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Consequences of a Radiological Dispersal Event with Nuclear and Radioactive Sources

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A potential terrorist attack utilizing a Radiological Dispersal Device would spread fear and panic on a massive scale, in addition to creating a widespread, severe, and long-lasting economic burden. The main purpose of the present study is to make some assessments of the radiological impact of such an incident, with a primary focus on the public health risk. We discuss the possibility of utilizing radioactive sources or nuclear fuel in such an incident and make a comparison of the relative impact associated with each source.

INTRODUCTION

Radiological Dispersal Devices (RDDs), which are commonly referred to as "dirty bombs," operate on the principle that conventional explosives are employed in order to disperse radioactive material. For the purpose of evaluating the consequences of an incident involving detonation of an RDD or another type of Radiological Dispersal Event (RDE), one must be able to account for the amount and the type of radioactive material present (α, β, γ radiation or neutrons), for its potential impact on exposed members of the public, and for the

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manner in which it will be dispersed under given meteorological conditions. In all possible scenarios the damage to the biosphere is primarily due to radioactive contamination, whereas the health effects that could potentially be inflicted on a section of the population exposed to radiation depend critically on the type of radioactivity and on the irradiation pathway (internal vs. external exposure). Of concern for the latter pathway is γ radiation, whereas α and β radiation are only primarily a concern in the case of internal irradiation following ingestion or inhalation.

A number of studies have already addressed the problem of potential RDDs that could be constructed from isotopes that are in common usage for medical and industrial purposes[1–5], such as ^{60}Co, ^{90}Sr, ^{137}Cs, ^{192}Ir, ^{238}Pu, ^{241}Am, or ^{252}Cf. Both the nature and the intensity of the radioactive material contained in such sources vary considerably. For example, smoke and moisture detectors contain ^{241}Am with an activity of about 10^4 to 10^8 becquerel (Bq),[3] respectively, whereas the activity of a typical ^{60}Co source in a clinical irradiation facility for cancer treatment is almost one million times higher ($7{\cdot}10^{13}$ Bq).[3]

In comparison, the total activity associated with 2 kg of spent nuclear fuel after discharge from a nuclear reactor with a burn-up of 50 GWd/tHM is of the order of 10^{16} Bq. However, a simple comparison of the activity associated with these sources does not present a full picture, as the radiation emitted in the decay of these isotopes differs in both type and energy. In order that this difference can be fully taken into account and the impact on human health can be fully assessed, it is important to deal with the radiotoxicity associated with a particular isotope.

The main aims of the present study are to present an overview of RDE phenomena, to develop a clear, physically intuitive approach for dealing with such phenomena, and to apply this approach in order to obtain a series of estimates of the consequences that are likely to arise as a result of an incident involving dispersion of radioactive sources, spent nuclear fuel, and fresh Mixed-OXide (MOX) fuel. The probability, and therefore the risk, of a scenario involving nuclear fuel occurring is clearly much smaller than in scenarios involving the interception and utilization of sources containing the aforementioned medical and industrial isotopes, because physical protection of nuclear fuel is one of the industry's top priorities, and also because it is only transported in shielded equipment weighing dozens of tons. On the other hand, if one considers that the definition of the risk associated with an event is given by the product of the probability that the event will occur multiplied by its consequences, then the threat of an RDE involving nuclear fuel cannot be neglected, due mainly to the high activity content of various radioisotopes.

It is worth emphasizing at this stage that dispersion of radioactive material in the fine aerosol form required to induce the most serious adverse health effects through the inhalation pathway is far from trivial. The mass of the radioactive material and the matrix in which it is contained have a huge

impact on the percentage of the original material that will be transformed into fine aerosol form through the use of explosives. The physical dimensions and composition of a nuclear fuel rod render it a highly unsuitable target for an RDD, with a very small probability of dispersing even a small fraction of the radioactive material as an aerosol. However, studies involving potential sabotage of spent fuel casks during transport[6] or sabotage of spent fuel pools at reactor sites[7] have shown that it is possible to produce relatively large fractions of radioactive material in a respirable aerosol form.

In order to estimate the consequences of an RDE, one needs to quantify the type of radioactive material (the source term) and to model the dispersion under the meteorological conditions at the time of the detonation. The following section will develop the source terms corresponding to an attack with spent nuclear fuel and fresh MOX fuel and compare them to an RDD based on a widely available γ-emitter, ^{60}Co, which is taken to be representative of medical/industrial isotopes. The article will then describe the consequences of an RDE based on approximate calculations involving a simple model to account for dynamic dispersion of the resultant radioactive plume (the wedge model). As an example, these calculations are performed for ^{60}Co and for the most important radionuclides present in both spent UO_2 and fresh MOX fuel. Generalization to any other radionuclide is straightforward.

SOURCE TERM EVALUATION OF A RADIOLOGICAL DISPERSAL DEVICE

Commercial Radioactive Sources: ^{60}Co Example

Radioactive sources are used for various medical and industrial applications, such as cancer treatment, medical diagnostics, food sterilization, smoke detectors, and so on. Such sources, if not properly secured, could be employed for the construction of RDDs.[3,8] For this reason, the IAEA has recently developed guidelines for securing such radioactive sources.[8] Commercial sources of particular concern are those containing significant amounts of radioactivity, typically more than a few curies, which roughly corresponds to gram quantities. These include sources such as ^{241}Am, ^{252}Cf, ^{137}Cs, ^{60}Co, ^{192}Ir, ^{238}Pu, and ^{90}Sr. Some of these isotopes—^{241}Am, ^{252}Cf and ^{238}Pu, in particular—are highly radiotoxic following ingestion or inhalation, whereas the others present additional health hazards due to external exposure to highly penetrating γ-radiation. For example, ^{60}Co is the most common radioisotope used in radiotherapy applications worldwide, followed by ^{137}Cs, which is more prevalent in developing countries in less advanced treatment facilities. A capsule of ^{60}Co used for cancer treatment typically contains about $7.4{\cdot}10^{13}$ Bq (2000 curie), which corresponds to a mass of about 1.8 grams.

Nuclear Fuel

The most common type of nuclear reactor worldwide is the Pressurized Water Reactor (PWR). Typically, fresh UO_2 fuel with an enrichment of around 4.2% in ^{235}U is irradiated in a PWR to a burn-up of about 50 GWd/tHM. After discharge, spent fuel is usually cooled for six years to allow for the decay of the short-lived fission products, thus rendering the spent fuel less radioactive and therefore easier to handle. The total mass of fresh fuel in a single rod is in the range 1.5–2 kg. The following calculations consider only solid and unprocessed fuel rods. The source term for a spent fuel rod with the parameters given earlier has been calculated with the KORIGEN depletion code,[9] which yields the isotopic inventory and the respective activities. Subsequently, decay calculations have been performed in order to determine the activity of each nuclide after six years of cooling, as well as 24 h after discharge for comparison purposes. The total activity results for both these cases are summarized in Table 1. After six years of cooling the activity has decreased by two orders of magnitude. One should note, however, that although around 60% of the fission products will decay after six years cooling, the total amount of highly radiotoxic actinide isotopes does not decrease significantly.

Sabotage of spent fuel transport casks, by missile attack for example, and the corresponding release of radioactive aerosol has been studied in detail.[6] Respirable fractions, that is, aerosol particles with an airborne diameter less than 10 μm, have been measured for actinides, fission products, and fuel cladding materials (using in some cases surrogates such as cerium). The actinide respirable fraction released from spent fuel casks as a result of sabotage has been found to vary between 0.5–1% of initial affected inventory, whereas the volatile fission product fraction can be as high as 30%. For the purposes of the present study one takes a reference value of 2 kg of respirable aerosol for all calculations involving nuclear fuel. This corresponds approximately to the radioactive inventory of a single fuel rod. Scaling to scenarios involving smaller or larger releases is then trivial.

For completeness, we also consider the same attack on fuel casks scenario described earlier but with fresh MOX as opposed to spent UO_2 fuel. There are currently over thirty reactors operating in Europe and Japan with partially loaded MOX cores. MOX is a mix of plutonium dioxide and uranium dioxide powders with a typical plutonium content of around 5–10%. Its use in today's

Table 1: Source term for 2 kg of spent uranium fuel (4.2 percent ^{235}U enrichment, irradiated up to 50 GWd/tHM and followed by a cooling time of 24 h and 6 years.

Source term	Activity (Bq)
Spent fuel after 24 h	$6{\cdot}10^{15}$
Spent fuel cooled for 6 years	$4{\cdot}10^{13}$

reactors has been prompted by growing stockpiles of separated plutonium from reprocessing of spent fuel and the absence of commercial fast reactor plants. Fresh MOX is transported from manufacturing facilities to reactor sites in the same manner as UO_2 fuel and is therefore prone to the same kind of sabotage, as has recently been studied.[10] What makes the fresh MOX case interesting is the much higher plutonium content in the fuel, which is significant because plutonium isotopes have high radiotoxicities. MOX fuel also emits far less neutrons and gamma rays than spent fuel and is therefore much easier to handle. Although the risk of interception of MOX is small, the fact that it is easier to handle makes it a scenario that cannot be entirely neglected. As in the spent fuel case, 2 kg of MOX respirable aerosol is taken as a reference value.

Comparison to Large-Scale Radioactivity Releases

In order to put the activities of the source terms considered previously into some kind of perspective, the article now presents a number of cases where radioactivity has been released to the environment. Such cases include nuclear weapons explosions, reactor accidents, accidents involving nuclear materials, and radioactive sources. Table 2 shows the two source terms described earlier (2 kg of spent fuel and a ^{60}Co capsule) compared to the release of the Chernobyl reactor explosion (1986), the Nagasaki fission bomb (1945), the Vladivostok submarine accident (1985), and the Windscale reactor fire (1957). Of course, both the physical and the isotopic composition of the radioactivity released in these incidents vary dramatically, depending on the type of material present and the mechanism of release (i.e., explosion vs. reactor fire). A fuller description of each of these environmental releases can be found in Appendix A.

The Goiânia Incident—A Benchmark for Radiological Dispersal Events

The Goiânia radiation incident is the most serious event recorded to date involving a medical radiation source.[11–14] Goiânia is the capital of the Brazilian

Table 2: Comparison of 2 kg of spent fuel and ^{60}Co source terms to large-scale atmospheric radioactivity releases.

	Year	Activity (Bq)
Spent fuel (2 kg) after 24 h	—	$6.3 \cdot 10^{15}$
Spent fuel (2 kg) cooled for 6 years	—	$4.4 \cdot 10^{13}$
^{60}Co Source (2000 Ci)	—	$7.4 \cdot 10^{13}$
Chernobyl Reactor	1986	$1.1 \cdot 10^{19}$
Nagasaki Fission Bomb	1945	$7.6 \cdot 10^{19}$
Vladivostok Submarine	1985	$1.9 \cdot 10^{17}$
Windscale Reactor Fire	1957	$7.4 \cdot 10^{14}$

state of Goias in south-central Brazil, with a population of 700,000 (1980). In September 1987, approximately one year after the Chernobyl accident, a radiation source contained in a metal canister was stolen from a radiotherapy machine in an abandoned cancer clinic and sold to a scrap dealer. Some five days later, the dealer opened the metal canister to find a fluorescent powder: radioactive cesium (^{137}Cs) chloride, with an activity of 50 TBq (approx. 1400 Ci). It is believed that the blue glow from the powder, caused by the absorption of gamma rays by chlorine and reemission of visible light, made it appear valuable to the dealer. In the following days, the powder was circulated among family and friends. A six-year-old girl rubbed the powder onto her body and ate a sandwich contaminated with the powder from her hands. In total 244 people were exposed, and 4 died. Approximately 100,000 people were screened for contamination. The incident in Goiânia was the second largest radiological accident after Chernobyl and is regarded as a key benchmark for discussing the potential consequences of radiological dispersal devices. The socioeconomic impact as a result of a sharp decline in tourism revenue resulted in an appreciable drop in the region's gross domestic product, which took more than five years to return to pre-1987 levels.

HEALTH EFFECTS OF RADIATION

The basic working principle for an RDD is based on the assumption that upon detonation the conventional explosives in the device will cause some of the radioactive material present to be converted into a fine aerosol form, which will then be dispersed. Obviously, the precise nature of the resulting aerosol cloud will vary dramatically, depending on the nature of the dispersal event and the radioactive material used. No attempt is made in the present study to model or characterize the initial explosive process.

In order to estimate the serious adverse health effects that will result as a consequence of an RDE, one has only to consider the risk to the exposed population through inhalation of the radioactive aerosol. This is because the dose received from radiation in the deep lung can be very large due to the amount of ionization in the lung tissue, which will result over a very short distance, especially for α-emitters, which have a typical range of just a few cell diameters. Regardless of the type of radioactive material used in the RDD, even for very strong γ-sources like ^{60}Co, serious health risks due to external radiation exposure are almost negligible compared to the risk through the inhalation pathway and are therefore ignored. In addition, radioactive intake through ingestion, which carries a similar high-dose potential as the inhalation pathway, is also neglected, as it is assumed that the relevant sections of the population will have been evacuated after a suitable time period and will not therefore be exposed to any contaminated food or water. When considering the risk due to

inhalation, it should be noted that the authors assume that all the radioactive material in the dispersal event is as an aerosol of respirable size ($<10\ \mu$m).

Consider an individual exposed to this airborne activity through intake by inhalation. First, acute radiation effects are not a particular concern because the levels of radioactivity involved are not normally large enough to induce such effects, even in the immediate vicinity of the detonation point. This means that the primary focus should be on the chronic, stochastic effects caused by such intake. In order to characterize the long-term health risks, the total intake activity must be converted into committed dose by multiplying by the relevant effective dose coefficient.[15] These coefficients have been calculated by the International Commission for Radiological Protection (ICRP) for the most important radionuclides, and take into account the effects of radiation on different organs over a fifty-year period after the initial exposure, as well as effects due to the possible transport of radionuclides in the body. Thus, if the total intake activity by inhalation of a particular radioisotope is given by A_{inh}, then the corresponding committed dose in man.Sv is simply:

$$D_{inh} = A_{inh} e_{inh}(50) \tag{1}$$

where $e_{inh}(50)$ is the ICRP effective dose coefficient over a fifty-year time period following intake. For situations involving inhalation by more than one person, a summation over all the individual inhalation doses must be taken to calculate the total collective dose (CD_{inh}). This collective dose can then be related to the number of excess cancers that are expected to develop among the affected population due to the radiation exposure in question by a risk factor (RF)[15] of 0.05 Sv^{-1}. This procedure for estimating the likelihood of cancer development is predicated on the critical assumption that the radiation effects are directly proportional to the radiation dose without any threshold effects (the Linear Non-Threshold or LNT hypothesis[11]).

DISPERSION OF RADIOACTIVITY AND DOSE ESTIMATES: THE WEDGE MODEL

Originally developed in the mid-1970s by a group studying light-water reactor safety,[16] the wedge model gives a rather simple account of the dynamics governing the dispersal of an aerosol cloud under various meteorological conditions. Fetter and von Hippel employed this model in the context of plutonium dispersal in nuclear-warhead accidents in their 1990 paper.[17] Its utilization with respect to radiological dispersal devices raises the potential for forming a basic understanding of dispersal of a radioactive plume after an RDE. This is because both the temporal and the spatial evolution of the radioactive cloud are fundamental features of the wedge model. Thus, identification of the key physical processes at work in this model makes it possible to make estimates

of the public health risk associated with the source terms described earlier, and to improve on the understanding of the phenomena associated with RDEs. Work is currently underway on an online implementation of the wedge model for scenario building in the European Commission's Nucleonica framework (see www.nucleonica.net).

Model Parameters

The wedge model assumes that the detonation of an RDE will involve a plume of finely dispersed aerosol moving downwind with a given wind velocity u and dispersing in the cross-wind direction with a characteristic opening angle Θ. The aerosol in the plume is assumed to be uniformly distributed in both the cross-wind (lateral) direction and throughout the constant plume height H. Figure 1 gives a schematic representation of the manner in which the plume evolves as the downwind distance r, and therefore the time since detonation ($t = r/u$), increases. Crucially, the amount of aerosol in the cloud at a given time after detonation—i.e., the shaded volume of radial width dr in the figure—will decrease as time increases. The mechanism responsible for this decrease involves the radioactive aerosol falling to the ground with a given deposition velocity (v), and therefore no longer posing an inhalation risk.

The list below gives the wedge model parameters:

- distance downwind of detonation point r
- height of the cloud H

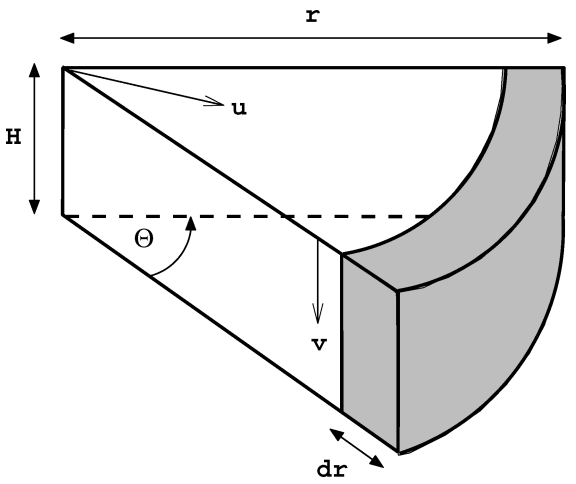

Figure 1: The wedge model involves the dispersal of radioactive aerosol as a function of downwind distance r given an initial wind speed u, opening angle θ and height H. The shaded volume highlights the position of the aerosol at a given time t after detonation.

- wedge opening angle Θ
- wind velocity u
- rate of inhalation for an average individual R_{inh}
- population density at a particular point $\rho(r)$
- deposition velocity v

The wedge model has an intrinsic characteristic timescale, defined by the deposition mechanism. This allows access to detailed information on the airborne activity and the committed dose by inhalation as a function of time and distance. The fundamental parameter is the deposition velocity: the speed at which aerosol particles fall to earth, which, when combined with the height of the cloud, defines an average time, τ, and average range, L, before an aerosol particle falls to the ground:

$$\tau = \frac{H}{v} \quad L = u\tau = \frac{uH}{v} \tag{2}$$

Calculation of Committed Dose

Assuming that the amount of aerosol deposited on the ground is proportional to the amount present in the air, one finds that the activity in the plume at a distance r is:

$$A(r) = k_a A_0 \exp\left(-\frac{r}{L}\right) \tag{3}$$

Consider the shaded plume volume in Figure 1 at a distance r from the detonation point, which has a radial spread dr and a volume $rH\Theta dr$. A factor k_a has been included in the equation in order to account for the difference between the geometries associated with the wedge model and the more complex Lagrangian model used to benchmark the wedge parameters (see later text and Appendix B). The introduction of this correction leads to a reduced value (41%) for the airborne activity. The radioactive aerosol will have an activity per unit volume of:

$$A_{vol}(r) = \frac{k_a A_0}{rH\Theta dr} \exp\left(-\frac{r}{L}\right) \tag{4}$$

An individual located within this plume volume at a distance r will inhale a total volume of aerosol in the time it takes for the plume to pass ($R_{inh.}dt = R_{inh} \cdot dr/u$). This corresponds to an individual activity intake by inhalation of:

$$A_{inh}(r) = \frac{k_a A_0 R_{inh}}{urH\Theta} \exp\left(-\frac{r}{L}\right) \tag{5}$$

The resulting total activity intake inhaled by the total population in the entire area affected by the plume, assuming a constant population density per unit area ($\rho(r) = \rho_0$), is then given by the integral:

$$A_p = \int_0^\infty A_{inh}(r)\,\Theta r \rho(r)\,dr = \frac{k_a A_0 R_{inh} \rho_0}{v} \tag{6}$$

Recalling that the inhaled dose is related to the intake activity by an effective dose coefficient (Equation 1), the total collective inhalation dose, which corresponds to a distance from detonation of $r = \infty$, is given by:

$$CD_{inh} = \frac{k_a e_{inh}(50) A_0 R_{inh} \rho_0}{v} \tag{7}$$

Notice that the above expression for the collective dose received by the affected population in the wedge model (and hence the number of cancers) is independent of the wind speed, opening angle, and plume height, and for a given population density and source term it depends only on the deposition velocity. Indeed, one of the great advantages of the wedge model is this very simple relationship for the collective dose (the best measure of public health consequences), which is not the case in the more complex models.

Results for RDEs involving Nuclear Fuel and a ^{60}Co Source

Fetter and von Hippel[17] recognize that the deposition velocity can vary dramatically depending on weather conditions, and that the value taken in the calculations then critically affects the outcome of the results for the all-important collective dose estimates. They have adopted a range of values in that paper—0.003 ms^{-1} for very dry atmospheric conditions, 0.03 ms^{-1} for average dry conditions, and 1 ms^{-1} for heavy rain—for estimating the risk to the public through inhalation of plutonium aerosol. A different approach is adopted for calculating collective dose estimates for inhalation (using Equation 7) and the corresponding number of excess cancer fatalities. These calculations are based on the isotopes with the highest dose contributions in spent nuclear fuel (^{238}Pu, ^{239}Pu, ^{240}Pu, ^{241}Pu,^{241}Am, ^{137}Cs, and ^{90}Sr), in fresh MOX fuel (^{238}Pu, ^{239}Pu, ^{240}Pu, ^{241}Pu) and for the ^{60}Co source.

Several advanced models have been developed for characterizing dispersal phenomena based on single particle Lagrangian techniques for solving the full fluid dynamical problem. One such model is utilized in the LASAIR code[18,19] developed by the German Federal Office for Radiation Protection (BFS). In order to calibrate and fix the parameters in the wedge model—the most important of which in terms of the collective dose estimates is the deposition velocity (v) (see Equation 7)—the authors have performed a calculation using the LASAIR code for detonation of an RDD with the ^{60}Co source term described earlier, in Commercial radioactive sources, under average

meteorological conditions. The authors have then made a quantitative comparison between the dose results from both LASAIR and the wedge model in order to find which value of deposition velocity produces the best agreement, which turns out to be $v = 0.03\ ms^{-1}$. This benchmark technique is described in detail in Appendix B. Using this value of deposition velocity the collective dose estimates and the associated number of fatalities for the radionuclides listed earlier have been calculated with the wedge model (using Equation 7) and are shown in Table 3. A standard MOX isotopic composition has been assumed, based on reprocessing of spent PWR fuel with a discharge burn-up of 50 GWd/tHM and a plutonium content of around 8%. We have assumed a wind speed of $5.4\ ms^{-1}$, an average breathing rate (R_{inh}) of $3.3 \times 10^{-4}\ m^3s^{-1}$ and a population density (ρ) corresponding to a large city like Frankfurt of $2600\ km^{-2}$. A full list of parameters used in the calculations can be found in Table 4.

As can be seen from Table 3, the public health consequences due to RDEs involving spent uranium fuel, fresh MOX fuel, and a ^{60}Co source are quite different. The total number of excess fatal cancers through stochastic effects over a period of 50 years due to the dispersion of spent fuel (2 kg) in a large city is around 65, whereas it is 335 for similar dispersion of fresh MOX fuel (2 kg) and only 1 for a scenario involving a ^{60}Co source (2000 Ci). Above a lifetime base cancer rate of around 25% in the developed world, these deaths do not have a large enough statistical significance to be detected. In the spent fuel case, the highest contributions to the collective dose are from the ^{238}Pu, ^{241}Pu, and ^{244}Cm radionuclides. The contributions to the collective dose from the fission products are relatively low. In the MOX case, as expected, the higher overall plutonium content and contributions from ^{238}Pu and ^{241}Pu, in particular, lead to a much higher collective dose.

THE AFTERMATH OF AN RDD INCIDENT

Protective and Clean-Up Actions

The authorities in charge of an emergency response in the case of a radiological dispersal event should develop measures or already have measures in place to deal with the following radiological hazards:

- direct exposure to radiation from a plume of airborne radioactive material or from radioactive material deposited on the ground,
- internal or external contamination caused by direct contact with the plume,
- inhalation of radioactive material,
- ingestion of radioactive material.

Table 3: Collective Doses CD_{inh} and excess cancer numbers resulting from the inhalation of high-activity isotopes present in 2 kg of spent uranium fuel (burn-up 50 GWd/tHM), for the plutonium isotopes present in 2 kg of fresh MOX fuel, and for a 2000 Ci source of ^{60}Co (final column). These calculations use the wedge model with a value of 0.03 ms^{-1} for the deposition velocity, which represents average, dry weather conditions.

	^{238}Pu	^{239}Pu	^{240}Pu	^{241}Pu	^{242}Pu	^{241}Am	^{244}Cm	^{90}Sr	^{137}Cs	Total	^{60}Co
					Spent UO_2 Fuel						
Mass (g)	0.6	11.8	5.5	2.6	1.7	0.95	0.1	1.4	3.1	—	1.8
Activity (Bq)	3.9E+11	2.7E+10	4.6E+10	9.9E+12	2.5E+08	1.2E+11	4.3E+11	6.9E+12	1.0E+13	—	7.4E+13
CD_{inh}(manSv)	503	38	65	267	0.3	135	287	13	5	1313	27
No. Excess Cancers	25	2	3	13	0	7	14	1	0	65	1
					Fresh MOX Fuel						
Mass (g)	4.2	80.5	37.9	24.3	11.9	—	—	—	—	—	
Activity (Bq)	2.7E+12	1.8E+11	3.2E+11	9.3E+13	1.7E+09	—	—	—	—	—	
CD_{inh}(manSv)	3480	253	450	2510	2	—	—	—	—	6695	
No. Excess Cancers	174	13	23	125	0	—	—	—	—	335	

Table 4: LASAIR and wedge model parameters used in all calculations.

Source term A_0	7.4×10^{13} Bq (2000 Ci) ^{60}Co
Effective dose coefficient $e_{inh}(50)$	3.1×10^{-8} $SvBq^{-1}$
Breathing rate R_{inh}	3.3×10^{-4} m^3s^{-1}(1.2 m^3h^{-1})
Amount of conventional explosives	2.5 kg
Corresponding cloud height (H)	35m
Wind speed (u)	5.4 ms^{-1} (constant)
Wind direction	45° (NE)
Dispersion class	D (neutral/stable weather conditions, weak solar irradiation)
Precipitation	0 mmh^{-1}(no rain)

In order to avoid, or reduce as much as possible, any contact between the public and the radioactive cloud immediately following an RDE, protective action should be taken in a series of different phases after the incident. During the early phase (defined from the time of detonation or radioactive release and extending until several hours later, when deposition of the airborne materials has ceased) it is advisable as soon as practicable to evacuate areas with total effective doses exceeding 100 mSv and to temporarily evacuate areas with effective doses exceeding 30 mSv. If local conditions prevent evacuation, the population should be advised to seek shelter. Indeed, as an alternative to evacuation, sheltering in a home, office, or other building with windows and doors closed may provide sufficient protection. This is simply because the shelter provides containment, distance, and shielding between individuals and radiological hazard. In the early phase, decisions must be based primarily on predictions of radiological conditions in the environment. Measures taken should include controlled access to the area that is likely to be contaminated, rapid communication with the public in the affected area in order to provide a warning to shelter inside of buildings with doors and windows closed, and a warning to eat and drink only from sealed bottles and packages. The main aims during this first phase are to reduce as much as possible external radiation exposure to the public and to take measures that prevent, wherever possible, inhalation and ingestion of radioactive material.

The period after an RDE when the source of radioactivity has been clearly identified and contained is the so-called intermediate phase, at which time more detailed environmental measurements should be available to aid the decision-making process on protective actions. During that time, relocation (temporary evacuation or continued exclusion of people from contaminated areas to avoid chronic radiation exposure) and decontamination are absolute priorities in order to protect the public from whole body external exposure due to residual gamma dose rates and from inhalation of potentially resuspended radioactive material. Another important protective action is the restriction on the use of contaminated food and water. Advisable maximum levels for food and

feedingstuffs are presented in Table 5. Suitable techniques for detailed environmental radioactivity measurements are γ- and α-spectrometry, which can be performed *in situ* or in a suitable facility off-site by making use of samples from the affected area. Exposure to small amounts of radioactive iodine, which would be of little concern if iodine spread throughout the entire body, is instead a major concern due to the fact that iodine concentrates in the thyroid glands, thus causing a very large organ dose. To prevent this exposure, it is advisable to take thyroid blocking agents when the organ dose exceeds 50 mSv. The intermediate phase extends until protective actions are completed and may overlap with the early and late phases lasting from several hours to several days.

The late phase begins after several days when recovery actions become the priority so as to reduce radiation levels in the environment to permit unrestricted, long-term use of the property. It ends when all recovery actions have been completed. The primary goal during this late phase is ensuring that the affected area is decontaminated. In an urban area this is likely to require considerable investment of time, manpower, and capital. The WAK case in Germany[20] may serve as a good example of the scale of such a clean-up activity. In 2000, about 10^9 Bq of α-emitters (in the form of various plutonium isotopes and ^{241}Am) and 10^{10} Bq of β-emitters (^{241}Pu and ^{137}Cs) were stolen from the WAK complex, which then resulted in the contamination of two apartments. The cost of decontamination for these two apartments alone was more than 2 million Euro, which is around five times the commercial value of the apartments.

Nuclear Forensic Investigations

In parallel with the efforts described above for protection of the public after an RDE, radioactive material which has been utilized in a RDD device, or indeed intercepted material which could be utilized in such a device, needs to be characterized. This information is critical in order to estimate the radiological hazard involved and to attempt to deduce the possible origin of the material—especially for the case of fissile isotopes. Knowledge of the origin allows one to improve physical protection measures and to prevent future thefts or diversions. Such nuclear forensic investigations rely on a number of analytical techniques in order to determine the various material components and properties.[21]

The primary analytical tool for the investigation of unknown nuclear material is γ-spectrometry. High resolution γ-spectrometry yields information on the isotopic composition of, for example, uranium and plutonium and reveals if any fission products are present. The isotopic composition can be determined in further detail by mass spectrometric techniques employing different ionization

Table 5: Maximum permitted levels for foodstuffs and feedingstuffs in Bq/kg.

	Baby foods	Dairy produce	Other foodstuffs	Liquid foodstuffs	Minor foodstuffs	Feeding-stuffs
Strontium isotopes	75	125	750	125	7500	
Iodine isotopes	150	500	2000	500	20000	
α–emitting isotopes, Pu and Am isotopes	1	20	80	20	800	
All other nuclides of half-life greater than 10 days, notably ^{134,137}Cs	400	1000	1250	1000	12500	1250[a] 2500[b] 5000[c]

[a]pigs, [b]poultry-lambs-calves, [c]other.

modes, depending on the type of material analyzed (thermal ionization for bulk samples, plasma ionization for trace amounts, and secondary ions for microparticles).[21] Electron microscopy can be used to determine the microstructure of the material, which can yield information on its production pathway. For example, scanning electron microscopy provides the particle size distribution as well as the elemental composition of single particles if combined with X-ray detection. In order to gain information on the origin of the nuclear material, the isotopic and physical composition results can be compared with reference data on nuclear materials wherever possible.

CONCLUSION

Inhalation of radioactive material (predominantly for α-emitters) dispersed as a result of detonation of a radiological dispersal device or sabotage of a nuclear fuel cask is the most significant health risk to the public and has the potential to cause fatalities over time through an increased cancer risk. Although the number of fatalities will be very low, even for the scenarios like the spent fuel and fresh MOX fuel cases considered in the present study, an RDE is likely to cause fear and panic due to the perception of threat related to radioactivity in general. In that sense, an RDD may be considered as a much more effective psychological weapon than a conventional weapon for maximizing the number of casualties. Moreover, the local economy will be severely affected in the event of a radiological dispersal event due to the disruption of commercial activities and the extremely expensive decontamination measures needed for both buildings and land. For these reasons it is absolutely critical that commercially available radioactive sources are secured and their traceability guaranteed. In the EU this is addressed by EC legislation.[22]

The consequences of an RDE are much less severe than those of a nuclear explosion. Compared to a fission or fusion weapon, the amount of radioactivity involved is much smaller as is the energy release from the conventional explosion used for RDD dispersal. For both model approaches, and for most of the RDDs scenarios conceivable, the casualties of an RDE are in the same range or smaller than those resulting from a "conventional" terror attack. It should be noted that the dose calculated for inhalation refers to the dose generated by inhaled radioactive material within the next 50 years. Therefore, for long lived nuclides such as many of the actinides considered the dose per year is much lower. In addition, in case of inhalation measures can be taken in order to eliminate the isotope from the body (DTPA injections or in very severe cases to wash the lung). This would reduce the dose deposited in the body.

An RDE is a localized phenomenon in space and time. The possibilities, therefore, of the earlier described systems for response to an RDE should hardly be seen in the context of operational emergency response systems, but rather as tools to be used *a priori* (e.g., for scenario building) or for estimating the effects

a posteriori (reassuring the public opinion). This is in contrast to a large nuclear accident, where the amount of radioactivity released can be extensive and can last for several days. In such a situation, support in real time via measurements and model predictions of the radioactive plume is useful to quickly provide decisionmakers with relevant information.

APPENDIX A

Large Scale Activity Release to the Environment

Section 2.4 gave a brief summary of a number of cases where radioactivity has been released to the environment in order to put the source terms considered for RDD construction into perspective. This appendix presents a short description of some of the events that led to these incidents. These cases include nuclear weapons explosions, reactor accidents, accidents involving nuclear materials, and radioactive sources. A detailed description of known criticality accidents between 1945 and 1999 has recently been completed.[23]

Nuclear Weapons Explosions and Weapons Tests

An estimate of the source term for a fission bomb, such as the Fat Man device that was dropped at Nagasaki with an equivalent of 22 kt of TNT yield, is 7.65×10^{19} Bq.[24,25] This is four orders of magnitude above the source term for a spent fuel rod after 24 h of cooling (Table 1).

Between 1966 and 1996, France conducted 193 nuclear tests above and beneath the atolls of Mururoa and Fangataufa in French Polynesia. Detailed IAEA studies were carried out[26–28] in order to ascertain whether radiological hazards exist at present or are likely to exist in the future as a consequence of these tests. The main conclusion was that a population permanently resident on the atolls and living on a diet of local products and seafood would not generally receive a radiation dose attributable to the residual radioactive material exceeding 0.01 mSv/year, which is equivalent to a very small fraction of the annual background radiation dose. The studies also estimated the radionuclide concentration in the ocean surrounding the atolls at various locations and over several different time frames. It was found that, in the absence of any hypothetical extreme disruptive (geological) event, the predicted long-term radionuclide concentrations will decrease to background oceanic levels beyond about 100 km from the atolls.

Reactor Accidents

Chernobyl, 1986[29-34]

By far the worst reactor accident to date, the Chernobyl accident, occurred during reactor tests on April 26, 1986. The release of radioactivity from the

Chernobyl plant was preceded by a steam explosion that caused the cover plate of the reactor to be blown off, resulting in an atmospheric release of volatile fission products. A secondary explosion threw out fragments of burning fuel and graphite from the core and allowed air to rush in, causing the graphite moderator to burst into flames. The graphite burned for nine days, during which time the main release of longer-lived radioactivity into the environment occurred. The initial large release was principally due to the mechanical fragmentation of the fuel during the explosion. It contained the more volatile radionuclides such as noble gases, as well as iodine and cesium isotopes. The second large release between day 7 and day 10 was associated with the high temperatures reached in the core melt. The sharp drop in releases after ten days is likely due to a rapid cooling of the fuel as the core debris melted through the lower shield and interacted with other material in the reactor.[32]

The total fuel mass in the core of the power plant was about 190 tons, about 3.5 ± 0.5 percent of which was released into the environment.[30] This corresponds to the emission of 6 tons of fragmented fuel. The analyses carried out on the core debris and the deposited materials within the reactor building have provided an independent assessment of the environmental release. These studies estimate that the release fraction of ^{137}Cs was 30 ± 10 percent ($8.5 \cdot 10^{16}$Bq) based on an average release fraction from the fuel of 47% with

Table 6: Current estimate of radionuclide releases during the Chernobyl accident.[31]

Core inventory on 26.04.86			Total release during the accident	
Nuclide	**Half-life**	**Activity Bq**	**Percent of inventory**	**Activity Bq**
^{133}Xe	5.3 d	6.50E+18	100	6.50E+18
^{131}I	8.0 d	3.20E+18	50–60	1.76E+18
^{134}Cs	2.0 y	1.80E+17	20–40	5.40E+16
^{137}Cs	30.0 y	2.80E+17	20–40	8.50E+16
^{132}Te	78.0 h	2.70E+18	25–60	1.15E+18
^{89}Sr	52.0 d	2.30E+18	4–6	1.15E+17
^{90}Sr	28.0 y	2.00E+17	4–6	1.00E+16
^{140}Ba	12.8 d	4.80E+18	4–6	2.40E+17
^{95}Zr	65.0 d	5.60E+18	3.5	1.96E+17
^{99}Mo	67.0 h	4.80E+18	>3.5	1.68E+17
^{103}Ru	39.6 d	4.80E+18	>3.5	1.68E+17
^{106}Ru	1.0 y	2.10E+18	>3.5	7.30E+16
^{141}Ce	33.0 d	5.60E+18	3.5	1.96E+17
^{144}Ce	285.0 d	3.30E+18	3.5	1.16E+17
^{239}Np	2.4 d	2.70E+19	3.5	9.50E+16
^{238}Pu	86.0 y	1.00E+15	3.5	3.50E+13
^{239}Pu	2.4×10^4 y	8.50E+14	3.5	3.00E+13
^{240}Pu	6.6×10^3 y	1.20E+15	3.5	4.20E+13
^{241}Pu	13.2 y	1.70E+17	3.5	6.00E+15
^{242}Cm	163.0 d	2.60E+16	3.5	9.00E+14
			Total Activity Released (Bq)	1.09E+19

subsequent retention of the remainder within the reactor building.[30] For ^{131}I, the most accurate estimate for release is thought to be around 50 to 60% of the core inventory of $3.2 \cdot 10^{18}$Bq. The amount of actinide elements released is a factor of 10^2 to 10^3 larger than an RDD constructed with a spent fuel rod. It is estimated that a total of $1.1 \cdot 10^{19}$ Bq was released to the environment. The isotopic inventory of the source term is summarized in Table 6.

Windscale, 1957[35]

In 1957 a fire occurred in the graphite-moderated reactor at Windscale in the UK, which was being used for plutonium production for nuclear weapons. At the time of the accident, the reactor was shut down and work was in progress on tests involving a so-called Wigner energy release in the graphite. These tests involve heating the graphite above a certain critical temperature. A fire broke out in the graphite that led to the destruction of some of the fuel elements. The fire was only discovered a day later. As the reactor had no containment structure and was only fitted with air-filters, volatile fission products were released into the environment. The official report states that about $7.4 \cdot 10^{14}$ Bq of the volatile fission product ^{131}I was released during the fire.[35] The fallout from the reaction was significant: large quantities of milk produced in the area around Windscale had to be destroyed, and measurable amounts of ^{131}I were found in large areas of Central Europe.

Vladivostock, 1985[36]

In August 1985 an explosion occurred in the reactor of a submarine docked in Vladivostock on Russia's pacific coast (Chasma Bay). The incident is similar to the SL-1 accident that occurred in the USA in 1961. Both these incidents can be classified as criticality accidents, where manipulation of fissile material started unwanted chain reactions with an associated release of energy and radioactivity. In the Vladivostock incident, the operators overlooked safety regulations while exchanging fuel in the reactor core. An energy explosion suddenly blew pieces of the reactor core out of the submarine and caused a fire that lasted for an hour or so. Fragments of burning material containing fission and activation products were thrown over an area some 50–100 m from the submarine. However, contamination was much more widespread, stretching out from the harbor to the seashore some 5.5 km away. The amount of radioactivity released is estimated at around $1.9 \cdot 10^{17}$ Bq.

Accidents Involving Nuclear Material and Radioactive Sources

Palomares, 1966[37]

On January 17, 1966 an American tanker plane and a B-52 bomber carrying four one-Megaton hydrogen bombs crashed while refuelling in the air over the

Spanish Mediterranean coast. Pieces of aircraft wreckage together with three of the bombs fell near the town of Palomares. The fourth bomb fell in the Mediterranean and was later found intact, as was one of the three bombs that fell to earth near Palomares. However, the conventional explosives in the warheads of the other two bombs exploded during the collision and their content of fissile material was scattered over an area of about 2.3 km^2, made up of agricultural, urban, and uncultivated land. No members of the Spanish public were directly injured in the accident but a large area of land had to be decontaminated. About 1000 m^3 of radioactively contaminated material was later removed for storage in the USA. To monitor the stochastic, long-term effects of the exposure, a total of 769 people underwent 1,190 examinations during the period from 1966 to 1990. For the 55 people who were identified as having internal plutonium contamination, urine samples have led to estimates of effective radiation doses (over 50 years) in the range 20 and 200 mSv. For comparison, a moderately high level of natural background radiation over 50 years leads to a total estimated dose of around 150 mSv.

Thule, 1968[38]

Similarly, on January 21, 1968 an American B-52 bomber carrying four one-Megaton nuclear weapons crashed onto the sea ice about 15 km from the Thule Air Base and about 11 km from the coast at Thule in north-western Greenland. The impact led to detonation of the conventional explosives in the bombs and spread their content of mainly ^{239}Pu over the so-called fire spot (approx. 750 $\times$ 150 m in size), where aviation fuel was burnt. The plutonium instantaneously oxidized to the insoluble oxide form. About 3 kg of the plutonium was found on or around the fire spot (the so-called black snow). An additional 3 kg was found among the many thousands of pieces of wreckage. An unknown amount was carried out to sea with the smoke from the fire that rose many hundreds of meters into the air. About 0.5 kg of ^{239}Pu, broken up into small particles, was later found on the seabed after the summer's radiological studies. In the time that has passed since then melting icebergs have left an increasing layer of gravel above the deposit. A Danish scientific expedition very soon found there to be no risk of contamination outside the so-called fire spot, which was rapidly shut off and later removed.

APPENDIX B

Cobalt-60: A Benchmark against LASAIR

In order to make collective dose estimates with the wedge model, the optimal parameters to be used must first be decided on. This is done by considering the case of an RDD that makes use of a widely available source: 2000 Ci of ^{60}Co,

which is in common usage throughout the world for radiotherapy applications in cancer treatment. By considering this particular source term, the LASAIR code is used to benchmark the wedge model and fix the free parameters.

Calibration of Deposition Velocity and Opening Angle

There are two essentially free parameters in the wedge model, the opening angle (Θ) and the deposition velocity (v). Fetter and von Hippel[17] recognize that the latter can vary dramatically depending on the environmental conditions, and that the value taken in the calculations then critically affects the outcome of the model results. In order to settle on reasonable values for both the deposition velocity and the wedge opening angle in the present study, a benchmarking exercise has been conducted with the aid of LASAIR. For the purposes of this benchmark, a calculation has been performed with LASAIR in which an RDD detonation is assumed to take place with an associated source term of 2000 Ci of ^{60}Co. A full list of the parameters used in this scenario is given in Table 4.

The aim of this benchmark procedure is to apply the wedge model to this same scenario and select values for the opening angle and deposition velocity in order to achieve the best possible agreement with the LASAIR results for the inhalation dose. However, before this procedure can be performed, a crucial difference between LASAIR and the wedge model must be taken into account. The inhalation dose at a given value of r in the wedge model is constant over the full lateral width of the wedge. This is not the case in LASAIR, as a simple inspection of the lateral profile of the inhalation dose reveals a peak value along the central axis with a symmetric decrease on either side of the central value. In terms of the physical model, therefore, LASAIR involves an additional loss mechanism which is not accounted for in the simple wedge model: a lateral loss mechanism. Since these lateral effects are absent in the wedge model, a correction must be applied to obtain a physically self-consistent cross-comparison, which requires introducing an effective deposition velocity. This simply means that the value of v obtained for the wedge model in the benchmark procedure will represent the total loss of aerosol as a function of r, which suggests a mimicking of the loss mechanisms in both the vertical and lateral directions. This is not simply a technical detail, but instead reveals a key mechanism that is not present in the wedge model.

This lateral profile discrepancy is significant, and a correction must therefore be applied to the wedge model to account for the difference. If the LASAIR lateral profile is approximated by a Gaussian function, the total airborne activity in the wedge model, based on the central axis values alone, will overestimate the corresponding LASAIR airborne activity as a result. This is simply because the area under a Gaussian function is much smaller than the area under a step function of the same width and height. To achieve agreement between the total areas under both of these functions, the magnitude (height) of the step function

Table 7: Inhalation dose (central axis values) as a function of r in LASAIR.

Dose (mSv)	Distance (km)
100	0.1
30	0.4
10	0.7
3	1.9
1	3.9
0.3	8.1

representing the wedge model profile must be multiplied by a correction factor. In order that both functions have the same width (3σ where σ in this case is the standard deviation of the Gaussian distribution), a simple exercise in algebra reveals that this correction factor (k_a) is:

$$k_a = \frac{A_{gaus}}{A_{step}} = \frac{\sqrt{2\pi}}{6} \tag{8}$$

With this correction taken into account, it is now possible to make a rigorous comparison between LASAIR and the wedge model. The inhalation dose along the central axis as a function of distance from the detonation point (r) obtained from the LASAIR calculation is shown in Table 7.

The corresponding formula for the inhalation dose in the wedge model can be rearranged such that:

$$log[r D_{inh}(r)] = log\left[\frac{k_a e_{inh}(50) A_0 R_{inh}}{uH\Theta}\right] - r/L \tag{9}$$

This means that if one were to plot the LASAIR values of $rD_{inh}(r)$ against r on a log scale, the wedge model equivalent results will be in the form of a straight line, and that a fit to this data will therefore allow access to the variables, Θ and v (through Equation 2 for the latter). The upper graph in Figure 2 shows precisely this type of plot for the LASAIR data. One can see that the points close to the detonation point (at 0.1 and 0.4 km) lie somewhat off the straight line that is expected in the simple model. This is due to the fact that LASAIR produces a distributed source at the detonation point, as opposed to the point source associated with the wedge model. Nonetheless, the remaining points exhibit very strong linear behavior, and have been used for performing the fit given by Equation 9, which is also shown in the figure. The results for the wedge model parameters that have been obtained from this procedure are:

$$\Theta = 12^\circ \text{ and } v = 0.03 \text{ms}^{-1}$$

Fetter and von Hippel[17] consider a large range of values for the deposition velocity, ranging from 0.003–0.03 ms^{-1} in dry conditions, up to 1 ms^{-1} in the presence of precipitation; they also state that a typical range of opening angle

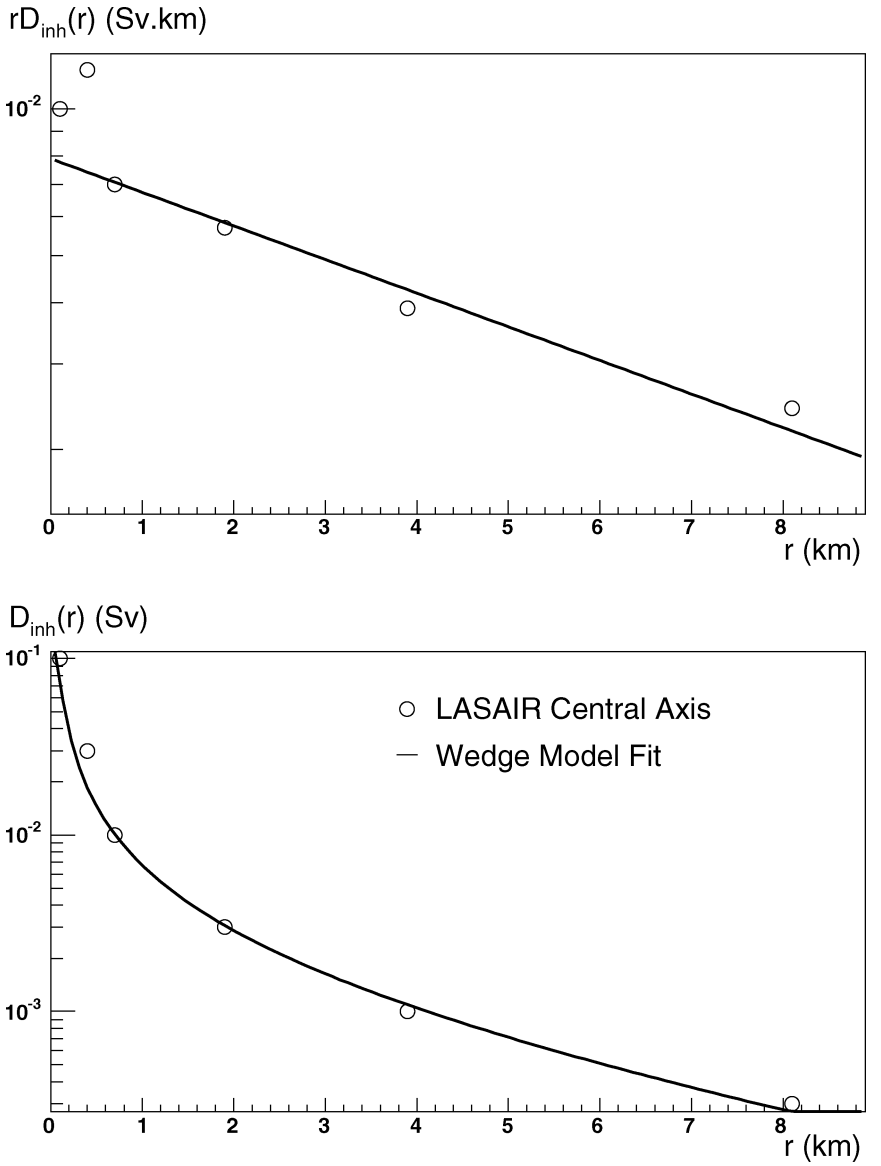

Figure 2: Central axis LASAIR data for the inhalation dose Table 7 and a fit based on the wedge model (Equation 9).

values is 3°–17°. Therefore, the values given earlier from the fit to the LASAIR data seem more than reasonable. However, one should bear in mind that the deposition velocity of 0.03 ms^{-1} is an effective parameter that takes into account loss of airborne activity in both the vertical and lateral directions. It is also very important to stress that one must apply the correction given in Equation 8 when using these deposition velocity and opening angle values in Equations 2–7.

What has been shown in the benchmark exercise, therefore, is that although the wedge model does not include a mechanism for lateral loss, excellent agreement between it and LASAIR can be achieved for the crucial inhalation dose profile by introducing an effective deposition velocity, and by making a simple correction to the wedge model lateral geometry. This agreement can be seen in the lower graph in Figure 2.

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