Requirements for estimation of doses from contaminants dispersed by a 'dirty bomb' explosion in an urban area

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Abstract The ARGOS decision support system is currently being extended to enable estimation of the consequences of terror attacks involving chemical, biological, nuclear and radiological substances. This paper presents elements of the framework that will be applied in ARGOS to calculate the dose contributions from contaminants dispersed in the atmosphere after a 'dirty bomb' explosion. Conceptual methodologies are presented which describe the various dose components on the basis of knowledge of time-integrated contaminant air concentrations. Also the aerosolisation and atmospheric dispersion in a city of different types of conceivable contaminants from a 'dirty bomb' are discussed.

Keywords: 'dirty bomb', terror attack, explosion, radioactive contamination, decision support, dose model, atmospheric dispersion, particle size, countermeasure, urban areas.

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1. Introduction

Following the attacks on the US World Trade Center and Japanese and Spanish subways, as well as on embassies and hotels, authorities over the world have in recent years become increasingly aware of the need for an efficient and focused preparedness against terrorism. A number of 'new threats' have attracted attention, including the possible malicious airborne dispersion of bacteria, vira, toxic chemicals and/or radiological substances. Urban areas would be likely to be major targets, as high numbers of people and important infrastructural elements could be affected there. Consequence assessment tools are needed both for predicting the consequences of different types of conceivable attacks so that an effective and focused preparedness can be established, and for optimising intervention in the event of an actual attack. The tool would need to be complex, as the consequences of an attack would depend on a multitude of case-specific factors, such as the means of dispersion, characteristics of the contaminants, meteorological conditions and features of the urban environment. It is considered possible that an attack might involve dispersion of several different types of agents, and a consequence assessment tool should therefore enable analyses of combined releases. Such features are currently being developed for the ARGOS system, which was originally created for decision support in connection with nuclear power plant accidents, and is currently operational in 11 different countries over the world (ARGOS, 2008). This system will also enable incorporation of measurement data, as they become available after an incident, in improving consequence prognoses (Kaiser et al., 2008). The focus in this paper is on the methodologies that are being implemented to estimate the distribution of doses received through different pathways from contaminants dispersed by a 'dirty bomb' explosion. The immediate vicinity of

the site of a ‘dirty bomb’ detonation would be likely to be monitored very extensively, and in this rather small area it would also be impossible to reliably model the distribution of contaminants, due to the dominance of large contaminated shrapnel. However, depending on, e.g., the explosive mass and physicochemical form of the contaminant, a considerable part of the contamination could be dispersed as fine particles and contaminate a rather large area. Here consequence assessment models can be highly valuable. Simplified calculations have indicated that at least under some circumstances, a well-staged ‘dirty bomb’ attack could over several kilometres distance result in significant exposure from dispersed contaminants (Andersson et al., 2008).

2. Methods and results

In connection with the EURANOS project supported by the European Commission, a new modelling tool, ERMIN (EuROpean Model for INhabited areas), was established for the estimation of dose in inhabited areas contaminated as a result of a large accidental release from a nuclear power plant (Andersson et al., 2008a; Jones et al., 2008). This model has now been integrated in the ARGOS emergency decision support system. ERMIN enables estimation of external and internal dose contributions received in the intermediate to late phase following an accident, with the exception of doses from ingestion of food products, which are often only produced in limited amounts in the inhabited environment. On the basis of air concentrations delivered from an atmospheric dispersion model and, if available, monitoring data, ERMIN calculates the likely initial distribution of contaminants on the different types of surfaces in the environment. A complex of transfer processes are reflected in the equations describing the subsequent contaminant migration. Using a library of conversion factors from surface contamination to dose rate contributions for different geometries and radionuclides, the radiological implications of an emergency can then be estimated. The basic concept of the ERMIN model is clearly applicable outside its intended scope of decision support for nuclear power plant release scenarios, as essentially the same types of processes govern many of the dose contributions in connection with for instance a ‘dirty bomb’ scenario. However, these two different types of emergencies differ greatly in a number of ways that strongly influence model parameterisation and the importance of including different dose pathways, while at the same time setting different demands to atmospheric dispersion modelling. In the following a discussion is given of some important issues that are being dealt with in the development and parameterisation of the new extension of the ARGOS decision support system for ‘dirty bomb’ scenarios. On this basis, the requirements for calculation of each dose component of possible significance in connection with atmospheric dispersion of contaminants from a ‘dirty bomb’ attack are discussed.

2.1. Radioactive sources of concern

A first requirement in modelling the consequences of any contaminant release scenario is an identification of the types of sources that are most likely to be dispersed. Decision support systems designed for dealing with nuclear power plant accidents are generally equipped with source data libraries that describe reactor inventories and likely release fractions of the different radionuclides for different accident categories. Physico-chemical forms of the radionuclides are, at least prior to

the accident, well known. By contrast, the contaminants that might be dispersed by a 'dirty bomb' could at least in principle be any one or more of a wide variety of radionuclides, and their pre-explosion physico-chemical forms would be less predictable. However, in reality, the list of relatively well suited and possibly available sources would for various reasons probably be quite short. If the possibility of terrorists obtaining access to facilities for production of strong radioactive sources is disregarded as unlikely, terrorists would need to apply existing sources. The availability of existing sources with sufficient strength to cause real harm if effectively aerosolised and released over an inhabited area would largely depend on how widely they are used for peaceful purposes. To obtain existing sources, terrorists would need to either locate an abandoned or lost ('orphaned') source, steal it from a licensed user or manufacturer, succeed in buying it by pretending to be a legitimate user with a peaceful industrial purpose (Anon, 2007), or buy it on a black market that might exist considering the many strong sources that are annually reported lost (Ferguson et al., 2003).

A classical example of a strong 'orphaned' source that suddenly appeared completely out of context in an inhabited area, causing severe health problems, is the Goiânia case in Brazil. Here a 50.9 TBq ^{137}CsCl source (in readily soluble powder form) caused contamination of a considerable area and many people, of which four were reported to have died within weeks (Leao & Oberhofer, 1988). The Goiânia experience is particularly interesting in relation to 'dirty bomb' scenarios, as this accident led to the contamination of a city area of a size equivalent to what might be affected by the contamination from a successful 'dirty bomb' attack (Andersson et al., 2008). Although a multitude of transportation and migration pathways contributed to the spreading of the contamination, it has been reported that the primary cause of contamination of housing in the area was atmospheric dispersion (Da Silva et al., 1991). The Goiânia accident resulted in extensive panic, social disruption, great efforts and expenses for personal monitoring, decontamination of humans and area clean-up, for instance of roof tiles with ^{137}Cs contamination levels ranging up to some 700 kBq m^{-2}.

Nevertheless, significantly stronger 'orphaned' sources might be considered for use in connection with a 'dirty bomb' attack. For instance, in December 2001, three residents of Tsalenjikha in Georgia suffered severe radiation sickness and skin burns after having found a 1300 TBq ^{90}Sr source in a forest (Wedekind, 2002). This source had in the Soviet days been in use to power a radioisotope thermoelectric generator (RTG) for a communication tower in a remote nature area. Hundreds of RTG's with ^{90}Sr sources with strengths typically ranging between 1000 and 10000 TBq were distributed over the Soviet Union to provide power for lighthouses, beacons, and other unmanned facilities, and many of these are no longer kept track of (Ferguson et al., 2003).

According to the IAEA (Yusko, 2001), 'orphaned' sources had by 2001 been involved in 60 severe radiological accidents, due to which 266 persons had been overexposed and 39 had died. Over the world, thousands of potentially harmful sources are annually reported lost. However, probably only a very limited fraction of these sources would be suited for use in a 'dirty bomb' attack. For instance, a number of radionuclides can essentially be excluded because their physical half-lives are so short that it could be argued that any contamination of an area would disappear by itself over a short period of time. Moreover, to cause any real concern by atmospheric dispersion over a city area, a source would need to be quite strong. On the other hand, very strong sources would often require substantial shielding

arrangements to allow handling prior to detonation in a 'dirty bomb', which would make them difficult to transport and position. Table 1 shows conversion factors that can be used to calculate the external dose without shielding to persons standing for some time at a distance of 1 m from various types of gamma-emitting sources that might possibly be applied in a 'dirty bomb'. Obviously, sources in the TBq range require very special handling arrangements.

Table 1. Estimated effective dose conversion factors for a person standing at a distance of 1 m from an unshielded source. Values are given for a number of gamma-emitting radionuclides that might be used in a 'dirty bomb'. (Lauridsen, 1982).

Radionuclide	Sv/h/TBq
^{60}Co	$2.6\ 10^{-1}$
^{137}Cs	$6.4\ 10^{-2}$
^{192}Ir	$1.1\ 10^{-1}$
^{226}Ra	$1.8\ 10^{-1}$
^{241}Am	$3.1\ 10^{-3}$
^{252}Cf	$3.2\ 10^{-4}$

It has been inferred that beta/gamma sources with strengths in the range of 0.1-100 TBq might be best suited to fulfil the two conflicting criteria of high source strength and transportability (Sohier & Hardeman, 2006). However, weaker sources might still be instrumental in creating some degree of social disruption, and stronger sources might be applicable, depending on the device design, the means for transportation, and the perpetrator's desire to survive. Pure beta or alpha sources would of course be less problematic to shield, and stronger sources might thus here be applied. Nevertheless, the detonation of a bomb to effectively aerosolise even a very small fraction of the radioactive material from within a shielding arrangement could well be exceedingly problematic (Harper et al., 2007). Table 2 shows the characteristics of a selection of existing strong sources that might be of particular concern (Harper et al., 2007; Ferguson et al., 2003; Argonne, 2005). For decision support, dose conversion factor libraries for different relevant dose pathways and geometries need to be established for at least these radionuclides. It should still be noted that offenders may if possible strive towards the unexpected, as illustrated by the use of ^{210}Po in 2006 in connection with the murder of the former KGB officer Alexander Litvinenko.

Table 2. A selection of potentially important existing strong sources, including typical physicochemical forms of large existing sources and maximum activity estimates.

Radionuclide	**Typical physicochemical form of large existing sources**	**Existing sources and their strengths**
^{60}Co	Metal - soluble in acid	Sterilisation irradiator (max. 400.000 TBq). Teletherapy source (max. 1000 TBq).
^{90}Sr	Ceramic ($SrTiO_3$) - insoluble, brittle, soft	Radioisotope thermoelectric generator (1000-10.000 TBq).
^{137}Cs	Salt (CsCl) - readily soluble	Sterilisation irradiator (max. 400.000 TBq). Teletherapy source (max. 1000 TBq).
^{192}Ir	Metal – soft, insoluble in water	Industrial radiography source

		(max. 50 TBq)
^{226}Ra	Salt ($RaSO_4$), very low solubility	Old therapy source (max. 5 TBq)
^{238}Pu	Ceramic (PuO_2) - insoluble	Radioisotope thermoelectric generator (max. 5.000 TBq).
^{241}Am	Pressed ceramic powder (AmO_2) - insoluble	Well logging source (max. 1 TBq).
^{252}Cf	Ceramic (Cf_2O_3) - insoluble	Well logging source (max. 0.1 TBq).

2.2. Aerosolisation and atmospheric dispersion

Recent experimental work by Harper et al. (2007) has demonstrated that the contaminant particle size distribution that can be expected after an explosion depends strongly on the elemental properties of the contaminants, their physicochemical forms, and the more or less successful construction of the explosive device. As indicated by Table 2, typical physicochemical forms can be broadly distinguished in four categories: ceramics, metals, powders/salts and liquids. It is clear from the work of Harper et al. (2007) that during shock aerosolisation some types of sources can, if the pressure is high enough, undergo phase changes to either liquid or vapour form. Thereby, small liquid or vapour condensation particles can be formed, which have a low deposition velocity to surfaces in the environment. Such particles will thus remain airborne over long periods of time and may contaminate a quite large area, even though their effective release height is in general very limited compared with for instance what was observed in connection with the Chernobyl accident. In other types of explosions, for instance involving contaminants in ceramic form, the particles are generally formed by physical fractionation, and therefore larger, leading to distribution of the contaminants over a somewhat smaller area. The sizes of the dispersed particles in general have great importance for the consequences of any atmospheric release of contaminants. For instance, the dry deposition velocity to a lawn is typically higher by a factor of about 30 for 20 µm particles than for 2 µm particles, and due to the different processes (Brownian diffusion, impaction, interception, gravitational settling, etc.) governing the dry deposition of particles with different sizes on environmental surfaces of different materials and orientation, also the distribution of the contamination on the various surfaces in the inhabited environment will be highly dependent on particle size. This issue is addressed in the new ARGOS system for RDD's (Radiological Dispersal Devices) through the inclusion of a series of data libraries describing the deposition of aerosols in intervals of the relevant size ranges for five different deposition 'modes': dry deposition, deposition in light rain, deposition in heavy rain, deposition in snow, and dry deposition to a snow-covered environment.

An other factor of interest in connection with the aerosolisation process is of course the *fraction* of the contaminants that can be aerosolised in a plausible explosion process. For ceramics, Harper et al. (2007) have recorded aerosolisation of between 2 and 40 % of the contaminant mass. The figures strongly depend on the construction of the explosive device, and in general, values in the lower end of the interval would be expected. The figures can on average be expected to be slightly higher for powders, whereas for some sources in metallic form, including ^{60}Co and

^{192}Ir, less than one percent aerosolisation would be expected. Metals are in general only aerosolised if phase transition occurs (Harper et al., 2007). The new ARGOS contains data libraries of aerosolisation fractions and particle size distributions that would be expected in connection with explosions of different types of 'dirty bomb' devices with contaminants in different physicochemical forms.

In modelling the dispersion in an inhabited complex of aerosolised radionuclides following a 'dirty bomb' attack, a requirement is to determine the initial plume dimensions and elevation height distribution of the contaminant aerosols generated by the blast. This is needed as input to atmospheric dispersion models to estimate how the contamination will in time spread out over an inhabited area. Plume rise obviously has great influence on the size of the affected area, and determines the significance of plume interaction with environmental structures (e.g., buildings, trees). The initial rise of the contaminated material after the blast will occur due to buoyancy and initial momentum. As the cloud rises, its movement will cause turbulent mixing with non-buoyant ambient air. Deceleration through decreasing buoyancy will reduce the boundary turbulence to that of the ambient air, by which point the 'initial' plume will have been formed.

In the new ARGOS feature dealing with 'dirty bombs', the parameterisation of plume rise relates to several independent blast studies. Figure 1 shows a 'qualitative' example of curves describing the initial plume top height and cap diameter as a function of the TNT equivalent explosive mass, on the basis of data from a US blast test series conducted in 1963 to investigate the effect of accidental conventional explosions spreading radioactive material from a nuclear device. Based on among other things this experience, the initial plume configuration is modelled in ARGOS by five vertically stacked plume elements (ellipsoids/spheres) constituting different parts of the 'cap' and 'stem' of the plume. Each of these elements is modelled as an individual puff containing a defined fraction of the total aerosolised contaminant mass. It should be noted that in accordance with the applied experimental work, the contaminant concentrations in the various stacked volume elements differ greatly.

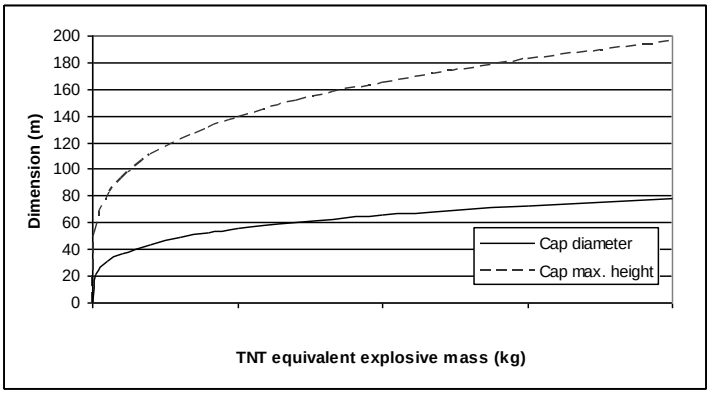

Figure 1. Plume top height and cap diameter following the detonation of a 'dirty bomb', as a function of TNT equivalent explosive mass. 'Qualitative' curves based on the US test data from 1963.

For modelling the subsequent airborne dispersion of the contaminants over the environment, it is important to apply methods that adequately account for the mechanisms governing the flow and deposition in relation to the given dispersion altitude and scale. Here a considerable degree of simplification is traditionally applied in decision support models, which are designed to describe the long-range transport of contaminants from high-altitude releases following large nuclear power plant accidents. Since the plume here generally passes well over the various environmental obstacles, the influences on the plume propagation in inhabited areas

can be reasonably simulated through the use of different overall roughness and deposition rate parameters compared with, e.g., rural areas (Päsler-Sauer, 1997; Mikkelsen et al., 1984). However, recent investigations have shown such simplifications to be problematic, when the contaminant dispersion partially takes place at street level, as would be the case following a 'dirty bomb' explosion. This is particularly true for scenarios involving shifting wind directions (Astrup et al., 2005). Here, higher resolution models are required to address the issues of plume interaction with and flow along obstacles in the inhabited environment. Therefore, a new atmospheric dispersion module for inhabited areas, URD (Urban Release and Dispersion), based on Gaussian puffs at high resolution (Fackrell, 1984), has been developed at Risø-DTU for the ARGOS system. This new implement is in some of its basic features inspired by the UDM code developed by the UK Defence Science and Technology Laboratory (Hall et al., 2002), and incorporates plume interaction with environmental obstacles in three different ways. The obstacles form barriers limiting the magnitude of horizontal eddies in the atmosphere, which in turn reduces the large-scale horizontal dispersion. At the same time, the interaction will increase the small-scale turbulence over a city, which will lead to greater small-scale dispersion. Finally, obstacles like downstream building walls constitute barriers that will to some extent delay the further dispersion of the contaminants.

Calculations for a test scenario involving an airborne release in central Copenhagen at a height of 2 m above ground level have illustrated that particularly the plume interaction effect over short distances, which influences the larger scale dispersion, is important (Andersson et al., 2008). Overall, a much more diffuse dispersion pattern was in this case observed when the modelling was alone carried out with ARGOS' longer range dispersion module RIMPUFF (Mikkelsen et al., 1984), and time-integrated downwind air concentrations were underestimated within the nearest few hundred metres of the release point, whereas the dispersion pattern at greater distances was distorted and could mislead decision making.

2.3. Resulting dose components

The dispersion of the contaminants from a 'dirty bomb' attack in an inhabited area can result in doses to the population through a number of pathways, including:

- Internal dose from inhalation during plume passage
- Internal dose from inhalation of resuspended contaminants
- External dose from the passing contaminated plume
- External dose from contamination on outdoor surfaces
- External dose from contamination on indoor surfaces
- External and internal dose from contamination deposited on humans
- External dose from contaminants transferred onto humans by contact
- Internal dose from ingestion

The relative importance of these various dose contributions will strongly depend on the scenario in question. For instance, some conceivable scenarios would involve dominant fractions of aerosols in the non-inhalable size range, and in some situations, the dispersed radionuclide could be a pure alpha-emitter (e.g., ^{238}Pu or ^{252}Cf), in which case all external dose contributions can be excluded. This section is aimed at providing an overview of the requirements for calculating each dose component from

knowledge of the relevant time-integrated contaminant air concentration. The modelling in ARGOS will follow this pattern.

2.3.1. Internal dose from inhalation during plume passage

This type of dose component was not considered in the ERMIN model, which does not provide estimates of doses that are received exclusively from exposure in the early phase (first few days) after a contaminating incident. In general, particles smaller than about 10 µm are considered inhalable, whereas larger particles will be cleared from the respiratory tract through natural ciliary action (ICRP, 1994). If people are outdoors during the entire period of the plume passage (t_1), the relevant effective committed inhalation dose can be calculated from the formula:

$$(1) \qquad D = \Gamma \cdot BR \cdot \int_{t=0}^{t_1} C_{o,a}(t) \cdot dt \,,$$

where Γ is the age-specific inhalation dose coefficient relevant to the dispersed physicochemical form(s) of the contaminant in question (notably particle sizes and absorption rate into the blood have importance; ICRP, 1996; ICRP, 2002), *BR* is the human breathing rate (varying between sexes, age groups, and exercise levels; California Air Resources Board, 2008; ICRP, 2002), and $C_{o,a}(t)$ is the outdoor air concentration time function at the relevant height above the ground. If persons were indoors during part of this exposure, the corresponding fraction of the integral dose should be multiplied by the relationship between indoor and outdoor air contaminant concentrations. This is at equilibrium given by:

$$(2) \qquad \frac{C_{i,a}}{C_{o,a}} = \frac{f \cdot \lambda_v}{\lambda_v + \lambda_d} \,,$$

(Roed & Cannell, 1987; Andersson et al., 1995) where f is the filtering factor (the fraction of aerosols, which are not retained in cracks and fissures of the building structure, as air enters the building), λ_d is the rate coefficient of indoor deposition (the fraction of aerosols in the building depositing per unit of time), and λ_v is the rate coefficient of ventilation (the fraction of air in the building that is exchanged per unit of time). It should be noted that both f and λ_d are highly dependent on particle size, and that the above relationship thus needs to be calculated for different sections of the relevant contaminant particle size spectrum. For instance, f will generally be close to unity for the particles smaller than 2 µm, but only about 0.1 for particles in the 10 µm range (Roed, 1990; Long et al., 2001). Over the same particle size interval, λ_d increases by more than one order of magnitude (Andersson et al., 2004; Long et al., 2001). The value of the rate coefficient of ventilation depends on building tradition and season, as well as on whether the inhabitants are aware that there might be a special reason to carefully close off air ducts. Also, the building might be equipped with a HVAC (heat, ventilation and air conditioning) system. With an efficient filter, these have been reported to remove of the order of 10-15 % of the particles in the 1 µm range, about half of the particles in the 2 µm range, and practically all particles greater than ca. 4 µm (Thatcher et al., 2001).

On average, it would under normal circumstances seem reasonable to assume that people spend some 85 % of the time indoors, although some climatic and seasonal variation occurs (Andersson & Roed, 2006). This pattern is likely to change when the nature of the incident becomes known.

2.3.2. Internal dose from inhalation of resuspended contaminants

Equation 1 is also applicable for the calculation of doses from inhalation of resuspended contaminants. The only difference is that the relevant outdoor air concentration is here given by the product of the contaminant concentration on the surface from which resuspension occurs by a resuspension factor. Investigations of the contaminant resuspension in cities following the Chernobyl accident (Garland & Pomeroy, 1994) showed that the time variation of the radiocaesium resuspension factor could be described by the equation:

$$(3) \qquad K(t) = K_0 \cdot \exp(-a \cdot t),$$

where K_0 was estimated to be $3.1 \cdot 10^{-8}$ m^{-1}, and the time constant, a, to be 0.51 y^{-1}. However, the first measurements applied to derive this formula were obtained some months after the accident. For the shorter term, Garland (1982) showed in a series of wind tunnel experiments with particles in the range of 2-30 μm deposited on grass that the resuspension over the first days or weeks after the initial deposition decreased according to the equation:

$$(4) \qquad K(t) = 1.2 \cdot 10^{-6} \cdot t^{-1},$$

where t is in units of days. At a time of about 40 days, equations 3 and 4 give the same result, and it has therefore been suggested to apply equation 4 for the earliest 40 days, and equation 3 for subsequent time intervals (Jones et al., 2008). It should be noted that large particles are in general more prone to resuspension than small particles, due to the different forces acting on them (Corn & Stein, 1965; Thatcher & Layton, 1995; Phillips, 1980). In connection with the Chernobyl studies, any resuspended radiocaesium would by the time that measurements were carried out have been likely to have become associated with larger environmental particles. By contrast, Garland's wind tunnel experiments of early phase resuspension were made with particles of different sizes, and as expected, the larger particles were found to be somewhat easier to resuspend. Also, different contaminants deposited in different physicochemical forms will have different affinities to soil and other environmental surfaces, which can influence their ability to be resuspended. The data of Garland & Pomeroy (1994) and Sehmel (1973) clearly show that resuspension is connected with considerable variation, e.g., according to wind speed and traffic. However, Sehmel (1973) found that even traffic-generated contaminant resuspension decreased by 2-3 orders of magnitude over the first month following the initial deposition.

It should be stressed that the total individual inhalation dose from resuspended contaminants would often be small compared with the corresponding inhalation dose received in the same area during an initial plume passage in dry weather (Andersson & Roed, 2006). On the basis of experimental data, Andersson et al. (2004) demonstrated that particularly indoor resuspension is likely to lead to low time-integrated inhalable contaminant air concentrations compared with that occurring due to the passage of the initial contaminated plume.

2.3.3. External dose from the passing contaminated plume

This dose component, which was not considered in the ERMIN model, can be estimated by multiplying time-integrated contaminant air concentrations by dose conversion factors for persons submersed in contaminated air of semi-infinite dimensions. Such factors are given by Hedemann Jensen (1992) and Kocher (1980) for a range of beta and gamma emitters, including ^{60}Co, ^{90}Sr and ^{137}Cs. These table values show that although different types of radiation are at play, equal air

concentrations of these three radionuclides lead to dose contributions of the same order of magnitude at the skin surface. Andersson & Roed (2006) demonstrated that for a dispersion scenario involving ^{137}Cs contamination in more or less readily soluble form, the external dose received during the initial plume passage would be likely to be small compared with the corresponding effective committed inhalation dose.

2.3.4. External dose from contamination on outdoor surfaces

The average external dose rate to inhabitants from the contamination on each surface in the environment can be calculated as the time-weighted average of dose rate contributions from the surface in question to a person present in different representative locations in the environment. Each dose rate contribution from the given surface contamination with a radionuclide with the physical half-life $T_{½}$ can be calculated from the equation:

$$(5) \qquad D'(t) = \Gamma \cdot C_0 \cdot r(t) \cdot \exp(-\frac{\ln 2}{T_{½}} \cdot t),$$

where Γ is the dose rate conversion factor appropriate for the particular surface and target location geometry, C_0 is the initial contamination level on the surface (time-integrated air concentration over the period of the plume passage multiplied by a deposition velocity), and *r(t)* is a time-function describing the reduction in dose rate from the surface due to naturally occurring migration processes.

The migration of contaminants on the different types of surfaces in an inhabited environment is a topic that has been dealt with extensively in ERMIN, as well as in previous urban dose models like EXPURT (Jones et al., 2006), URGENT (Andersson et al., 1995) and EDEM2M (Eged et al., 2006), primarily on the basis of measurements made after the Chernobyl accident. Although longer time-series information is practically only available for radiocaesium, it has been argued that the specific affinities of soil and common urban construction materials for retaining caesium would imply that, if based on the same migration data, long-term dose estimates for other radionuclides would be somewhat conservative (Jones et al., 2006; Andersson et al., 2008a). For instance, a significantly higher ionic migration rate in soil for strontium has been recorded than for caesium (Salbu, 2000), and since cobalt ions are sorbed on the surfaces of clay minerals, and not fixed in selective sites (Norrish, 1975), also these would have a comparatively higher migration rate, as recorded experimentally by Solovitch-Vella & Garnier (2006).

Early measurements made in Denmark after the Chernobyl accident on contaminated roof materials containing minerals known to have specific capacity to strongly fix and retain caesium showed that the fractions removed by the first heavy rain shower of a range of the different Chernobyl contaminant ions were similar (Roed, 1987). Only later, when the contaminant ions had moved into micro-cavities of the material, the specific sorption mechanisms became apparent, and caesium was found to be somewhat more efficiently retained already over the following month.

Experimental work is merited to improve estimates of the natural longer-term migration of other contaminant ions than caesium in the inhabited environment. However, it is even more important to distinguish between the different physicochemical forms of the deposited contaminants. For instance, as mentioned in Section 2.2, effective aerosolisation of metals in a 'dirty bomb' has been reported to involve phase transition, and the solubility and environmental behaviour of small particles created by condensation of evaporated contaminants would be expected to be very different from that of the larger particles generated by physical fragmentation of a virtually insoluble material. It is important to take into account that large particles

are considerably easier to remove from surfaces in an inhabited environment, both by natural and forced processes, than are small particles and contaminants in solution. This can be illustrated by results obtained by hosing water at the same pressure on similar sandstone walls that had been contaminated by the Chernobyl accident, in Pripyat only about 3 km from the power plant, and in Vladimirovka, some 65 km away. In Pripyat, where much of the contamination was in the form of large and insoluble particles, the treatment removed some two-thirds of the caesium, but as far away as Vladimirovka, where the contaminants were primarily in the form of small, soluble condensation particles, only about one-fifth of the caesium could at the same time be removed (Roed & Andersson, 1996). An other example is that Clark & Cobbin (1964) were able to remove as much as 98 % of a contamination with 44-100 µm particles from a street surface by firehosing. The result of early action with the same method in Swedish areas contaminated by the Chernobyl accident was typically a removal of 50-60 % of the radiocaesium (Brown et al., 2007; Andersson et al., 2003).

Kashparov et al. (2004) demonstrated that the dissolution in soil of deposited contaminant particles with high chemical stability is a process lasting over several years, depending on the soil pH. This will delay the migration in soil of the contaminants initially present in a low solubility matrix. However, a slight contaminant penetration into the soil might in any case occur with the first heavy rainfall (Roed, 1990; Ivanov, 1997), and for ^{90}Sr/^{90}Y, adding just 1 cm soil shielding would give an external dose reduction by about 3 orders of magnitude (Eckerman & Ryman, 1993).

Finally, it should be mentioned that additional dose conversion factors are needed, both for radionuclides that have in the past not traditionally been considered of importance in the context of environmental contamination emergencies (e.g., ^{60}Co and ^{192}Ir), and for ^{90}Sr, for which detailed factors for appropriate geometries are not available for the task.

2.3.5. External dose from contamination on indoor surfaces

Tracer experiments have shown that for the relevant particle size range, deposition velocities to the floor are typically at least one order of magnitude higher than those to the indoor walls and ceiling. The relationship between the time-integrated average deposition, C_i, per unit area in the room and the time-integrated dry deposition on an outdoor reference surface, C^r_o, can then be adequately described by the equation (Andersson et al., 2008a):

$$(6) \qquad C_i / C^r_o = (1/v_d^r)\, f\, H\, \lambda_d\, \lambda_v / (\lambda_v+\lambda_d),$$

where v_d^r is the deposition velocity on the reference surface, f is the filtering factor, H is the height of the room, λ_d is the rate coefficient of indoor deposition, and λ_v is the rate coefficient of ventilation. The external dose rate from the indoor contamination with a radionuclide with the physical half-life $T_{½}$ can then be calculated from the equation:

$$(7)\ D'(t) = \Gamma \cdot C_i \cdot \exp\left(-\frac{\ln 2}{T_{½} + T_{½,c}} \cdot t\right),$$

where $T_{½,c}$ is the natural clearance half-life of contaminants from indoor surfaces, and Γ is the dose rate conversion factor for the relevant indoor exposure geometry. C_i is

found as the time-integrated indoor air concentration over the period of the plume passage multiplied by an indoor deposition velocity.

2.3.6. External and internal dose from contamination deposited on humans

As it was assumed that this dose component would be received over a period of only few days, it was not considered in the ERMIN model. However, it has been estimated that it can at least in some 'dirty bomb' scenarios contribute considerably to the total dose (Andersson et al., 2008). The importance of distinguishing between the times spent indoors and outdoors is characterised by equation 2. The total dose received from each contaminant deposited on the human body can be calculated from:

$$(8) \qquad D = \Gamma \cdot \frac{1}{1/T_{½} + 1/T_{½,s}} \cdot v_{d,s} \cdot \int_{t=T_o}^{T_1} C(t)dt$$

(Andersson & Roed, 2006), where C(t) is the contaminant air concentration time function (taking into account time periods spent indoors and outdoors), $v_{d,s}$ is the deposition velocity to the human body surface in question (skin, hair, clothing, eyes), $T_{½,s}$ is the natural clearance half-life of the contaminant from the surface, $T_{½}$ is the physical half-life of the radionuclide in question, and Γ is the dose rate conversion factor. Potentially important dose contributions are beta dose to the skin and eyes and gamma dose to the body. It should be noted that both deposition velocities and clearance half-lives are strongly dependent on particle size. For instance, without forced removal (i.e., thorough washing and scrubbing), particles in the 10 µm range typically have a clearance half-life on human skin of some 3 hours, whereas the slightly submicroneous condensation particles that could also be produced by a 'dirty bomb' would tend to lodge in such cavities as hair follicles, where they may remain until the shedding of the stratum corneum after a few weeks (Hession et al. 2006). Even particles as large as several microns are not easily removed from the skin surface by force (Andersson et al., 2004).

In spite of the efficient protective barrier constituted by mainly the stratum corneum, it should be noted that particles in the submicron range may to a limited extent enter the body through intact skin (Schaefer & Lademann, 2001). However, a requirement for such transport is vigorous flexing of the exposed skin. The stratum corneum consists of a number of layers of randomly stacked corneocytes surrounded by lipids. These lipids form a winding, yet continuous, channel between the skin surface and the epidermis. The flexing force and its duration together with the particle size will essentially govern the possible extent of particle penetration through these channels. However, such particles only extremely rarely reach beyond the epidermis (Tinkle et al., 2003). Also the mobility of a range of radionuclides in solution has been found from in vitro experiments to be very low (Koprda et al., 2000). It can not be ruled out that skin contamination can under some circumstances be a source of some inner body contamination, but the available data is at present too sparse and not sufficiently quantitative to allow reliable consequence modelling.

2.3.7. External dose from contaminants transferred onto humans by contact

Contact transfer is not a pathway that is traditionally included in decision support models for atmospheric releases. Nevertheless, for instance the Goîania accident and the attack in London against Alexander Litvinenko demonstrated that this pathway can under some circumstances add rather unpredictable elements to an environmental contamination distribution. Evidently, people who have after a 'dirty bomb' emergency touched surfaces in a strongly contaminated area with their skin, clothing or shoes can inadvertently carry contamination with them out to less contaminated

areas. The extent of this problem is however virtually impossible to model reliably, since it requires exact knowledge on the behaviour patterns of individuals. In comparison with the direct contamination of humans in the *same area* after an airborne release, contact transfer by touching a surface is however probably of limited importance. Although as much as 87 % of a contamination on a surface has in experiments been found to be transferred to human skin by contact (Byrne, 2008), depending on, e.g., the surface type, particle size and skin moisture, it is estimated that contact transfer would for a given scenario involving ca. 0.5 µm particles typically maximally lead to about one-third of the skin contamination from airborne deposition, and the fraction is considerably less for larger particles (Andersson et al., 2004). Naturally, contact transfer to humans of contaminants is a process that can occur repeatedly, if contamination remains loosely held on the top of surfaces in the environment. However, the large contaminated environmental dust particles that could be transferred after a few days would have an exceedingly short natural clearance half-life on human skin (Hession et al., 2006).

2.3.8. Internal dose from ingestion

Food items are normally produced outside inhabited areas - in particular outside urban centres, which would be considered primary targets for a 'dirty bomb' attack. Although food is stored in urban areas, e.g., in shops and homes, it is often kept sealed, and since recommendations not to eat food that was unsealed at the time of the attack could rapidly be issued to the public in a contaminated area, the actual significance of this dose component might not be great. However, the deposition velocity to a plate of hot and moist food might for particles as large as 5-10 µm be as high as 10^{-2} m s^{-1} (Andersson et al., 2004). On a plate with a food surface area of, e.g., 0.03 m^2, a time-integrated air contaminant concentration of 1 Bq s m^{-3} would then give a total contaminant deposition on the food of 1 Bq s m^{-3} · 10^{-2} m s^{-1} · 0.03 m^2 = 3 10^{-4} Bq. If a breathing rate of 3 10^{-4} m^3 s^{-1} is assumed (corresponding to light exercising; ICRP, 2002), the same time-integrated air concentration of these particles would through inhalation also give an intake of 3 10^{-4} Bq. According to ICRP 72 (1996), the effective committed dose conversion factor for ^{137}Cs for ingestion is 1.3 10^{-8} Sv Bq^{-1}, whereas it is for inhalation (class F) only 4.6 10^{-9} Sv Bq^{-1}, indicating that the ingestion pathway can at least in some situations have importance in connection with an urban 'dirty bomb' scenario. An other ingestion dose scenario involves contact transfer to the tongue of toddlers licking on contaminated surfaces, which would however in many situations be likely to result in a more modest contaminant intake, due to the likely size of the contact area and the lower contaminant deposition velocity on many indoor surfaces.

2.4. Countermeasures

Decision support systems must also enable estimation of the effect of methods for forced dose reduction. Depending on the contaminant particle sizes, particularly inhalation doses and doses from deposition on humans may be reduced substantially, if people quickly shelter indoors with air ducts closed off. A reduction of the air concentration at equilibrium by a factor of 2 would typically be expected for particles in the 0.5 µm range, in comparison with a factor of the order of 100 for particles in the 10 µm range. However, considering that the period with the highest plume concentrations in the affected areas will be short (possibly only minutes), there may

well be insufficient time to effectively and consistently implement sheltering as a countermeasure.

An other countermeasure that might be considered is evacuation of people from an area immediately around the blast site, where contaminant air concentrations are likely to be particularly high. There would be a beneficial psychological effect in quickly moving away from the area known to be most contaminated. Although some authorities (e.g., New York City, 2008) recommend that people leave the immediate area on foot, traffic problems might still occur, which can slow down the process and extend the period spent outdoors. In this near-zone large particles would often dominate, and sheltering in the nearest intact building would thus be likely to effectively reduce the exposure. To evaluate the actual pros and cons of evacuating or sheltering people in an area of a given size, simulations for different types of blast scenarios with a decision support system like the new ARGOS could be valuable.

US authorities also recommend that potentially exposed persons shower and change clothes once they have moved away from the blast site. This is certainly a good idea, although particles larger than about five microns would anyway through natural processes be cleared off the human body over just minutes or hours. The effects of both natural and forced clearance processes greatly depend on the contaminant particle size. There may also be a public demand for personal monitoring, which could be an exceedingly demanding and time-consuming task, as demonstrated in connection with the Goiânia accident, where a total of 125,000 people turned up over 6 months at the local football stadium, demanding contamination checks (Steinhäusler, 2005). Also here, a decision support system like ARGOS could give valuable information on which areas would have become most affected by the contaminated plume (for instance not easily measured if the contaminant is a pure alpha or beta emitter), thus narrowing down the actual need for monitoring, although also the reassurance effect of monitoring persons that are not very likely to have become contaminated should not be underestimated.

As mentioned in Section 2.3.2, resuspension of contaminants would be particularly high in the early phase of an emergency, and for instance if the contaminant is a pure alpha emitter, the resulting inhalation dose rate component would be likely to be dominant once the initial contaminating plume has passed. It can therefore be useful to suppress resuspension using a fixing agent, which might be water, paint, bitumen or a solution of lignin (Brown et al., 2007), depending on, e.g., availability, size of the area and the period over which fixation is desired. Also here, the dose-reductive effect can be estimated for a specific scenario using the new ARGOS system.

A large number of clean-up measures might be considered for reduction of long-term contamination problems in the environment (Brown et al., 2007; Andersson et al., 2003). However, a problem in connection with previously published countermeasure databases is that the countermeasure effectiveness was generally evaluated on the basis of investigations involving small readily soluble radiocaesium particles originating from the Chernobyl accident. Some types of ‘dirty bombs’ would generate much larger and much less readily soluble airborne particles, which would as mentioned in Section 2.3.4 be much easier to remove from impermeable surfaces, but could stay in the very top layer of permeable surfaces over longer periods. It is therefore important to consider the contaminant particle characteristics in evaluating countermeasure effectiveness. A complex decision support system, where the effect of treating each type of surface can be modelled, is essential in justifying and optimising clean-up strategies.

It should be noted that in the event of a malicious action, which is by definition aimed at provoking severe psychological reactions in the public, a very high degree of anxiety and disruption would be expected (ICRP, 2005), and authorities would be likely to be pressed to go very far to remove even small traces of any substances that were placed in the environment to intentionally cause harm. This could mean that effective countermeasure variants which were deemed too expensive and thus not sufficiently cost-effective in connection with the Chernobyl accident might possibly become attractive means of restoring much more limited areas contaminated by a malicious dispersion event. For instance in connection with the Goiânia accident, where anxiety was also very high, practically all efforts possible were made on public demand to clean the affected areas as thoroughly as at all possible, in spite of the poor economical situation of the region in general. In fact, the concept of acceptable levels of contamination was in this case altogether rejected (da Silva et al., 1991), illustrating that methods that have actually been proven sufficiently effective in eliminating radiation hazards may still not have the potential to solve the societal problems satisfactorily. Although exceedingly difficult to quantify reliably in a decision support system, the psychological and social repercussions of a 'dirty bomb' attack are factors that somehow need to be carefully considered in consequence analyses and derivation of strategies for mitigation. Good communication strategies could here be exceptionally important, and for instance the setting up of information/advice bureaus has been suggested as a 'social' countermeasure for radioactively contaminated areas (STRATEGY, 2008).

Conclusions

A conceptual methodology has been described for estimation of the various internal and external dose contributions that may be received by inhabitants of a city area due to atmospheric dispersion of radioactive matter in the event of a 'dirty bomb' explosion. The derived set of formulae generally implies knowledge of time-integrated contaminant air concentrations, but also methods for estimating such concentrations are discussed, and data requirements pinpointed. It is clear that dispersed contaminants from 'dirty bombs' may affect a population in many different ways, and factors like the construction of the dispersion device, characteristics and amounts of contaminants, meteorological conditions and environmental features determine the overall severity as well as the relative importance of the various possible exposure pathways. Highly detailed and complex models are necessary to estimate the likely radiological consequences that different conceivable types of 'dirty bomb' explosion scenarios may have, so that an appropriate preparedness can be established well in advance of an attack. To ensure that resources are applied optimally in mitigating the consequences of an actual attack, this type of decision support tools are also needed to provide trustworthy prognoses of the radiological consequences of the attack, both in 'real time' during the progression of the contamination and over later phases, where countermeasures may be considered to reduce adverse long-term effects. The new ARGOS system addresses these needs, and will follow the concepts described in this paper. The establishment of the required data libraries is in progress. Naturally, disclosure of potentially security sensitive parameters in the ARGOS system is subject to strict clearance procedures.

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