ESTIMATION OF HEALTH HAZARDS RESULTING FROM A RADIOLOGICAL TERRORIST ATTACK IN A CITY

K.G. Andersson[1*], T. Mikkelsen[1], P. Astrup[1], S. Thykier-Nielsen[1], L.H. Jacobsen[2], L. Schou-Jensen[2], S.C. Hoe[3] & S.P. Nielsen[1]

[1] Risø National Laboratory - DTU, P.O. Box 49, DK-4000, Roskilde, Denmark
[2] Prolog Development Center, HJ Holst Vej 3C-5C, DK-2605 Brøndby, Denmark
[3] Danish Emergency Management Agency, Datavej 16, DK-3460 Birkerød, Denmark
* kasper.andersson@risoe.dk

Abstract

In recent years, the concern for protection of urban populations against terror attacks involving radiological, biological or chemical substances has attracted increasing attention. This sets new demands to decision support and consequence assessment tools, where the focus has traditionally been on accidental exposure. This paper is aimed at illustrating issues that need to be considered in evaluating the radiological consequences of a 'dirty bomb' explosion. This is done through a worked example of simplified calculations of relative dose contributions for a specific 'dirty bomb' scenario leading to atmospheric dispersion of ^{90}Sr contamination over a city area. Also the requirements of atmospheric dispersion models for such scenarios are discussed.

Introduction

In the event of a terrorist attack leading to airborne dispersion over a large city area of chemical, biological or radiological (CBR) contaminants, it is essential to decision makers, planners and operational units on different levels to effectively and rapidly obtain an overview of the situation, ensuring that resources are applied optimally to reduce hazards. It is equally valuable in advance of any contaminating incident to know what the hazards might be, so that a targeted operational preparedness can be developed and undue anxiety in the population avoided. This requires the use of reliable and detailed consequence assessment models. Plausible CBR scenarios often have a number of parametric similarities, e.g., with respect to particle sizes and environmental migration. For instance, bacteria typically have a size ranging from a few hundred nanometres to a few tens of microns, which is practically identical to the size range of particles that might be dispersed over a large area after a well executed 'dirty bomb' explosion, and also nebulisation of harmful chemicals would be expected to generate particles of about those sizes (Harper et al., 2007; Byrne, 1998). To illustrate the complexity of required considerations and parameterisation in modelling, a simplified example is in the following given of estimation of the relative contributions to radiation dose that might be received outside the immediate blast area of a 'dirty bomb' dispersing a strong ^{90}Sr source in a ceramic matrix, possibly taken from a radioisotope thermoelectric generator. Also the needs for detailed atmospheric dispersion models to adequately describe contaminant plume concentrations are discussed in relation to a specific dispersion case.

Methods and results

Scenario description
In the past, European standard decision support tools such as RODOS and ARGOS have practically exclusively focused on prediction of the radiological implications of accidents at nuclear power plants. Concerning contaminants that might be released to the atmosphere, the attention is here in general on a limited number of radionuclides known to be present in a given reactor with well-defined initial physicochemical forms, even though actual release fractions and characteristics of dispersed contaminants could vary considerably depending on the type and extent of the accident. In contrast, in the event of a 'dirty bomb' explosion, a very large number of different radionuclides could in principle be involved. However, in reality, the possibilities for using a source that would have a significant harmful effect if dispersed over a wide city area would probably for various reasons be very limited.

A first requirement is the availability of applicable sources. Acknowledging that the likelihood of terrorists gaining access to facilities for production of strong radioactive sources would in general be small, existing sources would need to be applied. Such sources might possibly be stolen from an authorised user or manufacturer, or bought, either by successfully pretending to be a legitimate user with a peaceful industrial purpose (Anon, 2007), or in a black market that that might exist, considering the number of strong sources that are reported lost each year over the world. For instance, in the former Soviet Union it is alone estimated that thousands of comparatively high risk category sources are currently 'orphaned' or lost without traces (Ferguson et al., 2003), although very likely only few of these are strong enough to potentially cause real harm through atmospheric dispersion over an inhabited area. Table 1 shows a series of examples of existing strong sources (Harper et al., 2007; Ferguson et al., 2003; Argonne, 2005).

In any case, the original application of the source would to a great extent determine its characteristics, including the strength and initial physicochemical form. The physicochemical forms, as well as the elemental properties of the radioactive matter, are evidently important in determining the dispersibility of contaminants from a 'dirty bomb'. According to recent experimentation by Harper et al. (2007), for instance extremely little of a metallic ^{60}Co source would be aerosolised in an explosion. However, if a source is in ceramic form, aerosolisation fractions ranging between 2 and 40 % have been reported, e.g., depending on the construction of the explosive device. This was in general found to produce a particle size spectrum with much of the contaminant mass in the 30-100 µm range, and a smaller peak in the range of a few microns. This type of size distribution was also measured in US blast experiments impacting on soil (Pinnick et al., 1983), and after the Thule accident in 1968. The Thule explosion was essentially similar to that of a 'dirty bomb': a conventional explosion dispersing a solid, quite possibly ceramic, radioactive material with a very high melting point. In the Thule case, only 1.3 % of the particles were larger than ca. 18 µm, but these carried nearly 80 % of the activity (Eriksson, 2002).

Table 1. Some important radionuclides that might be of concern in connection with 'dirty bombs', including typical physicochemical forms of large existing sources and max. source strength estimates.

Radionuclide	Typical physicochemical form of large existing sources	Existing strong sources and their strengths
^{60}Co	Metal	Sterilisation irradiator (up to 400,000 TBq). Teletherapy source (up to 1000 TBq)
^{90}Sr	Ceramic (SrTiO$_3$) - insoluble, brittle, soft (Mohs hardness: 5.5)	Radioisotope thermoelectric generator (1000-10.000 TBq)
^{137}Cs	Salt (CsCl), soluble	Sterilisation irradiator (up to 400,000 TBq). Teletherapy source (up to 1000 TBq)
^{192}Ir	Metal – soft - Mohs hardness 6.5	Industrial radiography source (up to 50 TBq)
^{226}Ra	Salt (RaSO$_4$) very low solubility	Old therapy source (up to 5 TBq)
^{238}Pu	Ceramic (PuO$_2$) - insoluble	Radioisotope thermoelectric generator (up to 5,000 TBq)
^{241}Am	Pressed ceramic powder (AmO$_2$)	Well logging source (up to 1 TBq).
^{252}Cf	Ceramic (Cf$_2$O$_3$) - insoluble	Well logging source (up to 0.1 TBq).

In the context of atmospheric dispersion over larger areas, it is the smaller particles that are of concern, since gravitational settling of large particles will occur very rapidly and over short distance. As an example, Hage (1961) performed a series of experiments dispersing glass microspheres with a mass mean diameter of ca. 50-100 μm from 15 m above a flat prairie area. These experiments were conducted under carefully monitored conditions and showed that with an average wind speed at the point of release of about 5 m s^{-1}, practically all the contaminants (>99 %) would deposit within a downwind distance of less than 300 m. This means that the air concentration of these large particles would decline greatly over the first minute. In contrast, small particles, on which gravity would have considerably less influence, would remain airborne over much longer periods of time and could reach correspondingly greater distances from the release point.

An example of strong and carelessly 'orphaned' sources is the ^{90}Sr elements (typically in the range of 1000 - 10,000 TBq), originally applied in radioisotope thermoelectric generators, which have in recent years on several occasions unwittingly been retrieved by village inhabitants in the former Soviet Union (Andersson, 2005). For comparison, it has been estimated that the total release of ^{90}Sr from the Chernobyl accident was about 8000 TBq (Sohier, 2002). It should however be mentioned that such strong sources might be exceedingly difficult to handle and bring to dispersion, even if involved terrorists may not consider it important to survive themselves. Even though

the beta particles from ^{90}Sr/^{90}Y might require less shielding than the gamma radiation from other strong sources, the Bremsstrahlung problem could also be substantial.

For the purpose of defining an illustrative scenario for a demonstration example of estimation of dose contributions, we will here assume that the contaminants originate from a strong ^{90}Sr source in a ceramic matrix. The contamination within the immediate vicinity of the blast would to a great extent be dominated by large shrapnel. Therefore the contamination pattern within that small area, which would anyway be very extensively monitored, would be impossible to model. However, as discussed above, a proportion of the radioactive matter would become aerosolised as fairly small particles, which might contaminate a large and complex city area. Monitoring and mapping of beta activity over a large city area would be an extremely demanding and time-consuming task, and reliable case-specific modelling would thus be very valuable, not only for pre-incident planning, but also for real-time impact evaluations, which is addressed in new developments for the ARGOS system. We will therefore here focus on doses received from the atmospheric dispersion of contaminant aerosols over a large city area. Based on the experience described above with explosions involving ceramic material, we will assume that it would be the peak in the lower end of the aerosol size spectrum that would be important to follow over this area. Since the largest of the particles in that peak would carry the greatest contaminant mass, we will assume that the particles of concern are in the 5-10 μm range. Such particles would in general be inhalable (ICRP, 1993). We will further assume that the weather is dry on the day of the attack. Internal doses from inhalation of resuspended contaminated dust as well as external doses from the passing contaminated plume would be likely to be of comparatively little significance (Andersson et al., 2004; Wagenpfeil et al., 1999; Andersson & Roed, 2006; Jones et al., 2008) and are not dealt with in this paper.

External dose from contamination on outdoor surfaces
The dose rate received at the depth of the basal layer of the epidermis of human skin from a large ground surface uniformly contaminated with ^{90}Sr has, probably highly conservatively, been estimated by HPA-RPD (2005) for decision-making purposes to be some $4 \cdot 10^{-11}$ Sv h^{-1} per Bq m^{-2}. This estimate is applied for the calculations in this paper, although the exact assumptions regarding geometries are not quite clear, and more detailed calculations for different distances and thicknesses of clothing are called for. A requirement to reach as high dose rates to the skin as this would be that the contamination lies on the very surface of the ground. If it were 1 cm down in soil, as would be expected rather shortly after an airborne contamination, the shielding effect would be so great that the dose rate to the skin would be about 3 orders of magnitude lower (Eckerman and Ryman, 1993). The dose rate contribution would thus probably be reduced to a negligible value at the time when the first heavy rain shower arises, and the contamination is washed slightly down in the soil (Roed, 1990). For the purpose of this example, we will assume that this happens 10 days after the deposition. Doses to inner organs would typically be at least 3 orders of magnitude lower, and thus be likely to be of little significance (Eckerman and Ryman, 1993). Here, Bremsstrahlung would contribute. Based on a number of independent surveys in Western Europe and in California (Jenkins et al., 1992; Andersson, 1996; Long et al., 2001; Kousa et al., 2002), it seems reasonable to assume that people on average spend some 15 % of the time outdoors. This would give a skin dose contribution estimate of 15 % · $4 \cdot 10^{-11}$ Sv h^{-1} per Bq m^{-2} · 10 days · 24 h day^{-1} ~ 1·10^{-}

[9] Sv per Bq m^{-2}. Contaminant particles will naturally also deposit on other outdoor surfaces. As gravitational settling will largely govern the process of deposition for these supermicroneous particles, horizontal surfaces will receive much higher levels of contamination than vertical. Also the limited time normally spent outdoors very close to a wall would make the contamination on these surfaces less important from a beta dose perspective. On paved horizontal surfaces, large and insoluble particles will much more rapidly be removed by natural weathering processes than will smaller particles (Sartor et al. 1974; Roed & Andersson, 1996). This is at the extreme illustrated by the very high decontamination factor of 50 obtained by Clark and Cobbin (1964) when hosing water on a street contaminated by insoluble 44-100 µm particles. If the contamination had instead been in the form of submicroneous soluble Cs particles, as seen after the Chernobyl accident, hosing at the same pressure would be expected to only remove about half of the contamination (Brown et al., 2008). Routine street cleaning together with natural weathering processes would therefore be likely to rather rapidly result in low street contamination levels. For simplicity, only dose contributions from contamination on the soil are considered here, but an assessment of the post-deposition behaviour of various types of supermicroneous particles on other surfaces in the urban environment is called for to provide an adequate background for decision-making.

External dose from contamination on indoor surfaces
It is here assumed that windows and doors are closed during the contaminating episode. For supermicroneous particles it is not unreasonable to assume that nearly all the deposited indoor contaminants will be distributed on the horizontal surfaces, primarily the floor (Lange, 1995). Based on the findings of Roed & Cannell (1987), it can be shown that the deposition, D_i, per unit indoor area can be estimated from the formula: $D_i \sim (D_0/ v_{d,0})$ h f $\lambda_d \lambda_v / (\lambda_d + \lambda_v)$, where D_0 is the corresponding deposition on an outdoor reference surface (e.g., a cut lawn), $v_{d,0}$ is the deposition velocity to that reference surface, h is the room height, f is the filtering factor, λ_d is the rate coefficient of deposition indoors, and λ_v is the rate coefficient of ventilation of the dwelling. For particles in the relevant size range, λ_d would be high compared with λ_v, so that $D_i / D_0 \sim$ h f $\lambda_v / v_{d,0}$. With typical values for ca. 5-10 µm particles used in the new version of the ARGOS system this here gives: $D_i / D_0 \sim 0.09$. A reasonable estimate of the dose rate can be obtained using the same dose conversion factor as for outdoor horizontal surfaces. In reality, indoor floor surfaces will be likely to be considerably smaller than open outdoor horizontal surfaces, but beta doses come from contamination within short distance. A likely half-life of the natural removal process of the smallest relevant particles on the floor would be of the order of a few months (Andersson et al., 2004). We will here assume a value of 60 days. The total external dose to the skin from the contaminated 5-10 µm particles deposited on indoor surfaces (taking into account the fractions of time spent indoors and outdoors) would then be estimated to be of the order of (60 d/10 d) · 0.09 · 0.85/0.15 = 3 times that from contaminants deposited on outdoor surfaces, or in other words, $3 \cdot 10^{-9}$ Sv per Bq m^{-2} on the horizontal outdoor reference surface. It should be noted that as even thin fabric shields well against beta radiation, the most critical situations would be those where unshielded skin comes into direct or close contact with a contaminated surface. One important situation is at night, when, e.g., the face is in direct contact with a possibly contaminated surface for hours. For this scenario, the applied dose conversion factor might well be much too low. However, by washing the pillow case regularly, these doses would be limited to a short period of time after the contamination took place, as

ordinary machine washing is efficient in removing the contaminants (Andersson et al., 2002).

External dose from contamination on human skin
The most critical exposure of human skin would occur to persons who are outdoors during virtually the entire period of deposition, and this is what is assumed. If people were indoors with windows and doors closed, the deposition on humans of 5-10 μm particles would typically be at least one order of magnitude less (Andersson et al., 2004). Deposition velocities of 5-10 μm particles to human skin, as parameterised in ARGOS, are high, but thorough washing will be rather efficient in removing these particles from skin. However, if nothing is done to actively remove the particles (e.g., when no early warning is given), the natural removal of 5-10 μm particles would be expected to occur with a half-life of ca. 0.2 day (Hession et al., 2005). Using this clearance half-life together with a data library value for absorbed dose rate at a depth in tissue corresponding to that of the basal layer of the epidermis, the dose from contamination with these ^{90}Sr particles on freely exposed skin would here amount to 8 $\cdot 10^{-6}$ Sv per Bq cm^{-2}. This, multiplied by the relationship between deposition velocities on skin and the grassed reference surface, here gives a skin dose estimate in ARGOS of $4 \cdot 10^{-9}$ Sv per Bq m^{-2}. Corresponding skin doses from contamination on clothing would be considerably smaller, both because clothing can, dependent on thickness, shield very well against the beta radiation, and because the typical deposition velocity to clothing of aerosols of this size is about a factor of 5 less than that to skin (Fogh et al., 1999). Although the natural clearance half-life of these contaminants from clothing would typically be somewhat longer than that from skin, the dose contribution from the ^{90}Sr particles deposited on clothing would be expected to be at least one order of magnitude less than that from deposition to uncovered skin (Andersson et al., 2004).

Committed dose from inhalation during plume passage
We will also here look at the dose contribution to the critical group: people staying outdoors during the plume passage. Inhalation doses can be estimated by multiplying *time-integrated* contaminant air concentrations by a dose conversion factor (ICRP, 1995) and an inhalation rate. The particles in the 5-10 μm range are all assumed to be inhalable (ICRP, 1993). The time-integrated air concentration outdoors, C_o^{int}, that would lead to a deposition of 1 Bq m^{-2} on an outdoor grassed reference surface can be found by dividing the 1 Bq m^{-2} by the deposition velocity to the reference surface. By multiplication of this time-integrated air concentration with an inhalation rate (3.3 $\cdot 10^{-4}$ m^{3}s^{-1}; ICRP, 1993) the total inhaled amount of contamination can be found. According to ICRP (1995), the committed dose by inhalation of 1 Bq of ^{90}Sr would be 1.6 10^{-7} Sv if the 'slow' absorption class 'S' is assumed. By multiplying the inhaled amount by this factor, it is found that inhalation in this case gives a committed dose of $3 \cdot 10^{-9}$ Sv per Bq m^{-2} on the reference surface. It should be stressed that this dose would be much smaller if people stayed indoors during the time period of elevated air concentrations, and if the aerosol had been more readily soluble. Also, if it rained during the plume passage, this could significantly deplete plume concentrations, while instead enhancing deposition on outdoor surfaces, as observed after the Chernobyl accident (Roed, 1990).

Atmospheric dispersion modelling

In the sections above we have looked at an example demonstrating some considerations required for estimating dose contributions from a 'dirty bomb' explosion, relative to a given contamination level on a reference surface in the environment. This type of considerations are used to define parametric requirements for the extension of ARGOS to estimate consequences of 'dirty bomb' explosions, and the creation of libraries of relevant parameter values has been initiated. Also, a new, highly detailed model for urban atmospheric dispersion and deposition of the different types of particles from 'dirty bombs' is being created for implementation in ARGOS. For this dispersion model, a methodology has been derived, primarily on the basis of experimentation, to determine the likely initial contaminant cloud dimensions from a 'dirty bomb' as a function of the applied explosive mass, and new data libraries have been generated describing the deposition of particles of the different relevant sizes on all different urban surfaces for five different weather categories at deposition: light rain, heavy rain, dry weather, dry weather with snow cover, and snowfall (Andersson, 2008). Other than accommodating a wider range of particle sizes, the reason for the development of the new dispersion model URD (Urban Release and Dispersion), which in is many ways inspired by the British code UDM (Urban Dispersion Model; Hall et al., 2002), is that it has become apparent that it would be problematic to use the mesoscale dispersion modules in decision support systems like RODOS and ARGOS for estimation of the contaminant dispersion after a comparatively low-altitude in-town release, which would be expected from for instance a 'dirty bomb' blast. In larger scale atmospheric dispersion models, such as ATSTEP (Päsler-Sauer, 1997) and RIMPUFF (Mikkelsen et al., 1984), inhabited areas are simply modelled as areas with enhanced surface roughness and different deposition rates compared with open areas. This is adequate for higher altitude longer-range dispersion, e.g., following a large nuclear power plant accident. However, over areas near the point of a comparatively low-altitude release, the plume interaction with buildings and other obstacles and dispersion through street canyons can be important factors in determining the plume shape and dispersion pattern. This is illustrated by the example shown in Figures 1, 2, and 3. All of these figures show calculations of time integrated concentration at ground level in an urban area (Frederiksberg, part of Copenhagen, Denmark) of aerosols released at a height of 2 m above ground. The amount of dry deposition of aerosols is proportional to the time-integrated concentration, the proportionality factor being a function of the land cover, while for larger particles gravity becomes dominant. In URD the presence of buildings has three effects: it limits the size of the horizontal eddies present in the atmosphere and thereby limits the large scale horizontal dispersion, it enhances the smaller scale dispersion by creating more small scale turbulence, and it delays part of the dispersion by retaining aerosols in more or less stationary recirculation zones on the downstream side of the buildings. Figure 1 shows an estimate made with the RIMPUFF mesoscale model. Figure 2 shows a corresponding estimate with URD, where buildings are only modelled to limit the large scale dispersion, whereas the other two building effects are not accounted for. In Figure 3 all building effects are included, and it is seen that especially the retaining of matter behind the buildings is of importance. Figure 4 is an excerpt from Google Earth of the area in which the calculation area is located.
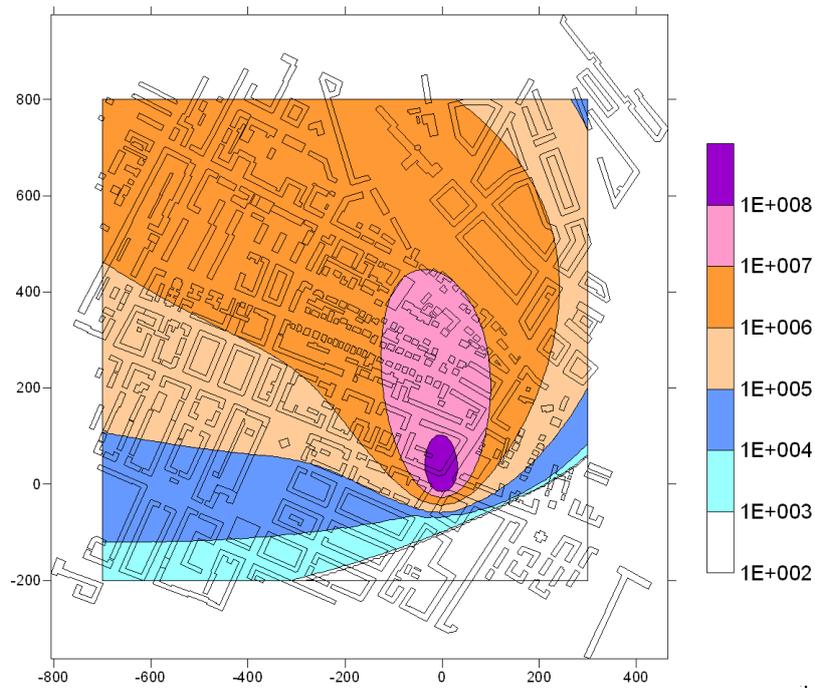
Fig. 1. RIMPUFF estimate of the time integrated concentration [Bq s/m³] at ground level. No building influence. Frederiksberg case.
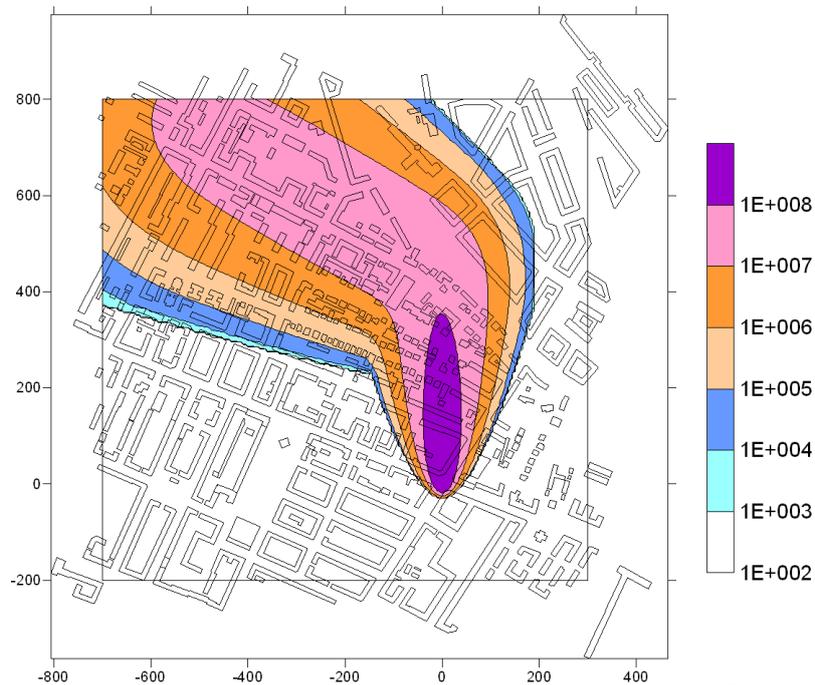

Fig. 2. URD estimate of the time integrated concentration [Bq s/m³] at ground level, with the buildings just limiting the large scale dispersion. Frederiksberg case.
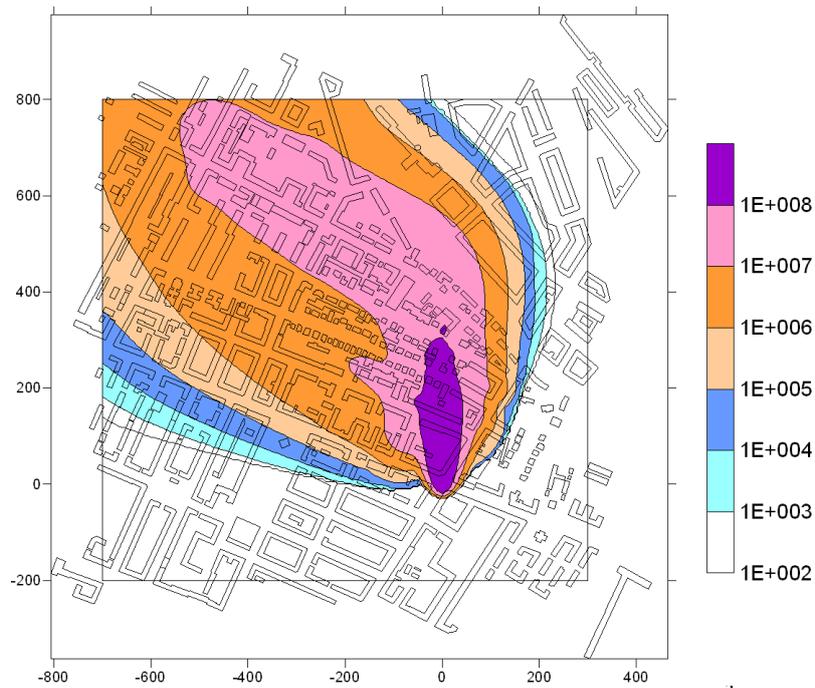
Fig. 3. URD estimate of the time integrated concentration [Bq s/m^3] at ground level, with all three building effects modelled. Frederiksberg case.
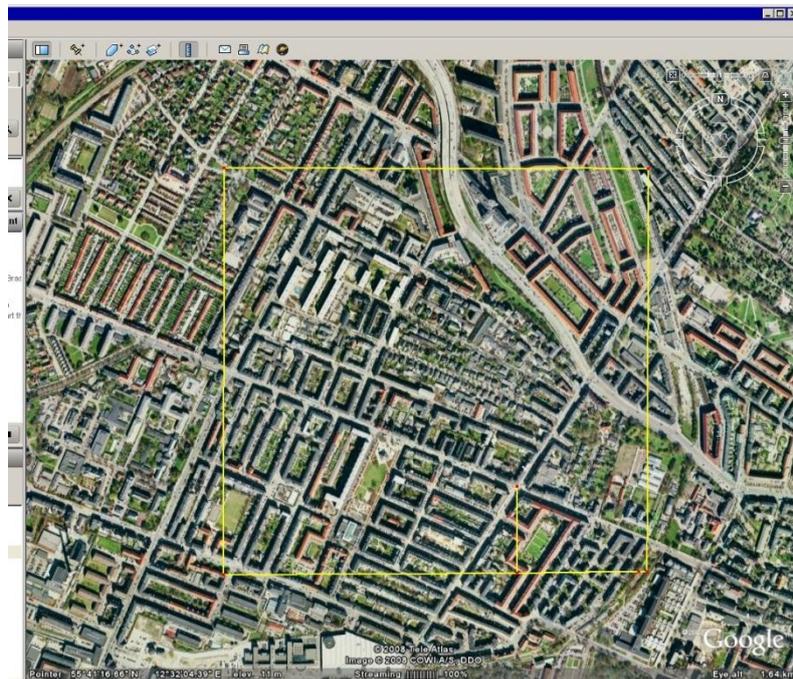

Fig. 4. Google Earth picture of part of Frederiksberg, the calculation area indicated by the yellow quadrant.

Discussion and conclusions

The above simplified calculations might give the impression that skin beta doses from contamination on outdoor surfaces, skin beta doses from contamination on indoor surfaces, skin beta doses from contamination on human skin and committed doses

from inhalation during the passage of the plume from a 'dirty bomb' explosion are of the same order of magnitude. However, it should of course be noted that the skin is not the most radiation sensitive of human organs, and the threshold for acute radiation effects by exposure of large areas of skin is generally assumed to be some 20 Sv, even though acute tissue breakdown has been reported for a dose as low as 15 Sv (ICRP, 1991). The most important lesson from the above considerations is however that the proper estimation of each dose component depends on the values of a number of parameters, which would vary considerably between different conceivable 'dirty bomb' explosion scenarios. The values calculated for the specific demonstration scenario reflect a wide range of case-specific assumptions, and a completely different distribution of the dose contributions would be envisaged for other 'dirty bomb' scenarios. To prevent against the results of this calculation being quoted out of their context, they are purposely not summarised in a table.

Importantly, however, the scenario evaluations and considerations demonstrate the need for a number of datasets to properly estimate the different dose contributions that would be received after the detonation of a 'dirty bomb' in a city area. For instance, as mentioned above, a data library of parameters describing the deposition to the different surfaces in an inhabited area of particles of different relevant sizes, for different weather conditions, has already been created for use in a new version of the ARGOS system for evaluating the dose implications of 'dirty bomb' attacks. Also a compilation of factors determining the post-depositional migration in the inhabited environment of contaminants with different relevant physicochemical characteristics (notably particle sizes) is needed. Further, there is a need for a series of dose conversion factors for inhabited environments, for instance for beta doses from ^{90}Sr on different surfaces in different geometries, but also for gamma doses from radionuclides that are not traditionally envisaged in connection with nuclear power plant accidents, but would be considered of relevance in connection with a terror attack. It is also important to note that the dose distribution following a 'dirty bomb' attack would depend on factors such as physicochemical forms of dispersed materials, radionuclides, weather, season, human behaviour, area type, and protection offered by dwellings and ordinary clothing, which can all be ideally addressed in a computerised decision support system.

Without the use of a detailed and complex assessment tool, it is exceedingly difficult to judge the significance of the direct health implications of a given 'dirty bomb' attack scenario. In any case, it is essential for decision makers and authorities to be equipped with as good a background material as possible for understanding the implications of different types of 'dirty bomb' scenarios. This would help in issuing early and reliable information that could prevent undue anxiety and social disruption in an emergency situation. As mentioned above, also real-time prognoses of the contamination of a complex city area could be very useful for decision-makers, planners and operational units in the event of an attack. This is targeted in an extended version of the ARGOS decision support system. The implementation in ARGOS of a series of state-of-the-art datasets based on a combination of experimentation and theory, together with the described high resolution dispersion features eliminating errors arising from the interaction of a contaminated low-altitude plume with structures in the environment, will be important instruments in securing reliable consequence prognoses. Disclosure of potentially security sensitive

parameters in ARGOS, such as plume dimensions and deposition relations on surfaces, will be subject to strict clearance procedures.

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