They are also the consequences of the processes the substance undergoes in the atmosphere or at the interface with the ecosystems. These processes may be measured at monitoring stations and/or taken into account in the chemistry-transport models. They are shortly described in the following sections.

II.2.4.1 METEOROLOGY

Meteorological parameters are required inputs for most critical levels or critical loads calculations, whether at site level or for modelling on wide geographical area. The data requirements and data provision will vary from country to country. Data are generally avaialable from national weather services. European data can be obtained from European Centre for Medium-Range Weather Forecasts (ECMWF[9]), who provides modelled data based on observations within Europe. There are other sources for some data such as the US National Center for Atmospheric Research global precipitation database[10].

Precipitation amounts are needed for critical load computations, for wet deposition mapping and for surface wetness parametrisations.

[9] http://www.ecmwf.int/

[10] https://climatedataguide.ucar.edu/climate-data/gpcc-global-precipitation-climatology-centre

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Fog and cloud occurence are needed for cloudwater/fog deposition estimates.

Wind speed, temperature and *radiation* are basic requirements for the inferential modelling of dry deposition. Additionally, *relative humidity*, *soil water deficit* and *atmospheric stability* are often required.

II.2.4.2   EMISSIONS

The precise knowledge of quantities as well as localisation of substances sources is essential to model their fate in the atmosphere. For less well known sources, emission factors may be used. Recent work has shown that the importance of biogenic emissions in the total budget of substance exchanges between the atmosphere and ecosystems (Fowler et al., 2009).

Within the LRTAP Convention, emissions are compiled by the Task Force on Emission Inventories and Projection[11]. Data reported following the "SNAP" nomenclature[12] from each party is compiled, provided to EMEP and integrated by assessment modelling teams. This Task Force also provides guidance on estimating emissions from both anthropogenic and natural emission sources, with default methods and emissions factors (TFEIP, 2013).

Total emissions include:

- Anthropic emissions are emissions issued from human acitivities: industry, transport, energy, urban activities… They are essential in pollution management since they are the sources that can most easily be reduced.

The availability of accurate local meteorological data is often a constraint to detailled local high resolution modelling. Therefore the success of models in improving deposition estimates to specific ecosystems may depend as much on the availability of quality meteorological data as on the quality of the local concentration estimates or measurements.

- Biogenic emissions: The vegetation and the soil are receptors for substances but are also emitters. They are dependant on surface types and meteorology. Biogenic emissions are of particular importance for mapping ozone because of VOC emissions and for mapping ammonia, which is emitted as well as absorbed by vegetation. Biogenic emissions are therefore linked to the type of vegetation and soil present over the area modelled. Their quality is therefore linked to that of the land use map.

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[11] http://www.UN-ECE.org/env/lrtap/taskforce/tfeip/welcome.html

[12] SNAP: "Selected Nomenclature for Air Pollutants" nomenclature, which consists of yearly masses per surfaces for various domains and resolutions.

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II.2.4.3   SEA SALTS

Primary marine aerosols are composed of sea salt and organic material, including sulphur compounds such as dimethyl sulphate (DMS). They are an important contributor to the global aerosol load (Barthel et al., 2014). In the framework of critical loads, marine aerosols are important as they include acid anions (sulphate and chloride) as well as base cations. Consequently, the base cation and chloride deposition in the charge balance from which critical loads are derived have to be corrected for sea salt contributions, since critical loads are compared with anthropogenic inputs only (thus excluding also dust from erosion, especially from Sahara).

There are two main mechanisms for sea salts generation: bubble bursting during whitecap formation and through spume drops under the wave breaking (Simpson et al., 2012). In the EMEP model, sea salts calculations includes particles with diameters up to 10 µm that originate mainly from the bubble mediated sea spray (Tsyro et al., 2011).

Depositions of base cations, sulphur and chloride (given in equivalents) are corrected by assuming that either all sodium or all chloride is derived from sea salts, and that the relations between ions are the same as in sea water (after Lyman and Fleming 1940, cited in Sverdrup 1946). Details on how to carry out this correction is given in Chapter V.

II.2.4.4   DEPOSITION OF AIRBORNE SUBSTANCES

Deposition is the various processes through which airborne substances are transferred from the atmosphere to the vegetation, the soil or the water. These processes occur at different time and space scales. It is once that they are deposited that these substances may impact ecosystem functioning and human health or damage materials.

Total deposition is the sum of all dry and wet deposition processes. Distinction between the different "depositions" type is as much a matter of physico-chemical characteristics of the deposition as a matter of measurement technique. Some of these are briefly described below. Information given below is largely inspired from two documents   :

The ICP Forests Manual (chapter 14: Sampling and analysis of deposition[13]) and the World Meteorological Office Manual (WMO, 2008 - updated in 2010). The interested reader will find more detailed information in these references. For information related to the simulation of these parameters, see Simpson et al., 2012 for the EMEP model, as well as for instance, Hertel et al., 2013; Hertel et al., 2006; Pacyna et al., 2008; Vet et al., 2013).

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[13] http://www.icp-forests.org/pdf/FINAL_Depo.pdf

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II.2.4.4.1   BULK DEPOSITION

Bulk deposition includes parts of particulate and gaseous deposition during dry periods as well as wet deposition. It is sampled continuously in an open area with a plastic funnel connected to a sample bottle. This cheap and simple method makes its implementation easy. However, the method is sensitive to dust from neighbouring areas. In regions with calcareous soils, bulk deposition gives incorrect information on the pH and chemistry of atmospheric deposition.

II.2.4.4.2   DRY DEPOSITION

Dry deposition is the settling due to gravity (sedimentation), impaction (due to turbulence) and interception (via chemical and biological processes) of aerosols and gases on a surface during dry periods. These processes are strongly influenced by the type of surfaces (leaves, needles, rocks, water. etc), the humidity of surfaces, the macro- and micrometeorology (stomata closure) and the bio-physico-chemical characteristics of the substances.

Dry deposition is a slow but continuous flux of contaminants to the soil, water or biogenic surfaces. It involves pollutants carried in the lowest part of the atmosphere and is thought to be of particular importance close to the sources of emissions.

These processes are simulated through a resistance analogy, including aerodynamic, surface and substance specific canopy resistances and stomatal conductance (Menut et al., 2013; Simpson et al., 2012). Resistances are modelled using observations of meteorological parameters and parametrisation of surface exchange processes for different receptor surfaces and pollution climates as described in Erisman et al (1994a), Smith et al. (2000), Nemitz et al. (2001), Emberson et al. (2000), Grünhage and Haenel (1997), Gauger et al. (2003).

II.2.4.4.3   WET DEPOSITION

Wet deposition occur when gaseous or particulate contaminants are scavenged from the atmosphere by rain, snow or cloud and fog and subsequently deposited to surfaces with the rain drops, the snow flakes or the fog droplets. Due to the scavenging process involved, wet deposition is a good integrator of the chemical content of the atmosphere and is potentially influenced by long range transport of chemicals. Also wet deposition leads to rapid delivery of pollutants, highly concentrated in precipitation, during the short times in which precipitation events occur.

The simulation of wet deposition integrates both in cloud and below cloud scavenging of gases and particles (Simpson et al., 2012).

There are different sampling methods possible for wet deposition. They are not exactly equivalent as they do not sample exactly the same fraction of wet deposition.

II.2.4.4.3.1   WET ONLY DEPOSITION

Wet only deposition determines the fluxes of dissolved components from the atmosphere in rain, snow and hail in the open field. It gives valuable information on the chemistry of atmospheric deposition and on long range transport of air masses. It requires collectors that open automatically at the onset of precipitation by the use of a sensor and close at the end after rain/snow/hail has stopped. It thus excludes particles and gases deposition during dry periods. These equipments require electrical power and maintenance.

They may not work properly in case of heavy snow.

Wet deposition is a function of precipitation rates. If wet deposition is assessed from field monitoring, it would be inferred from relatively dense meterological data and less dense monitoring of concentrations in rain, snow and hail.

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Then measured solute concentrations may then be interpolated and wet deposition may be estimated from a function of the mapped solute concentration and the precipitation amount, the latter provided by the meteorological service for the country (cf. for instance in Kopacek et al., 2012).

A very important increase of wet deposition occurs over wind exposed hills and mountains, particularly in the Northern Europe uplands, due to wash out of topographic cloud by falling rain or snow. As networks do not generally measure at high elevation in complex terrain, these effects are generally omitted from network measurements. The underlying physical process is well documented and the effects may be modelled using the network data (Dore et al. 1992, Fowler et al. 1995b, Kryza et al., 2011).

II.2.4.4.3.2 OCCULT DEPOSITION

Cloud, fog, rime, mist droplets also scavenge the atmosphere from aeorols and gases. When they come into contact of a surface (as the air mass meets a hill or passes through a forest), droplets are deposited, with their chemical charge. This is occult deposition. Although amounts deposited are relatively small, concentrations in occult deposition can be very high, which could lead to direct impacts.

This deposition can be estimated from concentration measurements of airborne substances by *micrometeorological measurements* at the process level (for SO_2: Fowler et al. 2001c; for NH_3: Flechard and Fowler 1998; for cloud: Beswick et al. 1991). Micrometeorologically based long-term flux measurements (i.e. continuous flux measurements over more than a year) are possible for O_3, NO_x and SO_2 (LIFE project, Erisman et al. 1998a) and for CO_2 and H_2O (Aubinet et al. 2000). Such measurements give information about the seasonal and interannual variability in the fluxes. Low cost micrometeorological methods, such as the Time Averaged Gradient (TAG) system (Fowler et al. 2001b), provide the means of obtaining deposition parameters for many more representative terrestrial surfaces in Europe.

The methods mentioned here only work if stringent prerequisites concerning micrometeorological variables (e.g. surface homogeneity) are fulfilled. They cannot be directly extrapolated but the process knowledge obtained from such measurements can be parametrized in inferential models and fluxes can be mapped using this information (see high resolution modelling above).

The deposition velocities of cloudwater/fog droplets can be similarly estimated by modelling momentum transfer (Fowler et al. 1993) and a similar technique has been used to estimate base cation deposition (Draaijers et al. 1995). Parameters determining the deposition velocity include atmospheric parameters (e.g. wind speed, temperature, radiation, relative humidity, atmospheric stability, cloud and/or fog frequency) and surface conditions (e.g. roughness, wetness, stomatal response, soil water).

II.2.4.4.3.3 THROUGHFALL AND STEMFLOW MEASUREMENTS

Throughfall deposition gives an estimate of the total deposition (bulk, leached through the canopy, dry depositions) that reaches forest floor.

Stemflow is wet deposition sampled on stems, usually at the trunk base. As water flows along the tree stems, it washes off gases and aerosols previously deposited by rain, snow or occult deposition. Stemflow also includes leachates from the bark and leaves. In beech forest, stemflow in an important contributor to the deposition do the forest floor.

Throughfall + stemflow give an estimate of the deposition to the forest floor. Throughfall is sampled beneath the canopy, stemflow near the base of the trunk. Both use via cheap, simple, low maintenance devices.

According to ICP Forests, the main drawback of the throughfall method is the interaction between the canopy and the

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throughfall water for nitrogen, potassium, calcium, magnesium, manganese and protons. However, the results from throughfall monitoring can still be used as a valuable indicator for the nitrogen and base cation deposition to the forest. Throughfall deposition can give information on the lower limit of the true deposition of nitrogen and the upper limit of true deposition of base cations other than sodium. For sodium and sulphur the canopy uptake and leaching is considered to be negligible and consequently the throughfall flux is used to estimate the total deposition.

The data will provide knowledge on the seasonal variation and the trends of deposition. In many cases throughfall monitoring is considered to be sufficient, and stemflow is only measured for some tree species, for which it is known to be of importance (e.g. beech trees).

Throughfall measurements give a good overview of the deposition situation in the forest, not only for sulphur but also for nitrogen compounds. Recent Swedish experiences have highlighted the problems with comparing throughfall measurements with wet deposition when the dry deposition contribution to the total is very low (Westling *pers. comm.*), as is now the case for sulphur in many areas of Europe. Large uncertainties in wet deposition at wind-exposed sites have been shown with field intercomparison studies (Draaijers et al. 2001). Even if it is not possible to estimate the total deposition of nitrogen with this method, a lower limit can be set. Sampling considerations (e.g. location of collectors, species composition, spatial variability) are very important for achieving good results and sampling requirements are described in detail in the ICP Forests Manual (UN-ECE 1999) and in review articles such as Draaijers et al. (1996a) and Erisman et al. (1994b).

II.2.4.4.4 TOTAL DEPOSITION

Total deposition is the sum of wet only and dry deposition. It excludes ion exchanges within the canopy. It may be inferred from the values of the depositions described above.

The relation between total deposition and throughfall can be expressed as:

(II.1)   Total DEP = DRY + WET + Cl/Fog
= THF - CEX

where:

THF = Flux in throughfall plus stemflow

$DRY, WET, Cl/Fog$ = dry, wet, cloudwater/fog deposition

CEX = canopy exchange; $CEX > 0$ for leaching, $CEX < 0$ for uptake

When CEX=0, the dry deposition can be estimated as the difference between total flux in throughfall and independent measurements of wet and cloudwater/fog deposition. If CEX differs from 0, dry deposition cannot be distinguished from internal cycling. This method can give large overestimates of the true deposition flux (CEX>0), due to canopy leaching (for some base cations), or large underestimates of the true deposition flux (CEX<0), due to canopy uptake (e.g., for nitrogen compounds and protons). The sum of throughfall and stemflow is considered to be equal to total deposition only for sodium and sulphur. For orther substances, throughfall plus stemflow fluxes may be interpreted as upper bounds of total base cation deposition and as lower bounds of total nitrogen and proton deposition.

In some cases, the total deposition to plant canopies can be deduced from throughfall and precipitation measurements in the open field using empirical *canopy budget models* (based on equation II.1, see Adriaenssens et al., 2013; Draaijers and Erisman, 1995). There are different possible approaches, some being more reliable than others. For instance, special care should be given to the choice of the main tracer.

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II.2.4.5   STOMATAL CONDUCTANCE

Stomatal conductance is the process through which gas and water may enter leaf cells. It is dependant on the plant species, its phenology (including leaf nitrogen content) and environmental drivers such as water, CO_2 and light availability.

Stomatal conductance increases when leaf surface are wet (due to humidity in air greater than 90%) and when soil humidity is high. Then the flux is mostly determined by atmospheric resistances (Erisman et al. 1994a).

II.2.4.6   LAND USE ROUGHNESS AND TOPOGRAPHY

Deposition increases with the roughness of the vegetation, as it filters airborne substances. As a consequence, forests receive larger quantities of dry and wet depositions than grasslands or bare soils. These effects will influence results from a monitoring station and may be assessed by modelling, with the help of land use maps. It is then essential that the land use map used for modelling deposition is the same as the one used for calculating critical loads and exceedances. Within the LRTAP Convention, land used maps are issued from Corine Land Cover[14] in Europe, Global Land Cover 2000[15] elsewhere. The development of work on biodiversity has led to map vegetation types at a very fine scale (100 x 100 m²) over Europe. The "harmonised land cover map" now used under ICP M&M is available from the CCE and is further described in Chapter V.

Topography influences the exposition of a site to air masses. In valleys, air may be stagnant. Coasts are submitted to high winds. Rainfall is more abundant on the winward side than downwind of mountains. At high altitude, the air mass might be "connected" to the substance (ozone in particular) reservoir in the boundary layer… Those aspects are therefore to be taken into account when setting up a monitoring site and inferring deposition from collected data.

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[14]

http://www.eea.europa.eu/publications/COR0-landcover

[15]

http://bioval.jrc.ec.europa.eu/products/glc2000/glc2000.php

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II.2.4.7   TEMPORAL VARIABILITY

Annual deposition rates are sufficient in order to determine critical load exceedances, whereas for critical level exceedances, short-term information is sometimes needed. However, since there can be substantial variability from year to year with deposition and climatic conditions, a 3 year average deposition for calculation of critical load exceedances may be used.

For some substances (ozone for instance), it is essential to monitor and simulate diurnal concentration variability. A time step may then be a minima of one hour.

II.2.4.8   GEOGRAPHICAL VARIABILITY

Geographical variability is really what mapping describes.

Geographical variability is induced by the spatial variability of climate, soil conditions, topography, land use and so on. These parameters are taken into account in critical loads and levels calculations as input data.

It is the result of the sources localisation as well as climatic, geo-pedologic, topographic conditions.

II.2.5   COMPLEMENTING METHODS TO ASSESS AIR POLLUTION: LONG RANGE MODELLING, NESTED MODELS AND MONITORING

Several methods are available to estimate boundary layer atmospheric concentrations and wet, dry, and cloud-water/fog deposition on different scales of time and space. In the context of long range air pollution, one main characteristic is whether the methods take emission inventories into account or are based on site measurements and monitoring. Two groups of methods can be defined on these characteristics:

- Group A: Long range chemistry-transport modelling

    o Objectives: (1) regional past, present and future situation analysis, (2) basis for scenario analysis, (3) contribution to field processes understanding.
    o Simulations are based on emission inventories.
    o Examples: EMEP, national long range transport models or hemispheric models.

- Group B: Monitoring based methods

    o Objectives: (1) model evaluation and validation, (2) site-specific effects analysis at a very fine scale (1 to 1000 ha), (3) monitoring and analysing trends.
    o Measuring directly concentrations and depositions

on sites: Air pollution representation is independent from inventories but interpolated (by krigging) from field measurements, when the monitoring network is dense enough.
    o Examples: Monitoring networks of air concentration and wet deposition including EMEP and national networks and their interpolation (for instance by kriging).

- Group C: High-resolution models

    o Objectives: (1) simulate air pollution at high spatial resolution (ca. 1 x 1 km² or higher), (2) analysis of local impacts of emissions scenarios.
    o Either based on increasing the griding of LRT models or on extrapolating (kriging) data from dense monitoring network.
    o Simulations are based on local emission inventories and on LRT models for boundary conditions.
    o Exemples: most LRT models have now the ability to run in nested mode, allowing zooming on specific areas, including the EMEP Model (Simpson et al., 2012).

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Nested models are Group A models, in which simulations cover a wide area on a coarse scale and a smaller area (nest) at a finer scale. Thus, they use LRT models for their boundary conditions, while focusing their calculations on smaller areas. EMEP has nowadays the ability to run in nested mode with for instance, its 7 x 7 km² grid. It allows zooming around urban areas (Cuvelier et al., 2013; Simpson et al., 2012).

These three approaches are essential and complementary to provide information to policy makers. Modelling may not be robust without validation via monitoring, while monitoring alone can not be used for scenarios analysis. Modelling and monitoring may be used to map concentrations and deposition. The accuracy of the results will be strongly dependant on the model grid size for the model and on the density of points in the monitoring network.

Well structured monitoring networks are set up and provide a wealth of results within the LRTAP Convention (EMEP, ICP Forests, ICP Waters, ICP Integrated Monitoring, ICP Materials). They provide measurements that are necessary to test, validate and improve steady state model parameterisations. They also provide long term trends that are essential for assessments of policy efficiency and sufficiency as well as dynamic modelling test, validation and improvement. Parties who are developing existing or new monitoring network may improve their data robustness and completion by combining their results to these existing networks.

II.2.5.1   LONG-RANGE CHEMISTRY TRANSPORT MODELS

Long range chemistry transport models are most suitable for scenario analyses and country to country budgets ('blame matrices') used in emission reduction negotiations (if the model domain is more than one country). They calculate patterns of concentration and deposition across large regions of the world. Withing the Convention, the EMEP model is used for the UN-ECE region.

Standard multiannual concentrations from long range chemistry transport models are given as one number per year per component per grid square and deposition fluxes are provided either as average deposition to the grid square or as ecosystem specific deposition estimates.

These model output can be provided for shorter time periods, but with the overall constraint that one of the major inputs, the emission inventory, is often provided only as an annual total.

II.2.5.2   UN-ECEMONITORING BASED METHODS

There are several techniques that may be used to carry out site sampling and monitoring of pollutant concentrations and deposition. It is beyond the scope of this manual to describe them here. The interested reader is referred to EMEP-CCC[16], to ICP Forests[17] and ICP integrated monitoring relevant documentations as well as specific litterature (setting up a network: Anshelm and Gauger, 2001). We will give here only some general aspects, essential in the framework of critical loads calculations.

In the framework of long range air pollution, monitoring stations are set at rural location, avoiding local sources, including farms (for NHy) and roads (for NOx and SOx). However, in order to get a monitoring network representative of the whole range of existing situations, stations may be located at various altitudes and/or at various distances from the emitters of pollutant sources and precursors (Table II.1). Stations at urban locations are not representative for extensive areas but they may be required for differentiating areas with rural and with urban pollution conditions. They are also essential for population and material exposure assessments.

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[16] http://www.nilu.no/projects/ccc/manual/
[17] http://icp-forests.net/page/icp-forests-manual

Table II.1: Minimum distance to emission and contamination sources as recommended by EMEP, 2001. All values are indicative as optimum distances also depends on sources intensities, meteorological and topographic site characteristics.

Type	Minimum distance	Comment
Large pollution sources (towns, power plants, major motorways)	50 km	Depending on prevailing wind directions
Small scale domestic heating with coal, fuel oil or wood	100 m	Only one emission source at minimum distance
Minor roads	100 m	Up to 50 vehicles/day
Main roads	500 m	Up to 500 vehicles/day
Application of manure, stabling of animals.	2 km	Depending on the number of animals and size of fertilized field or pastures
Grazing by domestic animals on fertilized pasture	500 m	Depending on the number of animals and size of fertilized field or pastures

The preferred sampling height is 3-5 m and the monitoring station requires an open aspect without the presence of trees or other tall vegetation in the proximity of the sample intake. The elevation of a particular location determines the extent to which it experiences the influence of air from the free troposphere and from the boundary layer.

Maps may be produced directly from point measurements, but only if the network is dense enough to account for spatial (and temporal) variations. This may be the case for networks measuring air concentrations of compounds with little spatial variation or for measurements of wet deposition in areas of simple terrain. Network (point) measurements should be interpolated using the kriging technique and it may be helpful to include monitoring data from neighbouring countries for interpolation. For some air concentrations, such as ammonia or ozone, or for rain concentrations in complex terrain, the required density of the measurement network could be too dense for practical application. In these cases, it is recomended that concentrations are obtained from less dense networks and that simple models are used to assist the interpolation, e.g. using altitude dependences. It is preferable to interpolate concentrations in rain or in air and then calculate the deposition at the receptor site using local estimates of rainfall and land-use specific ground-level dry deposition rates (see above).

Data assimilation, combining observed and LRT modelled concentrations, is another method applied to provide improved pollutant concentration fields (Flemming, 2003, Rouil et al., 2009).

In areas with clearly delineated watersheds, the "calibrated watershed method" integrates deposition fluxes over a scale compatible to critical load computations, for example for lakes and surface waters. Major fluxes to the groundwater and soil exchange have then to be accounted for. It is most useful for conservative elements (e.g., S, Na, Cl). The data are useful to validate deposition estimates derived from modelling.

Catchment mass balance is an alternative approach, valid at at small (catchment) scale. It combines information from monitoring to local modelling.

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II.2.6 CHARACTERISTICS OF SUBSTANCES MAPS IN RELATION TO EXCEEDANCES CALCULATIONS

Specificities to mapping of substances concentrations and deposition are given below, in the context of the calculation of critical loads and levels exceedances.

Usually the relevant information is provided either from the EMEP model, from small scale modelling or from monitoring network.

II.2.6.1 MAPPING OZONE (O$_3$) CONCENTRATIONS AND DEPOSITION

Data on ozone concentrations may be available from photochemistry/transport modelling or from monitoring networks.

The concentrations of ozone close to terrestrial surfaces (e.g. within 1 m) show a large spatial variability in both rural and urban areas. For urban areas, this variability is mainly caused by the rapid chemical consumption of ozone by NO, which is locally emitted. For rural areas away from local sources, this variability is largely caused by spatial and temporal changes in the degree to which individual sites are vertically 'connected' to the main reservoir of ozone in the boundary layer. Like in urban areas O$_3$ might be consumed by the reaction with NO which can be emitted from bacterial processes in the soil (PORG 1997).

An ozone concentration field at, at least, a grid of 1 x 1 km^2 cell-size is useful to provide a spatial resolution of the ozone exposure on a horizontal scale which reflects the variations in the orography. As the critical levels are based on the concentration measured in the turbulent layer near the receptor, ozone levels modelled or measured at higher distances from the ground are not directly related to the observed effects. The supply of ozone to vegetation is provided by atmospheric turbulence and hence wind speed and the thermal structure of air close to the ground are taken into account. The deposition of ozone on terrestrial surfaces and vegetation causes a vertical gradient of the ozone concentration, which is largely determined by the sink activity of the soil-vegetation system.

II.2.6.2 MAPPING SULPHUR DIOXIDE (SO$_2$) CONCENTRATIONS AND OXIDISED SULPHUR (SO$_X$) DEPOSITION

Data on SO$_2$ gas concentrations, sulphate (SO$_4^{2-}$) aerosol concentrations and SO$_4^{2-}$ concentrations in rain are available from long-range transport modelling, possibly coupled to small-scale modelling.

SO$_2$, in contrast to ozone or sulphate aerosol, is a primary pollutant. It is emitted by both high (e.g. power plants) and low (e.g. households) sources. Therefore the spatial variability of concentrations tends to be higher than that of ozone and sulphate aerosol but lower than that of ammonia. Close to urban areas, the concentrations of rural sulphur dioxide are elevated and this effect should be modelled explicitly where possible, for example by using urban concentration measurements and areas of urbanisation to model the urban effect.

For rural areas away from local sources, spatial variability is largely caused by spatial and temporal changes in the degree to which individual sites are vertically 'connected' to the main reservoir in the boundary layer.

As for ozone, the SO$_2$ levels measured 3-5 m above ground are not directly related to the observed effects, since dry deposition causes a systematic vertical concentration gradient towards the surface, while the critical levels are based on the concentration measured close to the receptor. However, surface-type specific corrections are not generally applied and measured/modelled values usually taken uncorrected.

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II.2.6.3 MAPPING NITROGEN OXIDES (NO_X) CONCENTRATIONS AND DEPOSITION OF OXIDISED NITROGEN (NO_Y)

Data on nitrogen oxides are available from long-range transport modelling, possibly coupled to small-scale modelling or from monitoring networks.

Like SO_2, nitrogen oxides ($NO_x=NO+NO_2$) are emitted by both high (e.g. power plants) and low (e.g. traffic) sources, mostly as NO and, to a lesser extend as NO_2. The spatial variability of NO_x concentrations is mainly controlled by reactions of NO with O_3. It tends to be higher than that of ozone and nitrate but lower than that of ammonia. In rural areas, emission of NO from soils (both agricultural and semi-natural) can likewise contribute to local NO_2 levels.

Many national modelling activities are able to provide estimates of surface concentrations of NO_2 at a high resolution of at least 5×5 km^2 and these can incorporate models to adjust concentrations for local emissions, for example by using distance to major roads.

The reaction products of NO_x in the atmosphere are collectively called NO_y. The main chemical species of NO_y are NO_x, NO_3^-, NO_2^-, HONO and HNO_3. Species of secondary importance for deposition (but essential if atmospheric chemistry is to be modelled) are organic oxidised compounds such as PAN (peroxy acetyl nitrates) and its homologues (peroxy alkyl nitrates, alkyl nitrates), dinitrogen peroxide N_2O_5, the nitrate radical NO_3 (Hertel et al., 2011; Seinfeld and Pandis, 1998)…

Dry deposition modelling includes principally NO_x, NO_3^- aerosol and HNO_3 (ideally also HONO). Reactions with ozone will affect both NO_2 and NO concentrations fields.

II.2.6.4 MAPPING AMMONIA (NH_3) CONCENTRATION, REDUCED NITROGEN (NH_X) DEPOSITION AND TOTAL NITROGEN DEPOSITION

Ammonia is emitted primarily from agricultural sources that may be grouped as (Hertel et al., 2011):

- Point sources, i.e. animal houses and manure storages,
- Application of manure and mineral fertilizers to the fields,
- Grazing animals,
- Other sources including plants.

Emissions from these sources vary with agricultural practices and meteorological conditions, as NH_3 emissions is a process that is highly temperature dependant. Gaseous NH_3 has a short atmospheric residence time (Erisman and Draaijers 1995) and as a result its concentrations in air may show steep horizontal and vertical gradients (Asman et al. 1988). Even in areas not affected by strong local sources, the ambient concentrations of ammonia may vary by a factor of three to four on scales less then a few kilometres.

The very localised pattern of ammonia concentration, and also of ammonia dry deposition, has consequences for mapping procedures. Mapping of ammonia concentrations by interpolation from measurements alone requires extremely high measurement network density and the method is only feasible over small areas.

A long-range transport model with, for example a 50×50 km^2 spatial resolution, will not resolve these large variations either for ammonia concentrations or for the dry deposition of ammonia which will be the major fraction of total reduced nitrogen deposition close to an ammonia source. So assessments of the exceedances of critical loads will be biased when using such LRT models. In the absence of very detailed emission data (on the level of the individual farm), measurements in a dense network are needed to obtain accurate exceedance levels (Asman et al. 1988).

It is also important to note that ammonia may be emitted by as well as deposited onto vegetation, and therefore surface–atmosphere exchange modelling must be used to quantify the net exchange over the landscape. The background developments to allow these processes to be simulated use a compensation point approach (Schjorring et al. 1998; Sutton et al. 2000).

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II.2.6.5 MAPPING TOTAL REACTIVE NITROGEN

Total reactive nitrogen (N_r) released in atmosphere consists in three main parts (NH_x, NO_x and N_2O). Organic compounds form a fourth, poorly quantified, fraction of Nr. N_2O takes part of the atmospheric cycle of nitrogen and is important as a green house gas but not for the critical loads and levels calculations. Organic compounds and N_2O will not be discussed further here.

The deposition of total nitrogen is needed for many applications in the critical load framework. It is defined as the sum of total deposition of reduced (NH_x) nitrogen [NH_3 dry deposition, NH_4^+ aerosol deposition, NH_4^+ wet deposition, NH_4^+ cloudwater/fog deposition] and oxidised (NO_y) nitrogen [NO_2 dry deposition, HNO_3 dry deposition, NO_3^- aerosol deposition, NO_3^- wet deposition, NO_3^- cloudwater/fog deposition]. The methodological considerations concerning NH_x and NO_y depositon mapping apply accordingly.

II.2.6.6 MAPPING BASE CATION AND CHLORIDE DEPOSITION

The deposition of physiologically active basic cations (Bc = Ca + Mg + K[18], i.e. the sum of calcium, magnesium and potassium) counteracts impacts of acid deposition and can improve the nutrient status of ecosystems with respect to eutrophication by nitrogen inputs. Sodium (Na) fluxes are needed for estimating the sea-salt fraction of sulphur, chloride (Cl) and Bc inputs, and as a tracer for canopy and soil budget models. In addition, inputs of Bc as well as sodium and chloride determine the potential acidity of deposition.

Emissions of base cations are from anthropogenic and natural processes (such as rock weathering, sea salts, biomass burning, volcanic dust, industrial emission, vehicle emissions…). Base cations occur in the air in the particulate phase and are deposited in dry and wet processes. In precipitation, they are largely dissolved and occur as ions. In Europe, depositions of base cations are srongly influenced by Saharan dust, especially in countries around the Mediteranean. Sea salt depositions tend to be correlated with the distance to the sea, with highest depositions at western European coastal sites (Torseth et al., 2012; Vet et al., 2013).

As the aim of the Convention is to minimize acid deposition irrespective of other man-made emissions, base cation inputs not linked to emissions of acidifying compounds (for example from emissions of Sahara dust, large-scale wind erosion of basic topsoil particles, etc.) should in principle not be accounted for within the critical loads framework. The non-anthropogenic, non-sea-salt atmospheric input of base cations is defined as a property of the receptor ecosystem and indirectly enters the critical load equation for acidity (see Chapter V, section V.3).

Base cation particle deposition can been estimated from concentrations in wet deposition and empirical scavenging ratios (Eder and Dennis 1990, Draaijers et al. 1995). Dry deposition velocities can be inferred as for SO_4^{2-} aerosol and the obtained dry deposition estimates added to measured and interpolated wet deposition estimates (e.g. Gauger et al. 2003, RGAR 1997, CLAG 1997).

Deposition of base cations have been estimated for Europe, and especially for the Nordic countries based on monitoring data on concentrations of base cations in precipitation and air-borne particles (Draaijers et al., 1997; Hellsten et al., 2007; Van Leeuwen et al., 1996; Van Leeuwen et al., 1995; Van Loon et al., 2005; Vet et al., 2013; Werner et al., 2011, Lövblad et al., 2004).

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[18] Na is not taken up by plant (and therefore not « physiologically active »). It is not included in the Bc sum. However, it is a base cation and included in the critical loads calculations in Chapter V in the BC sum : BC = Ca + Mg + K + Na).

II.2.6.7 MAPPING TOTAL POTENTIAL ACID DEPOSITION

Total Potential Acid Deposition is defined as the sum of total deposition of strong acid anions plus ammonium minus non-sea-salt base cations.

As stated in the preceding subchapter, most chloride inputs are assumed to be of sea-salt origin, and these are removed from the equation by removing all other sea-salt inputs (i.e. of sulphate and base cations including Na) using a "sea-salt correction" with Na as a tracer. The implicit assumption is that sea-salt is neutral and contains no carbonates. Surplus chloride inputs (Cl^*_{dep}) are assumed to be due to anthropogenic HCl emissions.

The sum of critical load (for sulphur) and background (non-anthropogenic) base cation deposition has formerly been defined as critical (sulphur) deposition, as used for the negotiations for the Second Sulphur Protocol (Oslo, 1994). For comparison to CL(S) + CL(N), as defined in Chapter V.3.2 (eq. V.19), only deposition values of S and N are needed. However, if the amount of total acid input is of interest (e.g. for comparison to $CL(Ac_{pot})$, as defined in Chapter V.3.2), non-sea-salt base cation and chloride deposition has to be included into the input side of the potential acidity exceedance equation:

(II.3) $Ac(pot)_{dep} = SO^*_{x\ dep} + NO_{y\ dep}$
              $+ NH_{x\ dep} - BC^*_{dep} + Cl^*_{dep}$

where:

$SO^*_{x,\ dep}$ = non-sea-salt sulphate deposition

$NO_{y\ dep}$, $NH_{x\ dep}$ = total oxidized/reduced nitrogen deposition

BC^*_{dep} , Cl^*_{dep} = non-sea-salt base cation / chloride deposition

In areas strongly affected by sea spray (high sea-salt Na, Cl, S inputs), the "total potential acid" definition of eq. II.3 becomes problematic, since base cations have a beneficial nutrifying effect irrespective of their chemical form (e.g. CaCl vs. $CaCO_3$). At the Grange-over-Sands Workshop 1994 it was concluded that *total* Mg+Ca+K deposition rates should be used for the determination of critical loads for acidity (Sverdrup et al. 1995) (see Chapter 5.3.2).

As stated in Chapter V.3.2, eq. II.3 assumes that deposited NH_x is completely nitrified and exported from the system as NO_3^-, thereby acidifying the system. Thus, with respect to soil acidification it is assumed that 1 mol of SO^*_x is forming 2 moles of H^+, and 1 mol of NO_y, NH_x and Cl^* each 1 mol of H^+.

It is important to be consistent when determining total acid inputs: If results are determined on a site and process level, and if H^+ deposition rates are determined separately, NH_4^+ inputs (max. 2 eqivalents H^+ per mol) have to be distinguished from NH_3 inputs (max. 1 eqivalent H^+ per mol). The same applies to SO_2 (2 eqivalents H^+ per mol) vs. SO_4^{2-} (0 eqivalents H^+ per mol). On a larger scale, this may be neglected: Note that the emission and subsequent deposition of 1 mol SO_2 and 2 mol NH_3 yields the same potential acid deposition as the deposition of 1 mol of their reaction product $(NH_4)_2SO_4$, namely 4 eqivalents.

II.2.7 USE OF DEPOSITION AND CONCENTRATION MAPS

II.2.7.1 ISSUES RELATED TO MAP SCALES

Deposition and concentration maps are designed to be used in combination with critical loads and critical levels maps to show where and by how much critical loads and critical levels are exceeded. The use of deposition data with critical loads data very often involves different scales of the different data sources and, in most cases, the critical loads data are provided at a finer resolution than the deposition data resulting in an underestimation of the critical load exceedance. These issues have been discussed above and improved deposition estimates, for example by using national models at a finer spatial resolution, can improve the quality of the critical load exceedances. One important point re-iterated here is that it is essential to note any different scales in the legends to figures and maps.

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The scale at which critical levels/loads and concentration/deposition are mapped greatly influences the magnitude of exceedance values (Spranger et al. 2001, Bak 2001, Lövblad 1996, Smith et al. 1995). For example, if the average value from a 50 x 50 km^2 grid square is matched to critical loads on the 250 squares of 1 x 1 km^2 within the 50 x 50 km^2 grid square, there will be generally be less critical load exceedance than if the deposition were available at the 1 x 1 km^2 scale. The only circumstance in which this underestimate would not occur would be if the high deposition locations matched the high critical load locations. Over many areas of Europe, exactly the opposite occurs. In many areas of complex terrain the parts of the landscape receiving the largest deposition, such as the higher areas in the mountains of North-West Europe, are also the most sensitive to the effects of deposition, for example acidification. The same holds true for forested areas, which tend to correlate with poor soils in large parts of Europe. This problem is worse for components with local sources (NH_3, NO_x) because the within-grid distribution of sources is not reflected in the grid average estimation from a LRT model, but does markedly increase the within-grid variability of deposition and hence potentially increases critical load exceedances over local sensitive areas. As the current deposition estimates from EMEP are provided at a scale which is much larger than the scale of this spatial variability, depositions are averaged over large areas and "hot spot" are smoothed down. The critical loads exceedances for these areas tend to be underestimated.

These effects are minimised by estimating deposition to the smallest spatial scale possible. They are therefore becoming less acute with the use of finer scale in new chemical transport models (the latest version of EMEP is now based on a 0.1° x 0.1° grid). However, there is an underlying relationship that the critical load exceedances will increase as the spatial resolution of the deposition gets closer to that of the critical loads. Beyond the effects caused by a change in grid resolution, it has been shown that the constant improvement in scientific knowledge can modify air pollution risk assessment results. When comparing assessments of environmental situation, it is therefore important to make sure that different assessments are carried out with similar tools and based on similar hypotheses (Hettelingh et al., 2013).

The comparison of national maps and model outputs with data from the EMEP model is a source of science and policy relevant information. Identification of differences between model outputs may lead to model, parameterization and methodology improvements. These may be of particular significance at borders between countries and may contribute to avoid "border effects". National data may also complete the information provided to policy makers through EMEP. Whether national data confirm or infirm European integrated assessments, they are part of the information national representative may use during protocol negotiations or their implementation. Models and approaches intercomparisons are commonly carried out within the LRTAP groups and subgroups.

II.2.7.2 SOME PRELIMINARY REMARKS REGARDING THE USE OF MODEL RESULTS BY PARTIES

Within UN-ECE countries, the expertise and facilities for measurement of concentrations and fluxes of pollutants is very variable. The extent to which the methods presented in monitoring manuals can be applied is therefore variable. In each country, it is necessary to assess the range of options available. It is important to stress that involvement in measurement alongside modelling activities is highly desirable, as both are complementary.

Besides, the cooperation of all parties to air pollution impact assessment process is necessary to develop satisfactory and efficient strategies for control of pollutant emissions. These require full participation in the underlying science as well as in the political process.

National Focal Centres are strongly advised to ensure that the monitoring and modelling methodologies described in the

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publications listed above are documented in the development or validation of a database for national concentration and deposition (and critical level and load exceedance) maps. Compatibility of these maps with other national maps within the Mapping Programme as well as with monitoring methods employed within other deposition monitoring programmes under the Convention (ICPs on Forests and on Integrated Monitoring; EMEP) is very important.

II.2.7.3   IDENTIFYING ECOSYSTEMS POSITION FOR CRITICAL LOADS CALCULATION AND THEIR EXCEEDANCES

The land use maps used for deposition modelling should be identical to the stock-at-risk maps used for critical levels/loads mapping. Land use maps based on Corine Land Cover in Europe, Global Land Cover 2000 elsewhere are used within the LRTAP Convention (cf. Section II.2.4.6 and cf Chapter V, section V.6.3). Vegetation types are then identified using EUNIS nomenclature. In addition to the geographical position of sensitive ecosystems, land use type/vegetation type, vegetation height and crown coverage are mapped as well on a scale that allows for correct allocation of deposition to all ecosystem types in the model domain.

For all maps, the most recent available data should be used. Should data not be available for a given year, data set filling should not go back in time further than five years (unless, of course, the objective is to prepare time series).

II.2.7.4   UNCERTAINTIES OF MAPPING METHODS

Since the 2004 version of the Mapping Manual, modelling has significantly improved. The EMEP/MSC-W 50 x 50 km^2 Eulerian LRT has been used, tested, compared to other chemistry transport models, calibrated to a greater number of monitoring stations over longer periods. At the time of the update of this Manual, finer grid scales at around 28 x 28 km² (0.5° Long x 0.25° Lat), or finer, are being implemented in the EMEP model.

There are, of course, a number of persistant issues in chemistry transport model uncertainties that lead to uncertainties in critical load exceedances calculations (there are of course also some uncertainties in critical load calculations. They are discussed in relevant chapters and sections). The following points are mentioned in recent literature, including that related to the EMEP Model, as potential sources of uncertainties (Cuvelier et al., 2013; Hertel et al., 2011; Simpson et al., 2012):

- Emission data
- Climate
- Atmospheric processes (and their non-linearities)
- Pollutant dispersion and vertical resolution
- Deposition and its relation to interactions of pollutants with surfaces and vegetation and aerosol size distributions

When evaluating model-measurement intercomparisons, it is important to recall that

a) there are also uncertainties with the measurements,
b) the model may be estimating something rather different from what is being measured.

For instance, the NO_2 concentration at a single site in a (50 x 50 km^2) grid square is only an estimate from a sample of size one of the 'average' NO_2 concentration in the square, which is the value the EMEP/MSC-W model is attempting to match. An evaluation of the overall uncertainty of the model requires that some further information is available on the effects of the spatial distribution of measurement sites.

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II.2.8   WHO DO YOU ASK FOR FURTHER ADVICE?

For questions on ...., please contact :

EMEP long-range transport models:

Erik Berge, The Norwegian Meteorological Institute, P.O. Box 43 - Blindern, N - 0313 Oslo, Norway, Tel. +47 2296 - 3000; Fax. +47 2296 - 3050

High resolution modelling of dry and cloud/fog deposition, combination with maps of interpolated wet deposition:

Jan-Willem Erisman, RIVM-LLO, P.O. Box 1, 3720 BA Bilthoven, The Netherlands, Tel. +31-30-274-2824; Fax. +31-30-2287531

David Fowler, Centre for Ecology and Hydrology, Bush Estate, Penicuik, Midlothian, EH26 0QB, United Kingdom, Tel. +44-131-445-4343; Fax. +44-131-445-3943

Combination of high-resolution models with long-range transport models:

Jan-Willem Erisman, RIVM-LLO, P.O. Box 1, 3720 BA Bilthoven, The Netherlands; Tel. +31-30-274-2824; Fax. +31-30-2287531

Erik Berge, The Norwegian Meteorological Institute, P.O. Box 43 - Blindern, N - 0313 Oslo, Norway, Tel. +47 2296 3000; Fax +47 2296 3050

Measurement and interpolation methodology (Ambient air concentrations, wet and bulk deposition):

EMEP CCC, NILU, Postbox 100, N-2007 Kjeller, Norway, Tel. +47-6389-8000; Fax. +47-6389-8050

Evaluating total deposition maps with throughfall measurements:

Gun Lövblad, Swedish Environmental Research Institute, Box 47086, 40258 Göteborg, Sweden; Tel. +46-31-725 6240, Fax. +46-31- 725 6290, gun.lovblad@ivl.se Jan-Willem Erisman, RIVM-LLO, P.O. Box 1, 3720 BA Bilthoven, The Netherlands; Tel +31-30-274-2824; Fax +31-30-2287531

Diffusive samplers for air pollution monitoring:

Martin Fern, Swedish Environmental Research Institute, Box 47086, 40258 Göteborg, Sweden; Tel. +46-31-725 6224, Fax. +46-31-725 6290 martin.ferm@ivls.se

General information on mapping excercises can also be obtained by contacting the Coordination Center for Effects, CCE, Netherlands: Jean-Paul Hettelingh, Tel. +31-30-74 30 48; Fax. +31-30-74 29 71

General information on modelling:

Ron Smith, Centre for Ecology and Hydrology, Bush Estate, Penicuik, Midlothian EH26 0QB, Tel. +44 131 445 4343; Fax. +44 131 445 3943

General information on NO_x, NO_3:

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Annexes

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