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Confidence
Coping with uncertainty for improved modelling and decision making in nuclear emergencies

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D9.7 – Report on uncertainty reduction in exposure assessment based on environmental monitoring data, including concept for identifying critically exposed groups

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Abstract

Emergency preparedness and response systems for nuclear and radiological emergencies have to deal with decision making in situations with relevant uncertainties. Consistent and appropriate protective measures must be decided before, during and after emergency situations.

This report gives an overview on relevant environmental monitoring data available in European emergency preparedness and response systems. Data from stationary (see sub-report D9.7.1) and mobile monitoring systems (see sub-report D9.7.2) quickly give relevant information about affected areas, the level of contamination and the actual and future exposure. The report regards aspects of potential releases to atmosphere and discuss methods to improve monitoring strategies including stationary monitoring and monitoring by mobile teams.

The focus of the report is the decision making process in the urgent response phase and the early response phase of an emergency. The main objective of the report is to discuss approaches for reducing the uncertainty of exposure assessment for the population in the early phase or emergency response phase of major release scenarios. Approaches for advanced dose assessment tools using monitoring data and concepts for identifying critically exposed groups are introduced and discussed.

Contents

1. Introduction

Emergency preparedness and response systems for nuclear and radiological emergencies have to deal with decision making in situations with relevant uncertainties. Consistent and appropriate protective measures must be decided before, during and after such emergency situations. This requires information about the affected areas, the level of contamination and the actual and future exposure.

Nuclear power plant accidents in Chernobyl (1986) and in Fukushima Daiichi (2011) prove that environmental monitoring data are very helpful to assess exposure to population in affected area. Both accidents were characterized by released radioactive materials with activities above 100 PBq and the duration of the release exceed one week.

In both cases, data from stationary monitoring systems provided only little information about the situation. Extended mobile monitoring programs were needed to provide detailed information on exposure situation. To improve the monitoring capacity, national dose rate monitoring systems were improved after the Chernobyl accident. In addition, the European data exchange platform EURDEP was introduced to trigger exchange of information for emergency situations with transboundary aspects.

Work package WP2 of the European project CONFIDENCE deals with the reduction of uncertainty in dose assessment for the population. This report aims to improve the preparedness and response capabilities in the early phase of a major accidental release situation.

This report gives an overview on relevant environmental monitoring data available in European emergency preparedness and response systems. The report regards aspects of potential releases to atmosphere and discuss methods to improve monitoring strategies including stationary monitoring and monitoring by mobile team. The main objective of the report is to reduce the uncertainty of exposure assessment for the population in the early phase or emergency response phase of major release scenarios.

2. Release scenarios and definition of phases of emergency response

Most countries introduced emergency preparedness and response systems based on different assumptions about types of release scenarios as planning basis. National threat analysis investigations depend on historical events as well as on safety analysis for existing facilities.

In some cases a **pre-release phase** may occur - like in the Fukushima Daiichi accident, where a major release of radioactive material into the atmosphere occurred nearly one day after the initiating event. Decision support systems allow to assess the impact of a potential release especially in such a pre-release phase. Furthermore, such systems allow to assess the potential exposure to the public and to propose adequate protective actions. Most relevant protective actions in this phase are evacuation, preparing sheltering and distribution of stable iodine tables.

The **release phase** of an event starts, when significant amounts of radioactive material are released to the atmosphere (or to water systems like rivers). The release phase ends, when the release has stopped, and the risk of additional further releases is small. Most relevant protective actions in this phase are sheltering and administration of stable iodine tables. In addition, consumption of potential contaminated local produced agricultural products and drinking water should be avoided.

IAEA GSG-11 report denotes pre-release and release phase together as the urgent response phase (1], see Fig. 1).

The **post-release phase** or **early response phase** starts, if the contaminated cloud has passed the region of interest. Most relevant protective actions in this phase are sheltering (for a short period of

time) and / or relocation of people living in area with assessed exposure exceeding a given intervention level. In addition, consumption of contaminated local produced agricultural products and drinking water should be avoided. If there is no need for further emergency response actions, the post-release phase can be terminated and a "**transition phase**" may start (c.f. Work package 4).

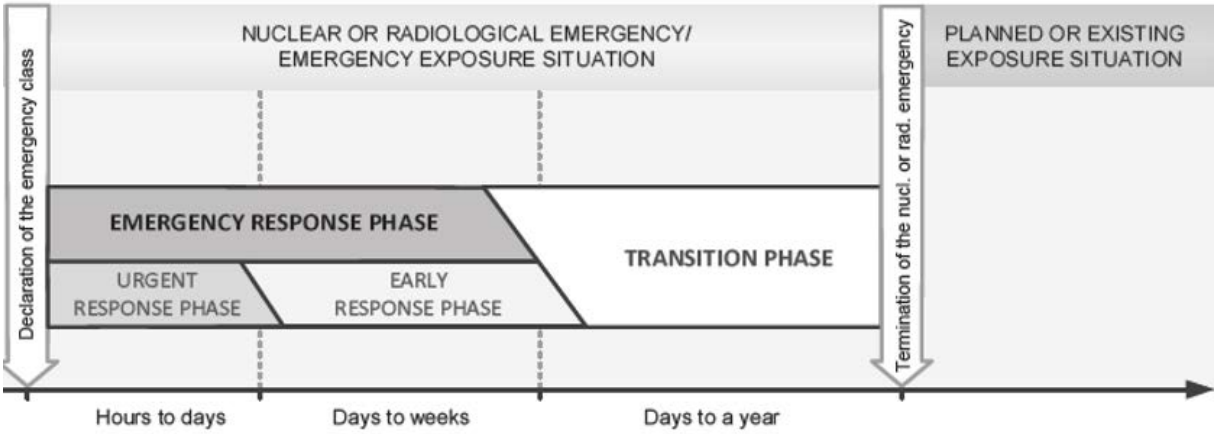

Fig. 1: Phase concept of emergency preparedness and response planning.

To compare, in the Nordic countries – Denmark, Finland, Sweden, Norway and Iceland, the scope of definitions is broader and in places more specific than those existing in the e.g. the IAEA guides. In the Nordic Guidelines [2], phases of a nuclear or radiological emergency are defined as:

1. **The early phase** – the threat and initial phase of a radiation hazard, including the initial events before deterioration of the shielding of a radioactive source or before radioactive material is released into the environment as well as the actual release. The early phase may result in an emergency exposure situation. The early phase ends when the radiation level in the environment no longer increases, and there is no further threat of additional, significant releases.
2. **The intermediate phase** – an emergency exposure situation in which the radiation level is no longer increasing, and no new major radionuclide releases are reasonably anticipated. The intermediate phase can be, for example, the time after which the radioactive plume has passed, and where the main part of the radioactive material is already on the ground and fallout is no longer increasing significantly. The intermediate phase does not necessarily need to be preceded by an early phase. Alternatively, the duration of an early phase could have been very short, e.g. in the case where a contamination is identified after an unintended melting of a radioactive source, or in case of an explosion of a dirty bomb. In the intermediate phase is possible to decide whether to lift, alleviate or change the early phase's protective measures and to introduce new protective measures. In addition, necessary actions to reduce long-term radiation exposure and decrease the amount of radioactive material in the environment should be initiated in this phase. The duration of this phase depends on, among other things, the cause of the radiation hazard. The intermediate phase can last from a few days to a few years.
3. **The recovery phase** – an existing exposure situation where the activities of people and the society are adjusted to the prevailing radiation situation and where the focus is to bring society back to a new normal situation. Recovery typically includes actions made by citizens to reduce their own exposure, based on expert recommendations and advice as well as local and social knowledge. Long-term restrictions regarding the use of land and water areas are given when necessary, or the use of areas and production is redirected. The decontamination of the environment from

radioactive material may be continuing, as may the management of radioactive waste. The duration of the recovery phase can be from months to decades depending on the situation.

The transition between phases will most likely be gradual. Furthermore, actors may be in different phases at the same point in time: for instance, one actor may be in one phase applying measures different to those applied by another actor in the following phase, at another location [2].

Nuclear power plant accidents are characterized by significant releases of noble gases, Iodine and aerosol bound radionuclides like Cs-134 and Cs-137. Most released activity contributes to dose rate and the long term exposure is dominated by external exposure. Typically, internal exposure is dominated by inhalation of iodine isotopes in air. Internal exposure due to ingestion can be effectively reduced, if contaminated water and food is banned from the market. For nuclear power plant accidents, IAEA has proposed three types of emergency planning zones (see Table 2).

Table. 1: Types of release events

Type	Description	Pre-release	Release	Post-release	Historical event
NPP	Nuclear power plant accidents	Hours	Hours, days	Weeks	Chernobyl (1986) Harrisburg (1979)
NF	Nuclear facility accidents		Hours, days	Weeks	Tokai mura (1998) Kyshtym (1957)
SC	Satellite crash	Days	Hours	Weeks	Canada (1989)
DB	Dirty bomb		Hours	Days	
TA	Transport accident		Hours	Days	

Table 2: IAEA proposal for NPP emergency planning zones [3]

Emercency zones	Suggested distance from NPP
Precautionary action zones (PAZ)	3 to 5 km
Urgent protective action planning zones (UPZ)	15 to 25 km
Extended planning distance (EPD)	50 to 100 km

NPP accidents are typically well characterized by gamma emitters in the release. However, release scenarios with dominant beta or alpha emitters and only minor gamma components are possible. For example, the Kyshtym accident in 1957 lead to an explosion of storage facility for liquid radioactive waste [4]. In such cases, dose rate monitoring is less important and emergency workers have to avoid internal contamination. First responders should be able to avoid especially inhalation and skin contamination.

For most European countries, emergency preparedness and response fact sheets give an overview on the national framework for decision making process in case of a radiological emergency [5]. These fact sheets give an overview on relevant nuclear facilities, on planning zones, protection strategies and intervention levels.

3. Environmental monitoring

Systematic environmental radiation monitoring programs were introduced since more than 60 years. In most cases, national environmental monitoring programs were designed together with the planning of research reactors, NPPs and other nuclear facilities. In most European countries, additional monitoring programs were introduced for civil protections topics. Last not least, additional regional monitoring programs of NPP operators are needed. According to basic IAEA Safety Standards (2005), specific objectives of monitoring programs in case of an emergency situation are defined:

- To provide accurate and timely data on the level and degree of hazards, in particular, on the levels of radiation and environmental contamination
- To assist decision makers on the need to take protective actions
- To provide information for the protection of emergency workers
- To provide information for the public on the degree of hazard
- To provide information for the to identify people who warrant long-term medical screening

3.1 Summary of stationary monitoring

Recommended parameters for monitoring programs in the early phase of an emergency situation are given in Table 3.2 in ICRU Report 92 [6]. Data for ambient dose equivalent rate (ADER) are very helpful to assess external exposure and data for radionuclide specific activity concentration are helpful to assess exposure due to inhalation. During accidental releases from NPPs, exposure from external radiation and from inhalation is dominated by noble gas and Iodine radionuclides and contributions from other gamma emitters.

In such emergency situations, data from stationary dose rate monitoring give timely and very useful information about affected area, the level of contamination and the level of exposure to public in the vicinity of probe location. During an accidental release, information from stationary monitoring will reduce uncertainties about the actual release situation and the levels of contamination. Data from stationary air and fallout monitoring will contribute to the helpful information.

In such situations, stationary and mobile dose rate monitoring data can be used assess exposure to the public using dose reconstruction method described in chapter 6.2.

Relevant contributions from alpha and beta emitters without relevant gamma emitters may occur in other release scenarios (see Table 3). In such cases, data from stationary air monitoring have to be combined with data from fallout monitoring and with mobile monitoring with alpha/beta surface detector systems (e.g. Fig 4.16 from ICRU Report 92 [6].

Table 3: Monitoring programs - nuclide types and relevant environmental media

	Air	**Ground**	**Water**	**Food and feed**
NPP accident				
Noble gases	**Release phase***	No	No	No
Iodine	**Release phase***	**Yes**	Yes	Yes
Other gamma emitters	**Release phase***	**Yes**	Yes	Yes
Alpha / Beta emitters	(Release phase*)	(yes)	Yes	Yes
Possible scenarios without relevant gamma emitters: Satellite crash, Dirty bomb, Transport accident				
Gamma emitters	**Release phase***	**Yes**	Yes	yes
Alpha / Beta emitters	**Release phase***	**Yes**	Yes	yes

*including cloud passage

As an example, table 4 gives an overview on stationary monitoring networks in different European countries.

Sub-report D9.7.1 gives an overview on the current status of stationary monitoring in European countries. This sub-report describes proper designed monitoring networks regarding emergency preparedness and response aspects. The report describes relevant sources of uncertainties of stationary monitoring data and gives information for procedures to reduce these uncertainties.

Table 4: Stationary monitoring networks in four European countries

	Area km²	Population	NPP locations /research reactors	Dose rate monitoring stations	Air monitoring stations
Finland	338448	5503000	2 / 1	260	8/9
France**	551695	65167000	19	~ 600	
Germany	357578	83000000	7 / 1	~ 2000	~ 50
Norway	323802	5328212	**0 /** 2	33	7

*** The Finnish research reactor is currently being decommissioned. Air monitoring consists of 8 particulate / gas and 9 total beta monitoring stations.**
****France métropolitaine**

3.2 Stationary dose rate monitoring

Most of European countries have installed dose rate monitoring networks since more than 30 years. For member states of the EU, it is obligatory to provide data from these networks via EURDEP data exchange platform.

During an accidental NPP release situation, data from stationary dose rate monitoring give very helpful information. During the urgent response phase, data from properly designed monitoring networks will indicate the start of a relevant radionuclide release and give helpful information about atmospheric transport and deposition conditions, on affected area and the level of contamination. Uncertainties about actual source term and weather conditions will be reduced.

Very helpful is the timely combination of data from national and regional monitoring networks. Data from spectroscopic dose rate probes in the vicinity will improve the quality of stationary monitoring in case of an emergency situation. Data from such probes will reduce uncertainties about nuclide vector of actual released radionuclides.

Measurement uncertainties of stationary dose rate monitoring systems are in principal understood. Large efforts are made for characterization of different probes and detectors and discussing harmonization aspects [7]. In most cases, measurement uncertainties for data from stationary dose rate probes are less than 20%.

However, additional uncertainties due to disturbing objects in vicinity of probes should be taken into account. A MetroERM deliverable discuss classification of different sites with respect to representativeness of measured data [8]. The proposed site characterisation technique is clearly linked to network harmonisation aspects and to an uncertainty model for ambient dose rate measurements under environmental conditions including disturbing contributions from non-standard probe locations. The proposed technique is adequate for post-release situations, where freshly deposited activity dominates the total dose rate.

3.3 Stationary air and fallout monitoring

Monitoring of airborne radioactivity is carried out routinely and continuously by most European countries. Under normal circumstances the monitoring results consist of observations of cosmogenic and naturally occurring radionuclides, as well as the remains of the Cs-137 fallout from the Chernobyl accident still present in the soil and subsequently in the air due to resuspension. Trace detections of artificial radionuclides are made occasionally with major detection events occurring in 2011 and 2017 with the detections of the radioactivity released from the Fukushima accident, detections of iodine released from what was most likely a radiopharmaceutical production facility and detections of ruthenium from what is conjectured to be related to production of a radionuclide source.

The routine air monitoring programs operate at a time scale that introduces a considerable delay between the passing of a radioactive release and the arrival of the result (see D.2.1.1 for more details). The collection time of a sample is usually one week, further delay is caused by the transport and the measurement of the sample. This delay is alleviated by two approaches:

1. Maintaining the readiness to shorten the sample collection period in an emergency situation.
2. Installation of *on-line* monitoring equipment (spectrometers or dose-rate probes) on the sampler to monitor the sample collection.

In an emergency situation, approach 1 will provide accurate knowledge of the air concentration of the released radionuclides. It will also lead to a greatly increased volume of samples (even an hourly sample change is prescribed), which will, in turn, place demands on the capacity of the measurement laboratories. This approach will not alleviate the delay of sample transport, which can be substantial when considering the furthest reaches of the monitored area. Important considerations are:

- Awareness of the measurement capabilities and capacity in the country / region.
- The capability to deploy a mobile counting laboratory closer to the sampling location.

Approach 2 provides timely early warning of a plume of airborne radioactivity entering the monitored region. The limit of detection is lower than in the spectroscopic monitoring of the environment in conjunction with stationary dose rate monitoring. It is still much higher than the laboratory counting of approach 1. Quantitative analysis of the air concentration of radionuclides is more complicated, and software packages and methods for doing this are not widely available. The use of scintillators and dose rate probes also provides information only on gamma emitters. The dangerous alpha emitters remain much more difficult – or even impossible - to detect.

Observed data for activity concentrations in air help to reduce uncertainties of dose reconstruction methods for inhalation dose. The uncertainties related to the monitoring of airborne radioactivity are sufficiently low to provide an estimate of the dose due to inhaled radioactivity, especially when the sampling time can be shortened as is the case in an emergency situation. The important issue of iodine speciation can be answered by maintaining the capability to sample and measure the gaseous and particulate fractions separately. However, there are only few stationary monitoring locations. For example, German Weather Service (DWD) operates about 40 stations measuring nuclide specific activity on aerosols. (e.g. about 50 locations in Germany). Nuclide specific air monitoring is done in only 8 locations in Finland, and only 6 locations in Norway.

In fallout monitoring the uncertainties can be much higher, especially in the case of dry deposition monitored with passive deposition gauges as used in many places. Theoretically, the deposited activity together with the measurement of air concentration can be used to establish the deposition velocity, and thereby the deposition flux, of radioactivity to the ground. The related uncertainties are large – to the order of 100 %. In order to use a passive fallout gauge to estimate dry deposition it has to be characterized carefully, a demanding task requiring special equipment and facilities. But even then, the dependence on the atmospheric conditions and the task of characterizing the natural surface to which the deposition density is to be estimated remains. Inferential methods relating measured air concentrations to theoretically or semi-empirically derived models of deposition velocity as a function of terrain and atmospheric conditions are preferred in the case of dry deposition. Wet deposition on the other hand can be monitored more accurately. Care must be taken when estimating deposition density resulting from a radioactive plume, especially the precipitation conditions have to be considered.

3.4 Data exchange and harmonization

On behalf of the European Commission, the data exchange platform EURDEP collects and provides ambient dose rate data from all European early warning networks in almost real-time. Data are collected from all member states of the European Union and from additional countries: e.g. Bela-Russia, Russia, Ukraine, Turkey, and Canada.

Since more than 20 years, data exchange of stationary dose rate monitoring is well established. The monitoring networks using a considerable number of different detector types and following different national policies. However, the comparability of these data is crucial for a meaningful interpretation.

This includes correct interpretation of the data under natural background conditions and it is especially important during a nuclear accident with trans-boundary implications. Data assimilation techniques used in decision support systems like RODOS [9] strongly depend on harmonized data. The physical characteristics of the detectors has to be complemented by an appropriate uncertainty budget.

At the moments, exchange of data from stationary air monitoring via EURDEP is poor. In case of an emergency with trans-boundary implications, exchange of these data would be helpful to understand the situation. Therefore, data exchange from stationary air monitoring should be established on the European level. Other relevant monitoring networks are described in sub-report D9.7.1.

4. Prognostic calculations and models (Connection to WP1)

4.1 Uncertainties of prognostic calculations

In the very early (pre-release) phase of accidental release scenarios, decision support systems like RODOS and ARGOS have to rely on prognostic model calculations [9]. Fig. 2 gives an overview of the RODOS model chain and related uncertainty budgets. The impact of a major release scenario mainly depend on the total amount and the nuclide composition of the release. The source term describes the amount of the released material and start, duration and time dependence of released activity concentration for relevant nuclides. In principle, source term estimation is possible from NPP status information using knowledge from licensee and regulatory authority. In most cases, the uncertainty budget of the source term is dominated by the lack of knowledge about actual release and its further development.

For a given source term, meteorological parameters like wind direction and wind speed as a function of time and position are relevant for atmospheric dispersion calculations for the released radionuclides. The output of the model is the nuclide specific air activity concentration and the time integrated air activity concentration. The uncertainty budget for these values is dominated by uncertainties of source term estimation, of the atmospheric transport process modelling and of the underlying meteorological data (see WP1).

Meteorological parameters like amount of local precipitation may be very important to assess activities deposited on ground from modelled air activity concentration. Thus, model based calculation of activities deposited on ground is influenced by the additional uncertainty budget due to the deposition process [10].

Decision support systems like RODOS and ARGOS use output from the discussed dispersion models as basis for dose assessment. Inhalation dose and external dose from cloud shine can be assessed from (time integrated) air activity concentration and assumptions on living conditions during passage of contaminated cloud. External exposure from ground shine can be assessed from activities deposited on ground using assumptions on living conditions and additional knowledge of environmental transfer

processes (e.g. migration into the soil, run-off from paved surfaces) of relevant radionuclides. In addition, ingestion dose can be assessed using knowledge from radioecological models. However, models for the assessment of ingestion dose use simplified assumptions on consumption of relevant food and drinking water, on agricultural production and on transport and trade routes for the produced food- and feedstuffs. In case of an emergency, ingestion dose will be largely influenced by countermeasures on agricultural production and spontaneous impacts on consumer behaviour.

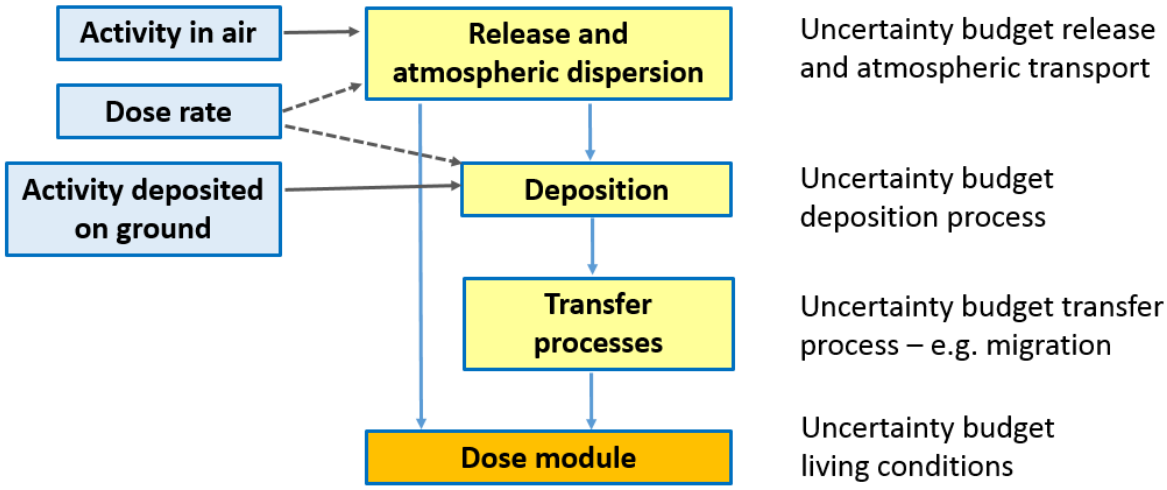

Fig. 2: Overview on RODOS model chain and connected uncertainty budgets and most relevant monitoring data to reduce uncertainties

During emergency response phase, WP2 analysis should focus on the following endpoints for prognostic calculations:

- Population based dose assessment and its uncertainty for populated and affected area from
  - external radiation
  - from inhalation
- Activities deposited on the ground

4.2 Reduction of uncertainties due to environmental monitoring

During release and cloud passage phase, measured air activity concentration can be directly compared with calculated data from atmospheric dispersion models. Measured activities on the ground can be compared with data from atmospheric transport and deposition models. At locations with measured data, the corresponding uncertainties are reduced to the uncertainty budget of measured data.

Figure 2 shows, how data from environmental monitoring can help to reduce the uncertainties of predicted calculations. Measured dose rate from locations with stationary probes helps to reduce the uncertainties of prognostic data. During the release and cloud passage phase, exposure due to external radiation can directly be assessed from time integrated measured dose rate. In addition, measured air activity concentration from stationary monitoring systems give helpful information about release and atmospheric transport process. Data from these monitoring stations may be used to indicate the end of the urgent response phase and nuclide specific air monitoring data improve information on relevant radionuclides.

During the early response phase, measured dose rate can be used to assess activities deposited on the ground using information about the nuclide vector. Spectroscopic dose rate probes may provide

additional information about the nuclide vector and can help to reduce uncertainties of the assessed activities deposited on the ground. In addition, nuclide specific data from fall-out monitoring and data from stationary in situ gamma-spectrometry monitoring support the assessment of radionuclide activities deposited on ground.

5. Mobile measurements during emergency situations

5.1 Summary of mobile monitoring report

During the urgent response phase of an emergency, data from stationary monitoring give very helpful information to improve understanding of the situation and to reduce uncertainties for dose and risk assessment.

In the early response phase, the affected area could be derived from the combination of prognostic data and from stationary monitoring data. Additional measured data from mobile teams are needed in regions, were spatial interpolation techniques of observed data lead to large uncertainties.

This additional information is relevant especially in populated and severely affected area. To improve the data basis for dose assessment for the public, mobile teams with the following equipment are helpful:

- Air-borne dose rate monitoring team – e.g. helicopter based aero-gamma systems for long range surveys
- Car-borne dose rate monitoring teams – e.g. plastic scintillator or NaI scintillator
- Teams equipped with in situ gamma-spectrometry systems – e.g. HPGe detectors – and additional hand held dose rate probes

The planning process for additional measurements by mobile teams should reflect radiation protection issues for the teams, the needs of decision making process and the benefit of additional data for information of the public.

Sub-report D9.7.2 gives an overview on different mobile monitoring systems for radiation detection, their capabilities and applications in emergency response phases as well as discusses challenges related to the measurements' uncertainties. Suggestions and recommendations are given for an optimized monitoring strategy that will allow to reduce uncertainties of mobile measurements and get more accurate monitoring data for prognostic models and assessment of doses to population and the environment in post-release and transition phases.

5.2 Reduction of uncertainties using data from mobile monitoring

Major release scenarios for radiological emergencies typically lead to large gradients of observed dose rate near the location of the release. In addition, large gradients can be expected in case of precipitation events during the atmospheric dispersion of the contaminated cloud. In both cases, spatial interpolation of measured data leads to large uncertainties. This is also true for calculated data derived from measured data – e.g. for assessed exposure for population of affected area using dose reconstruction techniques (see chapter 6.2). Thus, additional mobile monitoring is needed to reduce the corresponding uncertainties.

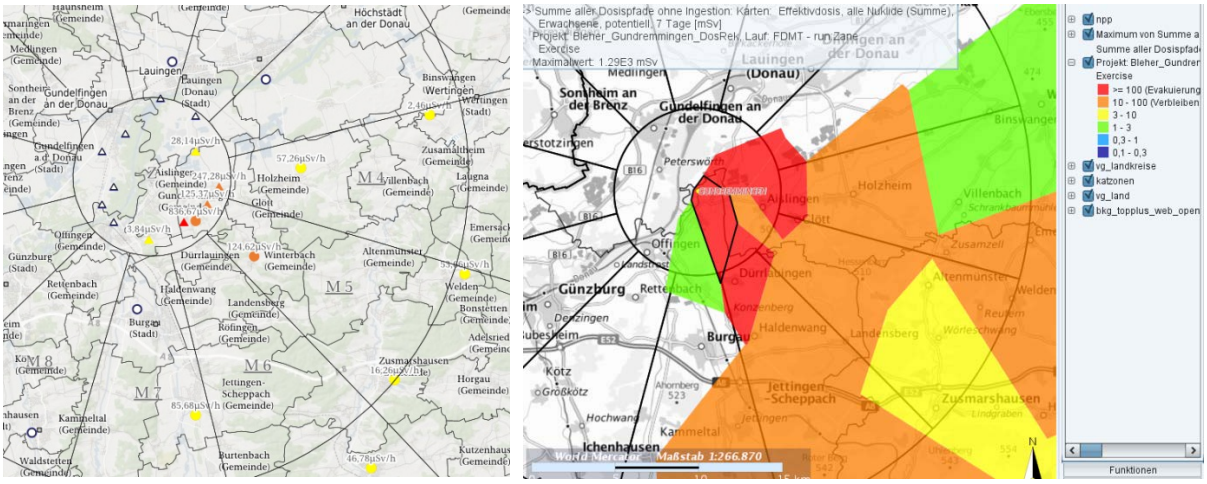

Fig. 3: Example of dose rate data from stationary monitoring (left) and results from dose reconstruction method (see chapter 6.2) using nearest neighbour interpolation technique (right).

Figure 3 shows an example from a German NPP emergency response exercise. For a situation after the urgent response phase, the left side shows simulated data from stationary dose rate monitoring. Planning zones up to a distance of 20 km are indicated. Obviously, spatial interpolation of observed dose rate will lead to large uncertainties. Typically, these uncertainties may be very large near the edges of corresponding Voronoi polygons from nearest neighbor interpolation algorithm.

5.3 Spatial interpolation of measured data and corresponding uncertainties

Very simple spatial interpolation of measured data can use nearest neighbour interpolation. However, this interpolation method lead to stepwise functions at the edges of the Voronoi polygons. The triangulation method for spatial interpolation overcomes this problem and is typically used for topographic surface models.

In principle, uncertainties due to spatial interpolation of measured data in area with large gradients can be assessed using Delaunay triangulation algorithms [11] used for nearest neighbour interpolation and connected Voronoi polygons (see Fig. 4).

Alternatively, geostatistical interpolation methods can intrinsically provide an assessment of uncertainties due to the spatial interpolation and also the uncertainty of the underlying monitoring data.

More advanced automatic working spatial interpolation models for monitoring data were developed and characterised within the EU funded INTAMAP R&D project [12]. During routine mode, spatial interpolation of monitoring data can use either automatic working kriging models (AUTOMAP) [13] or projected sequential Gaussian processes method (PSGP) [14]. However, monitoring data in emergency situations may show large gradients between measured data. For this case, a spatial interpolation algorithm using the copula method (COPULA) was developed [15]. The method can provide data for “best estimates” or predictive mean for spatial interpolation as well as predictions for typical uncertainties (e.g. standard deviation).

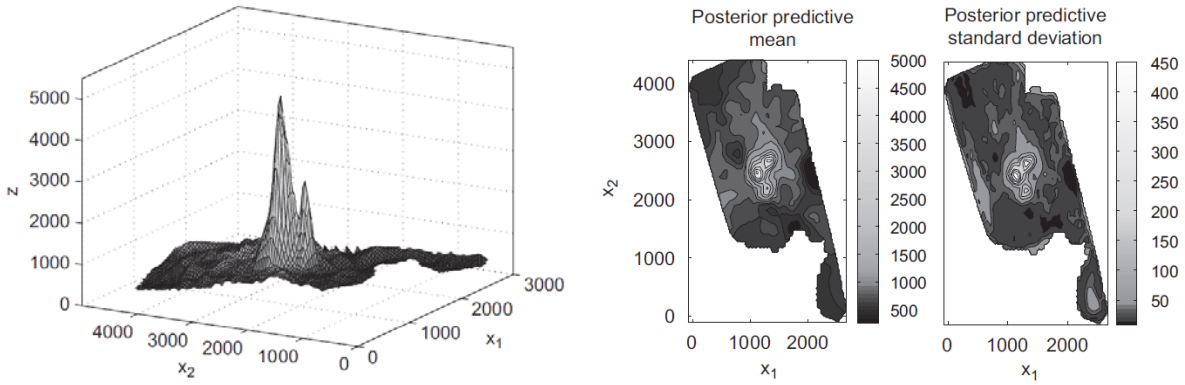

Fig. 3. Surface plot of Helicopter data.

Fig. 7. Predictive (a) mean and (b) standard deviation.

Fig. 4: Three-dimensional result of triangulation method for spatial interpolation of measured data in a hot-spot region (left) and COPULA interpolation results of the data (right, from [15]).

However, decision support systems should avoid more advanced spatial interpolation models. In most emergency situations, data from mobile monitoring should be used in combination with data from stationary monitoring according to figure 5. Simple spatial interpolation should be used to analyse the uncertainty budget of dose assessment methods. High uncertainties due to spatial interpolation can be effectively reduced by additional mobile dose rate monitoring data. Thus, optimization of the planning process for additional measurements by mobile teams should include information about the uncertainties due to spatial interpolation.

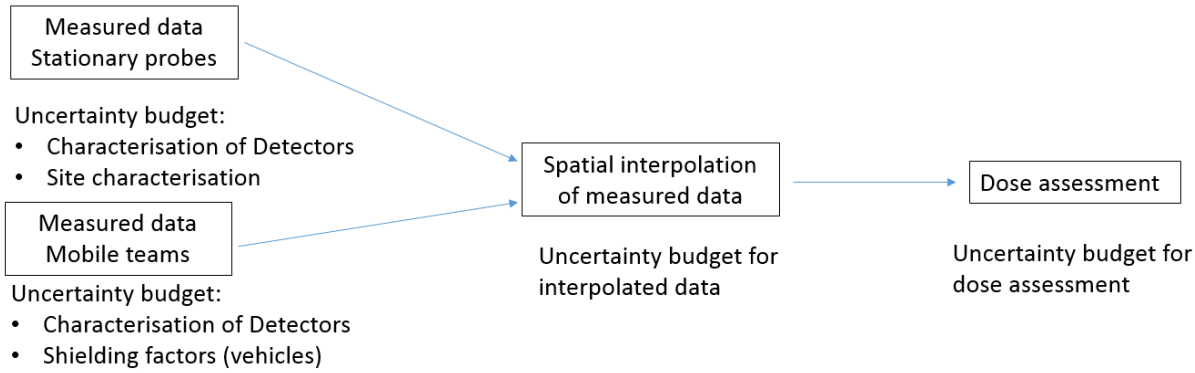

Fig. 5: Uncertainty reduction for dose reconstruction using mobile monitoring and spatial interpolation

During the release and cloud passage phase, data from stationary dose rate monitoring give helpful information on time evolution on exposure in the vicinity of probe locations. In most cases, uncertainty budget for measured data from stationary monitoring are dominated by the impacts of disturbing objects in the vicinity of the probes. These contribution to the uncertainty budget may be analysed using site characterisation procedures [8].

Aspects of mobile monitoring to improve the delineation function were discussed in a MetroERM deliverable [16]. Modern mobile dose rate monitoring techniques described in sub-report D9.7.2 are very helpful to improve the spatial resolution measured information in the affected area. For example, the affected area in figure 3 could be covered within on working day using date from airborne dose rate monitoring. Uncertainty budget of monitoring data have to regard information from type and probe testing procedures as well as information on relevant information on perturbation effects due to probe location in the vehicles or air-crafts (e.g. shielding factors).

6. Methods for dose assessment

6.1 Release phase: Combination of predicted and measured data

During the pre-release phase of an emergency situation, decisions on urgent protective actions for the public fully have to rely on assumptions or predicted information on the potential source term and on predicted information about weather conditions.

During the release phase, data from stationary monitoring give helpful additional information about the beginning of the release and on the time-scale and the amount of released radionuclides. In addition, the comparison of predicted and observed data may indicate deviations of real conditions for atmospheric transport and deposition process. Monitoring data should be used to check decisions on protective actions. If necessary, updated decisions e.g. for sheltering of the public should be made.

For example, modern geographic information systems can be used to overlap maps of predicted dose rate with observed dose rate. In addition, time evolution charts may be used to compare predicted and observed data.

More advanced methods for the combination of model predictions and monitoring data use data assimilation techniques (see Fig. 6).

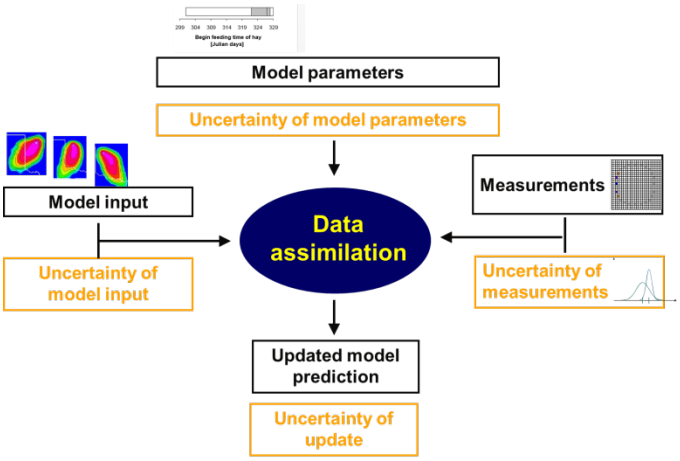

Fig. 6: Uncertainty reduction using data assimilation techniques

Within the RODOS model chain, data assimilation capabilities can be introduced at several modelling steps. In Appendix A the data assimilation is described for the Food Chain and Dose Module of RODOS, FDMT. The main focus here is on updating the total deposition on the ground and on all kinds of plant surfaces based on measurements of gamma dose rates. Furthermore, radionuclide concentrations in different feed- and foodstuffs – as modelled with FDMT – could be updated based on direct measurements of these quantities.

Appendix A contains a description of the underlying deposition model of RODOS, then the model uncertainties and the formulation of the deposition model in the so-called state space are shown. Furthermore, the Ensemble Kalman filter and it's initialisation within the deposition model is briefly described. Finally, an example for data assimilation with the Ensemble Kalman filter within the deposition model is presented. This example demonstrates how data assimilation approaches can be used for reducing the uncertainty of model predictions making best use of monitoring data.

6.2 Early response phase: Dose reconstruction using environmental monitoring data

At the German Federal Office for Radiation Protection, the tool DosRek was developed in recent years before the CONFIDENCE project started. This tool provides the basis for a population-based and for an individual dose assessment, based on environmental monitoring data. It can therefore be used to meet the aims of WP 2.1 in order (1) to generate ground contamination maps from radiological monitoring data, and (2) to assess individual dose histories based on all available radiological monitoring data. In addition, the aim of this tool is to ensure that the EU standards regarding the assessment of doses [17] and thus the determination of the effectiveness of implemented measures and the comparison of doses with existing reference levels are met. The DosRek tool was developed as a module of the German decision support system RODOS (*Real-time Online Decision suppOrt System*) and can be used within RODOS to identify critically exposed groups for subsequent medical surveillance, health long-term monitoring or epidemiological studies. A detailed documentation of the modelling concept and the fields of application of the DosRek tool is available in German [18]. A short description of the main features of the tool and the German dose assessment approach is summarized below.

Modelling concept and calculation of dose values. Fig. 7 shows a rough overview over the concept of the dose reconstruction within the DosRek tool. Radiological measurement data at its respective location and time serves as main input data. This includes gamma dose rate measurements

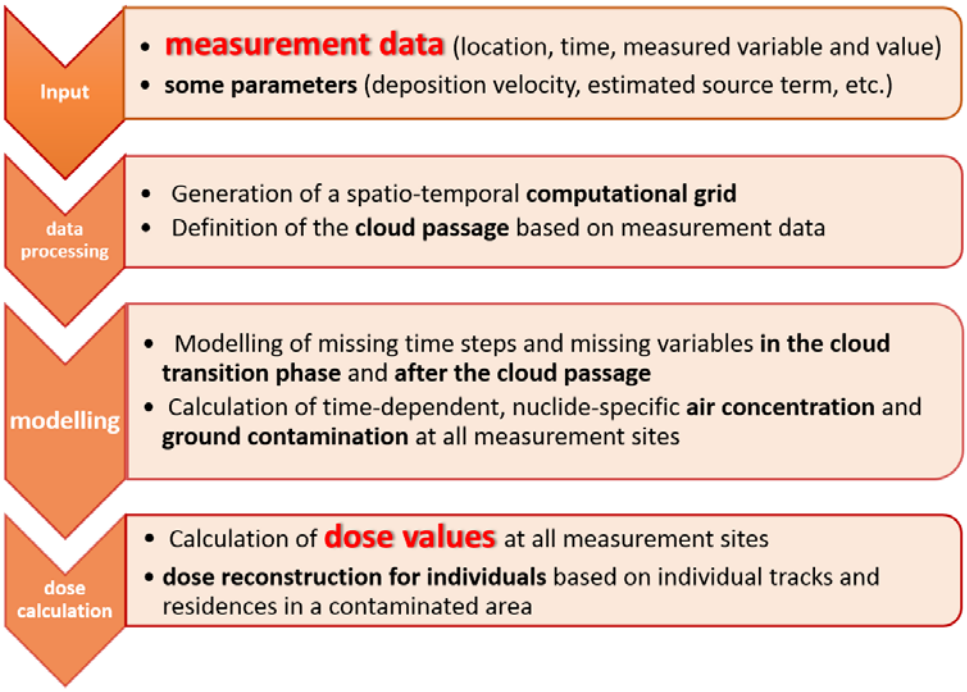

Fig. 7: Overview of the DosRek concept

and nuclide-specific measurements of activity concentration in air and ground contamination, both from stationary measurement devices and mobile measurements (helicopter-based or car-borne). In addition, several supplemental input parameters (such as an estimated source term and its nuclide vector) can be used during the calculation process in special cases if measurement data alone does not provide sufficient information to reconstruct nuclide-specific ground contamination or air activity values. This is used for example for the reconstruction of (often not directly measured) noble gases by

employing an assumed ratio between a measured reference air radionuclide to the noble gas radionuclide of interest.

This input data is then processed internally and sorted onto a spatial and temporal computational grid. This step includes the averaging of available data: The default values are 1 km for the maximum spatial distance and 1 h for the temporal averaging. The spatial grid has an irregular structure with unevenly distributed grid points based on the given input measurement coordinates. Once a specific coordinate is defined as grid point, all other measurement sites that are located within a certain distance (e.g. 1 km) are allocated to the first grid point and all available measurement values are averaged for each time interval.

Each time step of each individual grid point must be assigned to one of the three different cloud passage phases: before, during or after the radioactive cloud passage. This is an important part for the dose reconstruction and strongly influences the dose components from different exposure pathways, as it defines whether the measured gamma dose rate can be assigned only to ground radiation or whether it also contains parts of present air nuclides. This in turn changes the contribution of the different exposure pathways (e.g. inhalation).

In a next step, the modelling of missing measurement values on the basis of available measurement data is carried out. This is done separately for the individual cloud passage phases. Missing values include on the one hand values at missing time steps (e.g. the first hours of an accident, if mobile measurements are deployed later in the course of an accident), but also missing values for nuclide-specific air concentration or ground contamination at locations where only gamma dose rate measurements are available.

The dose calculation itself is then performed in the RODOS- Module FDMT (Terrestrial Food Chain and Dose Module) [19], based on the reconstructed time-dependent values for nuclide-specific ground contamination and activity concentration in air for each time step and each grid point. Doses are calculated for the three exposure pathways inhalation, cloud radiation and ground radiation. Ingestion is neglected as this path is not expected to be relevant during the emergency phase.

Importance of the nuclide vector. The nuclide vector plays an important role for a reliable dose estimation due to the highly variable dose rate factors of different radionuclides. Within the DosRek tool, measured nuclide vectors for ground nuclides and for air nuclides and the measured ratio between air and ground nuclides are transferred in space and time to neighbouring locations in order to provide a best possible estimate of the nuclide distribution in the whole modelling domain. The nuclide vector is crucial for an accurate assessment of doses at the measurement sites. If it is not available (as it is typically the case in the early phase, e.g. in the first hours after the radioactive release has started), the ratio between the released nuclides has to be estimated by standard source terms (see e.g. [20]) and their respective nuclide ratios that are then applied to the measured gamma dose rates. Therefore, the dose assessment is often associated with large uncertainties. For the special case of existing nuclide measurements in the cloud phase, the measured nuclide information is distributed to all measurement points by spatially averaged measured nuclide vectors.

Importance of mobile measurements. For the dose assessment, mobile measurements are essential for the uncertainty reduction both for the determination of the nuclide ratio and for the spatial resolution of the dose maps. If the trajectory of the radioactive cloud does not pass one measurement station that is able to detect nuclide-specific information, mobile in-situ measurements or stationary spectrometric probes are the only possibility to get nuclide information and thus improve the uncertainty of the dose assessment.

In addition, mobile measurements are essential for improving the spatial resolution of the dose maps for the contaminated area. In Germany, stationary gamma dose rate probes are typically 15 km apart from each other, which strongly enhances the uncertainty for the dose assessment on the small scale. Within the DosRek -tool, a maximum distance of 1 km between two measurement sites is recommended. This can only be achieved by mobile car- or helicopter-based measurements.

Spatial interpolation of values. After the previously described modelling procedure, dose information is only available at the grid points themselves, i.e. at the measurement sites. Around these locations, doses are initially not known because of the lack of area-covering measurements. Consequently, a spatial interpolation is needed in order to obtain dose values for the whole domain of interest. In the DosRek-tool, the **Voronoi-approach** (see e.g. [21]) is used for the spatial interpolation. Voronois are polygons that are formed based on available measurement data in the manner that each point of the area obtains the measurement data of the nearest measurement site. Consequently, within a Voronoi-polygon, the same measurement data is valid. An example of a ground contamination map based on the Voronoi-approach for the radionuclide I-131 after a virtual accident in the German nuclear power plant Gundremmingen is shown in Figure 8. The simulated release was calculated on July 6, 2018 with real weather conditions with the INES-7-rated source term FKA (for detailed information on this source term, please see [20]). Based on that release, simulated gamma dose rate measurements for all German operational measurement sites were created. This simulated

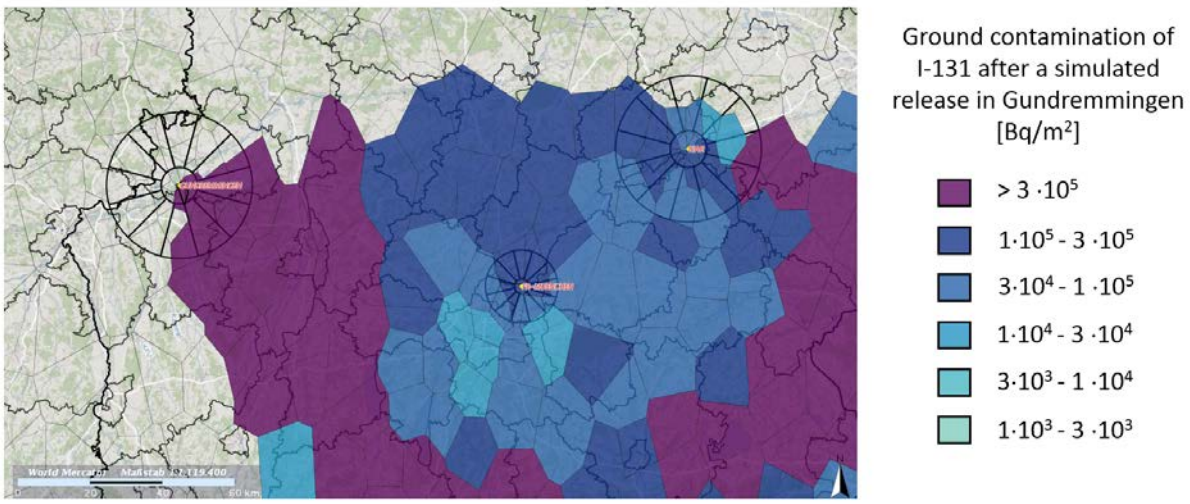

Fig. 8: Example of a Voronoi ground contamination map of the radionuclide I-131 after a simulated release in the German nuclear power plant Gundremmingen on July 6, 2018 with real weather conditions. Basis for the DosRek calculation were gamma dose rate measurements that were simulated throughout the release for all operational German gamma dose rate probes

measurement data formed the input for the shown ground contamination map after the radioactive cloud has left the shown area.

Dose maps for situation reports. The DosRek-Tool calculates doses based on environmental monitoring data and spatially interpolates the values on a Voronoi-based approach. With this tool, dose maps can be created that visualize areas where protective measures should be implemented from a radiological point of view. These dose maps based on environmental monitoring data serve as a situation assessment in German situation reports in order to identify areas of critically exposed population groups by means of a comparison of doses to the predefined intervention levels. Dose maps based on environmental measurements can be used as soon as sufficient measurement data is available for a first dose assessment. This is typically the case when the nuclide vector is determined either by a stationary air monitoring probe that is hit by the radioactive cloud, or by mobile in-situ measurements at the periphery of the contaminated area. If a measurement of the nuclide vector is available, this information can be spread in space and time, so that the uncertainty of the dose assessment is reduced significantly.

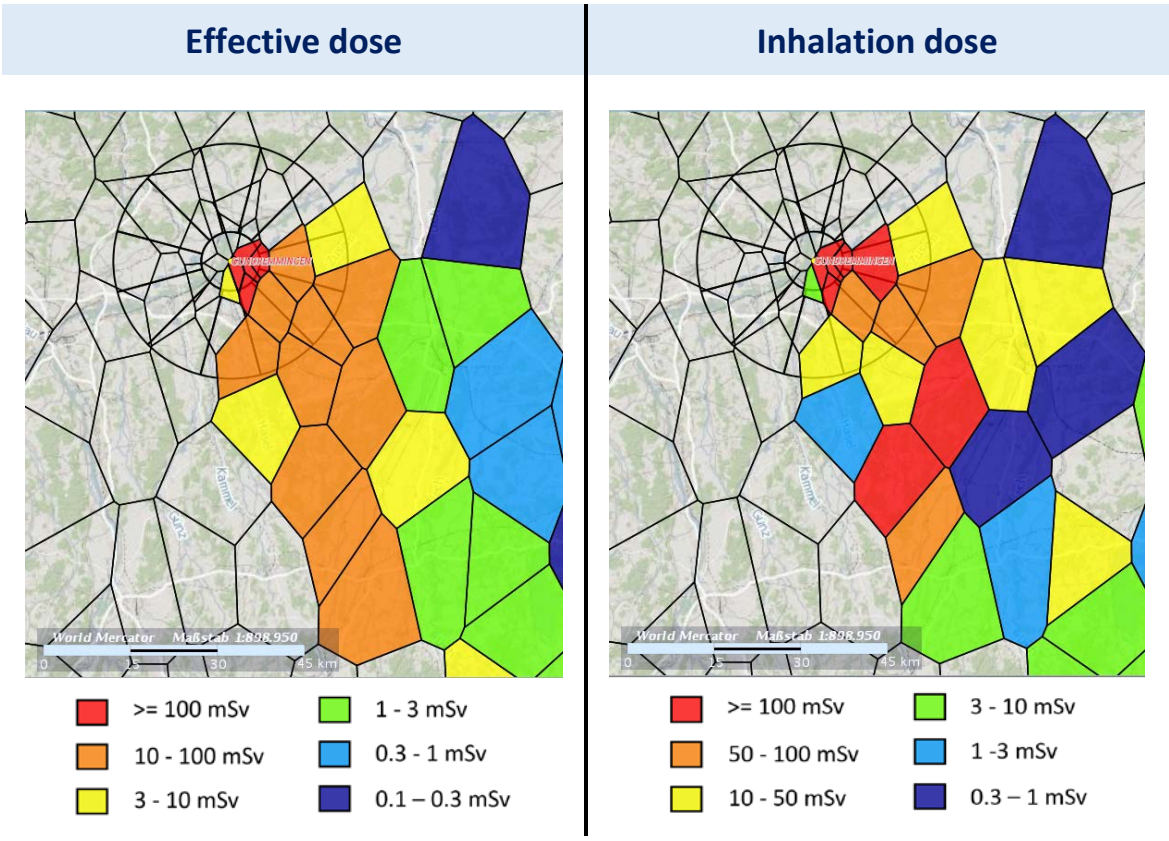

Fig. 9: (left) Effective potential 7-day dose for children for the exposure pathways inhalation, cloud shine and ground shine and (right) inhalation dose for children based on a DosRek calculation for the same scenario than shown in Fig. 8. The corresponding dose intervention levels for the left graphic are 100 mSv for evacuation and 10 mSv for sheltering; for the right figure the intervention level is 50 mSv for children [cf. Table 1]. Please note the different color coding for the two figures.

Figure 9 shows two examples of dose maps based on environmental monitoring data for the same scenario that was shown in Fig. 8, i.e. an INES-7-rated simulated release in the German nuclear power plant Gundremmingen on July 6, 2018. Typically, dose maps that are created with the DosRek-Tool for

German situation reports display values that can be compared directly to the German intervention levels (cf. Table 1). On the left part of Fig. 9, the effective 7-day potential dose for children for all exposure pathways without ingestion is shown. The red Voronoi-polygons indicate areas where 100 mSv are exceeded and therefore an evacuation must be considered. The orange Voronois indicate that the intervention level for sheltering is exceeded.

The right part of Fig. 9 shows the inhalation dose that occurs during the radioactive cloud passage due to the inhalation of air radionuclides (including long-term dose commitment). This relates mainly to the effect of iodine isotopes that accumulate in the thyroid and can cause health effects like thyroid cancer or other thyroid diseases. In addition to the shown two example dose maps, there are additional versions of different dose maps available, e.g. for a comparison of the 1-year dose under normal living conditions to the corresponding reference value, or for areas where food restrictions should be considered.

6.3 Individual dose assessment

Dose assessment maps from the DosRek -tool that are presented beforehand also serve as a basis for an individual dose reconstruction for a person-based assessment of the individual radiation exposure. For that purpose, the time-resolved dose information of the dose maps is combined with personal information of individual movement profiles in order to get a person-specific total dose. The basic

Fig. 10: Schematic illustration of the dose calculation for an individual movement profile (blue line) within a contaminated area. Voronois with increased dose values are indicated with different colors. The person moves from "start" to "end" partly through the contaminated area (colored Voronois) and stays during its journey in each Voronoi for a certain time. The entrance and exit points of Voronois that have increased dose values are indicated with a white star (☆). The duration times of the person in each Voronoi are then combined with the corresponding dose values that are valid for the respective time.

principle of this process is shown in Fig. 10 as a schematic overview to illustrate the underlying concept. A person moves from one location ("start") to another location ("end") and thereby passes through the contaminated area with increased dose values, which is indicated by colored Voronois. Voronois without contamination have been left blank. The Voronoi map shown in Fig. 10 is serves as an example for one time interval, but is internally available in a one hour interval. These time-resolved Voronoi maps serve background layers for the individual dose calculation. Therefore, the movement profile of the person is combined with the respective time-dependent background dose layer. The time the person spends within the area of a single Voronoi-polygon is merged with the corresponding dose value at the respective time step. This is done for all waypoints in all Voronois of the movement profile, and the sum of all segments then results in the total dose for the entire route. In order to refine the dose assessment, each person has to specify for each time step for the whole movement profile important factors that influence the radiation exposure. This includes information about the whereabouts (staying inside in a house or in a cellar, or staying outdoors or in a car) and about any protective measure that was taken at a specific time (e.g. wearing of a protective mask or the intake of an iodine tablet).

7. Concept for identifying critically exposed groups

National emergency preparedness and response fact sheets give an overview on different national frameworks for the decision making process in case of a radiological emergency [5]. These fact sheets give an overview on relevant nuclear facilities, on planning zones, protection strategies and intervention levels. For example, different intervention levels for sheltering of the people are used in different countries:

Table 5: National criteria for urgent countermeasures to protect public: Intervention levels (IL) for exposure and operational intervention levels (OIL) for dose rate.

	IL Sheltering	OIL for sheltering	Evacuation
Finland	10 mSv	100 µSv/h	Sheltering needed > 2 days
France	10 mSv		50 mSv
Germany	10 mSv	(100 µSv/h)	100 mSv
Norway	10 mSv	100 µSv/h	Sheltering needed > 2 days

This chapter discuss different OIL based and IL based criteria to identify critically exposed groups. Individual dose assessment discussed in chapter 6.3 should be applied for people from identified groups using criterion 1 or 2. Persons from identified groups using criterion 3 should be checked using methods of individual dose monitoring. Some methods were proposed and/or investigated within CONFIDENCE task 2. For example, the assessment of external exposure of individuals may be compared with the results from dose reconstruction methods described in Deliverables D9.8 and D9.10 or with results from biological dose assessment (see D 9.11). In addition, internal dose assessment for individuals should be compared with results from individual internal dosimetry (see D9.9).

7.1 Operational intervention levels

IAEA proposed a concept of operational intervention levels (OIL) based on measured dose rate data. Thus, IAEA proposed OIL 1 (1000 µSv/h) as an indicator to identify regions where evacuation or relocation of population should be taken into account. In addition, OIL 2 (100 µSv/h) was proposed as an indicator to restrict outdoor activities for population in the early phase of an accident.

The following criterion 1 is based on this OIL approach and may be used in situations, where output from decision support systems are very uncertain or not reliable. In this case, decisions on early countermeasures may be maid based on criterion 1.

Criterion 1: Population living in regions, where measured or interpolated dose rate exceeds operational intervention levels (OIL 1 = 1000 µSv/h or OIL 2 = 100 µSv/h)

In principle, this criterion may be adopted to different national rules for relevant operational intervention levels. Similar criterions could be derived for regions, where air activity concentrations or exceed corresponding OILs. For example, Norwegian regulation defines two OIL values for total activity deposited on ground:

- 1000 kBq/m^2 for gamma and beta emitters
- 10 kBq/m^2 for alpha emitters

7.2 Dose intervention levels

In addition to the application of operational intervention levels as described in the previous chapter, protective measures can be recommended based on the application of radiological criteria such as dose intervention levels. The most important **German intervention levels for emergency measures** (evacuation, sheltering, intake of iodine tablets) and for relocation are listed in Table 2 from [22].

In the early phase of a major release scenario, the most relevant output of decision support systems is information on potential doses to the population in the affected area. Thus, the focus during this time shall be on population-based dose assessment from external radiation from cloudshine, groundshine and from inhalation exposure. In addition, the uncertainty of the doses must be assessed. Critical for countermeasures (e.g. temporary evacuation of a populated area) are information about areas where relevant intervention levels are or may be exceeded. Decision support systems should therefore visualise areas where the local exposure exceeds the predefined intervention levels and ideally, integrate the uncertainty budget in the map, e.g. by using percentiles to show areas where the local dose exceeds relevant intervention levels with a certain probability. In addition, the system should provide data for nuclide-specific dose rate or activity concentrations deposited on the ground for relevant nuclides, e.g. for I-131 and Cs-137. These ground contamination maps should use information from monitoring data and appropriate spatial interpolation algorithms. Assessed resulting uncertainties may trigger additional mobile monitoring for the affected area. For populated and affected areas, this information should be visualised on a 1 km x 1 km grid.

In addition to the criterion 1 referring to operational intervention levels, there are two criteria concerning dose intervention levels to identify critically exposed population groups.

Criterion 2: Population living in regions, where assessed dose exceeds dose intervention levels (see Table 1)

This second criterion is applied to areas, where a certain dose intervention level is exceeded without taking into account the normal living conditions of the affected population, i.e. a person stays outdoors

at one spot 24 hours a day without any protective measures. This criterion is necessary to identify areas where emergency measures or measures for radiation protection should be recommended, i.e. evacuation, relocation or also decontamination measures etc.

Norwegian Intervention levels for emergency preparedness are based on the Nordic guidelines and recommendations 'Protective Measures in Early and Intermediate Phases of Nuclear or Radiological Emergency' given by Nordic Radiation Protection and Nuclear safety Authorities in 2014. As a non-nuclear power plant country, Norway follows this recommendation in radiological and nuclear emergency management.

The overall aim for planning for emergency response is that the annual residual radiation dose should not exceed 20 mSv. The total reference level includes the total residual dose expected as interventions in both the early and the intermediate phase (1).

The third criterion that can be applied to identify critically exposed groups is linked to the assessment of the individual dose / individual risk of single persons:

Criterion 3: People with individually assessed dose exceeding relevant dose levels

This assessment must be based on individual information of the affected persons, i.e. individual movement profiles, applied protective measures (e.g. the intake of iodine tablets) and information on indoor and outdoor times. Possible actions that are connected to that individual dose assessment include medical treatment or an early thyroid dose screening.

The EURATOM guideline 2013/59 Annex XI requires that the following points must be addressed during emergency response [17]:

- Assessing the effectiveness of strategies and implemented actions and adjusting them as appropriate to the prevailing situation
- Comparing the doses against the applicable reference level, focusing on those groups whose doses exceed the reference level

In order to meet the requirements stipulated by the EU and the three defined criteria above, an operational procedure is needed to reconstruct doses induced by radiation exposure of the affected population in order to identify critically exposed groups, to evaluate the efficiency of protective measures and to compare the reconstructed doses with operative intervention levels.

The identification of critically exposed groups based on an area-covering population-based dose assessment and an individual, person-based dose assessment as described in the above-defined criteria can be used to **pre-select critically exposed persons** and thus provide a filter function for the expectedly large number of potentially contaminated persons that have to be dealt with after a major nuclear accident. The pre-selected persons should then be overtaken for subsequent medical surveillance and health long-term monitoring.

Generally the German dose assessment approach (DosRek tool) that was presented in the previous chapter "6.2 Early response phase: Dose reconstruction using environmental monitoring data" shows

one example how to deal with these issues and can provide the basis to meet the above mentioned requirements.

7.3 Uncertainty management for identifying critically exposed groups

The concept of identifying critical exposed groups using the OIL approach (chapter 7.1) is straight forward. Observed dose rate data from stationary monitoring have two main sources of uncertainties: Uncertainties due to probe characteristics and to real probe location (see chapter 3.2). Data from mobile monitoring have the following sources of uncertainties: Uncertainties due to probe characteristics and to probe location and corresponding uncertainties (see sub-report D9.7.2).

Dose rate monitoring data from mobile teams improve the spatial resolution and reduce uncertainties due to spatial interpolation algorithms. In the very early response phase, mobile monitoring should

Table 6: List of relevant uncertainty aspects that should be included into a holistic dose assessment approach

	description of uncertainty components for dose calculation	idea of integration
Generic elements for the dose uncertainty management	**measurement** uncertainties	sampling of measurement values from a given uncertainty distribution using a Monte Carlo approach
	uncertainty of **dose calculation parameters**	sampling of values for all relevant parameters used during the dose calculation process using a Monte Carlo approach
	uncertainties due to the **spatial resolution** of measurement sites	additional mobile measurements by car and by helicopter to reduce the measurement gaps between stationary measurement sites
	uncertainties due to **spatial interpolation**	geostatistic approaches (Kriging)
Additional uncertainty elements for the individual dose assessment	**time** uncertainties of the movement profile	sampling of exact hours of exposure in one movement profile
	location uncertainties of the movement profile	sampling of exact coordinates of a person at each waypoint of its movement profile
	location factor (=reduction of doses due to shielding of buildings etc.)	sampling of shielding efficiency of the buildings within a given uncertainty distribution

focus on populated affected area (see table 1 in sub-report D9.7.2).

An uncertainty management system for the dose assessment and application of criterion 2 and 3 is proposed. In order to account for uncertainties within the dose assessment, a **holistic approach** is recommended for the dose uncertainty management. Table 6 comprises a list of relevant uncertainty information that should be integrated in a holistic dose assessment. This includes on the one hand generic uncertainty aspects that need to be considered in population-based, area-covering dose assessment, and on the other hand additional uncertainty issues that have to be taken into to consideration when assessing an individual person-based dose. In order to account for the first two aspects listed in Table 6, the DosRek-tool that was presented beforehand could be extended in a manner shown in Figure 11. As already described, the input data for the DosRek -tool is environmental monitoring data, i.e. stationary or mobile measurements of gamma dose rate and nuclide-specific air activity and ground contamination. If this measurement data is provided with an **uncertainty budget for each individual measurement**, a Monte Carlo approach could be used to integrate these measurement uncertainties into the dose calculation. Therefore, the DosRek calculation is carried out a variable number of times (e.g. 20 times, as suggested in Fig. 11), and for each single calculation, a sampling of measurement values from a given uncertainty distribution is performed. Additionally, the Monte Carlo approach is also applied for the **sampling of dose model parameters** such as dose factors in each of the runs.

This may also include a sampling of the estimated source term with nuclide ratio information that is used in case no measured nuclide information is available. The output of this uncertainty calculation is a perturbed ensemble of Voronoi dose maps, with each of them having slightly varying input measurement data and dose calculation parameters. This ensemble can be used to create "threshold

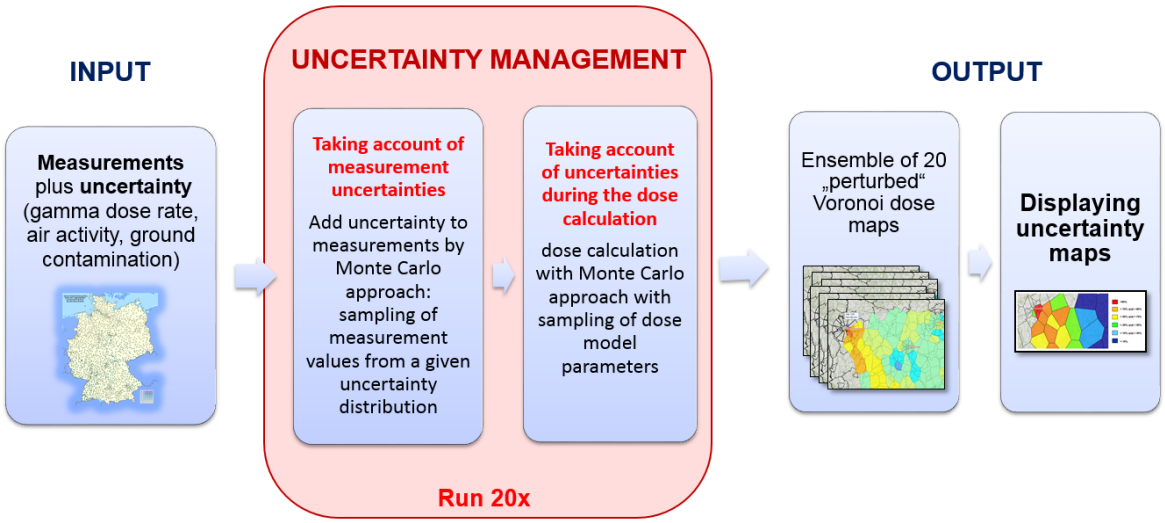

Fig 11: Procedure how to integrate measurement uncertainties and dose calculation uncertainties into the dose assessment model chain

maps", i.e. Voronoi-based maps that show the probability to exceed a certain threshold or intervention level within one Voronoi-polygon (e.g. exceeding 100 mSv effective dose the intervention level for evacuation). Different colour coding can be used to indicate the likelihood of exceeding a certain reference level in a way that e.g. 90% of all ensemble members show an effective dose greater than 100 mSv.

In addition, the **spatial resolution** of the background layer that is used for dose calculation provides a large source of uncertainty, which is a direct effect of the distance between measurement stations. Therefore, data from mobile monitoring by car or by helicopter in addition to data from stationary probes are very essential to reduce uncertainties from spatial interpolation algorithms (see chapter 5.3). In addition, data from mobile monitoring will also reduce the risk to miss highly contaminated hot spots.

The **spatial interpolation** of values between the measurement sites can be done by deterministic or geostatistic interpolation approaches. Within the DosRek-tool, the deterministic Voronoi-approach is implemented (cf. previous chapter "Methods for dose assessment"). However, deterministic methods typically do not include the assessment of uncertainties due to spatial interpolation. On the other hand, geo-statistic spatial interpolation methods using kriging methods or copula-based methods are able to assess corresponding uncertainties (see chapter 5 discussion and [23] and [24]).

One additional aspect that should be addressed when regarding the uncertainty of individual dose assessment is the uncertainty management of the movement profile, in particular **uncertainties in the location** (waypoint coordinates) **and in time** (exact time steps of waypoints, duration of driving routes etc.). Similar to the approach that was suggested for dealing with measurement uncertainties, a Monte Carlo approach can be applied here as well. Therefore, the time stamps and the coordinates of the movement profile are sampled within a given uncertainty range. By repeating the dose calculation for different perturbed movement profiles, an ensemble of individual dose values is created that can be used to assess the uncertainty. For example, deviations of the real route from the assumed route in Fig. 10 can be simulated by slightly changed waypoints within a certain radius around the assumed locations.

Last not least, the **location factor** of the individual whereabouts has to be taken into account. The location factor describes the reduction of doses of the different exposure pathways when staying inside due to the shielding of the surrounding building structures. The shielding efficiencies of different construction materials can vary considerably [25]. A Monte Carlo sampling of the location factor by using the most common building materials in the area of interest allows for taking into account uncertainties that are connected to shielding efficiencies.

8. Monitoring strategy in transition phase

In the transition (or recovery) phase, radiological situations on-site and off-site are better understood, and can be improved more effectively compared with the initial phase of the accident. The focus in this phase is to bring society back to a new normal situation.

In the post-release phase, the external gamma dose rate measurements are mainly attributable to radiation from deposition on the ground. The application of relocatable probes would allow to perform larger scale measurements, reduce area between probes in the network, get a better area coverage and more precise data on radioactivity levels at the relevant location as well as to reduce uncertainty of measurements within the measured area. This, will in turn allow for more precise and better dose assessment for population and workers (fire fighters, etc.). The aero-gamma data can be influenced by low air activity concentrations even in post-release phase. Thus, it is recommended to assess aero-gamma data in combination with ground-based monitoring data (when possible).

In addition to environmental monitoring of ambient dose rates, measurements of radionuclide concentrations (particularly caesium and iodine) in air should be made. This type of information enables the estimation of internal exposure due to the inhalation of radioactivity. Concerns regarding

internal and external exposures arising from deposited radioactive material in the environment require plans to measure soil surface concentrations as input to decisions on the implementation of both food and water restriction and extended protective actions (e.g. temporary relocation).

In general, the pattern of deposition is dependent on the magnitude of the event and on the prevailing meteorological conditions at the time of the release, particularly wind direction and any rainfall, occurring during passage of the plume. In the longer term, rainfall and weathering cause redistribution of radionuclides in the soil and their further migration. Plant uptake of radionuclides from soil varies according to the physical and chemical characteristics of the soil (e.g. moisture and fertility), and generally decreases with time. The levels of deposition may also vary greatly from one area to another. As an example, after the Chernobyl accident, surface contamination varied by factors of up to 10–100 within the same village. Usually, in the longer term, one or a few radionuclides will dominate as the principal contributors to both human and biota exposure [27].

The monitoring of soil, food, and water is likely to continue beyond the intermediate phase and into the long-term phase. In the intermediate phase, detailed environmental monitoring is essential for understanding the radiological situation of widespread contaminated areas, and for terminating the urgent protective actions implemented during the early phase. As radioactive releases are brought to a halt and more detailed monitoring becomes possible in affected areas, the availability of environmental measurement data increases. Experience from past accidents indicates that there is a possibility of radiation exposure from aquatic pathways due to the release of liquid radioactive material to the sea or surface waters, deposition of radioactive material directly onto the sea or surface waters, and from run-off into the sea or surface waters. For direct or indirect releases of radioactive material into the sea, people can be exposed externally from radionuclides in the sea or sea sediments. The doses from these pathways are not expected to make significant contributions to the overall exposure. Among them, the transfer of radioactive material into seafood should be considered as a possible primary source of internal exposure to the public [28].

Continuation of radiological characterization in affected areas should be complemented by the establishment of a system for monitoring the external and internal exposure of individuals. For the authorities, the monitoring system in the recovery process will help to fulfil several objectives: to obtain data on the actual contamination of affected areas and its evolution; to control the concentration of radionuclides in foodstuffs; and to provide information to the public on external ambient dose rates by using devices displaying the results in different places.

The control of ingestion pathways is an important component of the protection strategy for the public. Experience shows that maintaining radiological monitoring of foodstuffs in the long-term phase is useful to gradually restore the confidence of distributors and consumers inside and outside affected areas [29, 30].

The duration of the recovery phase can be from months to decades depending on the situation. The transition between phases will most likely be gradual (see details in Chapter 2 of this report). For other important aspects in monitoring strategy in the transition phase, see sub-report D 9.7.2, Chapter 6.

Appendix: Data assimilation

Within the RODOS model chain, data assimilation capabilities can be introduced at several modelling steps. In this section the data assimilation is described for the Food Chain and Dose Module of RODOS,

FDMT. The main focus here is on updating the total deposition on the ground and on all kinds of plant surfaces based on measurements of gamma dose rates ("data assimilation for the deposition modelling"). Furthermore, radionuclide concentrations in different feed- and foodstuffs – as modelled with FDMT – could be updated based on direct measurements of these quantities ("data assimilation for the food chain modelling").

The objective for data assimilation in general is to optimally estimate the state of a dynamic system from various data. This should be possible even if the state itself cannot be observed directly, the measured data contains noise, and the modelling of the system dynamics may be imperfect. Data assimilation in RODOS is based on the use of the Kalman Filter (KF) which is a recursive, linear, minimum mean-squared error estimator. The KF estimates a particular state of a system by linearly combining a prediction of the state with a set of measurements. The Kalman Filter was developed about 1960 [31] and has found widespread application since then, especially within atmospheric science [32], meteorology and oceanography [33]. For the use in a real-time system like RODOS cost-effective Kalman filter procedures that are especially tailored towards operational use have to be found, like e.g. the Ensemble Kalman filter (EnKF) [34].
In the following the approach for data assimilation in the deposition model of RODOS is described. First, the underlying deposition model, the model uncertainties and the formulation of the model in the state space are shown. Then the Ensemble Kalman filter and it's initialisation within the deposition model are briefly described.

Deposition modelling in RODOS

The deposition model in RODOS bases on the radioecological model ECOSYS-87 [35], extended by an atmospheric resistance model for the dry deposition. Input data are mainly results of the atmospheric dispersion models, e.g. concentration of radionuclides in air and rain water and atmospheric resistances. From this data the deposition model calculates the activity deposited on soil, lawn and up to 22 plant types.
In the model the total deposition to the ground is calculated under the assumption that the soil is covered by grass vegetation, i.e. it is composed of the following contributions:

$$A_s = A_{ds} + A_{di} + A_w \tag{1}$$

where

- A_s = total deposition onto vegetated soil (Bq m^{-2});
- A_{ds} = dry deposition onto bare soil (Bq m^{-2});
- A_{di} = dry deposition onto plant type i – here grass (Bq m^{-2});
- A_w = total wet deposition (Bq m^{-2}).

Dry deposition onto a plant is calculated from the activity concentration of a radionuclide in air and a deposition velocity which depends on the plant type. The deposition velocity is determined by the atmospheric resistance and the resistance of the plant canopy:

$$A_{di} = v_{gi} \cdot \overline{C}_{air} \tag{2}$$

$$v_{gi} = 1/(R_a + R_{ci}) \tag{3}$$

where

- $\overline{C}_{air}$ = time integrated activity concentration in air (Bq s m^{-3});
- v_{gi} = deposition velocity for plant type i (m s^{-1});
- R_a = atmospheric resistance near the ground (s m^{-1});
- R_{ci} = resistance of plant canopy for plant type i (s m^{-1}).

The atmospheric resistance depends on the wind speed and the surface roughness. For rapidly depositing radionuclides like elementary iodine the atmospheric resistance becomes the dominating factor for the deposition process. The resistance of plant canopy is assumed to depend on the stage of the plant's development. This causes a pronounced seasonality of the deposition velocity. The plant's stage of development can be characterized by the actual leaf area index (LAI), which is defined as the area of leaves present on a unit area of ground:

$$R_{ci} = R_{ci,min} \cdot \frac{LAI_{i,max}}{LAI_i} \tag{4}$$

where

$R_{ci,min}$ = resistance of plant canopy for fully developed plant i (i (s m^{-1});

$LAI_{i,max}$ = leaf area index for fully developed plant type i (m^2 m^{-2});

LAI_i = actual leaf area index for plant type i (m^2 m^{-2}).

Deposition to a plant consists of dry deposition to the plant surface and the fraction of wet- deposited radionuclides which is hold back on the plant (interception fraction). The calculation of the interception fraction considers the water strorage capacity of the plant's leaves, and the actual leaf area index (if eq. 6 results in a fraction > 1.0 it is set to 1.0):

$$A_i = A_{di} + f_{w,i} A_w \tag{5}$$

$$f_{w,i} = \frac{LAI_i \cdot S_i}{R} \cdot \left(1 - \exp\left(\frac{-\ln 2}{3 \cdot S_i} \cdot R\right)\right) \tag{6}$$

where

A_i = total deposition onto plant type i (Bq m^{-2});

$f_{w,i}$ = interception fraction for plant type i (1);

S_i = retention coefficient for plant type i (mm);

R = amount of rainfall of a rain event (mm).

The time evolution of deposited activity onto soil, lawn and up to 22 plant types is described in the following equations. The activity removed by harvesting is assumed to be recycled by organic fertilization.

$$A_s(\Delta t) = A_s \cdot \exp[-(\lambda_r) \cdot \Delta t] \tag{7}$$

$$A_i(\Delta t) = A_i \cdot \exp[-(\lambda_w + \lambda_r) \cdot \Delta t] \tag{8}$$

where

$A_s(\Delta t)$ = acitvity on soil (Bq m^{-2}) for the time period Δt (d) after deposition;

$A_i(\Delta t)$ = activity on plant type i (Bq m^{-2});

λ_w = weathering rate (d^{-1});

λ_r = radioactive decay rate (d^{-1}).

Uncertainties of deposition modelling

Results of the deposition model are always afflicted with some uncertainty, which can be attributed to uncertain model parameters, the uncertainty of the input data, and the general imperfection of the numerical model in describing physical processes. The first two sources of uncertainty can be described by assigning probability distributions to the uncertain quantities. The uncertainty of model results can

then be assessed by propagating these probability distributions through the deposition model. Probability distributions for model parameters have been derived based on earlier studies [36, 37] [38], data for some selected parameters can be found in Table 4. Uncertainty of the input data is described later in the section about initialisation of the Ensemble Kalman Filter.

Table 4: Probability distributions for some selected parameters of the deposition model with most probable value (mpv), standard deviation, minimum value (min), maximum value (max) and type of probability density function (pdf).

Parameter	mpv	sd	min	max	pdf
deposition velocity for fully developed grass for aerosol bound radionuclides (mm s^{-1})	1.5	0.5	0.001	3.0	normal
deposition velocity for fully developed grass for elementary iodine (mm s^{-1})	15.	5.	0.01	30.	normal
deposition velocity for fully developed grass for organic iodine (mm s^{-1})	0.15	0.05	0.0001	0.3	normal
LAI for fully developed grass (m^2 m^{-2})	7.	1.	4.	10.	normal
date when full LAI for grass is reached (Julian day)	135	10	105	165	normal
retention coefficient for cesium on grass (mm)	0.2	0.1	0.01	1.0	normal
half life of weathering [d]; ($1/\lambda_w$)	25.	-	15.	35.	triangular

State space formulation of the deposition model

Basis for the application of data assimilation methods in the deposition model is a state-space formulation of the model:

$$x_t = M_t x_{t-1} + B_t u_t + \eta_t \tag{9}$$

where

- x_t = state vector including all variables representing the full state of the modelled system at time step k;
- M_t = model operator describing the time evolution of the system according to the deposition model;
- B_t = operator describing the transformation of the forcing to the state space;
- u_t = external forcing of the system (e.g., input from other models);
- η_t = stochastic element representing the uncertainty of system dynamics (η_t is normal distributed with mean 0 and covariance Q_t).

Equation 9 is called the system equation of a model, which describes the dynamic of the considered system. The state vector represents the prediction of a numerical model. In the deposition model the system is represented by the following system variables in the state vector:

$$x_t = \begin{bmatrix} A_l \\ A_s \\ A_i \end{bmatrix} \tag{10}$$

where

A_l = contamination of lawn (Bq m^{-2}),

$$A_l^T = \left[\left(A_{l,k=1,m=1},\ldots,A_{l,k=K,m=1}\right),\ldots,\left(A_{l,k=1,m=M},\ldots,A_{l,k=K,m=M}\right)\right]$$

where K = number of grid cells and M = number of radionuclides;

A_s = contamination of vegetated soil (Bq m^{-2}),

$$A_s^T = \left[\left(A_{s,k=1,m=1},\ldots,A_{s,k=K,m=1}\right),\ldots,\left(A_{s,k=1,m=M},\ldots,A_{s,k=K,m=M}\right)\right]$$

A_i = contamination of plant type i (Bq m^{-2}),

$$A_i^T = \left[\left(A_{i,k=1,m=1},\ldots,A_{i,k=K,m=1}\right),\ldots,\left(A_{i,k=1,m=M},\ldots,A_{i,k=K,m=M}\right)\right]$$

Typically this state vector has a dimension of 10^5 x 1. Data assimilation in DeMM is based on an Ensemble Kalman Filter (EnKF). In the EnKF the statistical properties of the system state are represented by an ensemble of possible state vectors. About 100 such possible state vectors are required to approximate the statistical properties well enough. This means, that about 10^7 variables are necessary for the EnKF in the deposition model. The time evolution of the ensemble of state vectors follows equations 7 and 8, i.e. these equations define the model operator M_t in the deposition model.

The modelling of the uncertainty of the system dynamics - as described by η_t – is specified within the description of the Ensemble Kalman Filter.

The external forcing on the system can be expressed through the vector u_t:

$$u_t = \begin{bmatrix} \overline{C}_{air} \\ A_w \end{bmatrix} \tag{11}$$

where

A_w = det deposited activity (Bq m^{-2}),

$$A_w^T = \left[\left(A_{w,k=1,m=1},\ldots,A_{w,k=K,m=1}\right),\ldots,\left(A_{w,k=1,m=M},\ldots,A_{w,k=K,m=M}\right)\right]$$

$\overline{C}_{air}$ = time integrated activity concentration in air (Bq s m^{-3}),

$$\overline{C}_{air}^T = \left[\left(\overline{C}_{air,k=1,m=1},\ldots,\overline{C}_{air,k=K,m=1}\right),\ldots,\left(\overline{C}_{air,k=1,m=M},\ldots,\overline{C}_{air,k=K,m=M}\right)\right]$$

The operator B_t describing the transformation of the forcing to the state space is defined through equations 1 – 6. The deposition model is a localised model, i.e. only the local (for one grid cell) wet deposited activity and the local air concentration is considered for calculating the deposition at one grid cell.

Additionally to the system equation, the Kalman Filter bases on the observation equation, which transforms the observation process into a state-space formulation:

$$y_t = H_t x_t + \varepsilon_t \tag{12}$$

where

y_t = the observation vector, including all measurements at time t;

H_t = observation operator, which describes the relation between measurements and state variables (i.e. mapping of state space to measurement space);

ε_t = stochastic element representing the uncertainty of the observation process (ε_t is normal distributed with mean 0 and covariance R_t).

The prediction of the deposition model can be corrected by assimilating with three types of measurements:

$$y_t = \begin{bmatrix} \dot{D} \\ \dot{D}_m \\ A_i \end{bmatrix} \tag{13}$$

where

$\dot{D}$ = total net dose rate above lawn
(from all radionuclides, natural background subtracted; in nGy h^{-1}),
$\dot{D}^T = [\dot{D}_{j=1},...,\dot{D}_{j=J1}]$ with J_1 = number of total dose rate measurements;

$\dot{D}_m$ = nuclide-specific net dose rate above lawn
(e.g. from in situ gammaspectrometry; in nGy h^{-1}),
$\dot{D}_m^T = [\dot{D}_{m,j=1},...,\dot{D}_{m,j=J2}]$ with J_2 = number of nuc.-spec. dose rate measurements;

C_i = activity concentration of plant type i (Bq kg^{-1}),
$C_i^T = [C_{i,m=1},...,C_{i,m=J3}]$ with J_3 = number of concentration measurements.

The observation operator H_t relates the state variables with the measurements, it is defined by the following equations 14 and 15. The spatial mapping between state variables (given for grid cells) and measurements (can be given for any point in space) is defined in a simple way: each measurement site is mapped to closest grid cell.

$$\dot{D} = \sum_{m=1}^{M} \dot{D}_m = \sum_{m=1}^{M} d_{ground,m} \cdot A_{l,m} \tag{14}$$

$$C_i = \frac{A_i}{Y_i} \tag{15}$$

where

$d_{ground,m}$ = dose rate conversion factor for groundshine (nGy h^{-1} per Bq m^{-2});
Y_i = yield of plant type i at time of measurement (kg m^{-2}).

The modelling of the uncertainty of the observation process - as described by ε_t – is specified within the description and the initialisation of the Ensemble Kalman Filter.

Ensemble Kalman filter

Data assimilation with Kalman filters is a sequential two-step process: first, the numerical model is applied to calculate a forecast of the system state (forecast step). Then the model forecast is merged with the measured data based on a statistical criterion (analysis step). Both steps are described in the following for a Monte-Carlo version of the Kalman filter, the Ensemble Kalman Filter (EnKF). The EnKF is especially suitable for very large, non-linear systems. In the EnKF the statistical properties of the system state are represented by an ensemble of possible state vectors.

In the forecast step of the EnKF each of the initial state vectors is propagated through the deposition model one time step ahead:

$$x_{t|t-1,i} = M_t \, x_{t-1|t-1,i} + B_t u_{t,i} \tag{16}$$

where

$x_{t|t-1,i}$ = one state vector from the forecast ensemble at time t,
$x_{t-1|t-1,i}$ = one state vector from the updated ensemble at time t-1,
i = 1,2,...,I: index of state vector in ensemble (I = number of state vectors).

This means running the deposition model as many times as the number of possible state vectors in the ensemble is (typically about 100). For complex models this step is the bottleneck of the data assimilation process in terms of computational burden. Applying the full deposition model a hundred times would also make the data assimilation process to slow for an operational real-time system like RODOS. Thus, a simplified deposition model is used, in which a constant atmospheric resistance is assumed (i.e. no dependence of the resistance on wind speed and surface roughness). This allows to calculate global deposition velocities applicable for the whole RODOS grid instead of localised deposition velocities. The improvement in the computational burden is about a factor of 100.

Model errors are considered in this step by applying an ensemble of possible sets of model parameters according to their uncertainty distributions (see Table 4). With other words, each possible state vector is propagated with the model with a different set of model parameters. As a result, the initial spread of the ensemble will be enlarged by the model error coming from uncertain parameters.

The forecast of the system state can be represented by the first two statistical moments, the mean value and the covariance. In the EnKF the mean value of the system state is defined as the mean of all state vectors in the ensemble, and the covariance can be approximated from the deviation of each possible state vector from the mean:

$$\bar{x}_{t|t-1} = \frac{1}{I}\sum_{i=1}^{I} x_{t|t-1,i} \tag{17}$$

$$P_{t|t-1} = S_{t|t-1}\left(S_{t|t-1}\right)^T,\ S_{t|t-1} = \left[s_{t|t-1,1},\ldots,s_{t|t-1,I}\right],\ s_{t|t-1,i} = \frac{1}{\sqrt{I-1}}\left(x_{t|t-1,i} - \bar{x}_{t|t-1,i}\right) \tag{18}$$

where

$s_{t|t-1,i}$ = i-th column of matrix $S_{t|t-1}$,
$S_{t|t-1}$ = matrix approximating the square root matrix of matrix $P_{t|t-1}$,
$P_{t|t-1}$ = forecast covariance matrix at time t (representing the uncertainty of forecasted state of the system).

In the analysis step of the EnKF each state vector of the ensemble is updated separately using a common weighting matrix, the Kalman gain. The formulation of the weighting matrix is the most essential part of the data assimilation scheme. The update is a linear combination of the forecast and the so-called innovation, which is the difference between the measurements and the forecast of the measured data from the system state. A big advantage of the EnKF is that it is not necessary to calculate the full covariance matrix P, since the Kalman gain can directly be calculated from the much smaller matrix S:

$$x_{t|t,i} = x_{t|t-1,i} + K_t\left(y_{t,i} - H_t x_{t|t-1,i}\right) \tag{19}$$

$$K_t = \frac{S_{t|t-1}\left(H_t S_{t|t-1}\right)^T}{\left(H_t S_{t|t-1}\right)\left(H_t S_{t|t-1}\right)^T + R_t} \tag{20}$$

$$y_{t,i} = y_t + \eta_{t,i} \tag{21}$$

where

K_t = Kalman-Gain, that reflects the relative uncertainties of the model forecast and the measurements;
R_t = measurement error covariance matrix;
y_t = measurement vector, which contains all measurements at time t;
$y_{t,i}$ = possible measurement vector, which spreads around the measurement vector y_t according to the normal distributed measurement error η_t .

Measurement errors are considered by replacing the measured values through an ensemble of possible measurements generated from the measurement error covariance matrix [39]. The resulting ensemble of updated state vectors provides an estimate of the updated system state and the updated covariance matrix (as in eq. (17) and (18)):

$$\bar{x}_{t|t-1} = \frac{1}{I}\sum_{i=1}^{I} x_{t|t-1,i} \tag{22}$$

$$P_{t|t} = S_{t|t}\left(S_{t|t}\right)^T \;,\; S_{t|t} = \left[s_{t|t,1},\ldots,s_{t|t,I}\right],\; s_{t|t,i} = \frac{1}{\sqrt{I-1}}\left(x_{t|t,i} - \bar{x}_{t|t,i}\right) \tag{23}$$

In case that the measurement errors are uncorrelated (i.e. the measurement error covariance matrix R_t is diagonal), a sequential updating algorithm can be applied. In this algorithm only one measurement is processed in each step, which avoids the time-consuming matrix inversion in eq. (20). The updated state vector of one step ($x_{t|t,i,j}$) is used as forecast (as $x_{t|t,i,j-1}$) in the next step processing the next measurement.

$$x_{t|t,i,j} = x_{t|t,i,j-1} + k_{t,j}\left(y_{t,i,j} - h_{t,j}x_{t|t,i,j-1}\right), \quad x_{t|t,i,0} = x_{t|t-1,i} \tag{24}$$

$$k_{t,j} = \frac{S_{t|t,j-1}\left(h_{t,j}S_{t|t,j-1}\right)^T}{\left(h_{t,j}S_{t|t,j-1}\right)\left(h_{t,j}S_{t|t,j-1}\right)^T + \sigma^2_{m,j}} \tag{25}$$

$$y_{t,i,j} = y_{t,j} + \eta_{t,i,j} \tag{26}$$

$$S_{t|t,j} = S_{t|t,j-1} - \frac{k_{t,j}\left(h_{t,j}S_{t|t,j-1}\right)}{1+\sqrt{\frac{\sigma^2_{m,j}}{\left(h_{t,j}S_{t|t,j-1}\right)\left(h_{t,j}S_{t|t,j-1}\right)^T + \sigma^2_{m,j}}}}\;, \quad S_{t|t,0} = S_{t|t-1} \tag{27}$$

where

j = 1,2,...,J: index of measurements at time t (J = number of measurements).
$k_{t,j}$ = Kalman-Gain vector corresponding to measurement j;
$y_{t,i,j}$ = j-th measurement from the i-th possible measurement vector;
$h_{t,j}$ = observation operator for measurement j;
$\sigma^2_{m,j}$ = variance of the j-th measurement.

The Kalman filter updates the system state at the time of the measurements and then predicts this updated state into the future. This means, that the optimal estimation is available at the time of the last measurement.

Initialisation of the Ensemble Kalman Filter

Before starting the EnKF has to be initialised, i.e. inital values have to be set for the ensemble of state vectors, the measurements and the external model forcing. All initial state vectors are simply set to zero, since zero initial ground contamination is assumed before start of the deposition event.

Measurement data is available from radiological monitoring networks, but uncertainty of this data has to be estimated in terms of a covariance matrix. It is assumed that all measurements are uncorrelated amongst each other, i.e. their error covariance matrix is diagonal and contains the variance of each measurement in the diagonal. These variances are estimated from information about measurement uncertainties, which can be attributed to a number of sources: e.g., intrinsic uncertainty of the measurement equipment, the sampling procedure, interpretation of measurements in the sense of state variables (i.e. the use of local measurements as average value over grid cells), etc.

The external model forcing is determined by the results of the atmospheric dispersion modelling, uncertainty of the model forcing is given by an error covariance matrix. This covariance matrix is the outcome of propagating uncertainties through the atmospheric dispersion modelling. Due to the high-dimensional state of the system and the non-linearities in the modelling, mathematical approximations are needed for the propagation and exchange of the covariance matrix. Here an ensemble representation of the covariance matrix is used as an efficient approximation. This ensemble is being provided by the ensemble approach for the atmospheric dispersion models.

The system state in the deposition model consists of deposited activities. Typical probability distributions of the deposited activity follow a log-normal function. Thus, the system state is log-transformed before the EnKF is applied, a corresponding transformation is also performed for the measured data.

Data assimilation results

Some example results of data assimilation with the Ensemble Kalman filter applied in the deposition model are shown in the following. The performance of the Ensemble Kalman filter is tested with a twin experiment: data from one model scenario is assumed to be the “true” model prediction. Simulated measurements are generated from the “true” data, additionally some noise is added according to the assumed measurement errors. Then a second “false” model prediction is generated based on a modified scenario with disturbed model parameters and/or input data. The Ensemble Kalman Filter is used to correct (update) the “false” prediction by assimilating the simulated measurements. The resulting updated prediction can be checked against the “true” prediction. The smaller the differences between updated and true prediction, the better the EnKF is performing.
One example result for such a twin experiment for an EnKF applied in the deposition model is shown in the following:

- Fig. A1a/b show the cesium deposition to leafy vegetables (converted into Bq/kg fresh weight of leafy vegetables) as calculated for an ensemble of 50 model runs of the full JRodos model chain.
- In these figures the color blue indicates areas, in which the maximum permitted level (MPL) for cesium in leafy vegetables of 1250 Bq/kg is exceeded for at least one ensemble member. The color red indicates areas, in which the MPL is exceeded for all 50 ensemble members. In other words, the discrepancy between the red and the blue areas can be interpreted as an indicator for the spread of the ensemble members (the larger the difference between the size of the red and the blue area, the higher are the discrepancies among the ensemble members).
- Fig. A1a shows the situation before data assimilation, where the spread of the ensemble members is caused by the uncertainties of the underlying meteorological data, the uncertainties of the underlying source term and the uncertainties of the deposition model.

- Fig. A1b shows the situation after data assimilation, where the spread of the ensemble members – and thus the uncertainty of the model prediction as described by the full ensemble – is reduced by data assimilation with the Ensemble Kalman filter.
- The monitoring data used for the data assimilation is not shown here, it was artificially created by randomly assigning one of the ensemble members as the "truth" and simulating monitoring data for a few monitoring sites from these "true data".

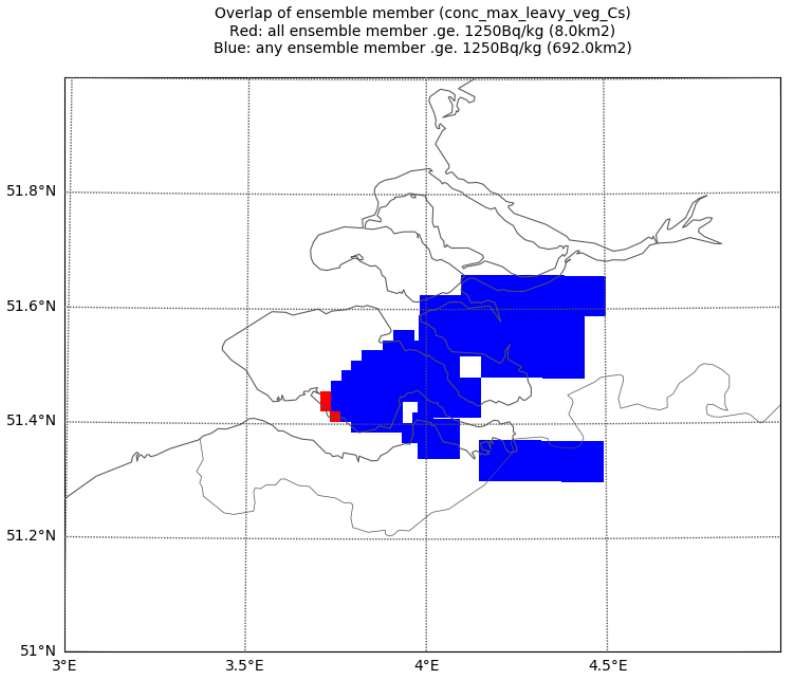

Fig. A1a: Ensemble results for cesium deposition to leafy vegetables (converted into Bq/kg) before data assimilation.

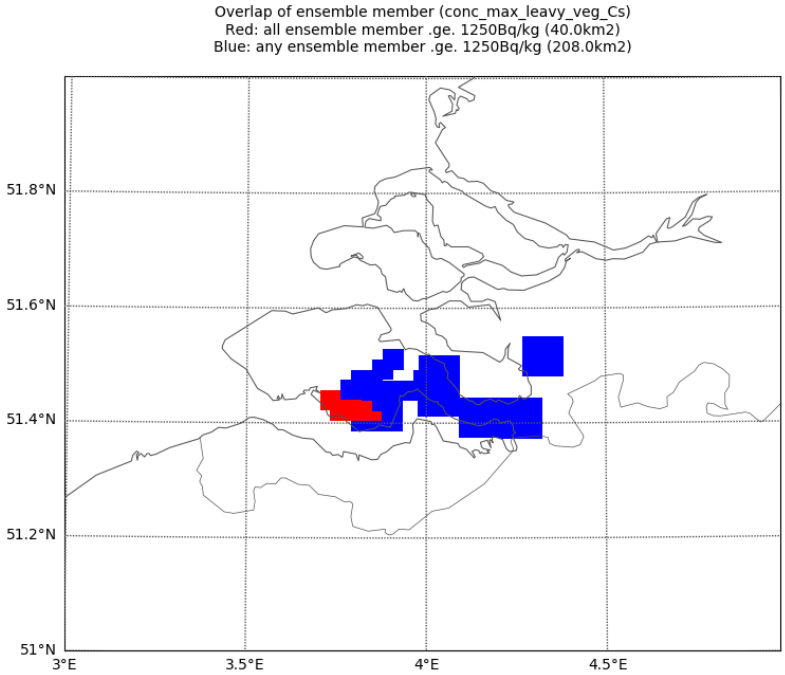

Fig. A1b: Ensemble results for cesium deposition to leafy vegetables (converted into Bq/kg) after data assimilation.

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D9.7.1 Stationary monitoring systems and their uncertainties

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Abstract

Methods, techniques and equipment used in environmental monitoring of radioactivity are presented and the uncertainties related to them are considered.

In this sub-report we will discuss stationary monitoring systems. The quantities monitored are ambient dose equivalent rate, activity concentration in surface air and deposition density of activity to the ground. The stationary monitoring systems of these quantities are considered to form networks of radiation monitoring stations (although this term is somewhat artificial when referring to air and fallout monitoring due to the sparseness of the monitoring locations). The uncertainty components resulting from statistical, systematic and epistemological sources are considered to provide a basis for evaluating the suitability of using the monitoring results for purposes given in the main report D9.7. Such purposes are data assimilation, interpolation across areas spanned by the network and estimation of doses due to external exposure and inhalation. The aim of using the stationary monitoring data is to reduce the overall uncertainties in decision making,

Contents

1. Monitoring of airborne radioactivity

Continuous monitoring of airborne radioactivity is carried out as a part of national environmental monitoring programmes of several countries. The member states of the EU have agreed to establish facilities for such monitoring as required by the article 35 of the EURATOM treaty. Surface air is monitored routinely for radioactive pollutants to ensure the health and safety of citizens and to detect events with potential radiological consequences.

The concentration of radioactive substances in ambient air is commonly monitored using air samplers. An air sampler draws air through a filter by means of a vacuum pump, causing the radioactive substances in the air to accumulate on the filter. The samples collected this way consist either of trapped particulate matter or gas. Glass fiber, polyethylene, polytetrafluoroethylene (PTFE) and Petrianov are common filter media used in the collection of particulate samples, whereas gaseous samples are commonly collected using activated carbon filters. Gas samples are most often collected in order to determine the concentration of gaseous iodine. The speciation of iodine is an important concern in emergency situations, and is discussed in section 1.5 *Speciation of Iodine*.

When the volume V of air drawn through the filter is known, the average activity concentration of given radionuclide in the air during the sampling time t_s can be determined by measuring the activity a accumulated in the filter. The average activity concentration C is then given by

$$C = \frac{a}{V}. \tag{1}$$

Given in this form, the sampling efficiency is neglected. The average activity concentration is based on the assumption that the activity concentration of radioactive substances remains constant during collection. For radionuclides with suitably long half-lives, the time integrals of the average activity concentration and time dependent activity concentration $C(t)$ over the sampling time interval (t_0, t_1) agree, leading to

$$(t_1 - t_0)C = t_s C = \int_{t_0}^{t_1} C(t)\, dt. \tag{2}$$

The time integral of the activity concentration is usually of interest in dosimetry applications. Radionuclides with short half-lives are discussed later.

The air samplers used in the monitoring of airborne radioactivity differ mainly in the filter media they employ and their capacity to draw air through the filter. This capacity is given as the volumetric flow rate $\equiv dV/dt$, i.e. the volume of air flowing through the filter in a unit of time. For a given sampling time t_s, the volume of air drawn through the filter under constant flow rate conditions is then

$$V = Qt_s. \tag{3}$$

More generally, to obtain the volume, the time dependent volumetric flow rate $Q(t)$ is integrated over the collection period, resulting in

$$V = \int_{t0}^{t1} Q(t)\, dt. \tag{4}$$

The measurement of the activity accumulated in the filter is performed using gamma or alpha spectroscopy. Beta counting is also done.

One consideration in the use of air samplers is the mode of operation; in *offline operation* a sample is collected and the radioactivity in the sample is then determined by transporting the sample to

counting, whereas in *online operation* the radioactivity in the sample is continually monitored using a radiation detector integrated to the sampler. Offline mode of operation has traditionally been much more sensitive in the detection of radioactivity due to laboratory counting of samples. The offline mode of operation suffers the delays of sample collection, transport and counting before the result is available. Both modes of operation can usually be employed simultaneously.

The sensitivity of the monitoring apparatus (i.e. the sampler and the measurement system) can be characterized by determining the *minimum detectable concentration* (MDC) - the smallest concentration of a given radionuclide that can be determined from a sample with given confidence in a given sampling time t_s, cooling time t_w and counting time t_c. The confidence is specified by defining the risk levels α and β corresponding to Type I and Type II errors respectively. The risk levels are used to derive the *minimum detectable activity* (MDA) of the filter measurement. The MDC is then derived from the MDA under the assumption of constant activity concentration during the sampling interval. It should be noted that the MDC is not subject to uncertainty, rather, it incorporates the consideration of uncertainties in the monitoring apparatus to produce the detection limit with respect to the given risk levels. The MDC depends on the background of the sample measurement.

For a given nuclide with decay constant λ, the MDA a_{min}, and the corresponding MDC C_{min}, for a filter measurement with standard deviation of the background (baseline) $\sigma_B = \sqrt{p_w b/w}$ estimated over energy range of w keV where the FWHM is p_w keV are given by

$$a_{min} = \frac{1}{\varepsilon\gamma t_c}\left(k_\alpha\sqrt{2}\sigma_B\frac{k_\beta^2}{2} + k_\beta\sqrt{\frac{k_\beta^2}{4} + k_\alpha\sqrt{2}\sigma_B + \frac{\left(k_\alpha\sqrt{2}\sigma_B\right)^2}{k_\alpha^2}}\right), \tag{5}$$

$$C_{min} = a_{min}e^{\lambda(t_w+t_c)}\frac{\lambda t_s}{V(1-e^{-\lambda t_s})},$$

where k_α and k_b are the coverage factors of the standard deviation corresponding to the predefined risk levels α and β obtained from standard normal distribution. The above formulas are from [1] and apply to offline mode of operation. This formulation neglects the various correction factors applied to the measurement (see Table 2).

For online mode the background does not stay constant for the duration of the measurement, instead, the background will increase as radon daughters are collected on the filter until their activity on the filter reaches its maximum. Eq. (5) can still be used if the evaluation time is defined.

As the determination of the average activity concentration (and thus the MDC) depends on the collected volume V, which in turn depends on the flow rate Q, it is obvious from Eq. (1), (2) and (3), that, for a fixed sampling time t_s, a higher capacity sampler will be able to achieve a smaller MDC than a sampler with lower capacity. The tradeoff is that in order to achieve a higher flow rate, a larger, more powerful pump is required, making the sampler larger and bulkier and placing greater requirements on the pumps power source. The MDC can also be reduced by increasing the sampling time t_s. This in turn will cause greater delay between the start of sampling and the arrival of the result. These factors are illustrated in Figure 1 and Table 1.

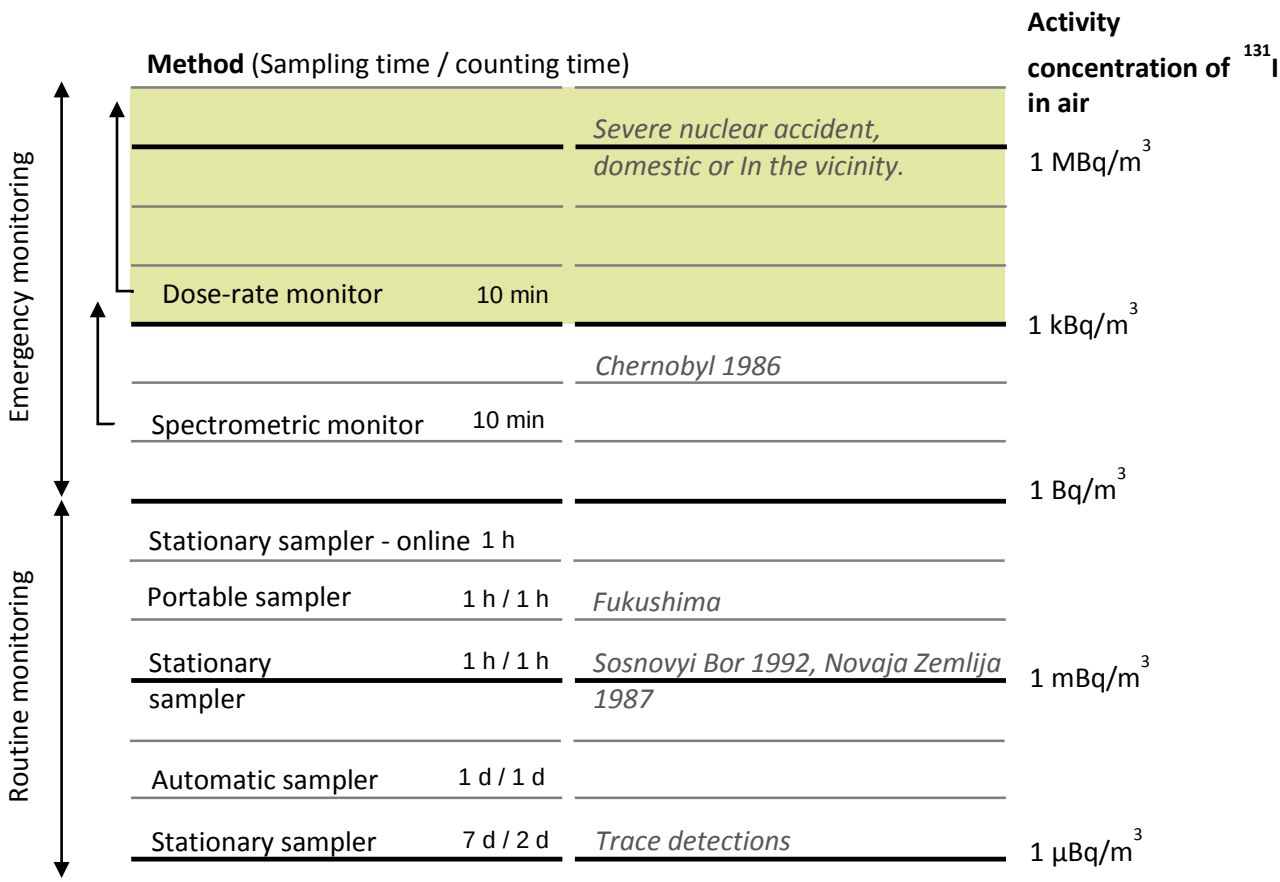

Figure 1 – The monitoring domains (routine and emergency monitoring), and their applicable monitoring methods in Finnish environmental monitoring. Some events and their magnitudes are also depicted.

Table 1 – Properties of stationary samplers under routine and emergency conditions. Offline mode measurements in a counting laboratory with a 50% BEGe detector, cooling time of 1 hour assumed. Online MDC is for $LaBr_3$(Ce) detector fpr collection a pediod of 1 hour at the end of collection (1 hour spectrum for collected volume Q). Risk levels $\alpha = \beta = 0.05$.

Sampler	**Q (m³/h)**	**Mode**	t_s	t_c	**MDC (µBq/m³)**		
					I-131	**Cs-134**	**Cs-137**
JL-150 Hunter	150	offline	7 days	48 hours	0.7	0.4	0.4
Cinderella G2	500	offline	24 hours	24 hours	3.4	3.4	3.6
JL-900 Snow White	900	offline	7 days	48 hours	0.2	0.1	0.1
Emergency monitoring					**MDC (mBq/m³)**		
JL-150 Hunter	150	offline	1 hour	1 hour	0.6	0.6	0.6
		online			53	198	198
Cinderella G2	500	offline	1 hour	1 hour	0.6	0.5	0.5
		online			476	668	668
JL-900 Snow White	900	offline	1 hour	1 hour	0.2	0.1	0.1
JL-10-24 Lilliput	12	offline	1 hour	1 hour	23	19	19

1.1. Uncertainties in air sampling

The average activity concentration is described by Eq. (1) in terms of the activity a and the volume V. To determine its uncertainty, the uncertainty components of both the volume and the activity need to be established. The uncertainties presented apply to the collection of particulate matter.

1.2. Uncertainty of volume

The volume of air drawn through the filter is determined by measuring the volumetric flow rate Q of air flowing through the filter. The flow rate is integrated by a flow computer to obtain the collected volume. The uncertainty of the volume therefore depends on the method of flow rate measurement. It should also be noted that, when stating the volume of a gas, the pressure and temperature should also be given. In order to state the gas volume in suitable conditions, the sampler needs to be able to measure the prevalent static pressure, temperature and relative humidity. The collected volume can then be stated in terms of prevalent conditions (based on the aforementioned measurements) or in STP - standard temperature and pressure (by transforming the volume reading from prevalent conditions to equivalent volume in predefined pressure and temperature). The gas volume in prevalent conditions and STP may differ, so it is necessary to be aware of the conditions in which the volume is stated.

1.2.1. Uncertainty of flow rate measurement

Orifice plate is a commonly used method of flow rate measurement. An orifice plate is a constriction with a carefully measured bore placed in tube of known diameter together with a pair of tappings on both sides of the constriction for pressure measurement. The method of measurement and computation is given in [2]. The pressure difference across the orifice plate is related to the volumetric flow rate

$$Q = \frac{C}{\sqrt{1-\beta^4}} \varepsilon \frac{\pi}{4} d^2 \sqrt{\frac{2\Delta P}{\rho}}. \tag{6}$$

The uncertainty of Q depends then on the measurement of pressure difference ΔP across the orifice, orifice diameter d, discharge coefficient C, the diameter ratio β of the orifice to the pipe, the expansibility factor ε and the density of the fluid ρ. A practical working formula for the uncertainty of the mass flowrate is given in [3]

$$\frac{u(Q)}{Q} = \left(\left(\frac{u(C)}{C}\right)^2 + \left(\frac{u(\varepsilon)}{\varepsilon}\right)^2 + \left(\frac{2\beta^4}{1-\beta^4}\right)^2 \left(\frac{u(D)}{D}\right)^2 + \left(\frac{2}{1-\beta^4}\right)^2 \left(\frac{u(d)}{d}\right)^2 + \frac{1}{4}\left(\frac{u(\Delta P)}{\Delta P}\right)^2 + \frac{1}{4}\left(\frac{u(\rho)}{\rho}\right)^2 \right)^{\frac{1}{2}}. \tag{7}$$

The measurement uncertainty for orifice plate measurement of mass flow rate has been estimated in [4], providing an example of the uncertainty evaluation. The relative uncertainty is typically found to be around 2%.

1.3. Uncertainty of activity

To determine the activity accumulated on the filter during sampling, the collected sample is measured. The uncertainties related to the measurement have to be considered and reflected in the uncertainty of the activity a.

In addition to measuring the activity on the filter, it is important to consider the degree to which the sample represents the particles suspended in the surrounding air. This is represented by sampling and collection efficiencies, describing the fraction of particles ending up on the filter.

1.3.1. Measurement uncertainty

The uncertainties related to sample measurements are generally well known and found in guides and literature such as [5] [1] and [6]. We will only summarize gamma spectrometry, alpha spectrometry is discussed in literature such as [7].

The activity present in a sample is given by

$$a = \frac{A}{\varepsilon \gamma t_c} \prod_i C_i^{-1}. \tag{8}$$

The relative uncertainty of activity is then given by

$$\frac{u(a)}{a} = \sqrt{\left(\frac{u(A)}{A}\right)^2 + \left(\frac{u(\varepsilon)}{\varepsilon}\right)^2 + \left(\frac{u(\gamma)}{\gamma}\right)^2 + \left(\frac{u(t_m)}{t_m}\right)^2 + \sum_i \left(\frac{u(C_i)}{C_i}\right)^2}. \tag{9}$$

Input quantities in Eq. (8) and their uncertainties are summarized in Table 2. The uncertainties in Table 2 are from [6] and [1], with the full energy peak area function modified to include the baseline. For air filter samples the uncertainty of activity is dominated by the uncertainties of efficiency, full energy peak area and background correction. Determining the uncertainty of efficiency is complicated. If the reference source emits several energies, the efficiencies determined using these energies are correlated. The correlations should be accounted for as discussed in [6]. Another complication is encountered in the extension of the experimental uncertainties to the whole efficiency curve. This is needed when the efficiency function is evaluated in order to obtain a suitable efficiency to use in Eq. (8).

Table 2 – The input quantities to Eq. (8) and their relative uncertainties.

Quantity	Symbol	Uncertainty
Full energy peak area	A	Fitting a Gaussian peak on a linear baseline $f(E; N, p_1, p_2, p_3, p_4) = N/\sqrt{2\pi p_2^2} \exp\left(\frac{(E-p_1)^2}{2p_2^2}\right) + p_3 E + p_4$: $\frac{u(A)}{A} = \sqrt{N + \sum_i \left(\frac{u(p_i)}{p_i}\right)^2 + 2\sum_i \frac{u(p_i, N)}{p_i N} + 2\sum_i \sum_{j \neq i} \frac{u(p_i, p_j)}{p_i p_j}}$ The standard deviations and covariances of the fitting parameters should be obtained from the fitting algorithm.
Full energy peak efficiency	ε	From measured peak area N_{ref} of reference source with activity A_{ref}, intensity of line energy γ_{ref} and correction factors $C_{ref,i}$: $\frac{u(\varepsilon)}{\varepsilon} = \sqrt{\left(\frac{u(N_{ref})}{N_{ref}}\right)^2 + \left(\frac{u(A_{ref})}{A}\right)^2 + \left(\frac{u(\gamma_{ref})}{\gamma_{ref}}\right)^2 + \sum_i \left(\frac{u(C_{ref,i})}{C_{ref,i}}\right)^2}$ Note: This formula is meant for the simple case of efficiency determined from a monoenergetic gamma emitter. The formula will underestimate the uncertainty in the presence of correlations.
Gamma yield	γ	From library data.

Measurement live time	t_c	The accurate determination of the measurement live time is extremely important. Unlike the real time, which can reliably be read from the device clock and has a negligible uncertainty, the live time can exhibit considerable uncertainty due to the way it is determined . For small count rates the uncertainty of live time is usually negligible, but can be significant with higher count rates. The subject is discussed in [8].
Sampling correction	C_1	For constant activity concentration, sampling time t_s and $\lambda = \ln 2 \,/\, t_{1/2}$ $$\frac{u(C_1)}{C_1} = \sqrt{\left(\ln(2)\frac{u(t_{1/2})}{t_{1/2}}\right)^2 + \left(\frac{u(t_s)}{t_s}\right)^2}\left(1 - \frac{\lambda t_s e^{-\lambda t_s}}{1 - e^{-\lambda t_s}}\right)$$
Cooling decay correction	C_2	For waiting (cooling) time t_w $$\frac{u(C_2)}{C_2} = \ln(2)\frac{t_w}{t_{1/2}}\frac{u(t_{1/2})}{t_{1/2}}$$ Not needed for online mode of operation.
Counting decay correction	C_3	For measurement time t_m $$\frac{u(C_3)}{C_3} = \ln(2)\frac{t_m/t_{1/2}}{1 - \exp(-\ln(2)t_m/t_{1/2})}$$
Self-absorption correction	C_4	Not needed if calibration performed on the type of filter used as discussed in [1].
Coincidence correction	C_5	The uncertainty of the coincidence correction factor is discussed in [6].

For the activity corrections $C_1 - C_5$; the sampling correction, cooling decay correction, counting decay correction, self-attenuation correction, self-absorption correction and coincidence correction, uncertainties are derived in [1] and [6]. The uncertainties of the corrections $C_1 - C_4$ are usually negligible as they depend only on the uncertainties of half-life, sampling time, cooling time and measurement time, all usually well known. For air sampling the self-attenuation correction is only needed if the filter material is highly variable in its density – this is usually not the case. The sampling correction can introduce epistemological uncertainty due to the assumption of constant activity concentration during sampling. This is discussed under *Temporal behavior of activity concentration*. The coincidence correction factors are an important source of uncertainty in laboratory measurements but not necessarily in online mode of operation. Their contribution to uncertainty in close geometry counting needs to be assessed and accounted for.

1.3.2. Sampling efficiency and particle size distribution

The collected sample does not necessarily accurately represent the particles suspended in the gas it was extracted from. As stated in [9], this is due to loss and deposition mechanisms caused by gravitational, diffusional and inertial forces acting on the particles as they are aspirated from their surroundings and transported to the filter. The influence of such losses and other factors are represented by various particle size dependent efficiencies. The most relevant is the *inlet efficiency*

η_{inlet} ,defined in [9] as the fraction of ambient concentration delivered through the inlet into the subsequent transport line. The inlet efficiency is a product of two components; *aspiration efficiency* η_{asp} and *transmission efficiency* η_{trans} , leading to

$$\eta_{inlet} = \eta_{asp}\eta_{trans}. \tag{10}$$

If the sampler construction is more complex, further multiplicative factors represented by various transport efficiencies through the transport lines need to be considered.

Collection efficiency describes the ability of the filter medium to trap the particles incident on the filter face. Particles are collected on the filter via five mechanisms: interception, impaction, diffusion, electrostatic attraction and gravitational settling. The mechanisms are illustrated in Figure 2 based on the theoretical models given in [10].

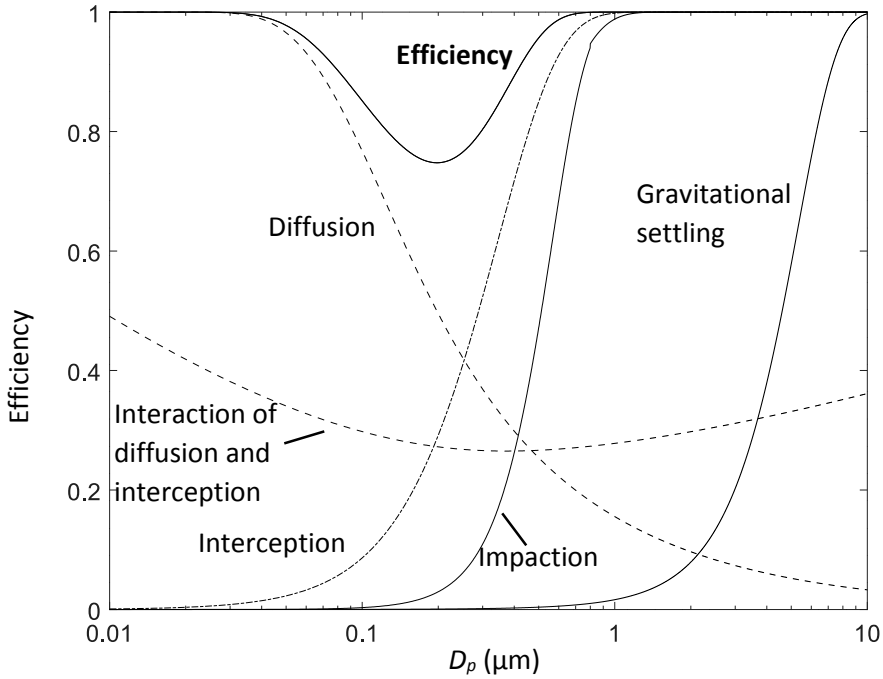

Figure 2 – The deposition mechanisms and the collection efficiency of a filter.

Filters suitable for sampling with face velocities ~ 1 m/s have been tested in [11]. In Figure 3 the theoretical model from [10] is fitted to the data points. The filtration efficiency of glass fiber filters is practically 100 % for diameters above 0.3 µm. For other media (Figure 3) the collection efficiency is 100 % for particle diameters above 0.6 µm. A drop in collection efficiency is seen for particles of diameter less than 0.3 µm.

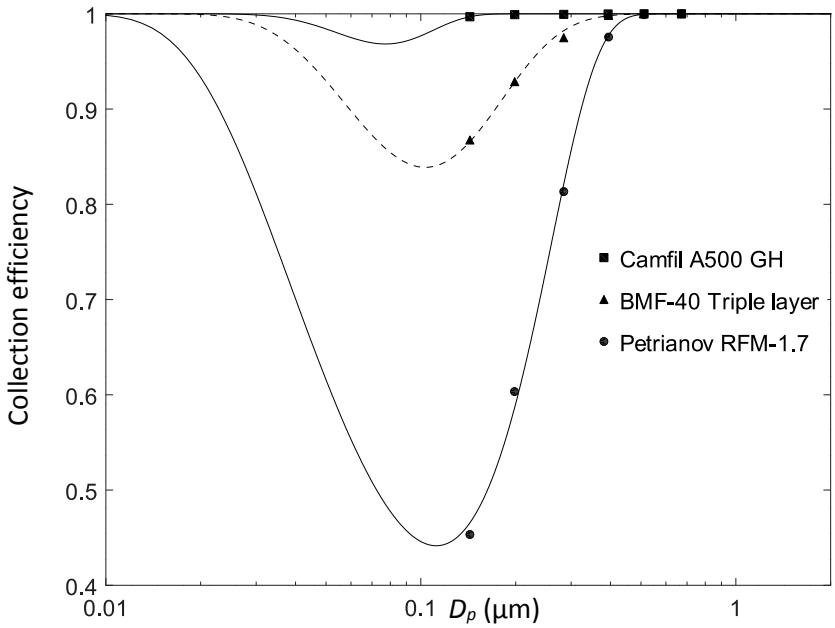

Figure 3 – Collection efficiency of different filter media as a function of physical diameter. The test is at 1 m/s face velocity on unloaded filters. The test aerosol was DEHS, with density 0.915 g/cm³. The efficiencies below 0.14 µm are purely theoretical.

Combining the aforementioned efficiencies yields the particle size dependent sampling efficiency η ,which describes how large a fraction of particles of given size are collected on the filter. These efficiencies are often neglected, leading to epistemological uncertainty in the average activity concentration determined using Eq. (1). Experimental determination of these efficiencies is demanding, requiring special facilities (ideally a wind-tunnel) and carefully calibrated equipment (an aerosol generator, isokinetic samplers, aerosol sizers). Such calibration is performed in [12] for JL-900 Snow White sampler, displaying the dependence of efficiency on not only the aerodynamic diameter of the collected particles, but on wind speed U and flow rate Q as well (Figure 4).

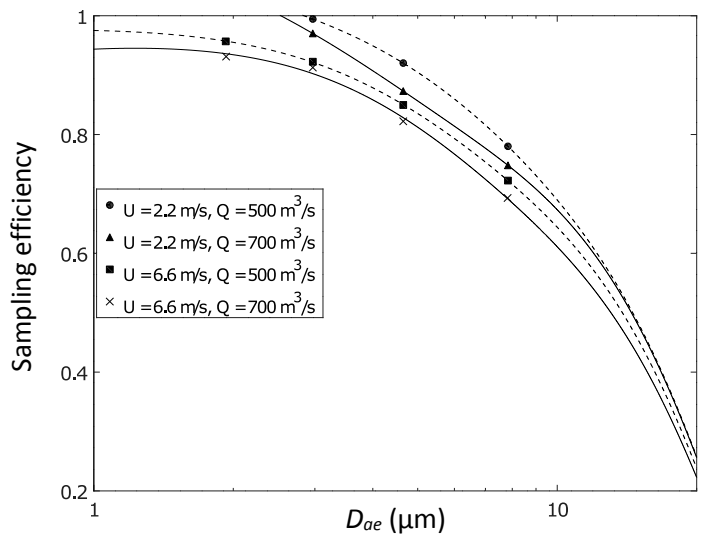

Figure 4 – The sampling efficiency of a JL-900 Snow White sampler. A product of losses due to misalignment, gravitational settling, interception and diffusion in the inlet is fitted to the experimental data points from [12].

As discussed above, the factors contributing to sampling efficiency depend on particle size. Therefore to relate the activity on the filter to the activity in ambient air, a correction has to be made based on the *activity size distribution* of the sampled particles. This is illustrated by the following example:

Example 1

Consider a monitoring apparatus with sampling efficiencies of 0.97 and 0.87 for particles of aerodynamic diameters of 5 µm and 10 µm respectively. Assume that 90 % of the activity in the air is in particles of 5 µm diameter and the remaining 10 % in particles of 10 µm. The correction relating the measured activity concentration to the activity concentration in ambient air is

$$\frac{0.9}{0.97} + \frac{0.1}{0.87} = 1.0428.$$

If, however, the proportions of activity in the particles are reversed, i.e. 10 % is in 5 µm particles and 90 % in 10 µm particles, the correction factor would be

$$\frac{0.1}{0.97} + \frac{0.9}{0.87} = 1.1376.$$

The probability density function of the activity size distribution $f_{asd}(D)$ and the sampling efficiency $\eta(D)$ are used to obtain the correction factor relating the sampled activity to activity in ambient air

$$\xi = \int_0^\infty \eta(D)^{-1} f_{asd}(D) dD. \tag{11}$$

The relative uncertainty of the correction factor ξ is obtained from Eq. (11) to be

$$\frac{u(\xi)}{\xi} = \sqrt{\sum_i \left(\frac{u(\eta_i)}{\eta_i}\right)^2 + \left(\frac{u(f_i)}{f_i}\right)^2}, \tag{12}$$

where η_i is the sampling efficiency of the i:th particle size class, and f_i the activity fraction contained in the i:th particle size class. The uncertainties of the efficiencies and the activity fractions of the size classes result from the efficiency determination procedure and the particle size measurements respectively.

We can now rewrite Eq. (1) to use the correction factor simply as

$$C = \xi \frac{a}{V}. \tag{13}$$

Neglecting the correction factor will result in an underestimation of the average activity concentration.

The activity size distributions vary depending on the generation mechanism of the particles. Determining the activity size distribution of the sampled particles is generally not possible in the case of trace detections made under routine monitoring.

Particle size has an effect on the atmospheric residence time of the particle as discussed in [13] and [14] (Figure 5), but also vice-versa – the transport of particles acts as a filter altering the size distribution of particles emitted from a source. Maximum residence time in the lower troposphere is achieved by particles with aerodynamic diameter D_{ae} of approximately 0.6 µm, with larger and smaller particles being deposited to the ground with greater probability during transport.

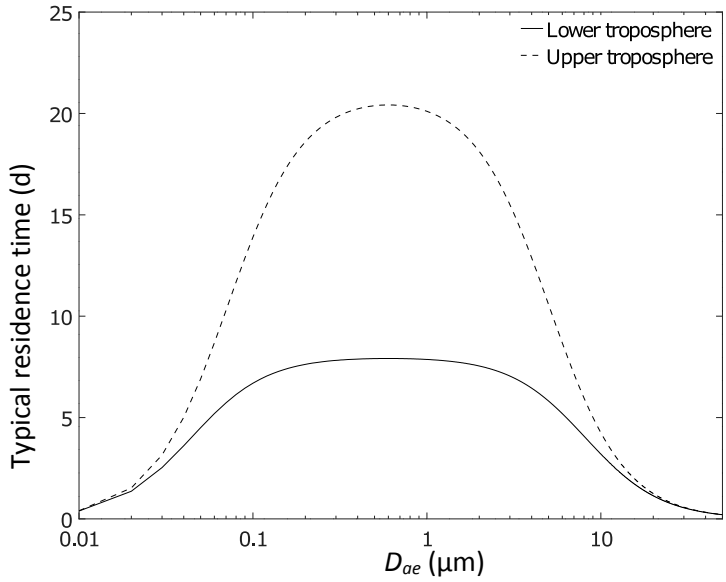

Figure 5 – Typical lifetime of atmospheric particles in the troposphere accounting for losses due to coagulation, dry deposition and wet removal. The model of residence time is given in [14].

Activity size distributions are found from literature following the Fukushima [15] [16] and Chernobyl [15] accidents.

A crude comparison on the effect of neglecting the use of the collection efficiency can be made using the efficiency data of Figure 3 (assuming $D_{ae} = D_p$) and Figure 4 together with activity size distribution of airborne particulate I-131 determined at Czech republic after the Fukushima accident in [15].

Table 3 – Activity size distribution and sampling efficiency used to determine the correction factor relating the activity measured on a filter to the activity in the ambient air. The activity size distribution is that of airborne particulate iodine determined after the Fukushima accident in [15]. The particle size distribution was lognormal with AMAD = 0.42 and GSD = 3.5.

Particle size interval [µm]	Fraction of activity	Inlet efficiency	Collection efficiency	Sampling efficiency	Correction factor
< 0.49	0.54	1.0	0.9	0.9	0.600
0.49 – 0.95	0.20	1.0	1.0	1.0	0.200
0.95 – 1.5	0.13	1.0	1.0	1.0	0.130
1.5 – 3.0	0.08	0.95	1.0	0.95	0.083
3.0 – 7.2	0.04	0.89	1.0	0.89	0.043
> 7.2	0.01	0.83	1.0	0.83	0.012
					1.071

1.3.3. Temporal behavior of activity concentration

The spatio-temporal distribution of activity concentration of airborne Cs-137 following the Fukushima accident was studied in [17]. From the gathered data several plumes could be identified, each passing the sampling stations in a matter of hours (
Figure 6).

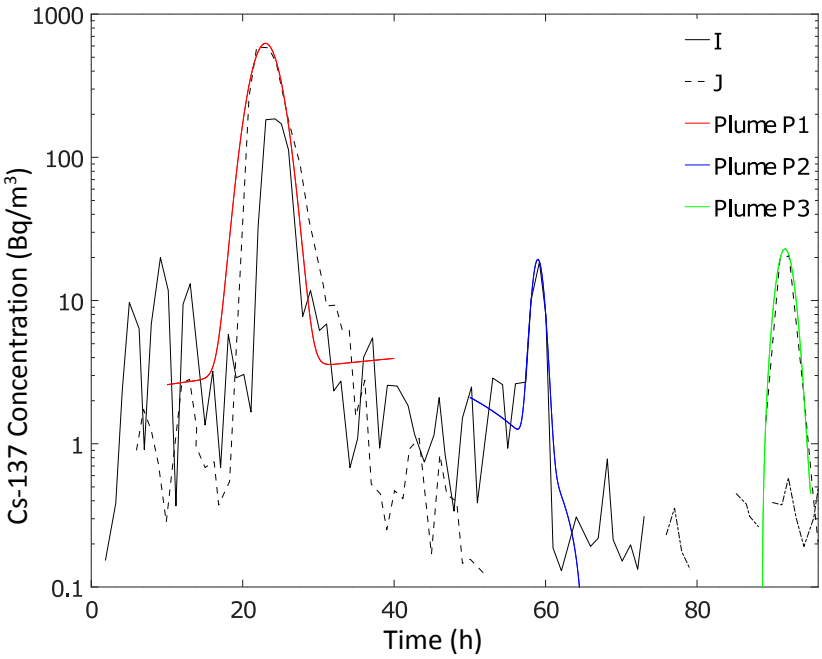

Figure 6 – Plumes P1, P2 and P3 identified from the samples collected at two different locations designated with letters I and J. The figure is based on the data from [17]. The plumes P1, P2 and P3 are Gaussian functions occurring at 23, 59 and 60 hours after 00:00 March 12. 2011, with standard deviations of 1.9, 0.7 and 1.2 hours.

As noted earlier, the time-integrals of the average activity concentration and time dependent activity concentration agree when the radionuclides in question have a suitably long half-life. For radio-nuclides whose half-lives are shorter than the sample collection period this does not apply. Under the long collection period of routine monitoring, such nuclides are for example I-131, Te-132 and La-140. An epistemological uncertainty is introduced by the lack of knowledge on the temporal behavior of the activity concentration.

Under the assumption of constant activity concentration during the sample collection, the accumulation of activity on the filter is given by

$$a = QC\int_{t0}^{t1}\left(1-e^{-\lambda t}\right)dt = \frac{QC}{\lambda}\left(1-e^{-\lambda t_s}\right). \qquad (14)$$

More generally, for time dependent activity concentration, the accumulation of activity is given by

$$a = \int_{t0}^{t1} Q(t)C(t)\left(1-e^{-\lambda t}\right)dt. \qquad (15)$$

Equation (15) is the convolution of activity production $Q(t)C(t)$ and the radioactive decay of the radionuclide $\left(1-e^{-\lambda t}\right)$

$$a = Q(t)C(t) \otimes \left(1-e^{-\lambda t}\right). \qquad (16)$$

The effect of different temporal behavior on the time-integral of the activity concentration can now be conveniently investigated numerically by requiring Equations (14) and (16) to agree at the end of sampling (Figure 7).

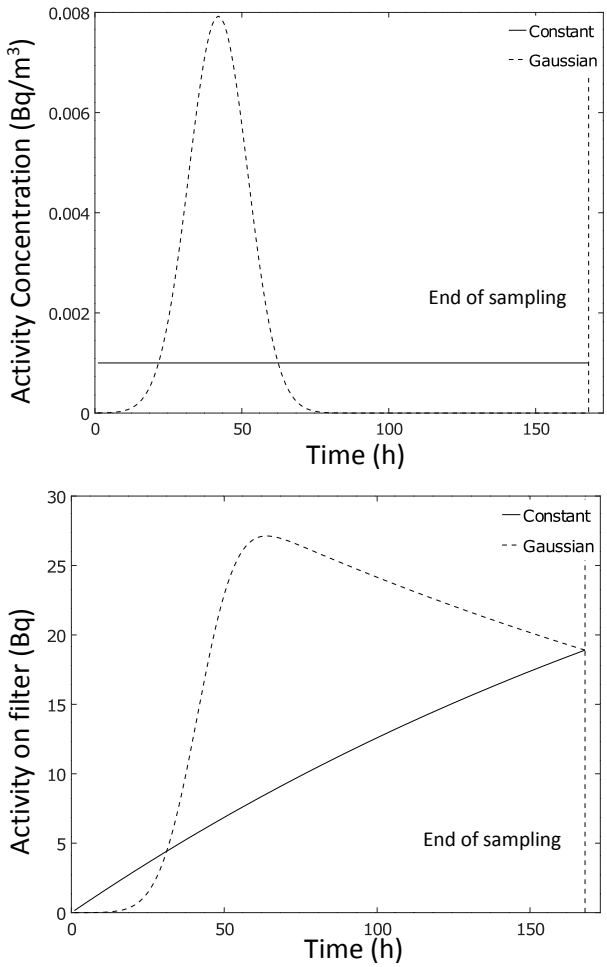

Figure 7 – The constant and Gaussian time profiles of activity concentration (Top) and the resulting accumulation of activity on the filter under constant flow rate conditions (Bottom). Half-life used is 8 days, the half-life of I-131. The end of sampling is denoted by the vertical dashed line at 168 hours (7 days). The maximum of the Gaussian occurs at 42 hours into sampling and 68 % of the activity concentration time integral is contained in a period of 20 hours.

The difference between the time integrals of activity concentration of constant and Gaussian time profiles depends on the occurrence time of the maximum of the Gaussian, and also on its width. For a given time profile, the difference is completely determined by the ratio of the half-life of the radionuclide to the sampling time.

Time integrals $I_{C(t)}$ of Gaussian and I_C of constant activity concentrations producing equal activity on a filter are compared in Table 4 for some short lived nuclides contributing significantly to dose rate (see [18], Figure 8 -from Wikimedia Commons), or released in a significant amount in an accident.

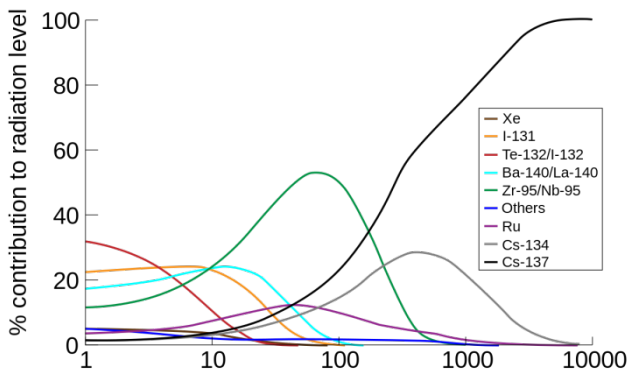

Figure 8 – Contributions of different radionuclides to absorbed after the Chernobyl accident.

Table 4 – Comparison of the time integrals of activity concentration under different time-profiles. Each time profile $C(t)$ shape is Gaussian with 68 % of the activity concentration time integral contained in a period of 20 hours around the occurrence of the maximum.

Sampling time t_s	**Time profile** $C(t)$ occurrence of max	**Ratio of time integrals** $I_{C(t)}/I_C$		
1 week		**I-131** ($t_{1/2}$ = 8 d)	**Te-132** ($t_{1/2}$ = 3.21 d)	**Np-239** ($t_{1/2}$ = 2.35 d)
	42 hours	1.18	1.61	1.99
	68 hours	1.02	1.10	1.19
	126 hours	0.87	0.75	0.71
1 day		**Te-132** ($t_{1/2}$ = 3.21 d)	**Np-239** ($t_{1/2}$ = 2.35 d)	**I-133** ($t_{1/2}$ = 20.8 h)
	6 hours	1.02	1.03	1.09
	12 hours	1	1	1
	18 hours	0.98	0.97	0.93

1.4. Uncertainty of average activity concentration

The uncertainty of activity, volume and the correction factor ξ are propagated from Eq. (13) to form the relative uncertainty of the average activity concentration

$$\frac{u(C)}{C} = \sqrt{\left(\frac{u(a)}{a}\right)^2 + \left(\frac{u(V)}{V}\right)^2 + \left(\frac{u(\xi)}{\xi}\right)^2} \qquad (17)$$

with the individual relative uncertainties of a, V and ξ given by Equations (7), (9) and (12).
The following example illustrates the evaluation of the uncertainty of average activity concentration, along with the effect of the epistemological uncertainties discussed in sections 1.3.2 and 1.3.3.

Example 2

Consider an automatic sampling station with 24 hour collection cycle and a plume containing I-133 passing it. The maximum of the plume is reached in 6 hours into the collection, with 68 % of its activity contained in 20 hours (the rest will spill over to the previous and next collections performed by the sampler). The I-133 in the plume exhibits the activity size distribution of AMAD = 0.42 and GSD = 3.5 (as in Table 3). The activity in the collected sample is measured with a Germanium detector. The average activity concentration is determined using Eq. (13), yielding 144 µBq/m^3. The uncertainty is determined by Eq. (17), with the individual uncertainties from Equations (7), (9) and (12), yielding relative uncertainty to be (assuming the relative uncertainty of the activity fractions to be 15% and th relative uncertainty of sampling efficiencies to be 8 %)

$$\frac{u(C)}{C} = \sqrt{0.08^2 + 0.02^2 + 0.38^2} = 0.39.$$

The result is then 144 µBq/m3 (+/- 39 %, k = 1). The time integral of activity concentration would be underestimated by a factor of 1.09. If Eq. (1) was used to determine the average activity concentration (neglecting the correction factor), the result of average activity concentration would be 134 µBq/m3 (+/- 39 %, k = 1) and the time integral would be underestimated by a factor of 1.17.

1.5. Speciation of Iodine

In the early phase of a nuclear accident, radioiodines are the most important radionuclides in regards to health impact [19]. Routine discharges originate from nuclear fuel processing and effluents from nuclear reactor operation. With increasing medical applications of radioiodine [20], releases from radiopharmaceutical production units and hospitals have also to be considered [21]. Radioiodines released to the atmosphere occur as particulate iodine (aerosol), molecular iodine I_2 (gas) and methyl-

iodine CH_3I (gas), each with a high dose coefficient because of their affinity to be accumulated in the thyroid gland. Most uncertainties in dose assessment results from the lack of knowledge concerning the partition between these three varieties, and the particle size of the particulate form. Each form has a different inhalation dose coefficient; whereas the gaseous forms have the highest (see Table 5). Besides inhalation, another important pathway, the deposition to vegetation (as input into the human food chain) is depending on the physio-chemical form: again the elemental Iodine shows the highest value (Table 5). The knowledge about the iodine speciation should be available at the place of potential exposure/contamination as the partition is not stable but changing with the meteorological conditions during the iodine dispersion in the atmosphere.

Table 5 - Inhalation dose coefficients for adults and 1 year old children and deposition velocities of several physio-chemical forms of ^{131}I.

Physical-chemical form	Particulate (AMAD= 1 µm; Absorption type F)	Gas Elemental (I_2)	Gas Organic (CH_3I)	Reference
Effective dose adult child (1 year) (Sv Bq^{-1})	7.4 E-09 7.2 E-08	2.0 E-08 1.6 E-07	1.5 E-08 1.3 E-07	ICRP Pub. 71 (1995)
Thyroid Equivalent dose adult child (1 year) (Sv Bq^{-1})	1.5 E-07 1.4 E-06	3.9 E-07 3.2 E-06	3.1 E-07 2.5 E-06	ICRP Pub. 71 (1995)
Deposition velocity to grass (m s^{-1})	1.0 E-03	2.0 E-02	1.0 E-04	Heinemann and Vogt (1980) [22]

Monitoring programmes for radionuclides were set-up in the 1960s in the framework of global fallout surveillance of nuclear tests in the atmosphere. Radioiodines were not in the focus of interest because of their short half-lives and the long residence times in the stratosphere where most of the radionuclides were injected after nuclear tests. After gamma-spectrometry replaced gross beta counting and NaI measurements, nuclide specific information (of e.g. ^{131}I) became available in such programmes, which continued with the development of nuclear industries. Sample media were aerosol filters and deposited material, which gave no specific information about the gaseous parts. There was also built an International Monitoring System (IMS) in support of the Comprehensive Nuclear Test Ban Treaty Organization (CTBTO). Here again (besides xenon isotopes) aerosol particles are monitored. The importance of gaseous radioiodines became obvious in the aftermath of the Chernobyl accident, where in many specialized laboratories a dominant proportion of gaseous radioiodine was measured in the Chernobyl plume (e.g. [23]). The aerosol size distribution of ^{131}I was different from other condensation type particulate radionuclides, and it was speculated that the high gaseous portion which can attach to the local aerosol might be the reason [24]. As consequence of the Chernobyl accident many national monitoring networks were built up, in which a part of the samplers were equipped with installations for trapping gaseous components of iodine. Therefore more information on the gaseous part of the Fukushima plume was available in 2011 and after. As an

example, the German Weather Service (DWD) operates today besides 48 high volume aerosol samplers about 40 iodine samplers which trap separately elemental and organic iodine. The flow rate is in the range 1-3 m^3 h^{-1}, allowing sufficient contact time with the adsorbent and resulting in a detection limit of below 5 mBq m^{-3} during routine operation [25].

During the Fukushima accident in European networks about 210 pairs of particulate and gaseous fraction were acquired in a single measurement period. In mean, about 77% of the total ^{131}I has been in gaseous form [26]. However, the variability in time was quite high: at one station it ranged from 79% to 90%, which makes a prediction from the aerosol part to the total iodine concentration quite uncertain [26]. Recent measurements of ambient ^{129}I in the vicinity of the La Hague reprocessing plant showed an even higher gas-to-particle ratio, ranging from 25 to 50 [27]. It can be explained by the high efficiency of the aerosol filtration in the stack of the plant: the release is mainly gaseous elemental iodine (I_2).

Due to the highly fluctuating proportions of the various forms of radioiodine, separate sampling of gas and particulate fractions are necessary for reducing the uncertainty in determining the total iodine concentration. In regard to dose assessment and the entry into the food chain, in addition separate sampling of the elemental and organic forms of the gaseous fraction is necessary for reducing the uncertainty. Usually aerosol samplers operate at a much higher flow rate compared to samplers for gaseous iodine. Therefore, the detection limit of the gaseous samplers is usually lower as for the aerosol samplers, even despite the higher gaseous proportion [28]. For that reason, it can be considered in monitoring networks to operate the samplers for gaseous iodine in emergency situations only.

1.6. Air monitoring systems and programs

Practically all radioactivity monitoring of the air in Europe is carried out using stationary samplers. Sample collection time ranges from one week to 24 hours for particulate samples. One month is common for gaseous samples, with the reservation for tightening the sample change interval in the event of significant detection of artificial radionuclides. Particulate samples are collected using a wide variety of filters. Glass fiber, polypropylene and Petrianov filters are widely used. Gaseous samples are collected using triethylenediamine (TEDA) impregnated activated charcoal with the specific aim of collecting gaseous iodine. The detection limits for particulate samples are generally on par with the values given in Table 1, iodine is discussed above. Radiochemical processing and alpha spectrometry are used to analyze plutonium isotopes and other trans-uraniums on the samples. Some stations are equipped with specialized equipment for the collection of noble gases.

1.6.1. Capabilities of monitoring systems

In order to assess the capabilities of air monitoring systems, the filters and samplers used to carry out the activity have to be characterized. The inlet efficiency of the sampler imposes an upper bound for the sampling efficiency of large particles (Figure 4). The collection efficiency of the filter imposes a similar bound for the collection of smaller particles (Figure 3). The location of the sampler has a large effect also, as evidenced by the dependence of the inlet efficiency on wind speed (and direction for directed samplers). The sampler should be located on a clear area far away from buildings and other obstacles that may have an effect on the air flow to the inlet. Determining the efficiencies of a monitoring system experimentally is a considerable task. Simulations made with CFD (complex fluid dynamics) software can be used to characterize samplers and to guide in the selection of a suitable location. Cascade impactors, aerodynamic particle sizers and electrical mobility analyzers can be used to determine the size distribution of particles. A cascade impactor allows for the measurement of the activity size distribution.

1.6.2. Quality control procedures

To ensure the correct computation of sampled volume, the flow rate measurement method of an air sampler should be calibrated with traceability to a national standard. Once calibrated the flow rates of the sampler should be monitored continuously. The tubings connected to the pressure difference measurement are subject to degradation with time and their connections may loosen due to vibration, leading to erroneous calculation of sampled volume. In order to ensure the proper operation of the device, it is suggested to monitor

- The pressure differences across the flow rate measurement orifice.
- The flow rates during sampling.
- The collected volumes.

The monitored values can be written down on a sampling data form (along with the sampling start and end dates, sampling time and sampled volume, which are all required to obtain a result) accompanying the sample. It is also possible to transmit these data electronically and store them in a database.

A sampler will require periodic maintenance (such as replacing the bearings of the pump) and a periodic check of the flow rate calibration is recommended. A period of five years should be sufficient for maintenance and calibration check.

1.6.3. Data exchange and harmonization

There are several European communities and instances for data exchange of air monitoring results, but many are either inactive, with low numbers of participation and low frequency of updates, or not appropriate for data exchange between countries and other interested parties. Some instances for data exchange are the European Radiological Data Exchange Platform (EURDEP), the Council of Baltic Sea States (CBSS) , the Ring of Five (Ro5) and the Radioactivity Environmental Monitoring (REM) database maintained by the Joint Research Centre (JRC). Additionally, the IAEA has plans to include air concentration results in its International Radiation Monitoring Information System (IRMIS).

The EURDEP collects air monitoring results from its participants, who send their results to a server as files of specific format. The results can then be browsed by any interested parties. The data exchange on air monitoring results sees considerably less participation than the corresponding data exchange on ambient dose rate (see section 3.2.4). The platform for browsing the ambient dose rate results was updated, providing a practical and easy to use map interface for the data display. The platform for accessing the results of the air monitoring was not updated however, and suffers from a complex and difficult to use interface. The data that can be browsed by this system also seems to be of low quality, with a low rate of participation.

The CBSS exchanges airborne radioactivity data between its 11 member states: Finland, Germany, Norway, Estonia, Sweden, Denmark, Poland, Latvia, Lithuania, Iceland and Russia. The data exchange format is the same as in EURDEP. Data is exchanged by each of the participants making their results available on their server. It is then up to the other participants to retrieve the data they are interested in. There is no central server to archive for the results, and no interface to browse them. The activities of the CBSS in the exchange of air monitoring data have gradually declined.

The Ro5 data exchange is informal, taking place on a mailing list maintained by the Swedish Defence Research Agency FOI. Detections of artificial radionuclides and alerts of releases or other interesting events are posted to the list by the participants of the Ro5. The participants are experts communicating their findings personally under no obligation or formal agreement. For events of longer duration, the Ro5 has established a reporting form as an Excel file, allowing multiple observations to be communicated more precisely. The Ro5 has been shown to be a quickly reacting source for high quality data in air monitoring. The results posted on the list have been used in collaboration to publish articles on notable events, such as the European

detections of the releases from Fukushima [26], the Europe-wide detections of I-131 in 2017 [21] and the detections of ruthenium made that same year [29]. Apart from such collaboration, the use of the data of Ro5 can be problematic due to the lack of formal agreements. There is also no central database or archive where the results could be browsed.
The REM database is used to store the radioactivity monitoring data provided by the EC Member States under the Euratom Treaty. On air concentration data, only Be-7 and Cs-137 concentrations are reported. The results are submitted yearly by the participants using a software tool provided by the JRC. The REM database data is of high quality, but only contains the air concentration data on two isotopes.
Harmonization is desired in all of the above data exchange instances. Due to differences in practices among the participating authorities and institutions, the approach taken by the JRC for the REM database or the Ro5 for its reporting form is the most usual; the result units, uncertainty types, coverage factors, MDC calculation methods and other assumptions are defined on the report format in order to ensure the correct interpretation of the results. This approach does not, however, allow the direct comparison of the raw reports, instead, they must first be transformed to a common form.

2. Monitoring of fallout

Deposition of radioactivity to terrestrial and aquatic environment is monitored by using fallout collectors. This is usually done in conjunction with the monitoring of airborne radioactivity. Deposited activity is transferred into the food chain via uptake by plants from soil and water and therefore presents a pathway for internal exposure. Deposition of large amounts of long lived radionuclides will also result in a long lasting external exposure situation.

Radioactivity is deposited in the form of wet and dry deposition. In wet deposition the airborne radioactivity is scavenged and washed down from the atmosphere by precipitation. In dry deposition airborne particles are sedimented on a surface by gravitational settling (large particles, D_{ae} > 10 µm), impaction and, in the case of gases, adsorption. Wet deposition is by far the more efficient form of deposition.

A passive fallout collector works simply by letting the deposited radioactivity settle into a container placed under a collection bowl with an opening of known area. When the sample is changed, the collection bowl is washed with acid to ensure collected particles end up in the container instead of the collection bowl walls. The container is then transported to measurement. A bulk collector collects both wet and dry deposition in the container, but some fallout collectors can differentiate between wet and dry deposition by letting the radioactivity settle into different containers based on the readings from a precipitation sensor. The resulting deposition density of a given radionuclide is then determined by the amount of activity a of the radionuclide in the container and the area A of the opening as

$$D = \frac{a}{A}. \tag{18}$$

In Eq. (18) the collection efficiency of the deposition gauge is neglected.

2.1. Uncertainties in fallout monitoring

The uncertainty components of the deposition density result from the measurement of the opening area and the measurement of the collected activity. The uncertainty of the orifice area is negligible, and the uncertainty of the deposition density is dominated by the measurement of activity. Gamma and alpha spectrometry is usually used to measure the activity, along with liquid scintillation counting. Gamma measurement of the sample is subject to the same considerations as in the monitoring of

airborne radioactivity. The chemical preparation of the sample for Sr-90 and H-3 analysis presents uncertainties in addition to the spectrometry method.

The uncertainties of fallout monitoring mainly relate to the epistemological uncertainty due to the collection efficiency and lack of characterization of the collectors. The following properties are given in [30] for bulk samplers:

- Sedimenting liquid and solid particles must be collected comprehensively.
- Matter collected must not be liable to subsequent loss due to resuspension or chemical or biological transformation.
- Gases or fine aerosol constituents must not be deposited on the sampler in significant amounts.

2.1.1. Collection efficiency

Collection efficiency of a deposition gauge is defined as the mass (or in this case, activity) collected in the gauge per unit area per unit time divided by the activity deposition on the ground per unit area per unit time in the absence of the gauge. The collection efficiency of a deposition gauge placed horizontally on horizontal ground depends on wind speed, wind pitch relative to the plane of the opening, physical properties of the particles collected and the design configuration and installation of the gauge.

The amount of activity deposited in the collector should be related to the natural surface in its surroundings. As stated in [31], for dry deposition, especially for particles that do not deposit mainly by gravitational settling (i.e. particles of diameter smaller than 10 µm), the dependency on the properties of the collection surface and the aerodynamic properties of the collector pose a difficulty in the interpretation of the result. The collection efficiencies of deposition gauges are studied in literature such as [32] and [33], but they mainly concentrate on large particles ($D_p > 50$ µm). Nevertheless, they point to very poor collection efficiency of dry deposition for funnel or bucket type passive collectors. The effects of atmospheric conditions are large even at moderate wind speeds around 1 – 3 m/s. An inverted Frisbee shape for a collector is suggested in [33] and found to have superior efficiency to funnel and bucket type collectors.

The collection of wet deposition is similar to the measurement of precipitation using a precipitation collector. Estimation of potential errors of precipitation measurements are given in [34], where evaporation, adhesion, non-horizontal aperture, splash effect are each determined to cause an error of around 0.5 % - 1 %. The most significant source of error is determined to be flow distortion, which is estimated between 5 % and 80 %. The airflow around the collector is distorted, causing small particles and rain drops to follow the streamlines and avoid entering the collector. A detailed discussion of these factors is given in [30]. In [35] the following error terms are listed for precipitation measurement:

a) Error due to systematic wind field deformation above the gauge orifice: typically 2% to 10% for rain and 10% to 50% for snow;
b) Error due to the wetting loss on the internal walls of the collector;
c) Error due to the wetting loss in the container when it is emptied: typically 2% to 15% in summer and 1% to 8% in winter, for (b) and (c) together;
d) Error due to evaporation from the container (most important in hot climates): 0% to 4%;
e) Error due to blowing and drifting snow;
f) Error due to the in- and out-splashing of water: 1% to 2%;

g) Systematic mechanical and sampling errors, and dynamic effects errors (i.e. systematic delay due to instrument response time): typically 5% to 15% for rainfall intensity, or even more in high-rate events;
h) Random observational and instrumental errors, including incorrect gauge reading times.

2.1.2. Resuspension

Activity deposited in the ground can become airborne due to resuspension. Such resuspended radioactivity may then end up in the fallout collector when it eventually settles back to the surface as wet or dry deposition. The resuspended and subsequently deposited material will then contribute to the activity in the collected sample, but does not truly represent the fallout from airborne pollutants. Resuspension will affect the estimation of overall deposition of activity.

3. Monitoring of ambient dose rate

Continuous monitoring of gamma dose rate using stationary dose rate probes is an essential tool for emergency preparedness and response. The member states of the EU have agreed to establish early warning networks using dose rate probes and to exchange data from these networks [36]. Data are collected from all member states of the European Union and from additional countries: e.g. Bela-Russia, Russia, Ukraine, Turkey, and Canada.

Ambient dose rate consists of different components: The secondary cosmic radiation component (SRC) mainly depends on the latitude and height above sea level. In addition, the variability of air pressure causes variability in SCR contribution to dose rate. Terrestrial component of dose rate mainly depend on natural radioactivity in soil or in building materials in the vicinity of the dose rate probe. In addition, natural activity – e.g. radon progenies - in air contributes to dose rate as well as artificial radioactivity from historical events (e.g. Cs-137 fallout from the Chernobyl accident).

Most dose rate probes used in European environmental monitoring networks are based on Geiger Müller counting tubes or on proportional counting tubes. Some few networks rely on scintillator based dose rate probes. In addition, modern scintillator based probes can deliver dose rate and additional spectral information [37]. These probes can help to assess nuclide specific dose rate contributions.

3.1. Uncertainties of external dose rate monitoring

3.1.1. Uncertainty due to probe characteristics

Measurement uncertainties of dose rate monitoring systems are in principal well understood. Different types of uncertainties are connected with

- probe response due to secondary cosmic radiation (SCR) component
- probe response depend on photon energies
- deviations from linear response (e.g. self-effect and saturation effects)
- influence of environmental conditions (e.g. temperature)
- statistical uncertainties

Typically, the characteristics of specific dose rate probes should be analyzed using type testing procedures. EURADOS Working group 3 organized different inter-comparison experiments for dose rate probes to compare and exchange data from dose rate probes used in European early warning networks [38]. The outcome of these inter-comparison experiments focus on self-effect and probe response due to secondary cosmic radiation (SCR) component and on different photon energies. In addition, BfS organized long-term inter-comparison experiments to investigate the influence of

environmental conditions [39]. Table 6 gives an overview on relevant source of uncertainties and methods and procedures to reduce uncertainties.

Typically, stationary dose rate monitoring use 10 minutes time intervals. Thus, statistical uncertainties of 10 minutes counting rates should be low. For example, Geiger-Müller based probes of German monitoring network typically deliver 1000 counts per 10 minutes with a statistical uncertainty of about 3% (standard deviation). Regarding typical natural fluctuation of out-door dose rate, this enables detection of increase dose rate in the order of 0.02 µSv/h. Very helpful is the calculation of net dose rate from observed dose rate and background level of the specific monitoring station [40].

Regarding the needs for emergency preparedness and response, dose rate probes have to be able to detect enhanced levels of radiation from background level up to an upper measurement level well above 1 mSv/h. However, most European early warning networks rely on dose rate probes with an upper measurement level in the order of about 1 Sv/h.

Large efforts are made for characterization of different probes and detectors, discussing the needs of harmonization aspects and to introduce recommendations for dose rate monitoring networks on the European level [41]. On national level, harmonization aspects have to be regarded, if data from national and different regional networks are used [40].

Table 6 - Overview on sources of uncertainties for data from stationary monitoring and methods to reduce uncertainties.

Sources of uncertainties	**Type of uncertainty**	**Methods and procedures**
Probe characteristics: • probe response depend on photon energies and secondary cosmic radiation • deviations from linear response (e.g. self-effect and saturation effects)	Parameter; model	Type testing procedures; calibration procedures; test procedures of individual probes
Probe characteristics: statistical uncertainties	Stochastic	
Influence of environmental conditions • e.g. temperature		Type testing procedures; long-term inter-comparison experiments
Probe location and site characteristics	Epistemological	Site characterizing procedures

3.1.2. Uncertainty due to probe positioning

Dose rate probes should be fixed under standardized conditions. This would help to compare measured dose rate at different locations. For example: Germany use the concept of ideal site: Dose rate monitoring stations should be located on infinite flat grassland on natural undisturbed ground. Probes have to be installed at the height of 1 m. In case of an accidental release, the ideal site concept helps to assess activity deposited in the vicinity of the probe from increase of dose rate [42].

However, real monitoring stations deviate from this ideal site concept. Disturbing objects in vicinity of probes should be characterized and the impact of these objects on measured dose rate should be known (Figure 9). A MetroERM deliverable discuss classification of different sites with respect to representativeness of measured data [43]. The proposed site characterisation technique is clearly linked to network harmonisation aspects and to an uncertainty model for ambient dose rate measurements under environmental conditions including disturbing contributions from non-standard

probe locations. The proposed technique is adequate for post-release situations, where freshly deposited activity dominates the total dose rate.

In the framework of the MetroERM project a questionnaire was circulated with the focus on aspects of the site characterization [43]. 14 countries returned the questionnaire.

Table 7 shows dose rate monitoring networks operated in 11 European countries follow the concept of an ideal site (Belgium, Finland, Germany, Greece, Irland, Italy, Lithunia, Netherlands, Poland, Slovakia and Switzerland). The other 3 countries (Austria, Latvia and Norway) do not use this concept for site selection. Some networks, especially in northern Europe, use elevated probe positions. Other networks use dose rate probes fixed on walls and on roof.

Table 7 – Overview on site characterization aspects derived from the responses of 14 countries.

Country	Concept of ideal site / excluding criteria	Height above ground	Documentation available	Sites with 5 m flat grassland in the vicinity of the detector
Austria	no	1 m	Buildings	
Belgium	not really (5m grass) always flat natural ground	1 m	4 probe photos	80%
Finland	lawn, no obstacles in 10 m (asphalt, walls, forests)	2 m	no systematic documentation	40 %
Germany	grassland in 20 m / no walls, roofs	1 m	systematic incl. photos	about 80 %
Greece	grassland in 10 m	1 m	no documentation	100%
Ireland	grassy area (flooding < 30 cm) / no building in 20 m	1 m	360° photos record	93%
Italy	not really (5m grass)	1.5 m	some photo	45%
Latvia	No		No	
Lithuania	smooth grassland / no building in 20 m	1 m		42%
Netherlands	Yes	1 m	Yes	15%
Norway	No	3 m	No	64%
Poland	flat grassland / no roofs, walls, trees, tall buildings	1,5 m	no documentation	77%
Slovakia	meteorological gardens	1 m	Yes	94%
Switzerland	grassland in 20 m / no walls, roofs	1 m	digital documentation	93%

In the MetroERM project a simple site characterization method was developed [43]. Basically this method divides the area around a given ADER monitoring station into four zones, where each zone contributes to the observed dose rate contribution by approximately 25% in case of a fresh deposition of radionuclides on flat grassland:

- zone 1: circle of 3 m surrounding the probe,
- zone 2: circular ring between 3 and 7 m,
- zone 3: circular ring between 7 and 20 m,
- zone 4: circular ring between 20 and 100 m.

The total area of each zone is divided into sub-areas of four “surface types”:

- type 1: grassland, agricultural used areas, areas with low vegetation,
- type 2: sealed areas (e.g. streets, paved areas),
- type 3: shielded areas, buildings, areas covered with water,
- type 4: trees, bushes, forest.

To apply this system, a documentation is required to characterize each station providing information on the surface types in each of the 4 zones. An excel sheet was developed to obtain an uncertainty interval and a bias depending on the meteorological situation (dry or wet conditions during the passage of the contaminated cloud) for each specific location.

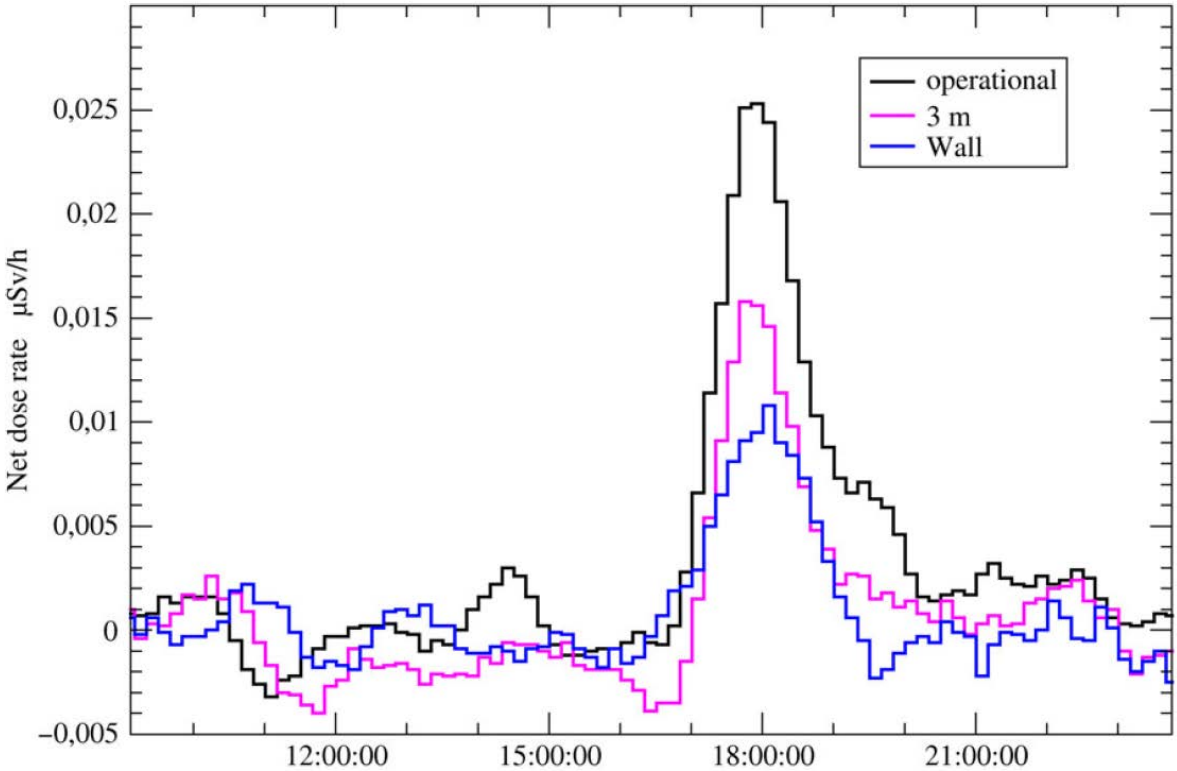

Figure 9 – Comparison of net ADER readings of three stations installed in Neuherberg during a precipitation event in June 2014. The data in black are from a detector installed at the free field, the data in magenta from a detector mounted in a distance of 3m away from a building and data in blue from a detector directly in front of the wall of the building.

Table 8 – Example of site characterizations of different locations

Probe is mounted	Lower uncertainty	Upper uncertainty	Dry bias	Wet Bias
1 m above flat grassland (ideal site)	0.80	1.20	1.00	1.00
On a wall of a building 1 m above flat grassland	0.40	0.70	0.55	0.55
In 3 m distance from a building	0.56	0.90	0.73	0.73
On a roof	0.35	1.10	0.57	0.76
Above flat grassland with forest in a distance of 10 m	0.91	5.52	3.10	1.30
1 m above flat grassland (50%) and sealed area	0.45	1.10	0.65	0.80
Typical site in the German network	0.75	1.96	1.33	0.97
Minimum	0.35		0.55	0.55
Maximum		5.52	3.10	1.30

In Table 8, the results are show for for 7 different location types. As can be seen, at a typical location in the German ADER monitoring network the measured dose rate can be enhanced by a factor of 1.33 (dry bias) under dry conditions and the uncertainty due to the site characteristics ranges from the observed dose rate multiplied by a factor of 0.75 (lower uncertainty limit) and a factor close to 2 (upper uncertainty limit). Under wet conditions, the measured ADER value can be reduced by a factor of 0.97 (wet bias) with the same uncertainty limits. The uncertainty from statistics and the physical properties of the probe was estimated as 20 %.

In case that the site conditions are not known, the lower and upper limit of the uncertainty as well as the observation bias can be seen in Table 8. In this case, a conservative approach would be to use the minimum and maximum uncertainty limits from Table 8 leading to the lower uncertainty limit of 0.4 and the upper limit of 5.5.

3.2. External dose rate monitoring systems and programs

3.2.1. Capabilities of monitoring systems

In case of an accidental release, measured ambient dose rate help to assess external dose. Stationary ambient dose rate monitoring data give direct information on the external dose for people living in the vicinity of the probe location. In principle, activity deposited on the ground can be assessed from measured ambient dose rate using knowledge about nuclide composition.

Stationary dose rate probes continuously measure the level of external exposure. Most systems are able to deliver 10 minute averages and additional 1 min data. Thus, the monitoring systems can be used for routine monitoring and early warning purpose and deliver very useful information in case of an emergency situation. IAEA proposed three operational intervention levels (OIL) for measured dose rate:

- OIL 1: 1000 µSv/h indicates the need for evacuation of exposed people
- OIL 2: 100 µSv/h indicate the need for sheltering of exposed people
- OIL 3: 1 µSv/h indicate affected area due to a major release scenario

Stationary dose rate monitoring probes should have an upper dose rate limit well above OIL 1. On the other hand, they should be able to indicate an increase in external radiation of more than 0.02 µSv/h within 10 minutes. Thus, released artificial radioactive material could be indicated. For example, ambient dose rate would increase by this amount due to 4 kBq/m^2 fresh deposition with Cs-134 or 10 kBq/m^2 fresh deposition with Cs-137.

Most European monitoring networks providing data to the data exchange platform EURDEP fulfil the mentioned conditions. Most of these networks use modern dose rate probe types with two different Geiger Müller counting tubes to meet the measuring range between background level and very high dose rate up to some Sv/h. These probe types are able to compare counting rates to indicate false readings or enhance level of dose rate. Other networks use probes based on proportional counters or on scintillators.

Table 9 – Typical natural influcences impact on ambient dose rate

Influence	Duration	Effect
Snow cover	**Weeks/months**	**Decrease of terrestrial component**
Soil moisture	Days/weeks	Variability of terrestrial component
Radon progeny (air)	Hours/days	Variability of terrestrial component (activity in air)
Rain events	**Hours**	**Increase of terrestrial component (Bi-214 and Pb-214)**
Air pressure	Hours /days	Variability of cosmic component

Obviously, it is very helpful to know the natural variability of measured ambient dose rate (Table 9). Routine data validation procedures using information about these natural processes indicate problems with dose rate probes and are useful to detect abnormal situations.

Some monitoring networks use modern spectroscopic dose rate probe based on NaI or $LaBr_3$ scintillator detector systems. These systems are able to detect artificial radiation and /or enhanced

level of dose rate due to wash-out of Radon progeny (Bi-214 and Pb-214) during natural rain events (Figure 10). Such probes improve the potential of early warning of dose rate monitoring networks. In case of an increase of dose rate, spectral information can be used to identify an artificial radionuclide (Figure 11). In case of an accidental release, such probes may help early identification of nuclide composition of released radioactive material (Figure 10).

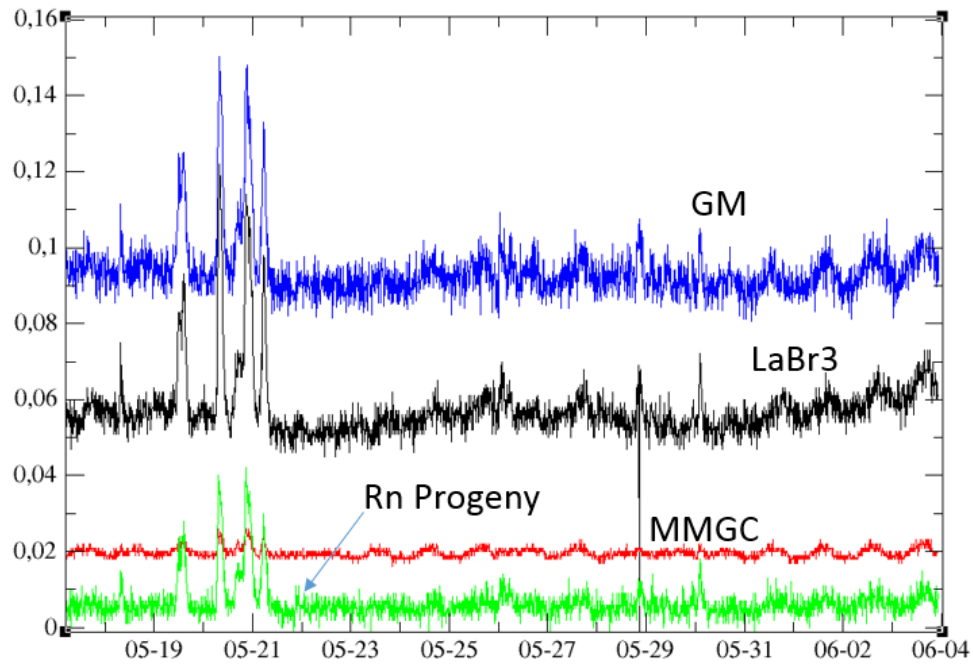

Figure 10 - Observed dose rate (in µSv/h) from GM based probe and calculated dose rate from a LaBr3 probe at the same site. The LaBr3 probe provide calculated dose rate from Rn-222 Progeny and a man-made gross count (MMGC) indicator for artificial radiation.

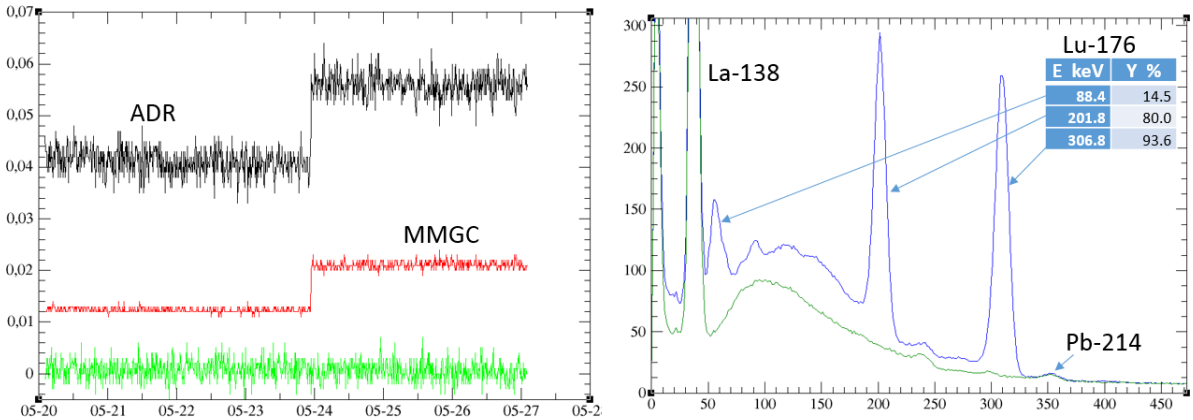

Figure 11 - Time evolution of calculated dose rate from a LaBr3 probe, calculated dose rate from Rn-222 Progeny and a man-made gross count (MMGC) indicator for artificial radiation (left). Spectra of LaBr3 probe enable identification of natural Lutetium with primordial Lu-176 (right).

3.2.2. Design of stationary monitoring networks

For emergency preparedness and response aspects, the design of stationary monitoring networks should meet the following issues:

The spatial distribution of probe locations should be linked to the needs of national thread analysis. Stationary probes should be regularly distributed to enable an early warning function. In addition, probes should be located in populated area with the risk of enhanced dose rate levels.

Monitoring data from monitoring stations should be timely transmitted to the central units. Enhanced dose rate data should be transmitted with no delay of time. Additional information on local precipitation events should be available – e.g. from precipitation sensors or from precipitation radar systems. This helps to check, if enhanced dose rate levels are due to natural precipitation events.

3.2.2.1. Monitoring networks in the vicinity of NPPs

Near nuclear power plants, the locations should refer to the emergency planning zones [42]. For example, Figure 12 shows the probe locations of a national and an additional regional monitoring network. In the Precautionary Action Zone (5km), one probe is located in each 30° sector. Each 20° sector of the 20km Urgent Protective Action Planning Zone should be equipped with one or two additional stationary probes.

The monitoring data from all networks should be timely available for decision support systems (like RODOS). It should be possible to combine monitoring data with data from atmospheric dispersion calculations (Figure 12).

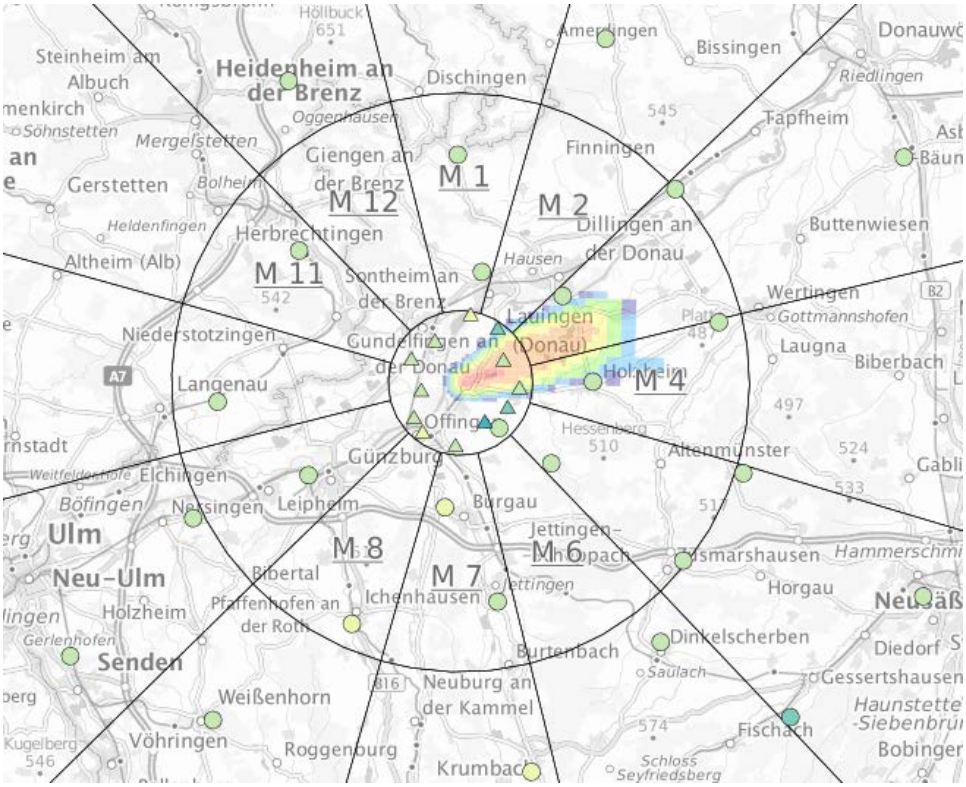

Figure 12 - Monitoring data from national (circles) and regional (triangles) monitoring network and prognostic dose rate data from RODOS in the vicinity of German NPP Gundremmingen.

3.2.2.2. Deployable dose rate probes

In most NPP release scenarios, the pre-release phase would last 10 hours or more. Thus, it would be possible to deploy additional mobile dose rate probes in this phase using first responder teams. Deployable probes useful for this type of application are operated independent from external power supply and use wireless data transmission techniques. Furthermore, integration of data from mobile probes in the database of the monitoring network is required.

The selection of the locations have to reflect information about the beginning and the duration of the potential release and information about atmospheric dispersion conditions. The selection process would benefit on the availability of a list of pre-selected locations with easy access conditions for deployment of mobile probes. Proper distributed deployable probes are helpful tools to improve the spatial resolution of the monitoring network e.g. in populated area at risk.

3.2.3. Quality control procedures

Quality control procedures for does rate probes are important to ensure a high quality functionality and to uncover possible failures. Type testing procedures are needed to characterize new probe types used in the monitoring network. Helpful measures to reduce uncertainty are test procedures for newly delivered individual probes.

3.2.3.1. Type testing procedures

Type testing procedures should be performed for each probe type used in a monitoring network. The procedure should use knowledge from metrological Institute (e.g. PTB).

Type testing procedures have to give information about:

- Dose rate measurement range (upper and lower limit) and typical parameters (self-effect, calibration factors, dead-time corrections …) used to calculate dose rate from primary data like counting rates.
- Angular response of probes
- Response of probes for different photon energies and for secondary cosmic component
- Impacts on environmental conditions on observed dose rate (e.g. influence of temperature)

3.2.3.2. Test procedures for individual probes

Standardized test procedures should be performed for each individual new probe and after exchange of probe components or probe firmware. Helpful to reduce uncertainties are the following standardized test procedures:

- Calibration tests with artificial sources (e.g. Cs-137) to check calibration factors
- Lead castle experiments to assess self-effect of individual probes
- Climate chamber experiments to identify probes with temperature drifts of observed dose rate

For example, BfS performs calibration tests with different Cs-137 sources to check probe readings at different dose rate levels between 10 µSv/h and 100 mSv/h.

Spectroscopic dose rate probes need additional information about parameters describing energy resolution, efficiency and energy calibration. Climate chamber test procedures using additional sources are helpful to investigate impacts of temperature on observed dose rate and spectral information.

3.2.3.3. Cyclic test procedures for individual probes

Additional cyclic quality assurance tests for each individual probe should be performed. Two types of test procedures can be used: The first method uses cyclic calibration checks at the measuring sites using an artificial source. This allows the inspection of the complete installation without taking the measuring system out of service for an extended period. If the test procedure identifies a failure, the probe on-site has to be changed. The second method uses laboratory test procedures for individual probes. On-site, probes to be checked have to be replace by properly checked probes.

Both methods can be used to compare long-term behavior of calibration factors of individual probes with their initial parameters. The cyclic test procedures can be completed with additional on-site measurements e.g. with reference devices.

3.2.3.4. Routine data validation procedures

Monitoring network operators should perform routine data validation procedures. Such procedures help to identify not proper working probes and can be performed in a two-step approach:

Step 1 rely on automatic pre-selection of suspicious data – e.g. dose rate data below or above specific levels for individual monitoring sites.

In step 2, pre-selected suspicious data are checked by experienced staff. Step 2 may be regularly performed on working days and should include experience on natural effects on dose rate (see Table 2.4) and information of typical problems of specific probe types. In step 2, experienced staff have to identify not proper working probes to be changed.

3.2.4. Data exchange and harmonization

On behalf of the European Commission, the data exchange platform EURDEP [43]collects and provides ambient dose rate data from all European early warning networks in almost real-time. Data are collected from all member states of the European Union and from additional countries: e.g. Bela-Russia, Russia, Ukraine, Turkey, and Canada.

The monitoring networks using a considerable number of different detector types and following different national policies. Monitoring network design of European countries differ significantly. Table 10 gives an overview on the spatial density of monitoring stations for European countries with active NPPs.

As already mentioned, EURADOS Working group 3 organized different inter-comparison experiments for dose rate probes used in different European countries. Information from these experiments are very useful to compare observed dose rate. This information is helpful for the comparability of these data and the correct interpretation of the data under natural background conditions (Figure 13). The correct interpretation of the data is especially important during a nuclear accident with trans-boundary implications.

Table 10 – Dose rate monitoring networks of countries with active NPPs: number of stations reporting to EURDEP platform in relation to total area and number of inhabitants [44].

	Stations	Area km²	Area/station km²	Inhabitants	Inhabitants/station
Belgium	125	30528	244	11142000	89136
Bulgaria	28	110994	3964	7305000	260893
Czech Republic	64	78864	1232	10515000	164297
Finland	266	338432	1272	5414000	20353
France	852	668763	785	65697000	77109
Germany	1887	357121	189	81890000	43397
Hungary	120	93036	775	9944000	82867
Netherlands	167	41548	249	16768000	100407
Romania	193	238391	1235	21327000	110503
Slovenia	27	20273	751	2058000	76222
Slovak Republic	134	49034	366	5410000	40373
Sweden	77	450295	5848	9517000	123597

Switzerland	168	41285	246	8139600	48450
United Kingdom	92	219331	2384	63228000	687261

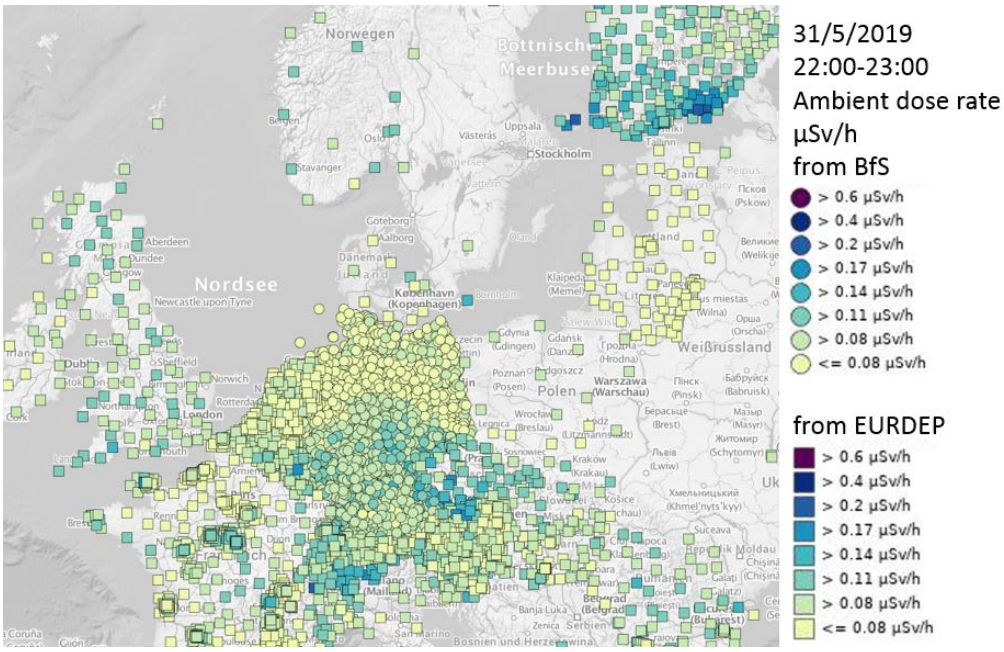

Figure 13 – Combination of observed dose rate from German monitoring network with data from EURDEP.

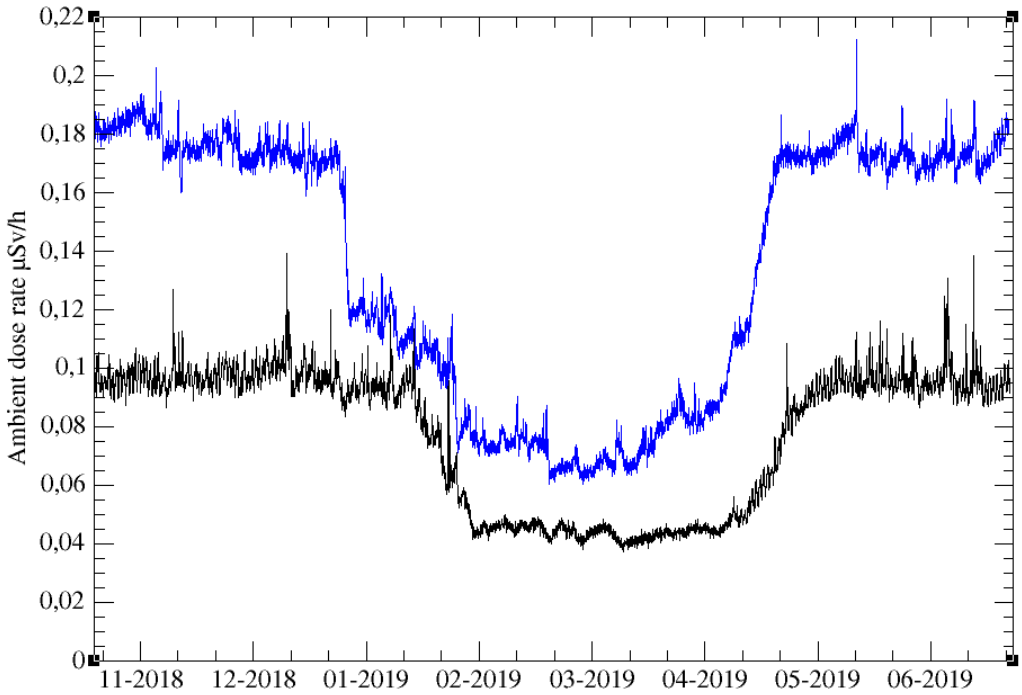

Figure 14 - Natural influences on measured on ambient dose rate (1 hour data) at station Reit im Winkel and Altenberg.

Data harmonization of dose rate data should be based on mechanisms to decompose the ambient dose equivalent rate in different components. Information is needed on the intrinsic background or self-effect and the response of the detector to secondary cosmic radiation. Using this information,

terrestrial gamma dose rate can be calculated and compared. In addition, the calculation of net dose rate from observed data is very helpful to compare data from different monitoring networks.

Data assimilation techniques used in decision support systems like RODOS [45]strongly depend on harmonized data. The physical characteristics of the detectors has to be complemented by an appropriate uncertainty budget.

4. Examples of national monitoring networks

In the following, we will take a more detailed look at examples of national radiation monitoring networks of Norway, Finland and Germany.

4.1. Norway

In Norway there are two different monitoring networks aimed at full time detection of radioactivity in air and air borne elements:

- Radnett: early warning network detecting dose rates in near real-time.
- Air filter stations – high volume samplers for radioactive isotopes.

For further surveys, the Norwegian Radiation and Nuclear Safety Authority (DSA) has two rainfall/precipitation collectors of the type RITVA 300. The one is located on the roof of the DSA's head office at Østerås and the other - at Svanhovd. DSA Svanhovd is also equipped with a car-borne monitoring unit (Møller et. al., 2018).

In addition, The Norwegian Civil Defense routinely conducts measurements of ambient dose rates at fixed stations for a national monitoring program to document levels of background radioactivity (https://www.dsa.no/temaartikler/91105/daglig-beredskap).

4.1.1. Radnett

Radnett is a nationwide alert network consisting of 33 automated stations that measure radioactivity in the surrounding area (Figure 15).

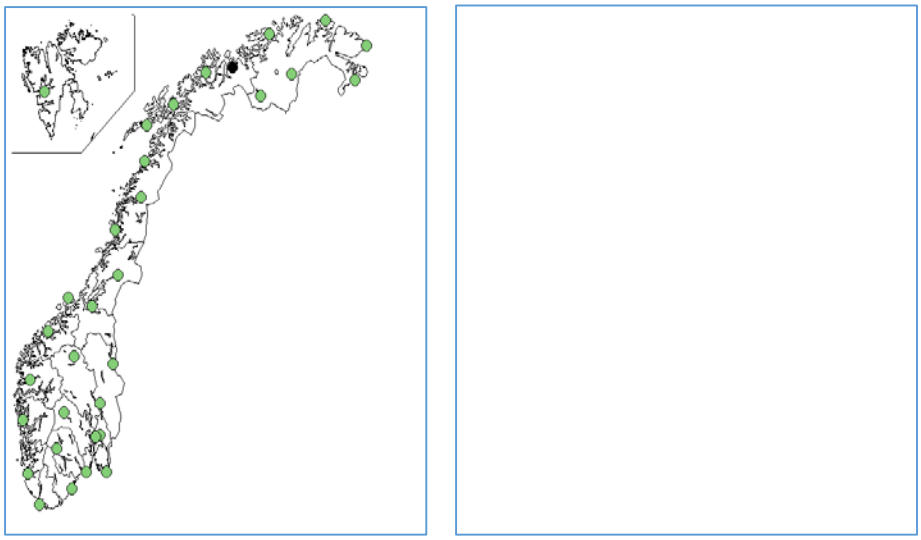

Figure 15 – Overview of Radnett stations located in Norway, including Svalbard (A); a Radnett station and a high-volume air sampler 'Snow White' at Svanhovd, North Norway (B).

The network was established in the years following the Chernobyl accident in 1986, and was upgraded and modernized in the period 2006-2008. The purpose of the network is to provide an early

warning/notification in the event of an unknown radioactive emission striking Norway. Furthermore, the measurements from the network will be an important part of the decision-making basis for the Crisis Committee for Nuclear and Radiological Preparedness at the early phase after the emissions occurred.

The stations consist of a detector for measurements of dose rates (µSv/h), a rainfall sensor and a data communication unit which transmits the information from the measurement sensors to the DSA. The detectors are antennae placed on a three meter high mast that stands on the ground, or on buildings. One detector measures radioactivity in the surroundings, the other detector is a rainfall sensor that detects whether there is a rainfall or not. This provides valuable information during an event as the ground becomes more contaminated by radioactivity when there is a rainfall. The precipitation information is also needed for verification of alarms caused by radon leaching/washout [46].

The information on radioactivity levels at all stations is available online at http://radnett.dsa.no.

4.1.2. Air filter stations

In Norway, there are six air filter stations today to monitor the air masses. Three of the stations are located in the north, two in the south and one in central Norway. There is also one air filter station located in Svalbard (Fig. 16). The filters in these stations collect dust from the air during a week, and are then analyzed at the Radiation Protection Laboratory. In addition, four of the stations are equipped with a detector of the same type as on the radar stations that can alert DSA if the radioactivity on the filter is too high.

All air filter stations have the same principle for taking samples of air, but vary somewhat in capacity and efficiency. Common to all stations is that large quantities of air are pumped through a special filter with high density where small particles (aerosols) are trapped (Table 11). The filter is changed every week and sent to DSA's laboratory for analysis [46].

Table 11 - Systems concerning sampling and analysis of radioactivity.

	Norwegian monitoring
Monitoring established	1993 (Svanvik)
Type filter	Whatman GF/A
Sampling period	1 week
Air volume	800 m^3/h
Option charcoal filter for uptake of gaseous iodine	Svanvik only
Instrument	Electrical cooled HPGE detector
Software	Maestro and GAMMA10
Filter treatment	Fold-and-roll
Analysis time (quick measurement)	1 hours
Analysis time (long measurement)	48 hours
Sampling beeker (analysis)	W1 (28 ml)
Monitoring of β-activity	No

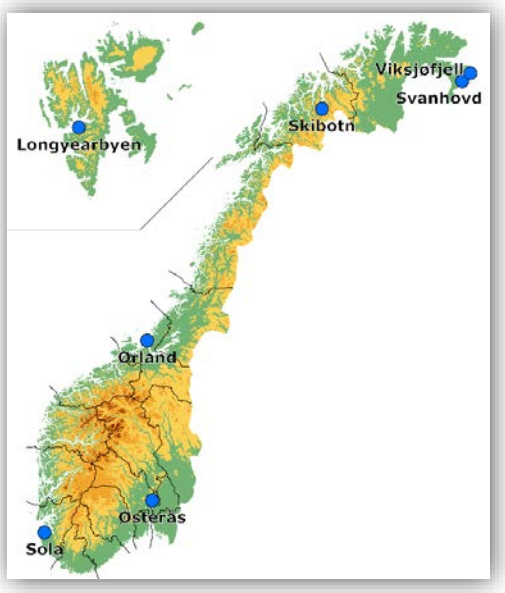

Figure 16 - Overview of the air filter stations in Norway, including Svalbard (DSA).

Some of the air filter stations are also equipped with a specially impregnated carbon filter that absorbs gaseous radioactive iodine. The carbon filter is replaced every month and analyzed in cases where radioactive iodine is detected on the particle filter or when a leak is suspected.

The stations are important for mapping radioactivity in the air and for assessing the magnitude and composition of emissions in the event of accidents and accidents. Corresponding stations are found throughout Europe, and cooperation between the countries makes it possible to detect any radioactive emissions [46].

4.1.3. Measurements by the Norwegian Civil Defense

The Norwegian Civil Defense's survey patrol service (RADIAC patrol) is an important part of the Norwegian nuclear preparedness. They also provide good reference measurements (background measurements) of radioactivity in the environment. The patrols are part of the national network for emergency measurements and carry out regular background measurements at around 350 fixed measurement points. The purpose of the measurements is to map the normal situation and to maintain the capacity for measurements in an emergency situation.

There are 126 nationwide patrols distributed across 20 districts. Each patrol comprises of a one patrol officer, two crew members and one reserve person. In addition to the regular surveys, the patrols could be activated at the request of the Crisis Committee for Nuclear and Radiological Preparedness, the county governors or the local emergency services in the civil defense district [47].

4.1.4. Assesment criteria and limit values

The Norwegian law defines a limit value concerning radiation stating that people should not be exposed to radiation so that effective dose exceed 1 mSv/y in addition to the background radiation [48].

The Norwegian Civil Defence has a limit value for 0.7 µSv/h for their RADIAC network (equal to approximately 10 times the background level) to alarm the DSA.

The RADNETT system triggers alarm to the officer on duty when exceeding 2 times the normal background in the area, measured as the average for the last 10 days.

4.2. Finland

Like its Norwegian counterpart, the Finnish national environmental monitoring program relies on two monitoring networks. A network of probes for monitoring the ambient dose rate and a set of air filter stations for monitoring the radioactivity in surface air. The dose rate monitoring network is called USVA. Additionally the nuclear power plants operating in Finland are required to have their own facilities for environmental monitoring. These facilities also consist of dose rate monitoring stations and air filter stations.

4.2.1. USVA network

The USVA network consists of 260 dose rate monitoring stations (Figure 17). All of the stations are equipped with a Geiger Muller tube and 25 stations with an additional LaBr3(Ce) spectrometer. The stations record the ambient temperature and precipitation in addition to the radiological data. The stations collect data in 10 minute intervals and communicate it to STUKs central server, where the data on the GM count rate, dose rate, environmental conditions and collected spectra can be browsed. The history and description of external dose rate monitoring in Finland is given in [49]. The general alarm value is 0.4 µSv/h, and additionally The central system USVA generates an alarm if the measured dose rate at a GM station exceeds the previous seven-day average by 0.1 µSv/h.

4.2.2. Air filter stations

The air filter stations used in the monitoring of airborne radioactivity are situated at 8 locations (Figure 17). Additionally total beta activity of air is monitored at 9 locations by the Finnish Meteorological Institute.

Table 12 – Summary of Finnish air monitoring.

	Finnish monitoring
Type of filter	Camfil A 500 GH
Sampling period	1 week particulate, 1 month gaseous. Helsinki 24 hour particulate.
Air volume	900 m^3/h, 500 m^3/h, 150 m^3/h
Activated charcoal cartridge	All locations, Helsinki only on demand.
Instrument	Germanium spectrometry
Software	Unisampo/Shaman
Filter treatment	Cut-and-stack
Analysis time (quick measurement)	3 hours
Analysis time (long measurement)	48 hours
Sampling beeker (analysis)	Rectangular 80 mm x 80 mm
Monitoring of β-activity	No (performed at 9 locations by the Finnish Meteorological Institute).

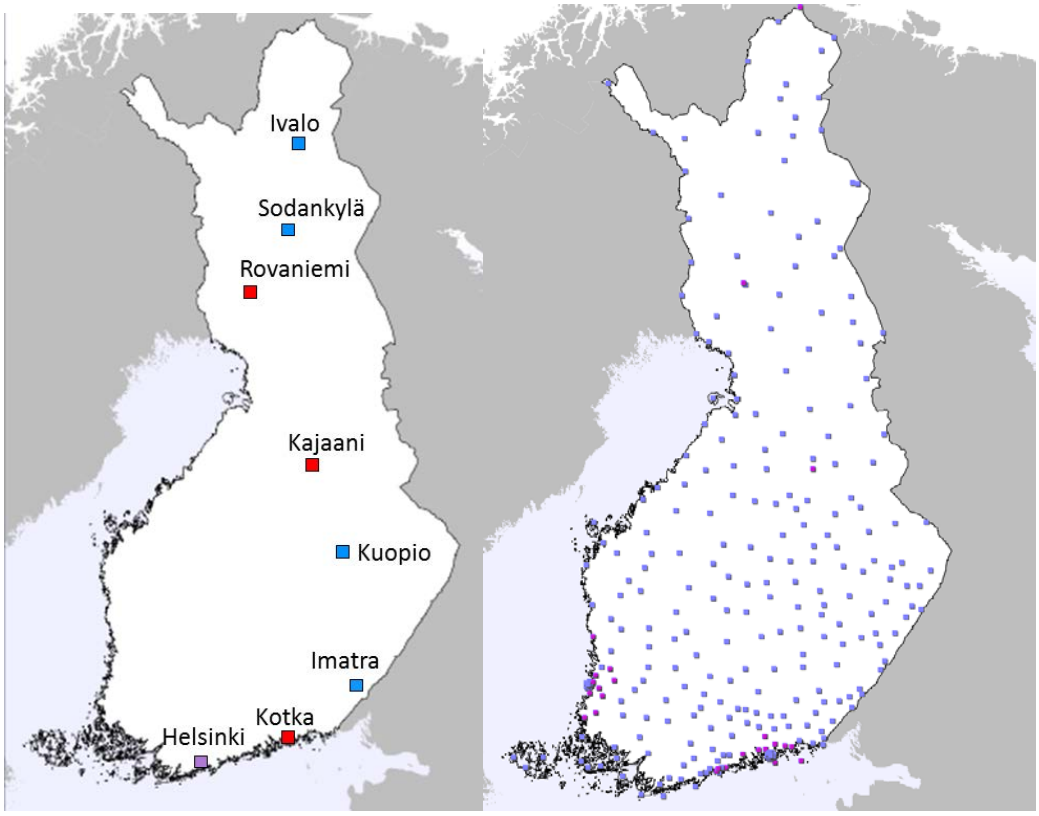

Figure 17 – Left: The Finnish air filter stations. Right:The Finnish dose rate monitoring stations. Spectrometers are marked with a red spot.

4.3. Germany

The national German dose rate monitoring network consists of about 1800 monitoring stations (see Fig. 13) and is complemented by additional regional dose rate monitoring networks. The regional networks in the vicinity of nuclear power plant or other nuclear facilities are operated by authorities from the federal states. Data from the regional monitoring networks are transmitted to the central unit of the national network. Observed data from regional networks can be visualised together with data from national network (see Fig. 12).

German Weather Service operates an additional air monitoring network with about 50 locations. About 40 monitoring stations use on-line particulate air monitoring devices equipped with automatic working gamma spectrometry detectors. Routine monitoring program provides nuclide specific daily average data (see Fig. 18). In case of an emergency, nuclide specific air activity concentration data are provided every two hours. In addition, data from beta and alpha monitoring systems are available – e.g. total and artificial beta and alpha air contamination.

Data from dose rate monitoring networks and from air monitoring network are combined using a common data base. Data from the networks can be visualised using a common graphical information system (GIS) software. In case of an emergency, data from station monitoring, from mobile monitoring and from prognostic information from decision support system may be combined using this GIS software.

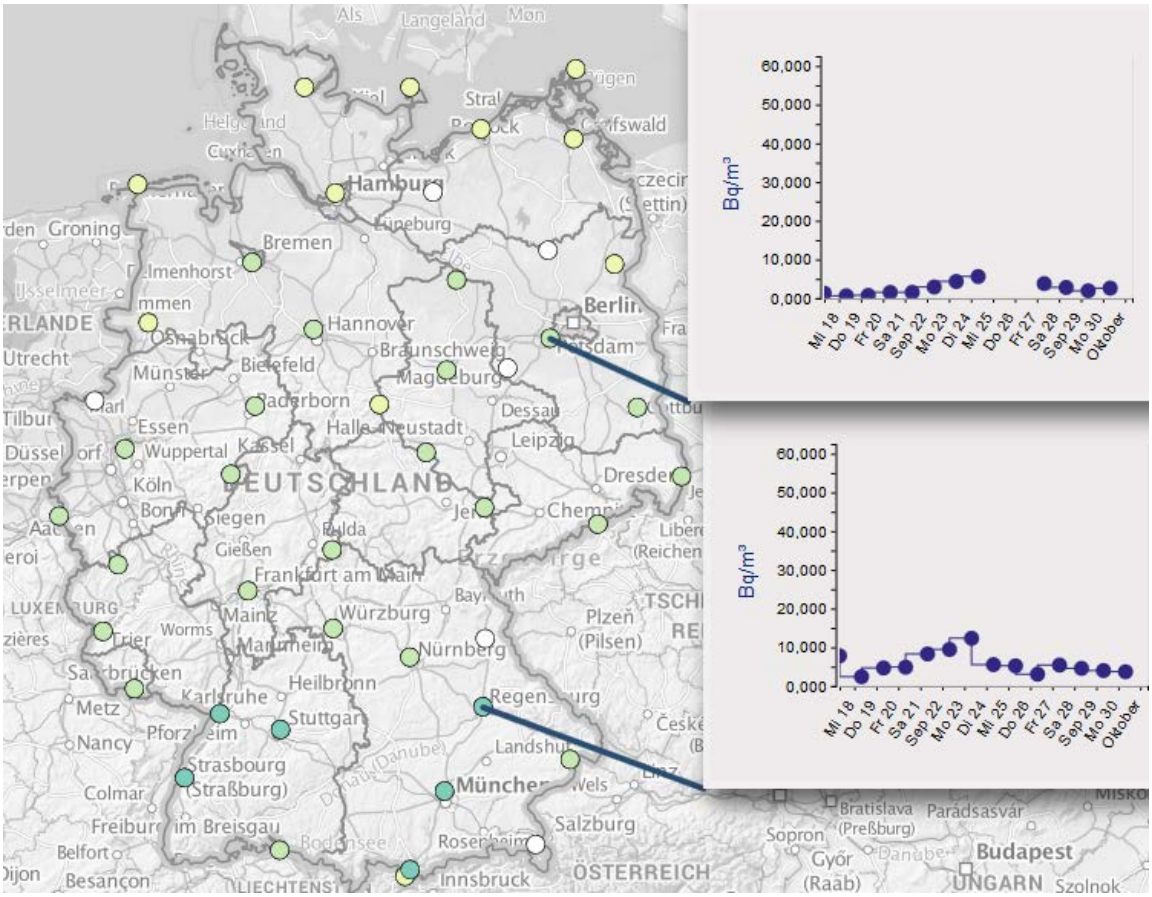

Figure 18 Locations from air monitoring network (operated by German Weather Service DWD) and observed time series of Pb-214 air activity concentration.

Appendix: Experience with BfS test procedures

For the determination of the intrinsic background of the LD and HD GM tubes, a method was developed using reference probes which were characterized in the underground laboratories of PTB, UDO resp. UDO II, during different EURADOS inter-comparison experiments [38]. The reference probes are used to determine the residual dose rate in a dedicated lead castle.

Calibration factors are determined for each probe individually at the irradiation facility of the Helmholtz center in Munich. This SSDL facility [50] allows irradiations with seven Cs-137 sources up to about 2 Sv/h and with five Co-60 sources up to 10 mSv/h. The mean background dose rate level inside the irradiation facility indicated by GM tubes is 0.06 µSv/h. For each individual detector, standardized calibration procedures are performed using three different Cs-137 sources in a distance of 4 m with corresponding dose rate levels of 10 µSv/h, 1 mSv/h and 100 mSv/h. For each calibration level, ten 1 minute measurements of both GM tubes are performed and compared with the 10 min background measurements before and after the irradiation. This procedure enables the calculation of calibration factors for both GM tubes individually.

A climate chamber is used to investigate the temperature dependence of the indicated dose rate of the probes at a background level of about 0.06 µSv/h. A standardized test procedure records counting rates of both GM tubes in a detector, in a controlled 48 hours measuring cycle. In one cycle, the temperature starts at 20 °C, reaches 55 °C and -20°C levels and returns to 20 °C. For each level, readings are recorded for 6 hours and the temperature drift of each probe is determined individually.

Cyclic test procedure of each individual probe are performed every three years. About 98% of the measuring systems passed the criteria of these tests. The error rate was constantly falling within the last years. In cases of errors, the causes were mainly specific effects in the Geiger Müller tubes.

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D9.7.2 Mobile monitoring systems and their uncertainties

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Abstract

This report gives an overview on different mobile monitoring systems for radiation detection, their capabilities and applications in emergency response phases as well as discusses challenges related to the measurements' uncertainties. Suggestions and recommendations are given for an optimized monitoring strategy that will allow to reduce uncertainties of mobile measurements and get more accurate monitoring data for prognostic models and assessment of doses to population and the environment in post-release and transition phases.

Contents

1. Introduction

In the initial stages of an emergency response, a number of different types of monitoring is needed. For the monitored quantities (ambient dose rate, concentration of radioactivity in the surface air, deposited radioactivity in the ground) there are stationary networks in place. However, the density of these networks is not necessarily sufficient to obtain the required information to determine the areas designated for protective actions or to estimate the dose to population due to internal and external exposure. The density of monitoring can be temporarily increased by the deployment of mobile monitoring systems. Mobile monitoring units are used to reduce uncertainties by deploying them to areas where uncertainties for decision-making based on data or models are large.

The goal of a field monitoring mission is to understand the radiological hazard and to map the outer border of the contaminated area. There are three major technical challenges:

• Detection of increased levels of radiation;
• Identification of the radionuclide composition causing the increased radiation levels; and
• Locating and mapping of the source/contaminated area and quantifying the source activity, activity deposition or activity concentration.

All these issues have their own characteristics which have to be fully understood and related measures implemented [TMT Handbook 2009].

Sensitive instruments are required for radionuclide *detection*. Field instruments should be robust and give fast answers. While moving, the monitoring team must use instruments which have a fast response, typically 1 - 5 s. These measurements are interpreted immediately, and the analysis must take into account the variability of the natural background which in many places varies by a factor of ten or more.

Radionuclide identification requires spectrometers. Devices should ideally be easy to operate and be designed for first responders, not for nuclear scientists. Automated software is needed to handle the analysis and related statistics correctly.

Location: in case of large contamination, the location task faces such a problem that the instrument readings and the coordinates have to be merged and an overall map needs to be generated.

The monitoring of *environmental gamma dose* rate within the affected areas (required for both external irradiation incidents and environmental contamination incidents) will allow to estimate doses for individuals who were within (or remain within) the affected areas. This monitoring will enable decisions to be made to remove any remaining people from areas with high dose rates and to restrict access to these areas. It will also help to confirm that zone boundaries have been correctly located (or indicate where changes to zone boundaries are needed).

Monitoring of *environmental contamination levels* within the affected areas will provide information on the spread of environmental contamination, and will help in the control of exposure to internal contamination (Figure 1). Contamination monitoring provides another important input to decisions about the placement of zone boundaries [TMT Handbook 2009]. That is why, the accuracy of all measurements and the obtained monitoring data is crucial.

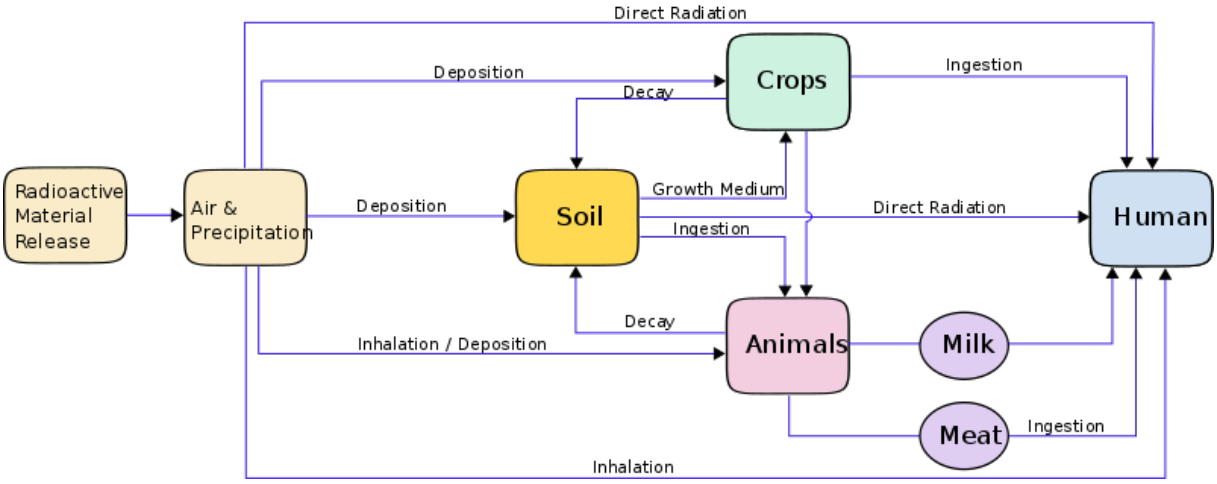


Figure 1 – Pathways from airborne radioactive contamination to humans.

Source: https://commons.wikimedia.org/wiki/File:Atmospheric_radiation_to_human.jpg

A range of instruments is available for emergency response. Dose rate measurement, contamination monitoring and concentration measurements of airborne radionuclides are of primary importance. Response to dispersion of radioactive material may also require fallout measurements.

Routine equipment used for emergency response includes:

- Geiger-Müller detectors (exposure rate, beta dose rate, contamination);
- Ionisation chambers (exposure rate, beta dose rate);
- Proportional counters (contamination, neutron dose);
- Scintillation detectors (exposure rate, alpha and beta contamination, ground deposition, concentration in air, nuclide identification); and
- Semiconductor detectors, also called solid state detectors (exposure rate, ground deposition, concentration in air, nuclide identification).

If the contaminated area is small, the measurements can be performed on foot with portable radiation detectors. If larger areas have been contaminated, specific field missions with mobile equipment may be needed. Such equipment could be detectors mounted in cars, helicopters or airplanes (TMT Handbook, 2009).

In the early phase after the passage of a contaminated cloud, the affected area could be derived from the prognostic data (modelling) and from stationary monitoring data (routine monitoring). Additional measured data from mobile teams can substantially improve the delineation procedure. For this task, mobile teams with the following equipment are helpful:

- Car-borne dose rate probes (e.g. plastic scintillator or NaI scintillator) and aerosol filters;
- Unmanned car-borne systems (using the same equipment as manned vehicles);
- Hand held dose rate probes;
- In situ gamma spectrometry systems – e.g. HPGe detectors;
- Drones-based aero-gamma systems for short range surveys;
- Helicopter-based aero-gamma systems for long range surveys.

The planning process for additional measurements by mobile teams should reflect radiation protection issues for the teams, the needs of decision-making process and the benefit of additional data for information on the public.

The function of an 'Environmental Monitoring Team' [TMT Handbook 2009] is to carry out environmental monitoring at the site of the incident to assess radiation (dose rates) and contamination levels (Becquerel values). The staff in these teams will be familiar with a wide range of monitoring techniques and have available instruments to perform the monitoring tasks. In most scenarios the following initial monitoring tasks are usually carried out:

- ✓ Mapping of the Red Zone boundary;
- ✓ Monitoring of the Red Zone expansion;
- ✓ Screening for hot spots outside the Red Zone;
- ✓ Monitoring of people coming out of the Red Zone; and
- ✓ Identification of the radionuclides present.

Obviously, exposure of a mobile team member depends on the dose rate level, and protection aspects have to be included in the decisions on organisation of a mobile monitoring (Table 1).

Table 1 - Proposed classification system of affected area and proposed priorities for mobile dose rate monitoring in the early phase.

Class	Dose rate level µSv/h	Tasks for mobile teams	Priority
1	< 1	Monitoring of contamination	Low
2	1 – 10	Improve database for dose assessment	Medium
3	10 – 100	Improve database for dose assessment	High
4	100 – 1000	Area, where OIL 2 (100 µSv/h) is exceeded	High*
5	> 1000	Area, where OIL 1 (1000 µSv/h) is exceeded	Very high*

*only, if additional measurements for decisions on the protection of the population are needed.

Monitoring tasks in the populated class 5 regions should be performed, if monitoring data are needed for decisions on evacuation or relocation of people. No additional monitoring is needed in the area where evacuation or relocation measures are already implemented. Dose rate monitoring tasks in the populated class 4 regions should be performed, if monitoring data are needed for decisions on sheltering or relocation of people.

The proposed classification system for mobile monitoring strategy is useful in the urgent response phase (see chapter 2 of D9.7). However, it should be adopted to the actual needs of decision makers in a real emergency situation. Useful devices for mobile monitoring and related uncertainties are discussed in chapters 2 – 5. In addition, chapter 6 discusses additional needs in the transition or recovery phase.

In countries, where the availability of fixed monitoring facilities is limited, or where existing facilities are centralized at a single location, transportable environmental monitoring units would be used. Normally, such a unit would have following basic outline specification for monitoring of radioactivity and data transfer:

- ✓ Dose rate monitoring;
- ✓ Radionuclide identification;
- ✓ Aerosol sampling;
- ✓ Sample and *in situ* gamma spectrometry;
- ✓ Alpha spectrometry; and
- ✓ Data communications tools.

Depending on the monitoring needs and types of contamination, additional instruments and capabilities would be integrated.

Critical for the improved delineation procedure from monitoring data and data from the mobile teams is the ability to visualize and to evaluate all types of data. Obviously, realistic exercises are very helpful to test this delineation process and to train mobile teams as well as back-office teams.

Below is an overview on different mobile monitoring systems for radiation detection, their capabilities and applications in the exposure assessment process. Challenges related to the uncertainties of measurements are described for each system. To optimize monitoring strategy and to reduce uncertainties of mobile measurements, suggestions and recommendations are given further in the report that will allow to get more accurate monitoring data for prognostic models and assessment of doses to population and the environment in post-release and transition phases.

2. Relocatable systems

Relocatable systems or probes are usually used to complement (or expand) the existing stationary network for gamma-dose rate measurements or to replace the existing probes. The relocatable probes are normally the mobile versions of stationary probes and could be easily mounted and being integrated into the existing stationary network. The biggest advantage is that ones being integrated, there is no more need for a human resource. The measured data is being automatically transferred to the central data server.

2.1 Relocatable dose-rate monitoring systems and their deployment

In an emergency situation, relocatable probes can be deployed to perform larger scale measurements, increase the spatial resolution of monitoring, increase the area coverage of monitoring and to get more precise data on radioactivity levels at the relevant location as well as to reduce uncertainty of measurements within the measured area. This, will in turn allow more precise and better dose assessment for population and workers (fire fighters, etc.).

If possible, these relocatable probes should be deployed in the pre-release phase following a strategy discussed in MetroERM Deliverable 1.5.6. On the other hand, the locations where the relocatable systems are deployed could be determined using the same approach as is taken in the optimization of stationary networks. This approach [Melles et al., 2011], [Abida et al., 2008] and [Helle et al., 2011] in general is in the following (see Figure 2):

1. Generate a data set of scenarios and determine the quantities of interest (external dose rate, deposition, activity concentration) for each scenario using simulated releases with realistic source terms and atmospheric dispersion models utilizing real weather data.
2. For each scenario in the dataset:
    a. Simulate the measurements of the network to optimize (i.e. for dose rate monitoring, obtain the external dose rate at station locations from the scenario currently considered).
    b. Interpolate the measurement from the step 2.a. over the optimization area.
    c. Calculate the difference between the field obtained by interpolation in step 2.b and the data from the scenario currently considered.
3. Adjust the positions of the stations and repeat step 2 and 3 until the best configuration of stations is found.

The crucial questions for decisions to make regarding the above optimization algorithm are:

- Choice of the cost function (defining which of the candidate networks is the best) to use in step 2.c. In [Abida et al., 2008] the L^p norm is used as a cost function, whereas in [Melles et al., 2011] the cost function is the combination of interpolation error variance (from regression kriging) and detection capability (given as a function of the depositional area, hour of detection and the number of detections).
- The weighting of the cost function by different factors. Population density is one of the factors used in [Melles et al., 2011]. A political constraint is considered in [Helle et al., 2011], where each administrative unit of the optimization region is required to have a monitoring station.
- The adjustment scheme in step 3. Simulated annealing is used in both [Abida et al., 2008] and [Melles et al., 2011].

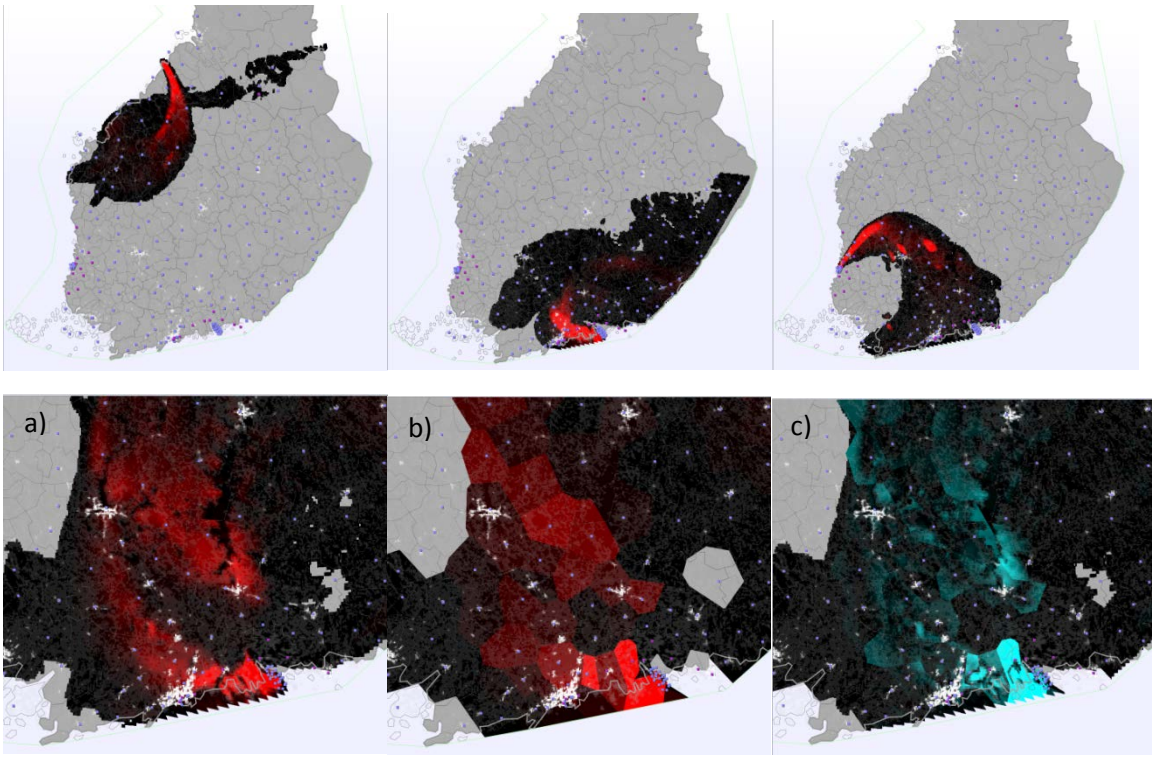


Figure 2 – Top: One simulated scenario from existing or planned Finnish NPP locations. The dataset of scenarios for a network optimization tool currently under development contains a large number of releases from each location. Below: The evaluation of a release scenario. a) The simulated release. The blue dots are external dose rate monitoring stations, the white specs are population density. b) Nearest neighbor interpolation of the measurements at current probe locations. c) The differences in external dose rate between the interpolation and the release. The calculation of the L^p norm is straight forward from this data.

3. For deployment of relocatable dose rate probes, the data set generated in step 1 would be an ensemble of prognostic data on the impending release. The monitoring network to optimize would naturally contain the stationary monitoring locations together with the relocatable probes. Only locations of the relocatable probes would be adjusted in step 3 of the above algorithm.

2.2. Sources of uncertainty

The sources of uncertainty for measurements are essentially the same as for identical stationary systems (see details in the sub-report D9.7.1). The difference is the lack of a background data for the incorporated/additional probes.

2.3 Challenges

Within related challenges it could be emphasized the need for an installation and performance of the probes within a short time (which requires good skills from a trained personnel) and assurance of the quality of the received data from these probes.

The deployed probes have to be integrated into the technical systems managing and storing field measurement data, this can be either an easy or difficult task depending on the probe type and its communication interface. As pointed out earlier, ideally a mobile version of a probe already integrated to a monitoring network can be used. To ensure the relocatable systems can be utilized in decision making, the data they produced should be made accessible in an emergency management system.

3. Airborne systems

Airborne measurement systems are used to provide an overview of the radiation situation over a large area. The range and coverage of airborne systems is unsurpassed and their deployment is not hindered by terrain. The required effort and capabilities to characterize and maintain an aircraft for this purpose are considerable. One approach to counter these requirements is the use of measurement units that can be placed in a general purpose aircraft. The characterization of this kind of systems is usually difficult, as the placement of the unit is more restricted. Capability for radiological sampling can also be integrated into an aircraft with specially designed samplers.

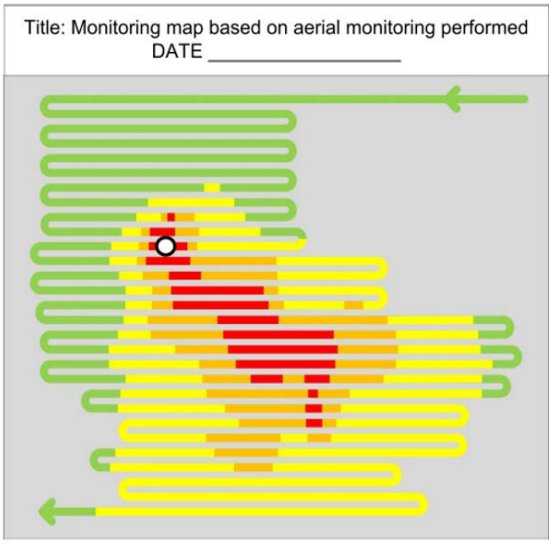
3.1 Air-borne dose rate monitoring and equipment

Air-borne dose rate monitoring equipment was designed for very different tasks. In USA, the first systems were established in the 1960s to support above-ground inspections at the Nevada Test site [L2012]. In Germany, routine-mode investigations in regions with enhanced dose rate due to mining activities were performed using air-borne dose rate monitoring techniques. After an accidental release, air-borne dose rate measurements are an important tool for rapid delineation of affected area [W2004]. Air-borne dose rate monitoring teams were established after the Chernobyl accident in different European countries and have experience from international comparison exercises since more than 20 years [PSI2009].

After the Fukushima accident, data from air-borne dose rate monitoring teams were very important to give an early overview on highly affected area [L2012]. A more detailed evaluation of the data was used to assess the amount of Cs-137 activity deposited on the ground [S2014]. Obviously, it is very helpful to combine data from the stationary dose rate monitoring with data from air-borne dose rate monitoring systems and data from the ground-based mobile monitoring teams.

The biggest advantage of an air-borne monitoring is in possibility of relatively fast measurements and mapping of the contamination in large populated areas, especially within a short time after a release, when the capabilities of mobile ground teams are restricted.

Air-borne dose rate monitoring equipment often rely on two or four large NaI scintillator detectors. In addition, high pure Germanium detector may be used to improve the possibility of nuclide identification. The monitoring data have to be combined with position data e.g. from GPS system and measurement information for the height above ground e.g. from radio altimeter.



EXAMPLE LEGEND FOR GROUND DOSE RATE MAP		
OIL exceeded	Dose rate at 1m above ground level	Immediately required protective actions[a]
OIL1	≥ 1000 µSv/h	▪ Instruct the public to take ITB agent; ▪ Safely evacuate; ▪ Reduce inadvertent ingestion; ▪ Stop consumption and distribution of all local produce, forest products (e.g. mushrooms), milk from grazing animals, rainwater, animal feed; and ▪ Stop distribution of commodities. ▪ Provide registration, monitoring, decontamination and medical screening for those in the area.
OIL2	≥ 25 µSv/h (for t > 10 days[b]) ≥ 100 µSv/h (for t ≤ 10 days[b])	▪ Instruct the public to prepare to relocate[c] while taking actions to reduce inadvertent ingestion; ▪ Stop distribution and consumption of local produce, milk from animals grazing in the area and rainwater; and ▪ Stop distribution of commodities.
OIL3	≥ 1 µSv/h	▪ Stop distribution and consumption of non-essential local produce, wild-grown products (e.g. mushrooms and game), milk from grazing animals, rainwater, animal feed until the concentration of radioactive material has been assessed using OIL7; and ▪ Stop distribution of commodities.
NONE	< 1 µSv/h	None, but further actions may be needed in this area based on later monitoring and sampling.

Figure 3 - Schematic results for assessed ground dose rate based on the data from air-borne monitoring.

The equipment can be mounted on fixed wings aircrafts and helicopters. At the moment, unmanned air-craft vehicles (UAV) equipped with different detector systems are being designed and investigated (see PREPAREDNESS WP5). In future, these UAV-based systems may help to get monitoring information from a very highly affected area and thus, the exposure to monitoring teams will be reduced (Figure 3).

Sampling particulate matter from the air is also possible using an UAV, allowing the sampling to take place at a desired location and altitude. An air sampler for a fixed wing UAV was designed in [Peräjärvi et al., 2008] and tested in [P2009] and [Pöl2009], showing that airborne trans-uraniums could be detected in concentrations of 0.2 Bq/m^3 (^{239}Pu and ^{241}Am) after 30 minutes of sampling and 1 h of counting (direct alpha spectrometry of the sample).

3.2 Uncertainties due to probe characteristics

Different measurement systems have different characteristics and advantages. Therefore, according to the given measurement task, an appropriate detector type has to be chosen. The crucial properties of detector systems used for radiation monitoring have to be characterized before any detector system can be used for monitoring and assessing reliable data.

The characteristics of detector systems used for air-borne monitoring should be analyzed using type testing procedures. Type testing procedures have to give information about relevant characteristics of probe types:

- Dose rate measurement range (upper and lower limit) and typical parameters (self-effect, calibration factors, dead-time corrections, etc.) used to calculate dose rate from primary data like counting rates,
- Angular response of probes,
- Response of probes for different photon energies and for secondary cosmic component,
- Impacts on environmental conditions on observed dose rate (e.g. influence of temperature),

- Short-term variability of observed dose rate.

Additional cyclic test procedures for individual probes should be performed to check relevant parameters – e.g. calibration factors.

3.3 Impacts of flight altitude and detector position

Obviously, data from air-borne dose rate monitoring is influenced by the flight altitude. The influence of a flight altitude on gamma photon flux and dose rate can be derived from Monte-Carlo calculations [ICRU53]. Air-borne dose rate monitoring systems have to be able to assess the dose rate under the standard conditions - e.g. 1 m above flat ground – from monitoring data. In addition, the teams should be able to assess the local nuclide specific ground contamination from the obtained monitoring data [S2014].

In addition, the location of detectors outside or inside the air-craft or a helicopter may have a large impact on monitoring data. Photon flux is affected by shielding and scattering effects of material in the vicinity of the detectors. For correct interpretation of monitoring data, Information about the impact of these effects is needed. In Germany, air-borne dose rate monitoring equipment is restricted to detector systems mounted inside helicopters. Angular dependence of detector response due to shielding and scattering effects of helicopter was investigated using point source experiments (see Figure 4 from [Str2018]). Thus, in the evaluation of a monitoring data the knowledge about the impact of the detector location have to be considered.

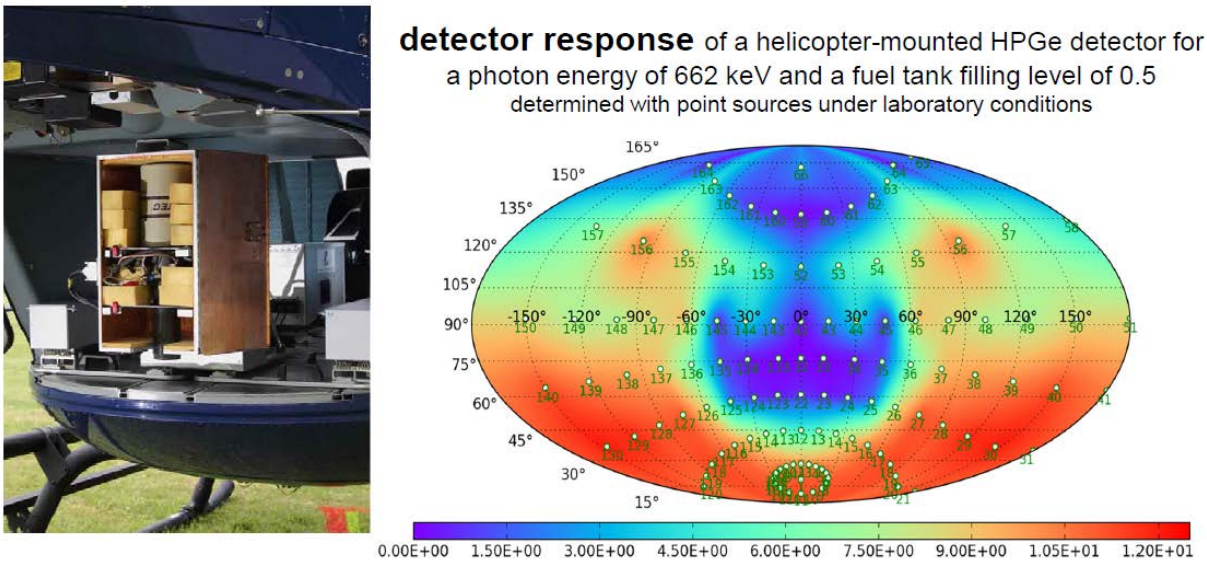


Figure 4 - Example for angular dependence of detector response due to shielding and scattering effects calculated from a Cs-137 point source experiments [Str2018].

3.4 Impact of activity in the air and potential self-contamination

Air-borne dose monitoring teams should be able to assess the impact of the activity concentration in the air on the monitoring data. In principle, the teams should be able to measure activity in the air e.g. using aerosol filters during the flights. The aero-gamma data can be influenced by low air activity concentrations even in post-release phase. Thus, it is recommended to assess aero-gamma data in combination with ground-based monitoring data (when possible).

To avoid self-contamination of the equipment, the teams should avoid measuring tasks in regions with relevant artificial activities in the air.

7.5. Amount of area that can be covered

Assuming an average speed of 100 km/h and a distance of 2 km between flight paths, one air-borne monitoring team would be able to cover about 200 km^2 per flight hour.

For example, the 5 km central zone (about 80 km^2) and one 30° sector up to a distance of 20 km (about 100 km^2) could be covered within one flight hour.

4 Carborne systems

Carborne systems can be thought as "filling the gap" between portable and airborne techniques [IAEA TECDOC 1364]. They can be deployed to perform monitoring and surveys in a desired location with greater area coverage than portable systems and better spatial resolution than aerial systems. A drawback of most carborne systems is that they must travel on roads, and most carborne systems display a limited off road capability. The characterisation of a carborne system for dose rate or activity measurement is more difficult than that of a portable system but easier than that of an airborne system. In addition to direct measurements, a radiological survey vehicle can also perform other tasks, such as transport of personnel and equipment, as well as the collection, storage and high resolution measurement of samples.

4.1 Carborne dose rate monitoring systems and equipment

Modern car-borne dose rate monitoring systems are used to quickly gain an overview over contaminated areas, especially in the urban area. Most of these systems combine dose rate monitoring with plastic scintillator probes or spectroscopic dose rate probes based on NaI or CsI detector systems with GPS systems. Normally, information on the dose rate being measured within 1 second intervals is combined with the car position data from a GPS system. Typically, position information have spatial uncertainties in the order of 5 m and statistical dose rate uncertainty is in the order of 10 % at background level of dose rate. Most dose rate systems used in emergency preparedness context have an additional Geiger-Müller or proportional counting tube for measurements of high level dose rates up to some mSv/h.

With regard for emergency preparedness and response aspects, car-borne dose rate monitoring systems should be classified for the following tasks:

- The systems monitor dose rate in high level radioactive area and are useful to identify such area where measured dose rate exceed operational intervention levels.
- The systems monitor dose rate and indicate artificial radiation (for example plastic scintillator systems using "natural background reduction" (NBR) algorithms).
- The systems monitor dose rate and enable the assessment of nuclide specific dose rate for relevant nuclides.

4.2 Sources of uncertainties due to probe characteristics

The characteristics of specific dose rate probes used for car-borne monitoring should be analyzed using type testing procedures. Type testing procedures have to give information about relevant characteristics of probe types

- Dose rate measurement range (upper and lower limit) and typical parameters (self-effect, calibration factors, dead-time corrections, etc.) used to calculate dose rate from primary data like counting rates.

- Angular response of probes
- Response of probes for different photon energies and for secondary cosmic component
- Impacts on environmental conditions on observed dose rate (e.g. influence of temperature)
- Short-term variability of observed dose rate

Additional cyclic test procedures for individual probes should be performed to check relevant parameters – e.g. calibration factors.

4.3 Impact of probe location

In principle, data from car-borne dose rate monitoring is influenced by the location of the probe inside or on the roof of the vehicle. Due to shielding effects, dose rate measured inside or on the roof of the vehicle is lower than dose rate measured outside under the standardized conditions – e.g. 1 m above the ground.

The concept of probe location factors may help to consider these shielding effects and to compare data from mobile and stationary dose rate monitoring. These location factors can be calculated from Monte-Carlo calculations using adequate models for probe location and relevant shielding materials. In addition, these factors can be derived from experimental investigations.

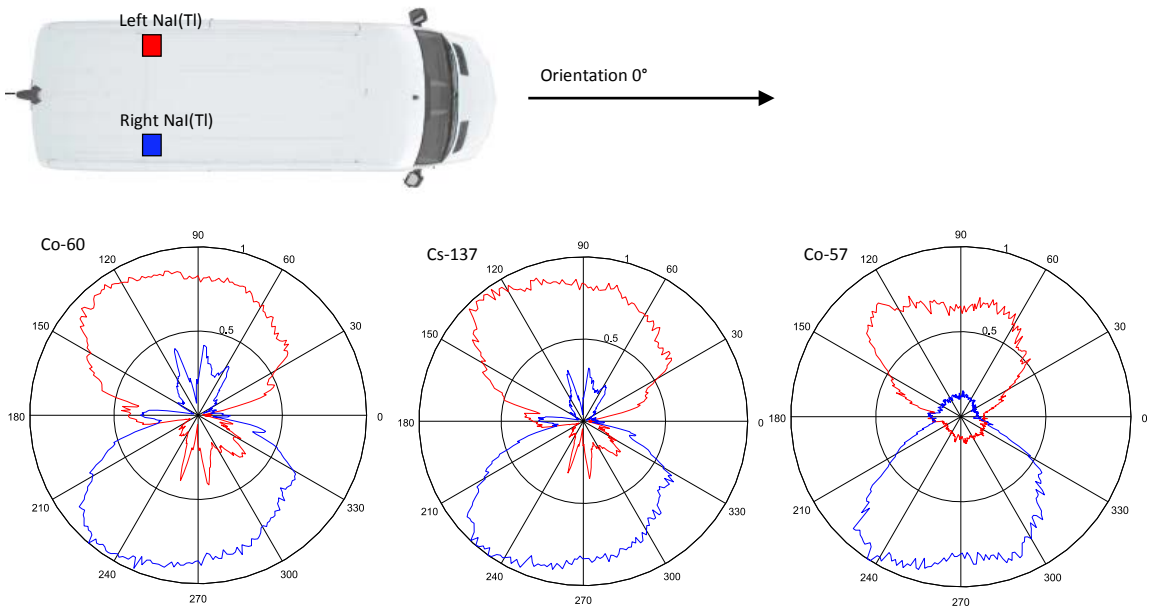


Figure 5 – Angular dependence studied with different point sources for a vehicle on a rotating platform. The count rates from the sources are relative to the highest count rate for given source as a function of the orientation of the vehicle.

The angular dependence of probe response inside or on the roof of a vehicle can be investigated with point source experiments (Figure 5). In addition, experiments in regions with enhanced natural or artificial level of radiation can be used to assess these probe location factors: At the same ground position, dose rate data measured in-side the vehicle are compared with observed dose rate 1m about ground without vehicle. During pronounced rain events, similar comparison experiments may compare net dose rate data from probes located inside the vehicle with net dose rate data from probes.

The shielding effects of cars have been studied by utilizing the naturally occurring radioactivity in the ground [Burson, 1974], placement of multiple point sources [Lauridsen and Jensen, 1982] and a combination of Monte-Carlo simulations and measurements in actual fallout situation [Takahara et al.,

2018]. The results show an increasing reduction in the dose rate from fallout with increasing mass of the vehicle (Table 2). The reduction in dose rate due to skyshine is slightly smaller than the reduction due to groundshine [Takahara et al., 2018]. This is most likely due to the fact that the main body of a car is considerably thinner than its undercarriage.

Table 2 – Reduction factors of dose rate due to fallout determined for various type of cars. The dose rate reduction factor is defined as the dose rate at probe location inside the vehicle divided by reference dose rate at the same spot without the vehicle.

Vehicle	Vehicle mass (kg)	Dose rate reduction factor	Contributing radionuclides	Reference
Daihatsu Mira	800	0.73	Cs-137	Takahara et al., 2018.
Toyota Vitz	1080	0.73		
Toyota Wish	1440	0.66		
Toyota Alphard	1930	0.64		
Fiat 127	675	0.64	Cs-137 & Cs-134	Lauridsen & Jensen, 1982.
Morris Mini	690	0.72		
Talbot Horizon	925	0.66		
Polski Fiat	1070	0.58		
Saab 99 GL	1090	0.62		
Mercedes 220 Diesel	1350	0.59		
Passanger car	-	0.67 – 0.59	Natural background	Burson, 1974. Mass of the vehicles not given.
Pickup	-	0.53 – 0.48		

4.4 Impact of potential self-contamination

Observed dose rate data may be influenced by radionuclide contamination of the car itself. Relevant self-contamination effects would lead to overestimation of the exposure situation. Such effects are possible under following conditions:

- Mobile team is in a low contaminated area after passage through a very high contaminated area,
- Passage through wet streets or streets with snow cover,
- Passage through un-paved terrain (as example, activity decreases much faster on the road vs. grassland, due to washout vs. migration).

After monitoring jobs, mobile teams should regularly check possible self-contamination of vehicles. Checks can rely on comparison of monitoring data outside contaminated area or using additional measuring equipment.

4.5 Amount of area that can be covered

Assuming an average speed of 30 km/h, one car-borne dose rate team can give an overview on dose rate levels of 240 km (line) or about 200 km^2 (area) within one working day. For example, the city of Munich (about 300 km^2) could be monitored by two teams within one working day.

Depending on the level of radioactive contamination, the car-borne measurements could be supplemented by measurements from a ground-based mobile team with hand-held instruments, thus allowing to include additional measurement points and expand the monitoring area along the proposed route for car-borne measurements.

5 Portable systems

Field measurements made using portable systems represent the most basic capability of environmental measurements. In [IAEA TECDOC 1363] field measurements are categorized as either *static mode* measurements, where radioactivity is measured at discrete points, or as *dynamic mode* measurements where the instrument is transported during the measurement. The dynamic mode measurement reflects the average radioactivity of the area traversed during the measurement time. The dynamic mode allows the operator to cover more ground and is thus more desirable in searching for, localizing and mapping areas of significantly increased radioactivity. Modern portable systems are often capable of storing a time series of dose rate or spectrometric data, reducing the uncertainty due to the averaging effect by storing the collected data at specific intervals. This on the other hand increases the statistical uncertainty due to the short counting time in each collection interval. Static mode measurements provide better sensitivity due to their long collection time and stationary nature, allowing for more precise quantification of dose rates, activities or activity concentrations.

5.1 Portable monitoring systems

In environmental radioactivity monitoring, portable monitoring equipment is used for:

- Dose rate monitoring of points of interest;
- Dose rate surveys for searching or mapping of areas;
- Contamination surveys.

The range, types and possibilities of portable instruments vary from the most simple ones (simple dose-rate meters) to the most advanced (modern backpacks). Technologies range from straight-forward radiation counting systems to advanced spectroscopic systems with fully-automated isotope identification and energy calibration.

In particular, portable systems could comprise of:

- Gamma dose rate monitoring equipment;
- Alpha contamination monitors;
- Beta contamination monitors. Instruments capable of detecting both alpha and beta contamination could be used, but must be able to distinguish between alpha and beta contamination;
- Gamma contamination monitors;
- Hand-held monitors for measuring neutron dose rate;
- X-ray and low energy gamma contamination monitors;
- Portable gamma spectrometry equipment;
- Modern backpacks (inkl. with wireless technology) with possibility of detection of alpha, beta, gamma and neutron contamination.

The mobile monitoring teams should also consider:

- Personal alarming dosimeters (these should be capable of measuring instantaneous dose rate as well as cumulative dose);
- Plastic bags to cover monitors (alpha contamination monitors must not be covered); and
- Personal protective equipment.

The portable instruments are applied for measurements in early phase after the radioactive cloud has passed. Measurements from these systems (i.e. in-situ measurements), combined with the data from the sampling from the affected areas, provide possibility for full and detailed overview on the borders of the affected area and radioactivity levels.

5.2 Sources of uncertainty

During in-situ measurements, a known source of uncertainty is the 'user effect' when measurement systems/equipment are close to the user's body and the measurement results could be affected by the attenuation due to user's body. This can be mitigated by use of a tripod.

The ambient dose equivalent rate (ADER) in a back-pack geometry for an operator with a body mass of about 70 kg and a height of 170 cm was 12% lower than the ADER registered in the standard measurement geometry with the detector placed on a tripod at a height of 1 m above ground. A similar effect was observed in [Buchanan et al, 2016] with a reduction of 16 % for a slightly larger operator (175 cm, 75 kg). The attenuation of the primary 662 keV photons from ^{137}Cs in the operator was measured as a 22% reduction of ADER. Much smaller effect, on average 3–4% reduction, was observed with respect to ADER due to scattered photons from ^{137}Cs in the environment [Ramzaev et.al, 2017]. In [Buchanan et al, 2016] a reduction of 25 – 30 % was found in the full energy peak intensities of ^{134}Cs and ^{137}Cs due to the operator when using a backpack system.

A light-weight tripod, made of wood or aluminum, is frequently used to support a gamma-ray dosimeter or spectrometer during measurements of environmental photon radiation. Due to the negligible weight of such tripods, their possible influence on gamma-rays emitted from sources distributed in the environment can be assumed to be insignificant. Therefore, the results of measurements obtained with the detector placed on a tripod may be straightforwardly used for calculation of effective dose to the human population [Ramzaev et.al, 2006]. In general, tripods are used for static mode measurements when precise quantification of dose rate or deposited activity (especially in the case of spectrometric *in-situ* measurement) is desired. Tripods may also be utilized to measure discrete radiation sources more carefully, ensuring good data quality for the subsequent analysis when, for example, a source out of regulatory control is encountered.

The application of the "on-tripod" measurement technique has practically no constrains for such outdoor locations as grasslands, arable fields and forests. At the same time, the appearance of a survey team with the measuring equipment at streets, pavements and other public places (especially at crowded sites) is an obstacle for pedestrians and may also disturb the traffic. A solution could be to let the operator carry the spectrometer-dosemeter instead of using a tripod to support the measuring instrument. Small hand-held dosemeters-spectrometers can be placed in a pocket of the operator [Onischenko et al., 2013], while a backpack worn by the user is more suitable for modern light-weight gamma-ray spectrometers with larger detectors.

Also, the measurement area and topography could affect the uncertainty of measurements due to the effects of terrain and spatial variability. The man-made source dominates ADER in forests while the contribution from the natural sources prevails in ADER recorded above paved areas [Ramzaev et.al, 2017].

Otherwise, these systems are subject to similar uncertainties as for other dose rate measurements.

5.3 Amount of area that can be covered

Compared with relocatable systems, the measurement area in this case is more restricted/limited, but on the other side these measurements provide more detailed information and possibility to identify nuclide composition of deposition in the affected area.

6 Monitoring strategy in transition phase

Nuclear accidents generate complex situations that affect all dimensions of individual and social life. Once an emergency situation is declared, decisions on protective actions on-site and off-site should be taken promptly during the early phase to be effective. Given the short time to react, these actions should be prepared in advance on the basis of plausible scenarios, and adapted as much as possible to the actual situation [ICRP, 2019].

During the intermediate phase, progressive characterisation of the radiological situation on-site and off-site is essential to guide decision making about the protective actions to be initiated, continued, or discontinued.

In the long-term phase, radiological situations on-site and off-site are better understood, and can be improved more effectively compared with the initial phase of the accident.

Environmental monitoring is required to provide an accurate picture of the radiological situation, both on-site and off-site. There is a need to establish zones and zone boundaries for contaminated areas. The Red Zone is the potentially hazardous area immediately surrounding the incident where extreme caution and safety measures are required. Responsibility for the health and safety of personnel working within the Red Zone remains with the individual agencies involved. The cordon surrounding this zone is called the "Safety Perimeter" [IAEA TECDOC 1162, 2000].

Predictions of exposure can be made using meteorological information, environmental monitoring data, and modelling. Sufficient meteorological stations should be available to characterise weather conditions in areas that might be of radiological concern (i.e. from close to the installation to surrounding areas where deposition may affect inhabited areas or agricultural land). Fixed and mobile radiological monitoring equipment can be used by trained operators to evaluate exposures with more precision. Radiation aerial monitoring also provides useful information on the degree and extent of environmental contamination in the case of widely affected areas.

The pattern of deposition is dependent on the magnitude of the event, and on the prevailing meteorological conditions at the time of the release, particularly wind direction and any rainfall occurring during passage of the plume. For an extended release, wind direction can be expected to vary over time. In the longer term, rainfall and weathering cause redistribution of radionuclides in the soil and their further migration. Plant uptake of radionuclides from soil varies according to the physical and chemical characteristics of the soil (e.g. moisture and fertility), and generally decreases with time. The levels of deposition may also vary greatly from one area to another. As an example, after the Chernobyl accident, surface contamination varied by factors of up to 10–100 within the same village. Generally, in the longer term, one or a few radionuclides will dominate as the principal contributors to both human and biota exposure [ICRP, 2019].

In addition to environmental monitoring of ambient dose rates, measurements of radionuclide concentrations (particularly caesium and iodine) in air should be made. This type of information enables the estimation of internal exposure due to the inhalation of radioactivity. Concerns regarding internal and external exposures arising from deposited radioactive material in the environment require plans to measure soil surface concentrations as input to decisions on the implementation of both food and water restriction and extended protective actions (e.g. temporary relocation).

Thyroid dose monitoring in the early phase is important for children and pregnant women. Environmental monitoring cannot provide an accurate estimate of individual thyroid exposures. Therefore, a specific effort should be made to monitor radioiodine content of the thyroid rapidly in children (up to approximately 15 years old at time of exposure) and pregnant women in order to get realistic estimates of thyroid doses. Thyroid measurements can be made by trained and properly equipped personnel at evacuation centres and post- accident centres established for health

surveillance. Given the 8-day half-life of iodine-131, it is important to make such measurements within a few weeks of exposure, ideally as soon as practical after exposure [ICRP, 2019].

The monitoring of soil, food, and water is likely to continue beyond the intermediate phase and into the long-term phase. In the intermediate phase, detailed environmental monitoring is essential for understanding the radiological situation of widespread contaminated areas, and for terminating the urgent protective actions implemented during the early phase. As radioactive releases are brought to a halt and more detailed monitoring becomes possible in affected areas, the availability of environmental measurement data increases.

Experience from past accidents indicates that there is a possibility of radiation exposure from aquatic pathways due to the release of liquid radioactive material to the sea or surface waters, deposition of radioactive material directly onto the sea or surface waters, and from run-off into the sea or surface waters. For direct or indirect releases of radioactive material into the sea, people can be exposed externally from radionuclides in the sea or sea sediments. The doses from these pathways are not expected to make significant contributions to the overall exposure. Among them, the transfer of radioactive material into seafood should be considered as a possible primary source of internal exposure to the public [EGNRS, 2011].

Continuation of radiological characterisation in affected areas should be complemented by the establishment of a system for monitoring the external and internal exposure of individuals. For the authorities, the monitoring system in the recovery process will help to fulfil several objectives: to obtain data on the actual contamination of affected areas and its evolution; to control the concentration of radionuclides in foodstuffs; and to provide information to the public on external ambient dose rates by using devices displaying the results in different places.

The control of ingestion pathways is an important component of the protection strategy for the public. Experience shows that maintaining radiological monitoring of foodstuffs in the long-term phase is useful to gradually restore the confidence of distributors and consumers inside and outside affected areas [Strand et al., 1992; Skuterud et al., 2012].

To optimize the monitoring strategy and to get credible data from measurements of radioactivity in the environment following a nuclear accident, as well as to reduce uncertainties of measurements and dose assessments in post-release and transition phases, the aspects described in the following paragraphs should be considered and recommendations implemented.

6.1 Background values

It is very important to have a good background data! The monitoring team should obtain and record representative background values before doing measurements in expected contaminated sites.

6.2 Protection of instruments

In general, it is recommended that dose meters, gamma probes and rate meters be enclosed in plastic film to protect them from becoming contaminated, as can the handles on alpha and beta probes. However, the detection surfaces (Mylar screen) on alpha and beta probes must not be covered so that to avoid puncturing and making the instrument light sensitive, that in its turn will seriously impair the instrument's performance. The extent of the damage can result in readings ranging from enhanced background to full scale deflection. Damaged probes should be replaced.

6.3 Measurement techniques and cautions for monitoring

Before leaving your base of operations:

- ✓ Check condition of batteries (take some spare batteries when applicable);
- ✓ Check instrument response with calibration sources;
- ✓ If more than one instrument is available perform cross checks to ensure a consistent reading;
- ✓ Measure and record background readings from instruments;

Due to the short range of alpha and beta particles in air, when monitoring for this type of radiation the probe should be as close as possible, and no further than 1 cm, from the surface being monitored. The contaminated surface must be dry and should be traversed at a rate of 10 cm per second or slower.

Alpha particles can not be detected with an instrument designed for beta/gamma radiation. Alpha emitting radionuclides often produce X-rays and photons, with a low abundance, which can be detected with beta/gamma sensitive instruments. The presence of alpha emitting radionuclides must be confirmed with an alpha sensitive instrument [TMT Handbook 2009].

Beta and X-ray contamination measurements may not be possible in the presence of significantly elevated gamma fields. Visual identification of a source may be very difficult. Make sure to turn on the instrument well before attending the incident location.

Rapid measurements, particularly gamma dose rates, must be taken at regular intervals (for example at 5 to 10 metre intervals). A minimum of 8 regularly spaced locations on the 'Safety Perimeter' must be monitored to confirm that the 'Safety Perimeter' is safely positioned.

Portable high resolution gamma spectrometers are usually not available in large numbers, so they should not be used inside the Red Zone due to the risk of instrument contamination and the fact that most spectrometers saturate at very high radiation fields and show zero or low reading, or show 'battery low'. This means that sampling and sample measurements are needed to assess the radionuclides present inside the Red Zone. In the case of an RDD, the different nuclides used in the device may have dispersed very differently from each other, which makes representative sampling difficult or practically impossible.

Measurements of X-ray emitting radionuclide contamination and beta contamination should be made by taking a surface wipe sample and monitoring it away from any other sources of gamma and X-rays. The sample should be taken by wiping an area of approximately 100 cm2. Light pressure should be used so that the wipe sample is not torn or rolled. Wipes may be taken from roads and pavements but only a small fraction of the total activity will be removed. Unless the exact fraction of non-fixed contamination by the wipe is known, a default value of 0.1 should be used. Use a portable contamination meter to assess activity on the wipe [TMT Handbook 2009].

6.4 Measurement Units

Contamination monitoring equipment displays activities as counts per second or cps and not Bq cm^{-2}. Conversion from cps to Bq cm^{-2} requires the response efficiency to be known, and this is instrument-specific (although similar equipment will have similar efficiencies and so a generic value may be applied). The monitoring team may be more familiar with cps and prefer to use these units. If so, they should convert the values (and double check the calculations) before starting monitoring. Reported results must include the reading, the units in which the result is expressed, the type of instrument and the conversion factor for the instrument. Record the conversion factor on the instrument with a waterproof marker pen.

6.5 External ambient gamma radiation monitoring

Because external ambient gamma dose rates represent ambient gamma radiation from all external sources, and because the instrumentation used is capable of continuous reporting, this sampling medium is highly relevant to estimating doses and to act as a warning system in the case of a radiological emergency.

During the release phase of a nuclear accident, external gamma dose rate measurements will include the contribution of radiation from the plume. In the post-release phase, the external gamma dose rate measurements are mainly attributable to radiation from deposition on the ground. The data can be

used to estimate doses to the public in both normal and emergency situations, and in the case of emergency preparedness, it can also offer additional important information, such as the movement of the radioactive cloud.
The application of relocatable probes would allow to perform larger scale measurements, reduce area between probes in the network, get a better area coverage and more precise data on radioactivity levels at the relevant location as well as to reduce uncertainty of measurements within the measured area. This, will in turn allow more precise and better dose assessment for population and workers (fire fighters, etc.).
The aero-gamma data can be influenced by low air activity concentrations even in post-release phase. Thus, it is recommended to assess aero-gamma data in combination with ground-based monitoring data (when possible).

6.6 Air monitoring

Airborne radioactive materials may occur in either gaseous or particulate form, and may originate either directly from discharges to the atmosphere or from the re-suspension of radionuclides deposited onto soil or other surfaces.
Air is being sampled by pumping air through filters and then removing and analysing the filters. The air measurements are not very frequent under normal conditions; however, most countries will increase their measurement frequencies in the case of an emergency.
The biggest advantage of an air-borne monitoring is in possibility of relatively fast measurements and mapping of the contamination in large populated areas, especially within a short time after a release, when the capabilities of mobile ground teams are restricted.
Airborne radionuclides is an important source for radionuclides that enter the human body, either directly through inhalation or by being deposited in water or on soil or plants and contaminate the media used for human consumption. However, whether the nuclides can enter the body or lead to exposure is dependent on the nuclide and the particle size. Information about the radioactivity of airborne radioactivity can therefore also be used when estimating doses to the public, and is particularly relevant for estimating doses received via internal exposure.

It is recommended the use of mobile aerosol filters that could be placed in requested places (the same way as relocatable gamma-probes) to substitute the existing network and reduce uncertainties of measurements for the relevant area.

6.7 Deposition

The term deposition includes both wet deposition (through precipitation) and dry deposition, and represents the transfer of radioactive particulates from the atmosphere to the earth's surface, and consequently to water, soil and the food chains. Radioactive deposition can therefore lead to both external radiation exposure and to internal exposure due to ingestion of radioactive contaminants via the food chain.
Precipitation increases the rate of deposition and therefore plays an important role in determining which areas receive heavy contamination after a release. Contaminated precipitation is created either when radionuclides present in the clouds become incorporated in cloud droplets that then precipitate, or when previously uncontaminated precipitation falls through contaminated air masses. Because precipitation often exhibits relatively local variations, this can cause heterogeneous patterns of contamination. As the particle size affects how far the airborne radionuclides can travel, this will of course also be reflected in the deposited radionuclides.

When radioactive deposition occurs through snow or settles on top of snow, it can behave differently than with other types of deposition. During the winter, the deposition accumulates in the snow, which can be moved, piled up, or dumped into the water, and which can create a sudden large release when the snow melts in spring.

Deposition data can be useful for estimating doses to the public, and it is particularly useful in the later phase of an emergency situation for assessing doses and evaluating the consequences of the release.

The measurements provide data for mapping the radioactive deposition following a nuclear incident and are therefore important to emergency responses.
The deposition data would be useful to other countries in the region, especially during radiological emergencies, seeing as the airborne radioactive particulates can travel great distances before being deposited and deposition data includes information about where these particulates are settling.
Metadata that includes the amount of precipitation during the sampling period is also important to the interpretation of radioactivity in deposition samples.

6.8 Soil monitoring

In the longer term, soil acts as a reservoir for long-lived radionuclides in the terrestrial environment. The interaction between soil components and radionuclides in large part determines the bioavailability for plants and animals. The soil characteristics therefore have a large effect on the radioactive contamination of the biota. Some radionuclides are more mobile in the environment than others, and can easily be transferred to the food chains. Other, insoluble radionuclides that are less mobile tend to remain in the soil.
Undisturbed natural soils tend to retain more radionuclides for longer than cultivated soil does, in part due to the different physical and chemical properties of the soil and the generally more nutrient-poor conditions in undisturbed soils (which increases the cesium–potassium ratio and in turn increases the relative uptake of cesium). Soil erosion and ruoff can redistribute the radionuclides in the contaminated soil across land areas or into water, and fires can release accumulated radionuclides to the atmosphere.

Measurements of radioactivity in soil is also to a large extent related to the transfer of atmospheric radionuclides to the food chains, but unlike the sampling media previously discussed, it also provides information about the accumulation of deposited radionuclides in the terrestrial environment over time. This offers additional insight into the cumulative environmental impact of several or diffuse sources of radioactive material, and can be used to monitor the movement of radioactive material in the environment as well. Although soil may not be directly relevant for dose assessments, it is a good indicator for the amount of radionuclides present in the terrestrial environment.

6.9 Food and water monitoring

Food and drinking water are of course very important for making assessments of radiation doses to the public. Extensive, regular monitoring is probably most relevant in contaminated areas, seeing as these levels can vary, for example due to variations in climatic conditions.

6.10 Sampling frequency and geographical density

Sampling and measurement frequencies and the geographical distribution and density of the sampling sites are partly determined by national priorities and objectives, and are particularly dependent on the financial resources available.

Geographical distribution and density should comply with standards for representative sampling when relevant. It would still be possible, however, to define a recommended minimum for frequency and/or density for some media (for example, minimum once per year). It could also be relevant to pre-define increased frequencies for sampling/measurement and reporting in emergency situations for the most relevant sampling media (perhaps this in some cases might be relevant for geographical distribution and density as well).

6.11 Format for data sharing

Today, many radiation safety authorities are reporting monitoring data in different formats to several different organisations. The exchange of comparable and interpretable data is important and the data sharing process in itself will also require careful consideration. One of possibilities is the use of IAEA IRIX (International Radiation Information eXchange) standard for sharing the monitoring data. IRIX is an xml-based format standard for data exchange that has been developed under the IAEA action plan, in close cooperation with the EC.

It is important to ensure that any report contains a proper definition of uncertainty in the reported data, in order to be able to interpret the data appropriately. Monitoring data can also be valuable for scientific purposes, and clear rules for the ownership of the data before any data is shared should be taken into account.

For some sampling media, the monitoring results are highly dependent on external factors that influence the radioactivity levels. These factors can in turn make interpretation of the data difficult. For example, for soil, characteristics like whether the soil is agricultural or undisturbed, the amount/type of nearby vegetation, soil type, dry weight, etc. can be significant to the comparability and interpretation of the results.
In the marine sampling media, salinity can similarly affect the radionuclide concentrations. To address this issue, one would need to include this information metadata for the relevant media in the data reporting format.
Definitions of what kind of metadata is required would therefore be an important step of a data exchange.

6.12 Communication

In an emergency situation, the commercial GSM and GPRS-systems may collapse or could be heavily loaded. Therefore, it is important to have more than one communication channel between the mobile teams and the headquarters. The additional communication link may be a satellite phone capable of data transfer or a TETRA-based professional mobile radio.
The selection of data is crucial, since one cannot count on the normal available bandwidth in an emergency situation; therefore, it is wise not to push the transmission speed near to its extreme capacity.

6.13 Data transfer

It is advised that data transfer should be highly automated. Crucial time is saved, when the position of the mobile teams is transferred automatically and in real time to a mapping system in the headquarters. When something is found, the location is already seen on the map and the communication between the mobile team and the headquarters can focus on what was found.
All functional data transmission pipelines needs well defined formats, protocols and send/receive procedures. These are defined by the data management of the system.

6.14 Mobile monitoring

The portable instruments are applied for measurements in early phase after the radioactive cloud has passed. Measurements from these systems (i.e. in-situ measurements), combined with the data from the sampling from the affected areas, provide possibility for full and detailed overview on the borders of the affected area and radioactivity levels.

Regarding for emergency preparedness and response aspects, car-borne dose rate monitoring systems should be classified for the following tasks:

- The systems monitor dose rate in high level radioactive area and are useful to identify such area where measured dose rate exceed operational intervention levels.

- The systems monitor dose rate and indicate artificial radiation (for example plastic scintillator systems using “natural background reduction” (NBR) algorithms).
- The systems monitor dose rate and enable the assessment of nuclide specific dose rate for relevant nuclides.

After monitoring jobs, mobile teams should regularly check possible self-contamination of vehicles. Checks can rely on comparison of monitoring data outside contaminated area or using additional measuring equipment.

6.15 Environmental Monitoring Team tasks

It is recommended to reduce each monitoring task to a set of simple instructions to be provided to a monitoring team. This task plan should specify the locations to be monitored and the type of measurements to be performed (e.g. gamma dose rate, contamination with beta-emitters, contamination with alpha emitters, etc.). For an area survey, the monitoring team should be requested to monitor at the intersection points on a suitably spaced grid.

Monitoring Control should divide areas requiring monitoring into sections using physical features such as roads, rivers and buildings where possible. It could be decided to further subdivide each section for monitoring using other physical features such house numbers, trees and drain covers [TMT Handbook 2009].

Initially, the Safety Perimeter will be established at a radius of 400 m surrounding the incident. At this distance, the Safety Perimeter extends for more than 2500 metres, enclosing an area greater than 500 000 m^2, and will require considerable resources to maintain. Ideally, the area of the Red Zone should be reduced as soon as practicable.

If the dose rates or contamination levels exceed those for the ‘Safety Perimeter’ (Tab. 3), then the monitoring team must perform a more detailed survey to confirm whether the high dose rate or contamination is widespread or a "hot spot" [TMT Handbook 2009]. If the high levels are widespread, the monitoring team must physically mark the location and record results before retreating to their previous (rapidly monitored) location (5 to 10 m back).

There they must rapidly confirm that dose rates and contamination levels are within limits and report their findings to ‘Monitoring Control’. They must then perform a detailed survey of their present location and report the result, plus the current cumulative dose for each member of the monitoring team, to Monitoring Control.

Table 3 - Maximum acceptable contamination levels and dose rates at Safety Perimeter for α–, β–, and γ –emitters (IAEA EPR-First responders, 2006).

Perimeter	*Type of measurement*	*Measured value must not exceed*
Red Zone (Safety)	Alpha contamination level	100 Bq cm^{-2}
	Beta contamination level	1000 Bq cm^{-2}
	Gamma contamination level	1000 Bq cm^{-2}
	Gamma dose rate	100 µSv h^{-1}

If the high dose rate or contamination is a "hotspot", the monitoring team must physically mark the location, record the results and report their findings to Monitoring Control.

Measurement teams should be able to detect hotspots of 100 µSv/h or more at one meter distance. Such a hotspot will produce only 1 µSv/h at 10 meters distance and could be missed if the most

insensitive dose rate meters are used and the team passes the source too fast (brisk walking speed). More sensitive GM-tube based dose rate meters or Personal Radiation Detectors (PRDs) with solid state detectors should be able to detect such hotspots from even greater distances. Detection levels are highly dependent of the ambient dose rate. Near the Red Zone, the ambient dose rate from the Red Zone could still be several tens of microsieverts and thus the search for hotspots must be done with much denser search grid than used further away from the Red Zone [TMT Handbook 2009].

If four or more monitoring teams are available, then monitoring should be undertaken to determine whether the size of the Red Zone can be reduced.

7. Summary and conclusions

To summarize, in order to get a substantial and precise overview on radiation situation and doses to the public and environment following a nuclear accident, important pre-release and post-release tasks should be implemented (together with strategies described above in Chapters 1-6).

In mobile measurements, special focus is required for the following aspects:

Deployable autonomous probes: additional mobile dose rate probes are recommended to deploy using first responder teams. If possible, these probes should be deployed in the pre-release phase following a strategy discussed in MetroERM Deliverable 1.5.6. The delineation function of monitoring networks would benefit considerably from such mobile probes deployed at adequate selected locations. It is advised to discuss different possible scenarios and analyze the shortages/gaps and possibilities in national monitoring networks in advance and have a list of pre-selected locations with easy access conditions for deployment of mobile probes, when needed.

Mobile teams with trained experts and the following equipment are helpful:

- Car-borne dose rate probes – e.g. plastic scintillator or NaI scintillator and aerosol filters;
- In situ gamma spectrometry systems – e.g. HPGe detectors; hand held dose rate probes;
- Helicopter based aero-gamma systems for long range surveys;
- Drones based aero-gamma systems for short range surveys.

Planning process for additional measurements by mobile teams should reflect:

- Radiation protection issues for the teams;
- The needs of decision making process;
- The benefit of additional data for information on the public.

This report is a sub-report for D 9.7 in WP2 of the EU CONFIDENCE project and reflects mainly mobile measurements, related uncertainties and provides suggestions and recommendations on reducing of uncertainties from mobile measurements in order to get more accurate monitoring data for prognostic models and assessment of doses to population and the environment.

For details on stationary monitoring systems with related uncertainties and for a complex overview on the uncertainty reduction in exposure assessments, see the sub-report D 9.7.1 'Stationary monitoring systems and their uncertainties' and the main report D 9.7 'Report on uncertainty reduction in exposure assessment based on environmental monitoring data, including concept for identifying critically exposed groups'.

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