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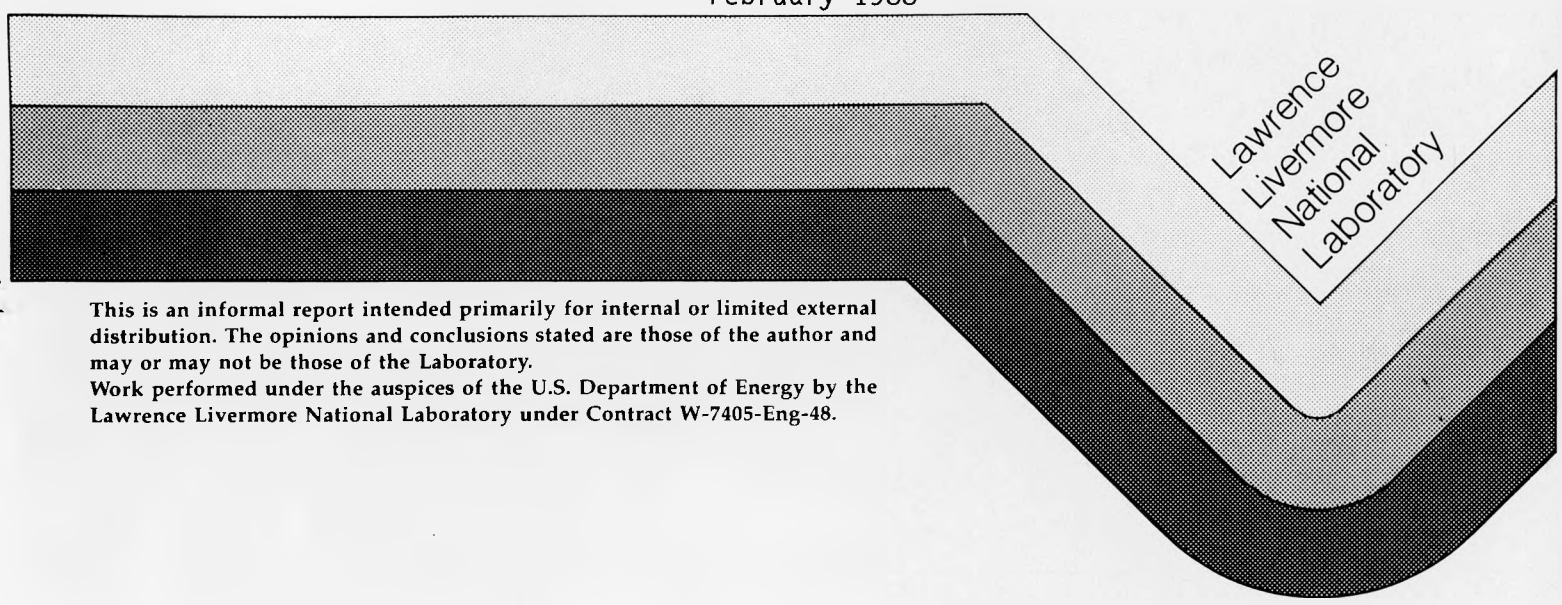
February 1988



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I. Introduction

The purpose of this report is to determine, from the available information, the potential environmental effects at Yuma Proving Ground (YPG), resulting from further testing of surface-burst artillery projectiles (XM785) containing amounts of beryllium (Be) and depleted uranium (DU).

In a previous review of the existing YPG Environmental Assessments by Los Alamos National Laboratory, Rodgers et al. (1984) found that there was insufficient data available to characterize the baseline, that the soil-sampling strategy used was inadequate, that the chemical analytical procedures followed were useful only for defining high impact areas, and that the source terms for Be and DU were not well defined. Rodgers et al. (1984) also recommended studies of corrosion, solubilization, and transport of DU and Be, determinations of mass balance, and a seven-point experimental effort.

There are a number of sources of information not yet utilized, however, and the Rogers et al. (1984) report did not quantitatively elaborate on its criticisms and recommendations. For example, it is not clear whether the Be and DU are deposited in an amount sufficient to be detected within the normal variability of background levels, and if Be and DU are assumed to be entirely of soluble form, then what relative risk are they to the environment of King of Arizona (KOFA) Firing Range, which already has been disturbed for two decades by 1000 to 4000 impacts of conventional projectiles per month?

A. Environmental Setting

This section will describe, in brief, the setting at Yuma Proving Ground (YPG). For an in-depth look at the existing environment, the site Environmental Assessment should be consulted (US Army, 1978). Yuma Proving Ground is located about 37 km northeast of the city of Yuma in southwest Arizona (Figure 1). The area of YPG that is being used for testing munitions is the KOFA Firing Range (Figure 2). The plain upon which this Range exists is a broad, sparsely vegetated area in the Sonoran Desert region of the United States. The area is approximately 240 m (800 ft) in elevation and is characterized by areas of compacted desert pavement. Precipitation occurs mainly in storms that can generate locally intense runoff, which can cause quite a bit of erosion.

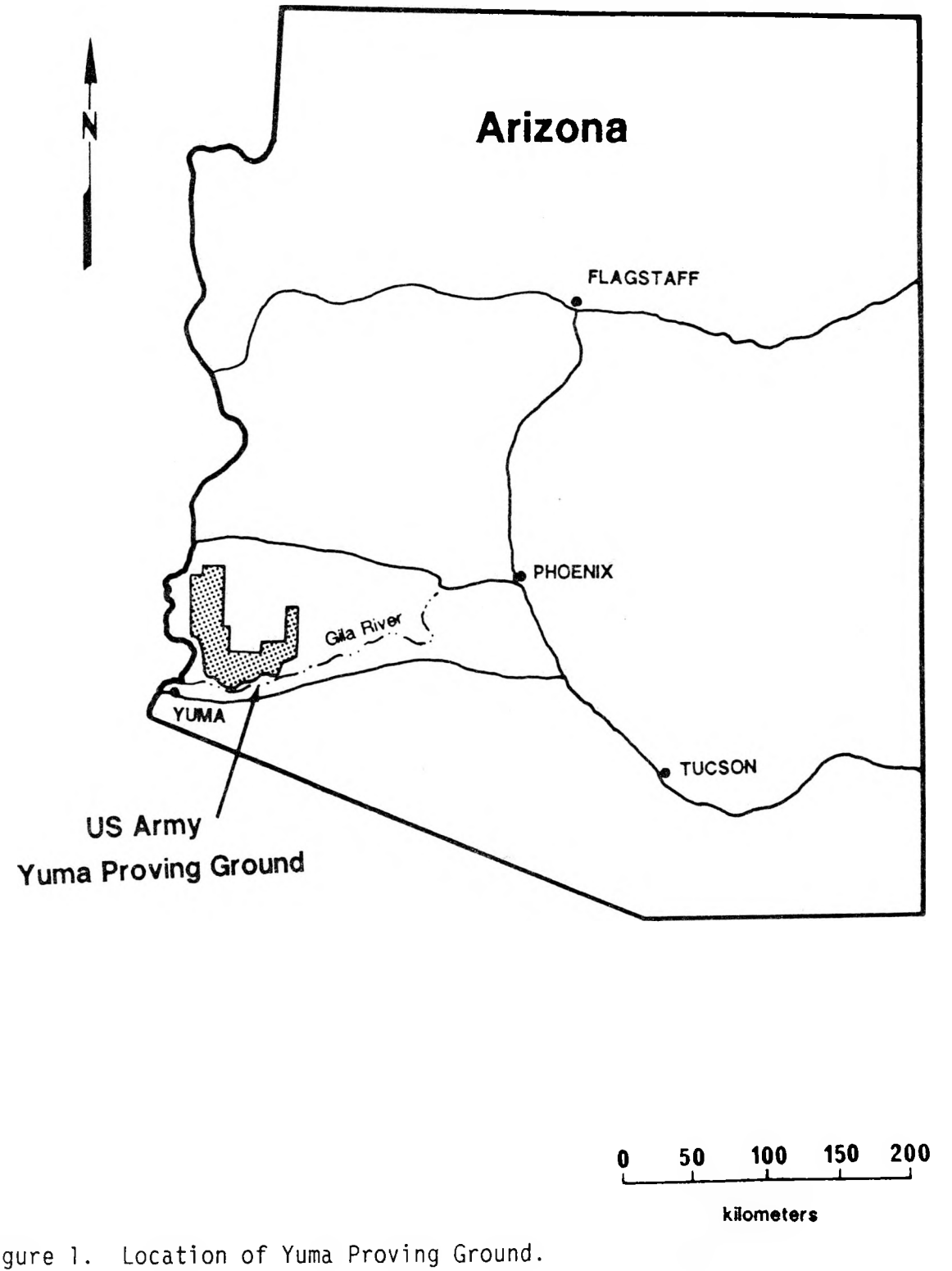

Figure 1. Location of Yuma Proving Ground.

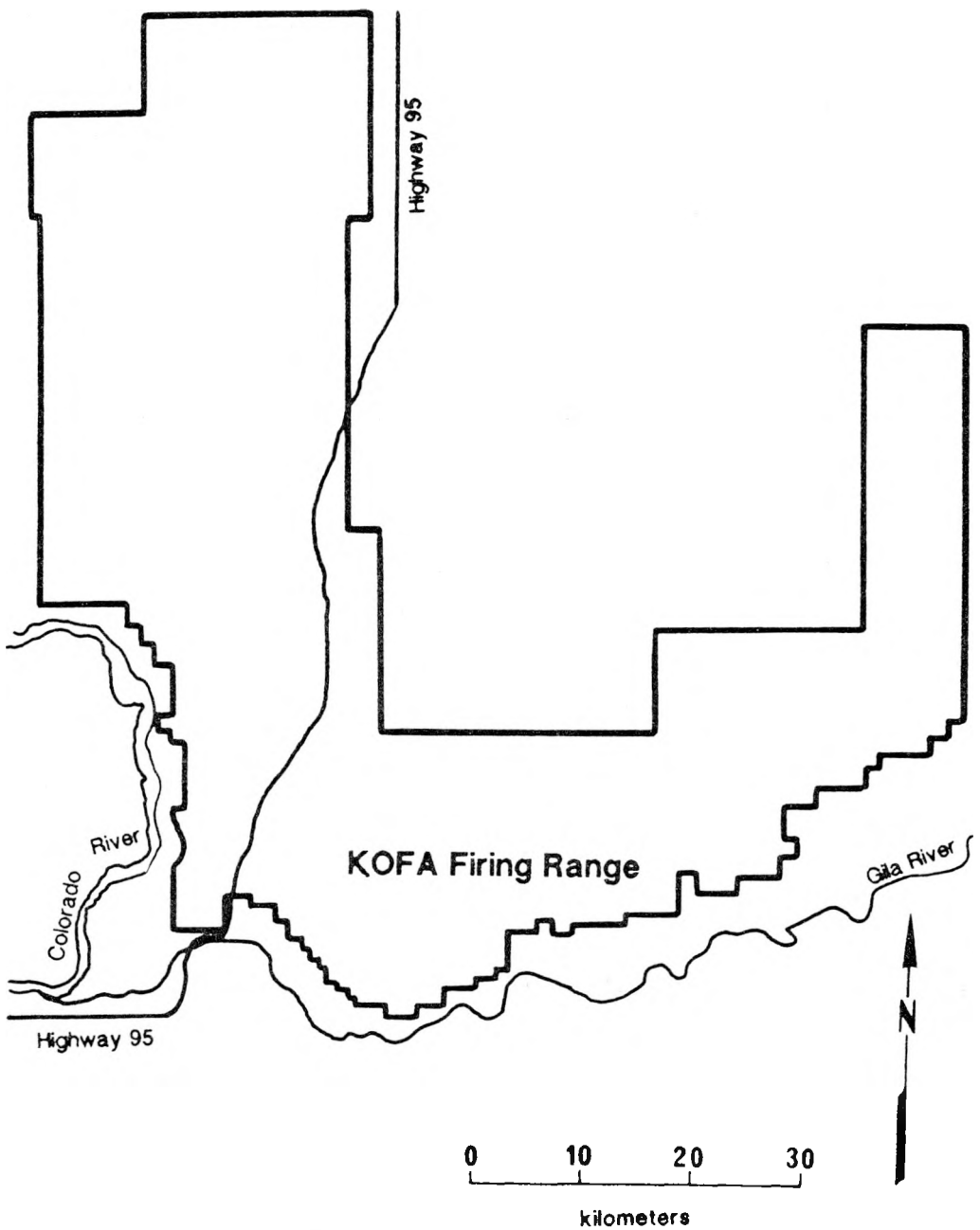

Figure 2. King of Arizona (KOFA) Firing Range location.

Runoff of the area is divided into two watersheds. The western portion of KOFA drains into the Castle Dome Wash, which drains into the Gila River near its confluence with the Colorado River. The eastern portion drains through washes off site until they intercept levees of an irrigation system just south of the Range boundary. The irrigation system feeds agricultural areas in the Mohawk Valley. From here, the runoff proceeds onto gently sloping alluvial fans near the Gila River, eventually flowing into the river.

Wind direction varies by season over the KOFA Range. In the winter and early spring, winds come from the northwest and north in the direction of the Mohawk Valley. It is during this season that the greatest occurrence of high wind speeds are found in the area. During the summer months, the wind is predominately out of the southwest. The potential for wind erosion on the sloping fan terraces is slight in the undisturbed areas; but disturbed areas with lost desert pavement are expected to be more easily erodible.

The predominant soil type in the portions of the Range used for testing is characterized as mixed alluvium of gravels, sand, silt, and clay. These soils are moderately permeable, have a moderate to low water-holding capacity, and have a pH of between 7.9 and 9.4 (US Army, 1978). Some of the soils in the eastern portion of the Range are the most productive rangeland soils in the area, especially where areas receive excess moisture from runoff. A study would have to be conducted to determine if the permeability of the soils in the area would allow contaminants to seep into ground water.

The wildlife in the area are most likely transient or have become somewhat accustomed to the sight and sound of people and shots. Tests contributing thousands of artillery rounds each month have been conducted in the KOFA Firing Range. KOFA is bounded on the north by the KOFA Game Range and to the east by the Palomas Mountains, which provides some habitat for larger mammals. Wild horses have been spotted at YPG; they likely use habitat along the border of the Mohawk Valley, located to the south (Rodgers et al., 1984).

B. Proposed Action

Under the direction of U.S. Army Test and Evaluation Command (TECOM), the XM785 atomic projectile is scheduled to be tested at YPG. This projectile

contains both DU and Be that will be deposited as a result of the testing. Data concerning the depostion of DU and Be at YPG have been obtained from previous tests of the XM753 projectile. The XM753 is a larger munition than the XM785, and a ratio has been determined to accurately predict the deposition from the XM785. The XM785 contains about one-half of the Be and the DU found in the XM753. All modeling and sampling data in this report are for the XM753 except where specifically noted.

The testing will involve groundburst detonations. The DU and Be will also be deposited as a result of mechanical breakup of the shell or shaped-charge induced detonation of dud rounds after impact. Much of the testing procedure is classified.

C. Summary of SAI Report

Science Applications, Inc. (SAI) by contract took soil and air samples of the area surrounding the detonation of XM753 tests in order to characterize the dispersion and deposition of Be and DU (Mishuck and Hartung, 1981). Because of small sample size and problems in the analytical method, limited conclusions can be drawn from these data (Rodgers et al., 1984). Further sampling should be conducted to better understand the dispersion characteristics. However, with these data, some preliminary estimates of environmental impacts can be calculated.

Three series of shots were analyzed by SAI: JB3, JFF3, and JFF4. The JFF3 series yielded only a trace of Be, whereas the JFF4 tests, probably because the rounds were shot into targets with very few rocks, yielded good data with a greater recovery of material.

Prior to completion of dynamic tests, static shots (3 rounds) were conducted to more precisely define the fallout pattern. A number of ground-based air samplers were placed in an array in the expected downwind direction, with a few also located upwind. Of the three shots, only the third one sent the plume through the center of the sampler array; the second shot missed most of the samplers. A helicopter was used to tow a sampler through the smoke cloud. According to SAI, the helicopter did not appear to disrupt the cloud, and it was possible to gather data on the Be content of the smoke cloud.

For the dynamic tests, which were fired from long range, an extensive program of ground-based air sampling and soil sampling was undertaken. Three-hundred sixty pre- and post-shot soil samples were analyzed. Helicopter samples were not taken because of the difficulty in chasing a smoke cloud, the origin of which would be unknown due to the inaccuracy of shell placement. The soil samples showed very little Be. Only 8 pre-shot and 16 post-shot samples had levels greater than 3 μg of Be/g of soil. This level of Be in soil is just above the natural background, considering the normal range of variability. That ratio of Be recovery to DU recovery was about 1:200, while previous tests yielded a ratio of about 1:30. This may indicate that Be values were undetected due to sampling or analytical limitations. Because so little Be was detected, additional soil samples were taken within 25 m of the blast. More Be than DU was found in this zone, which was opposite the pattern found in the outer soil samples.

Mass balance estimates were conducted to determine the fate of the Be from the blast (Table 1). Ejecta (small fragments) and fallout models were used to supplement the data on recovered Be and the helicopter-sampled smoke-cloud data. SAI states that the helicopter data and their fallout model likely underestimate the quantity of Be, while their linear and ejecta models could either overestimate or underestimate the predicted quantities.

Table 1. Beryllium Mass Balance

	Mean	Std. Dev.
Recovered parts	33.1%	15.5%
Plume-smoke	8.32%	8.86%
Dust fallout	7.53%	7.21%
Ejecta	54.0%	46.0%

The ejecta model predicts about 0.3 to 0.4 mg Be/m^2 at 186 m from the blast, which was determined to be the maximum throw distance for ejecta. Beryllium carries about 3.2 times as far as DU; inversely, larger particles of Be are found near smaller particles of DU. The mass balance data were then normalized (Table 2):

Table 2. Beryllium Normalized Mass Balance

	Sampled Avg.	Sampled Max.	Avg.	Estimated Std. Dev.
Recoverable parts	32.5%	32.1%	32.3%	22.9%
Plume-smoke	8.0%	18.9%	13.4%	14.1%
Dust fallout	7.3%	5.9%	6.7%	6.4%
Ejecta	52.2%	43.1%	47.7%	40.7%

SAI made some preliminary conclusions about the data they had collected:

- Toxic respirable Be remains in the vicinity of the craters.

- The amount of respirable and non-respirable toxic material leaving the boundary of the KOFA Range is below the accepted threshold limit values.

- Water quality is not affected.

- All significant contamination occurs within 500 m of blast.

- At distances more than 200 m from the crater, the amount of Be added to the soil by each shot is less than the background levels.

D. Additional Sources of Information

A report by Sandia National Laboratory (Luna et al., 1983) was largely overlooked in the review by Rodgers et al. (1984). The Sandia report supplies a framework to interpret the SAI data and provides quantitative estimates of downwind air concentrations and deposition from the XM753 tests. This information and that from other studies will be used to verify the levels of Be and Du that are of potential concern.

II. Exposure Scenarios

Possible adverse health effects to on-site personnel arising from exposure to Be as a result of the testing of the projectile containing Be and DU would only occur from three possible exposure scenarios:

A. Exposure to Be in the smoke cloud if it is carried, under unusual weather conditions, downwind to the personnel operating the cannon.

It is generally understood that the safety boundary from the site of detonation is 1200 m (Mishuck and Hartung, 1981), but even at this distance the possibility exists for potential exposure to elevated concentrations of Be. With the crew 1200 m away, under worst-case meteorological conditions, they could be downwind from the detonation site and in the centerline of the plume. With a 5 m/s wind speed, the smoke cloud would reach these soldiers in 4 min. We estimate here that a single 15-min exposure at 1200 m would be below the occupational 8-h exposure limit (2 $\mu g/m^3$) by a factor of ten.

The results of the 1981 study conducted by SAI (Mishuck and Hartung, 1981) show that the small size of the Be particles give them the potential to be carried for long distances. However, the SAI samples and calculations both indicate that due to rapid dilution, the Be concentration may only be potentially hazardous within the first 100 m from the site of detonation.

B. Inhalation of the previously deposited Be when it is resuspended by vehicles that travel to the impact site for purposes of cleanup, observation, or placing markers at ground zero.

This exposure possibility is the scenario to which the most attention should be given. Because the testing of munitions that contain Be has been conducted at Yuma Proving Ground over a period of thirty years, the amount of Be already on the desert floor may be locally elevated. Personnel travelling in motor vehicles over the surface of the desert to the impact site may resuspend particles of Be or dust particles to which Be compounds have adhered. The concentration of Be in this dust cloud together with freshly deposited Be from current testing may exceed permissible exposure levels.

C. Inhalation of previously deposited Be resuspended by wind erosion.

This scenario constitutes a long-term, chronic source of Be and begs for consideration of good management practice and of criteria for soil reclamation.

III. Dispersion Modeling

A. Introduction

The testing of the XM753 projectile involved the atmospheric release of Be and DU. In 1980-1981, SAI conducted a study of the emission, transport, diffusion, and deposition of material from the testing of XM753 munitions (Mishuck and Hartung, 1981). We used results of this field experiment to characterize the meteorology and source term that were input to an atmospheric dispersion model used previously for this application. The focus here will be on the computer model. In the next section we will summarize the modeling methodology and input parameters for model runs. Results of computer simulations will be presented in the final section.

B. Methodology

1. Atmospheric Dispersion and Deposition Model

The MATHEW-ADPIC three-dimensional particle-diffusion model is designed for calculating the dispersion and deposition of airborne material in the time- and space-varying atmospheric boundary layer. This model is really a system of submodels, as shown in Figure 3. The heart of the system is the atmospheric-diffusion particle-in-cell model (ADPIC). ADPIC solves the three-dimensional advection-diffusion equations in flux form by following Lagrangian marker particles within an Eulerian grid. The required mass-consistent wind fields are provided by a mass-consistent three-dimensional wind-field model (MATHEW). The details of these models and other supporting models, indicated in Figure 3, will be described below in the order they are used.

a. TOPOG

Block topographical surfaces used in MATHEW and ADPIC are created by TOPOG. The terrain to be simulated for this study was assumed to be flat. To adequately cover the study area, a 4 km x 2 km domain was chosen for ADPIC

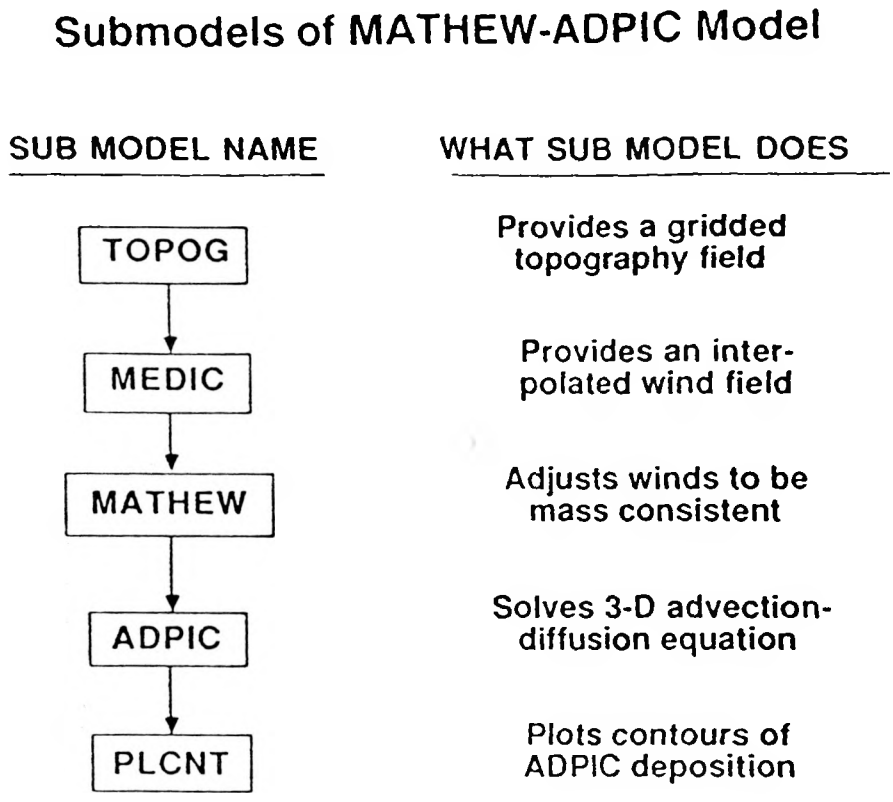

Figure 3. Submodel components of the MATHEW-ADPIC model.

calculations. The vertical extent of the model domain was set at 140 m above ground to encompass the debris cloud while providing as much vertical resolution as possible. Since ADPIC has 40 horizontal and 14 vertical cells, this leads to a horizontal cell size of 100 m in the downwind direction (50 m in the crosswind direction) and vertical cell size of 10 m. A second set of model runs were made with a 40 km x 20 km domain, with a vertical extent of 420 m.

b. MEDIC and MATHEW

The meteorological data interpolation code (MEDIC) creates a three-dimensional gridded wind field from surface and upper-air observations. Several parameters are used to achieve a realistic interpolated wind field. Extensive graphical output permits evaluation of the interpolated data to see if a rerun is necessary with revised parameters.

Mass-consistent wind fields are calculated from the MEDIC output using the three-dimensional, diagnostic wind-field model MATHEW (Sherman, 1978). This model uses a variational analysis technique (Sasaki, 1970) in adjusting the wind-field iteratively to be mass consistent (the constraint) within some specified level of tolerance, while minimizing the difference between adjusted and "observed" values. Allowable adjustments to velocity are governed by parameters that depend on atmospheric stability and grid-cell geometry. The initial interpolated wind field has no vertical velocity because only the horizontal wind components are given in the observations. Vertical velocities are calculated during the process of removing mass inconsistencies in the interpolated wind field; this, along with horizontal velocity adjustments, properly accounts for effects of terrain on the flow field.

c. ADPIC

The space- and time-varying distribution of released source material is calculated by the numerical, three-dimensional ADPIC model (Lange, 1978). This particle-in-cell code is capable of handling complex conditions, including strongly distorted wind fields, space-varying surface roughness, wet and dry deposition, gravitational settling, and radioactive decay. The

particle-in-cell approach is well suited for modeling the dispersion and deposition of radioactive particles; marker particles are used to simulate the transport, diffusion, and deposition of released material.

The governing equations for ADPIC are given in Figure 4. The three-dimensional advection-diffusion equation to be solved is Equation 1. By assuming incompressibility (Eq. 2) and defining a diffusivity velocity (Eq. 3), we can write the advection-diffusion equation in its pseudovelocity (flux conservative) form (Eq. 4); here the particle is moved according to both an advective velocity from the wind and a diffusive velocity from the concentration gradient.

Within the model, the computational time cycle is divided into Eulerian and Lagrangian parts, as illustrated in Figure 5. First, in the Eulerian part, gradients of the current concentration field are used to calculate diffusivity velocities, which are added to the advective velocities to give pseudovelocities at grid-cell corners. Next, in the Lagrangian part, the individual marker particles are transported by pseudovelocities to their new positions, defined in Lagrangian coordinates. Finally, the new concentration distributions are determined by locating the Lagrangian particles within the fixed Eulerian grid. This hybrid Eulerian-Lagrangian scheme eliminates the artificial diffusion associated with totally Eulerian schemes.

The source term can be described by several separate sources located anywhere within the grid and representing separate populations of particles. The spatial description of each source is given by specifying the center in three dimensions. A Gaussian distribution is assumed around this point, with separate standard deviations specified for each direction. Finally, bounds about the source center are specified that allow, for example, for a source to be located at the ground with an exponential decrease of particles with height.

ADPIC allows the specification of lognormal particle-size distributions for each source in terms of the median, geometric standard deviation, maximum, and minimum values. The particle-size information is most important for its role in gravitational settling and subsequent deposition.

Advection-diffusion equation:

$$\frac{\partial \chi}{\partial t} + U_A \cdot \nabla \chi = \nabla \cdot K_{ij} \nabla \chi \qquad (1)$$

Assume:

$$\nabla \cdot U_A = 0 \qquad \text{(Incompressibility)} \qquad (2)$$

Define:

$$U_D = -K_{ij} \frac{\nabla \chi}{\chi} \qquad \text{(Diffusivity velocity)} \qquad (3)$$

Pseudovelocity form of diffusion-advection equation:

$$\frac{\partial \chi}{\partial t} + \nabla \cdot (U_P \chi) = 0 \qquad (4)$$

where $U_P = U_A + U_D$

and U_P is the pseudovelocity vector
U_A is the advection velocity vector
U_D is the diffusivity velocity vector
K_{ij} is the diffusivity tensor
χ is the concentration

Figure 4. Governing equations in the ADPIC model.

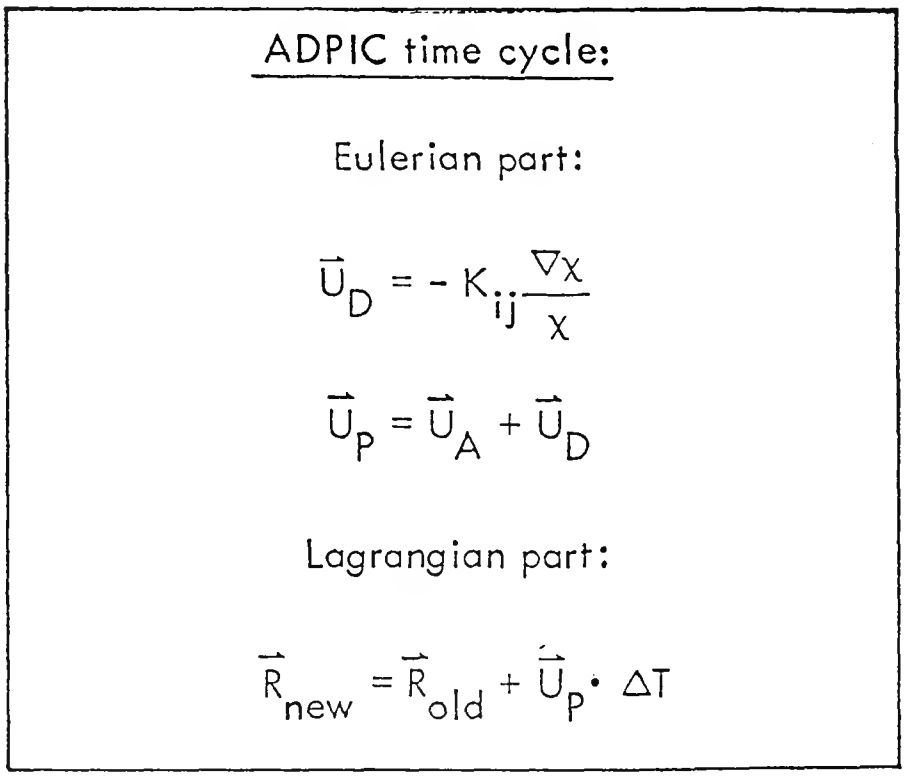

Figure 5. ADPIC time cycle.

Surface deposition is modeled by using a specified deposition velocity for each source. The deposition velocity is dependent on atmospheric stability, surface roughness, and particle size. The dependence of deposition velocity (and settling velocity) on particle size is shown in Figure 6. Values are often determined by experiments where the deposition rate (net flux at the surface) and surface concentration are measured; the deposition velocity is, then, the surface flux normalized by the surface concentration.

d. PLCNT

Most graphical output is generated by the plot-contour code, PLCNT, which uses an output file from ADPIC containing two-dimensional arrays of concentration values at cell centers. ADPIC has the capability for nested grids, with increased resolution near the source. In Figure 7 are shown the hierarchy of grids nested within the primary grid, and the horizontal resolution associated with each grid.

Contour levels may be specified, or chosen by the code to adequately describe the field. Contours may be drawn for instantaneous air concentrations, integrated air concentrations, and cumulative and time-integrated surface deposition. Using appropriate dose conversion factors, we can display exposure rates and total exposure. In addition to contours, PLCNT also provides a time history of values at up to 15 locations within the grid. This feature facilitates the direct comparison of model results with observations or the simulation of what individual samplers would have measured.

Although most of the graphical output for an ADPIC run is handled by PLCNT program, there is the option within ADPIC to plot the location of particles at specified times after release. This capability exists only within ADPIC. The three-dimensional cloud of particles is depicted by plots in three planes: x-y, x-z, and y-z. An example of such a plot for the x-z and x-y planes is given in Figure 8 at 2 min after detonation for the simulation.

The value of using a post-processor code, such as PLCNT, is that ADPIC need only be run once for a given case. Subsequent analyses can be generated by several runs of the efficient post-processor at greatly reduced computational expense.

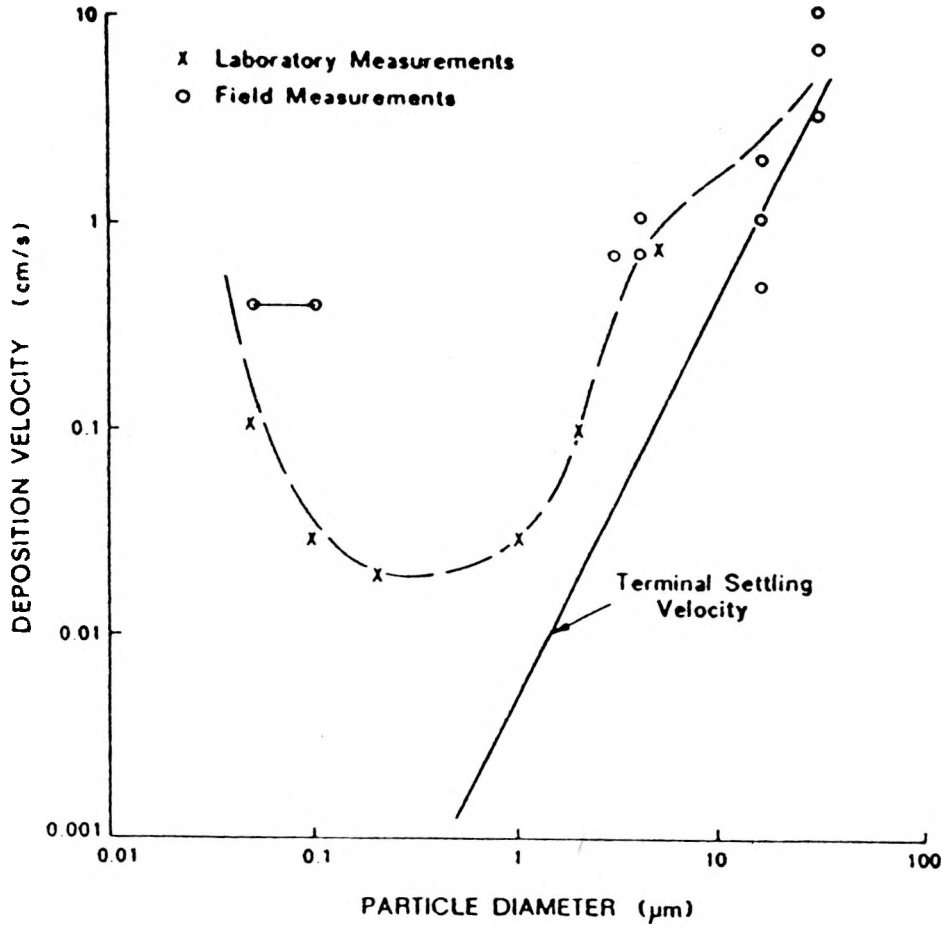

Figure 6. Laboratory and field measurements of deposition velocity to grass versus particle size (McMahon and Denison, 1979).

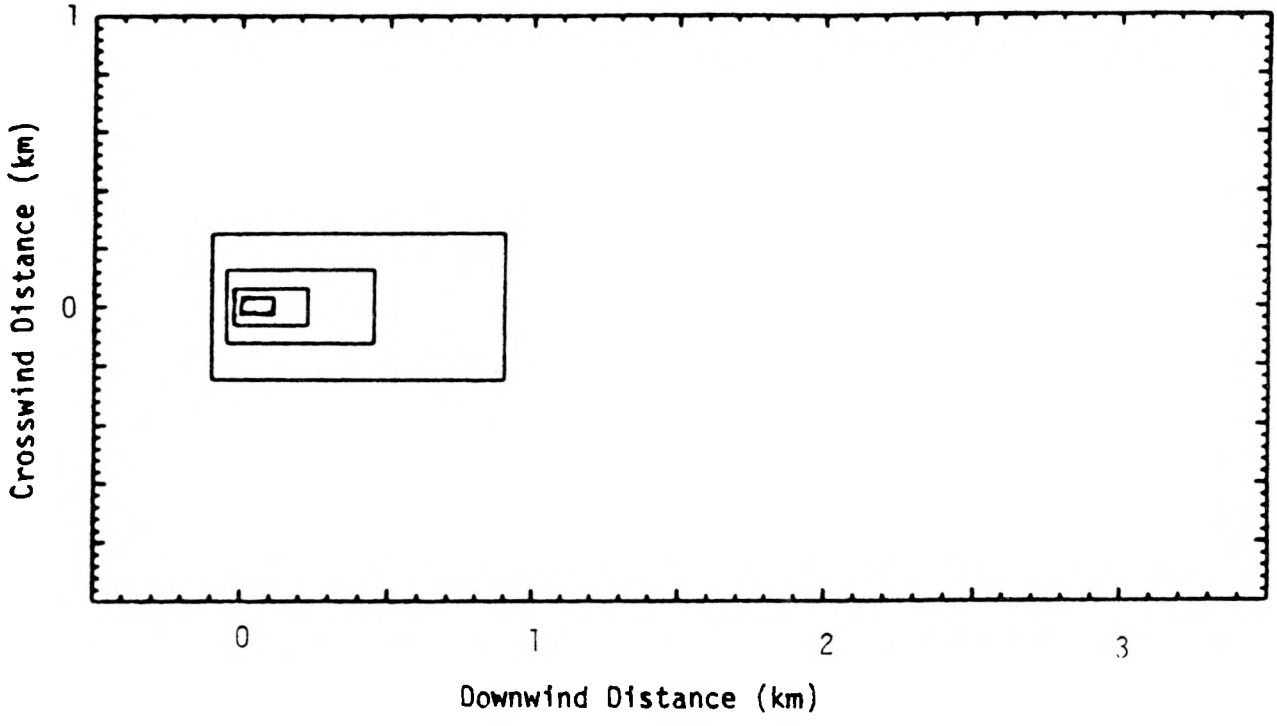

Figure 7. Nested grid plot showing heirarchy of grids.

Marker Particles Are Used to Represent the Debris Cloud

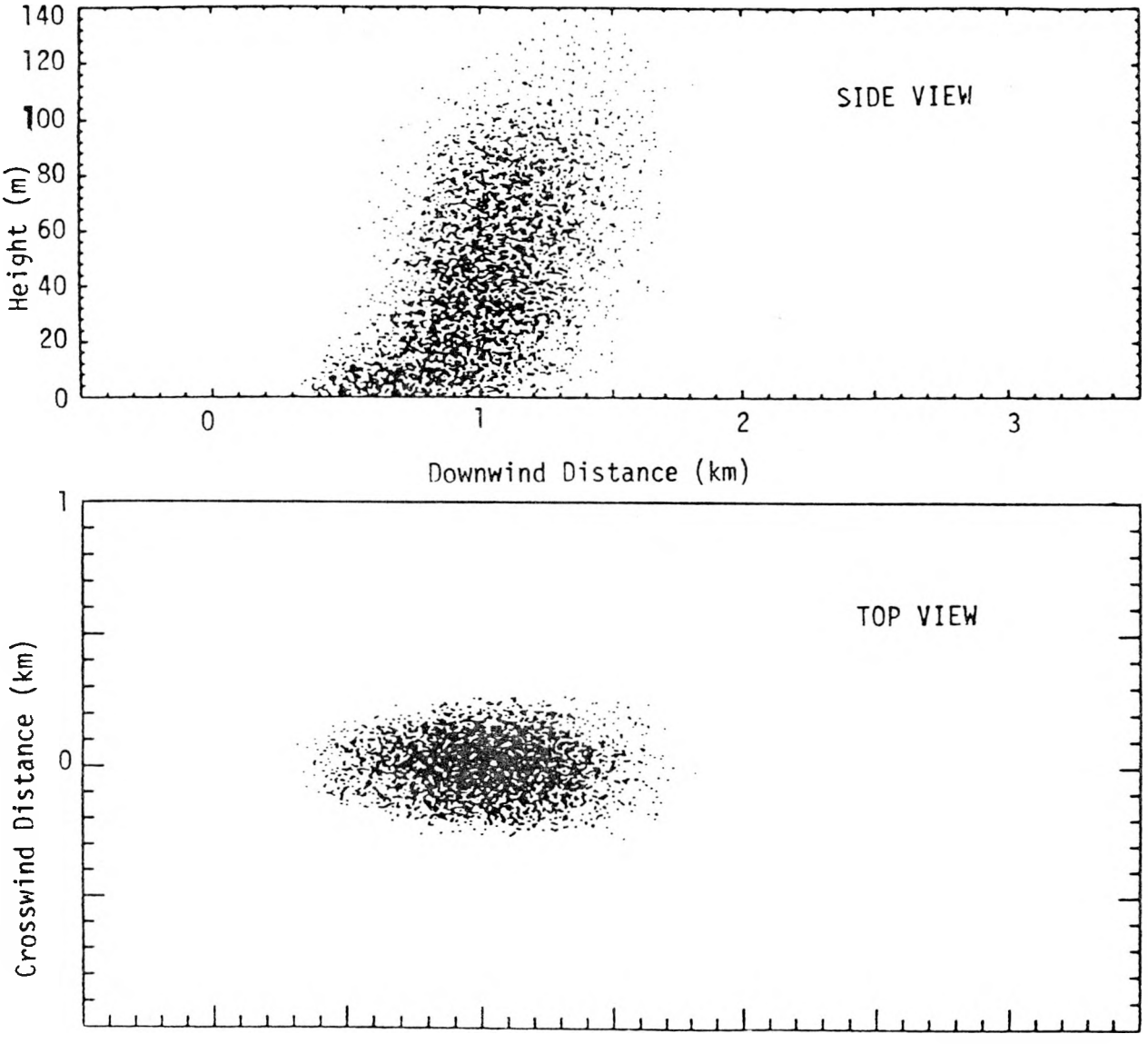

Figure 8. Marker particles are used to represent the debris cloud (2 min after detonation).

e. Model Validation

Tracer studies with data to a radius of 80 km were used to evaluate the accuracy of the MATHEW/ADPIC model (Dickerson and Lange, 1986). A summary of these model-data comparisons is shown in Figure 9. The two curves show the accuracy range depending on the type of terrain. From the figure, it can be seen that for 50% of the rolling terrain cases, the model results at specified locations were within a factor of 2 of the observations there; for the complex terrain cases, this level of accuracy was achieved only 20% of the time. For the XM753 test simulation, accuracy is expected to be as good or better than the "rolling terrain" curve.

Differences in modeled and observed values were due to several causes. Most importantly, the wind directions were reported by NOAA and FAA stations to the nearest 10 degrees. Often a shift in modeled contour patterns of only 5 degrees would have a noticeable effect on model-data agreement, especially where concentration gradients are large. Second, the coarseness of the grid often failed to resolve important small-scale terrain features, which effected the observed concentration patterns. Third, it is very difficult to specify the diffusion parameters in time and space to adequately represent turbulent diffusion found in typical atmospheric conditions. This can lead to factors of two to five difference between modeled and observed values. It must be recognized that comparison of modeled and observed values at specific locations is a severe test of model performance, especially because the model was not "tuned" to any given sites or conditions associated with the observations.

2. Source-Term Specification and Meteorology

The specification of input parameters for model simulations was guided in part by the results of the SAI study (Mishuck and Hartung, 1981). The cloud height was observed to rise rapidly to about 50 m in the first few seconds after detonation and subsequently stablilize at about 100 m after a few minutes. The top and bottom of the cloud were therefore set at 100 m and 50 m, respectively, with a diameter of 100 m. Material within the cloud had a Gaussian distribution centered about the center of the cloud, as shown in Figure 10. The stem under the smoke cloud was assumed to continue down to

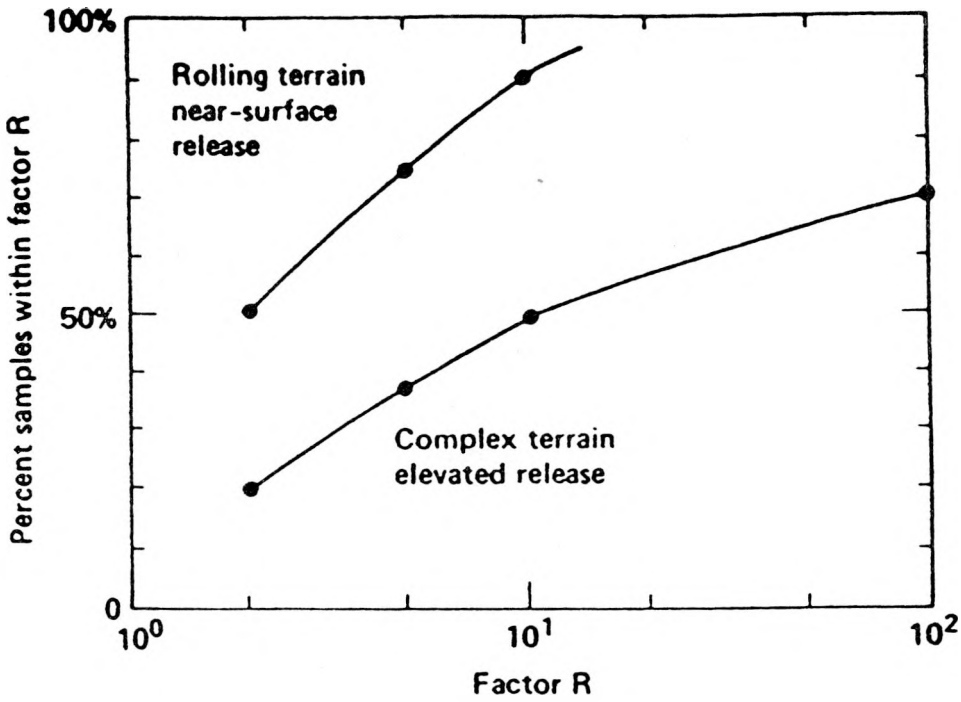

Figure 9. Model precision expressed as a percentage of model versus data comparisons within the factor R specified (after Dickerson and Lange 1986).

MODELED SOURCE TERM

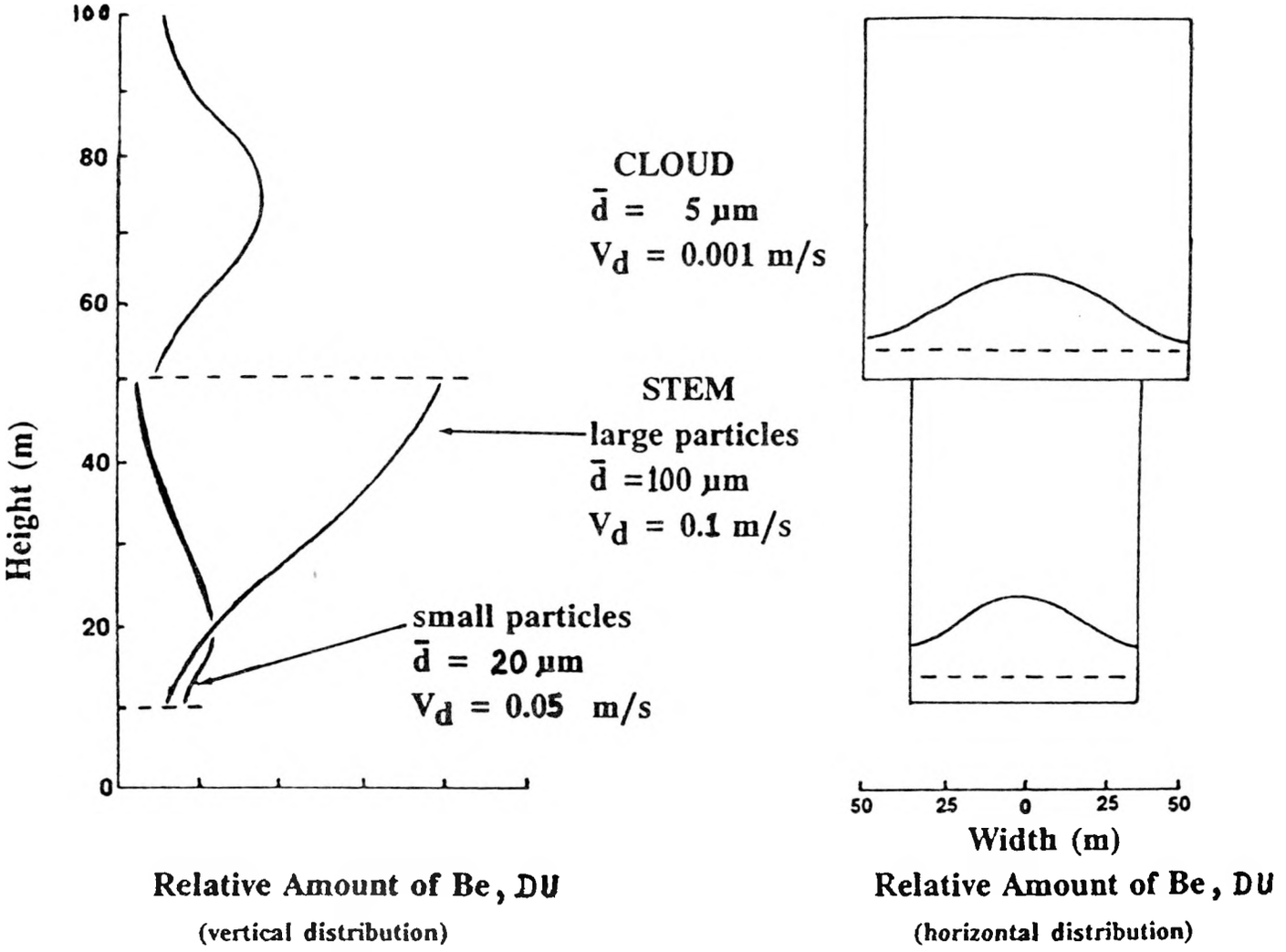

Figure 10. The relative amounts of Be and Du distributed in a two-part Gaussian cloud in the model.

within 10 m of the ground, with a width of 70 m. Material was also distributed spatially in a Gaussian fashion.

Results of helicopter sampling of the smoke cloud gave particle sizes with median diameters ranging from 3 to 5.5 μm, and with 90% less than 20 μm in diameter. The stem was assumed to contain lognormally distributed populations of both large and small particles, see Table 3,

Table 3. Particle-size Distribution Parameters in the Model

	Stem		Cloud
	Large	Small	
Median diameter (μm)	100	20	5
Geom. std. dev.	4	4	4
Maximum	500	100	20
Minimum	20	20	0.2

Deposition velocities were chosen to reflect the effect of particle size. A value of 0.001 m/s was used for the smoke cloud, while 0.1 m/s and 0.05 m/s were used for the large- and small-particle populations, respectively, in the stem.

Mass balances for the several SAI tests showed that an estimated 13.4% of the total Be was apportioned to the smoke cloud, while 6.6% was in dust fallout; for DU, the apportionment to smoke and dust were 19.2 and 11.1%, respectively. Assuming that all the dust was in the stem and that twice as much was in large- as was in small-particle populations, we specified the source for a typical XM753 test to be as given in Table 4.

Table 4. Source-Term Data Specified in the Model (grams)

	Be	DU
Smoke cloud	134	4800
Stem (large particles)	44	1850
Stem (small particles)	22	925

Meteorological data were selected to characterize a typical daytime situation in the Southwest. A wind speed of 5 m/s was used at the 10-m height, increasing to 9 m/s at 300 m. The stability category was set at C (slightly unstable), given the moderate wind speeds. The wind direction was assumed to be constant with height. Although observations commonly show some variation of wind direction with height, the assumption of constant direction is a conservative one because directional wind shear enhances diffusion.

The parameters outlined above were used for the simulations reported here, and are thought to be typical of XM753 testing conditions. Additional runs were made with varying source and meteorological parameters to test model sensitivity. The results will show conservative values as the amounts of Be and DU are about one-half those in the XM785 projectiles, for which this report is prepared.

C. Results

The MATHEW-ADPIC model was run to simulate the atmospheric transport, diffusion, and deposition of Be and DU released during a typical XM753 test. The simulation period was 150 min to allow marker particles time to travel across the large model domain. Data were saved every min for the first 12 min (on the small model domain), and then at 6-min intervals thereafter. These data included values for ground deposition and integrated air concentration at multiple heights.

1. Ground Deposition of Be and DU

Simulated fallout plates were placed at the surface in the model domain at various downwind distances along the centerline of the cloud track, where maximum deposition would occur for given distances. These maximum values for Be and DU are shown in Figure 11. A line created by visual approximation is fitted to each set of simulated data. Also included are measured fallout-plate data for SAI Test #7. At close-in distances, the model values fall below the measured data; it is thought that these measurements may include some ejecta material that is not included in the model source term. Beyond 80 m, the maximum model values are slightly larger than the measured data, which is expected. This model-data comparison gives credibility to modeled deposition values beyond the range of available data.

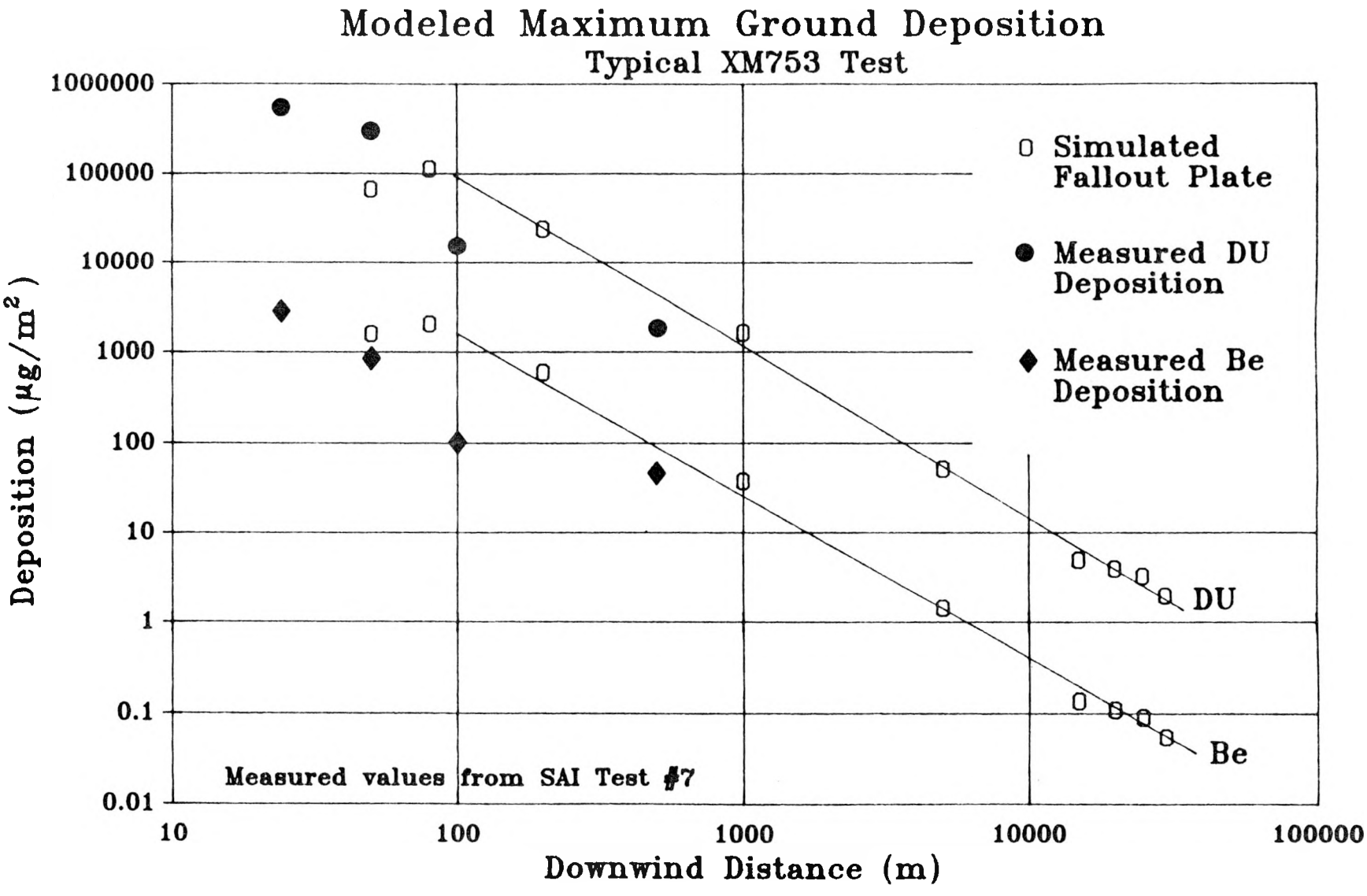

Figure 11. Maximum downwind deposition calculated versus observed at YPG.

Ground-deposition information is particularly useful in evaluating the impact of resuspension on air quality (discussed later; section V.A.3). For the air-quality standards governing Be and DU, the model results indicate that wind-driven resuspension of material from a single XM753 test will be a factor of 50 lower than the limiting value at any distance downwind. For early vehicle traffic, resuspended DU will be a factor of 10 lower than the limit, while resuspended Be will be at the limit at 80 m downwind (where the peak Be deposition occurs). At 300 m downwind, the resuspended Be will be a factor of 10 below the limit, and a factor of 1000 below the limit at 4 km downwind. It must be remembered that these model values are maximum values along the centerline; this implies that vehicle traffic would have to drive down the centerline to encounter these values. Also, these values are calculated for the XM753, a projectile containing more Be and DU than the XM785.

2. Averaged Air Concentrations of Be and DU

Short-term-average air concentrations of Be and DU were calculated for breathing height (1.5 m). Values for Be are shown in Figure 12, along with the 30-min threshhold limit value (TLV) of 25 $\mu g/m^3$. Modeled values remain a factor of 10 below the TLV at every distance downwind. Luna et al. (1983) report a measured value of 0.01 $\mu g/m^3$ in Yuma, about 37 km downwind of the test area; the 30-min average Be values from the model agree quite well with that measurement. The 0.01 air quality 30-d emission standard for Be refers to a much longer averaging time, however, and thus the 0.01 value here should not be confused with that standard; for example, the 24-h average Be concentrations for a typical XM753 are a factor of 50 lower than the 30-min averages shown here. The 15-min average DU concentrations from the model are shown in Figure 13, along with the 15-min TLV; here the modeled values are at least a factor of 10 below the TLV at all downwind distances.

Maximum air concentrations averaged over shorter durations within the simulated smoke cloud are shown in Figure 14. The cloud center was at 75 m above ground, while the initial cloud base was at 50 m. Values from the helicopter air sampling of the smoke cloud for SAI Tests #4 and #7 are also shown. The modeled maximum values are larger than the measured values; this is expected because the helicopter-sampling passes included material from regions throughout the smoke cloud, not just the center. The assumptions of vertical distribution are reasonably confirmed, however.

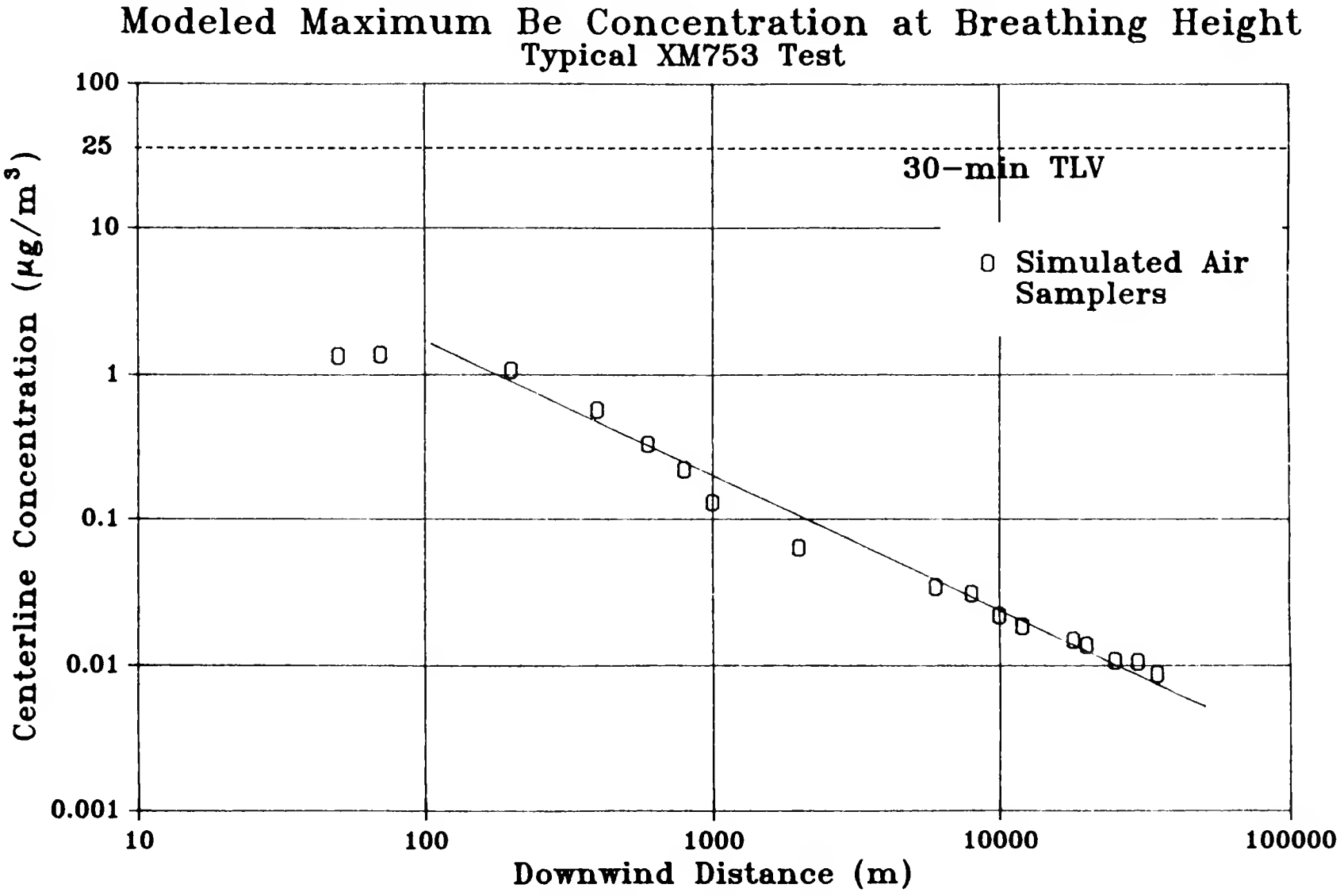

Figure 12. Maximum 15-min average concentrations of Be compared to the 30-min guideline.

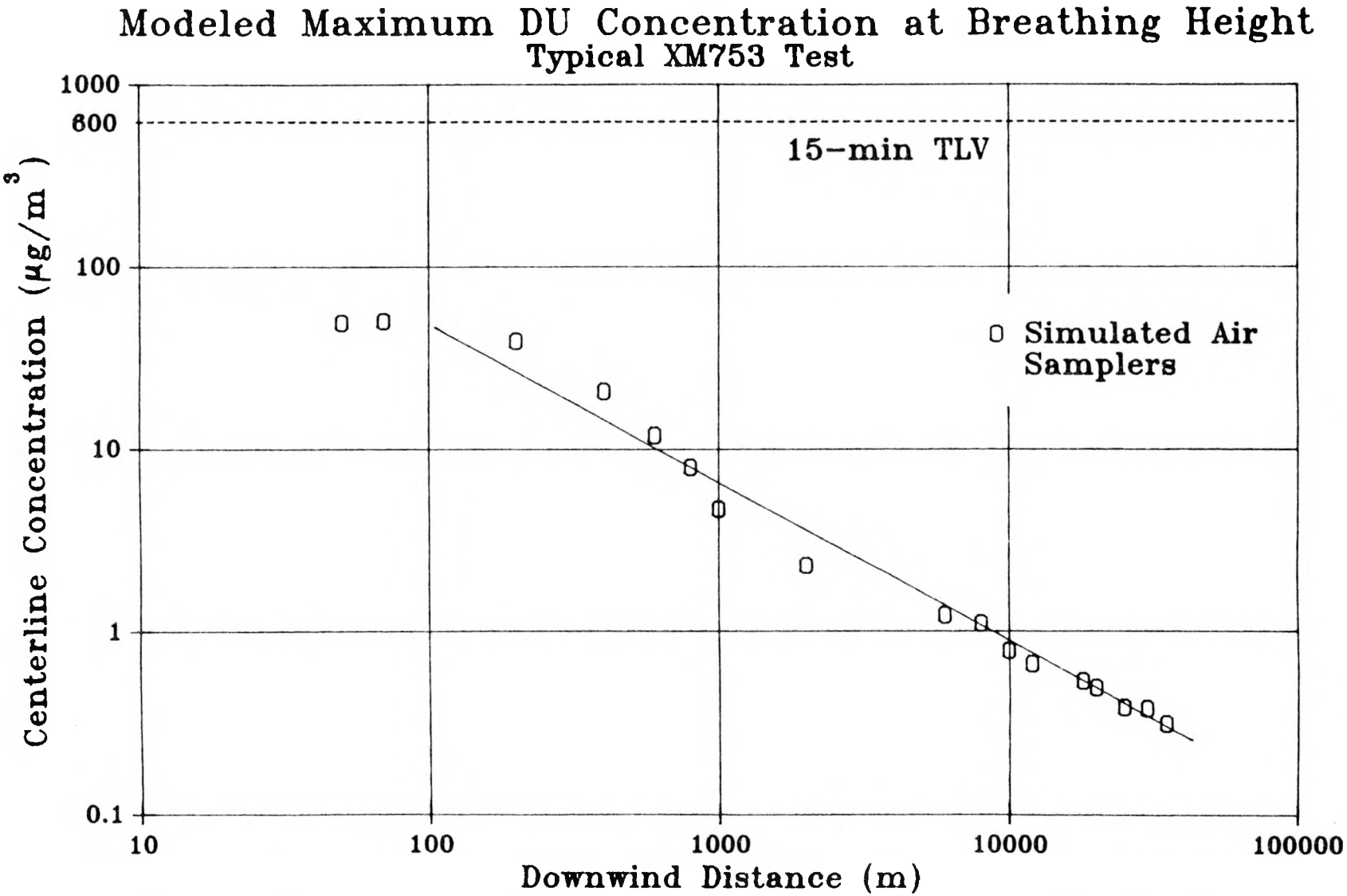

Figure 13. Maximum 15-min average concencentration of DU compared to the 15-min guideline.

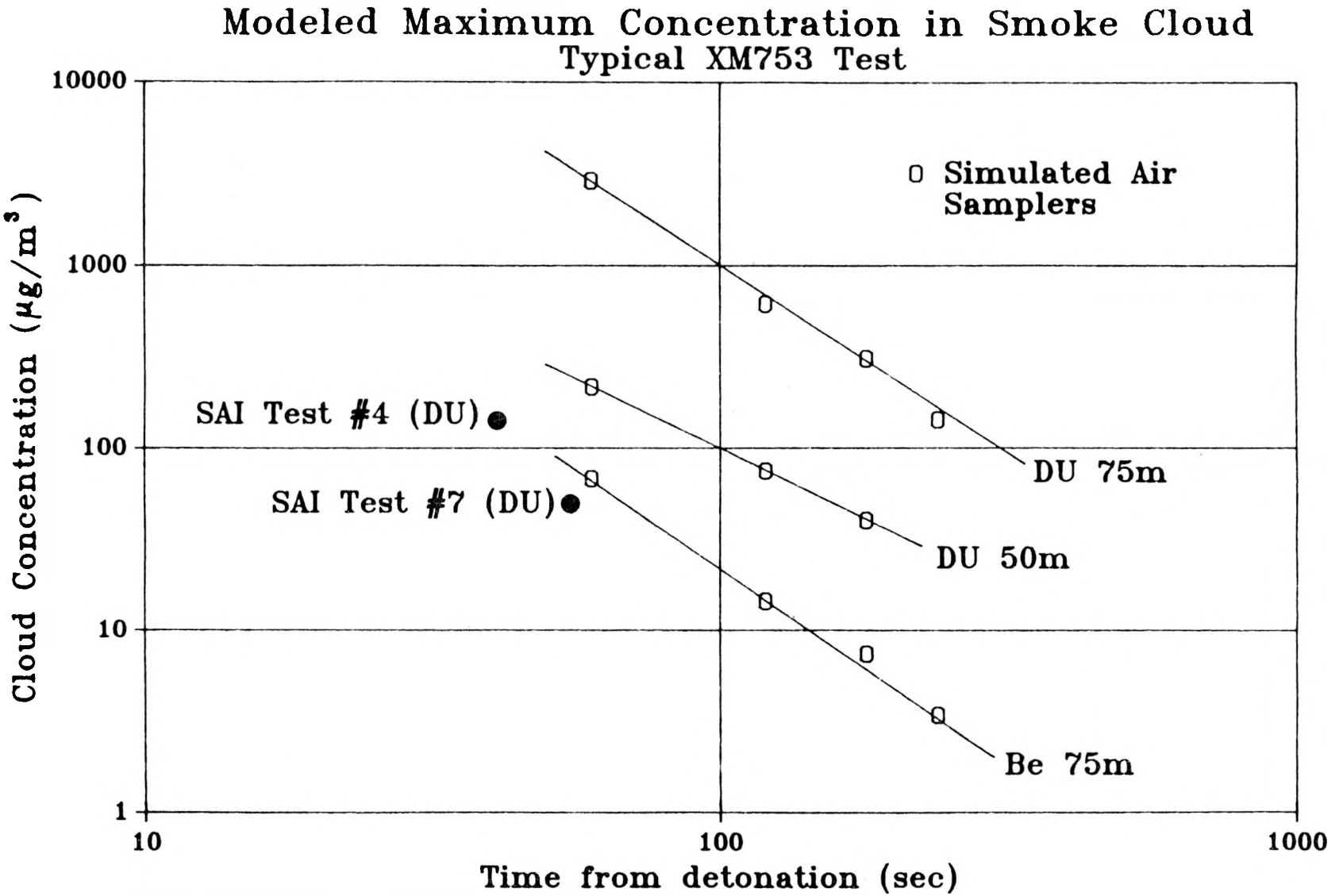

Figure 14. Maximum DU concentrations at different heights in the cloud compared to observations at YPG.

In general, the modeled maximum values compare well with observed data, being larger than the data in most cases where data are not from the cloud centerline. With this confidence in the model, especially that it is conservative in its estimates, we conclude that air concentrations from a single XM753 test, and therefore a XM785 test, are well below governing standards.

IV. The Dispersion from a Single XM785 Test at Tonopah, Nevada

Our investigation not only included modeling of dispersion, but also an examination of downwind dispersion from a test unit. The test unit, an XM785-type device, was detonated on the ground at Tonopah Test Range, Nevada. Although this was a single test, it serves as a validation and scale exercise.

Measurements were made of airborne particle concentration, particle deposition, and particle-size distribution of suspended aerosols. A 200-m by 500-m area was surveyed with the major axis north-south on a dry lake (playa) at Tonopah Test Range (TTR). This area was gridded into 100-m squares so that samplers could be placed in a graduated pattern downwind of the detonation point (ground zero).

A. Methodology

Particle concentrations were measured with high-volume air samplers (85 m^3/h) and cascade impactors (34 m^3/h). The time of exposure of each sampler during the presence of the smoke and dust cloud was determined by an optical scattering device (integrating nephelometer). The particles were collected on cellulose fiber filters (Whatman-41). Particle-size distributions were determined from the 5-stage, jet-type cascade impactors (Anderson 2000, Inc.). Filter media were conditioned in heated, dry rooms 72 h before weighing to avoid moisture retention errors. Passive samplers were also used to obtain a wider spatial distribution of concentration measurements. The passive sampling employed cheesecloth material held in plastic hoops, 20 cm in diameter. The hoops were attached to stakes from hanging swivels at the 1.2-m height. The stakes were separated by 25 m in two lines perpendicular to the plume centerline at the same downwind distance as the high-volume air samplers and cascade impactors--150 m and 300 m downwind from ground zero. The cheesecloth samplers were efficient for particle capture even though passively ventilated. They were used mainly to determine plume cross section in a relative sense and to fill in between absolute air samplers. They have been used successfully on similar tests at LLNL. The cheesecloth samplers were also suspended from the tether line of a 7-m^3 airfoil balloon. Cheesecloth samplers were flown to a height of 32 m at 300 m downwind of ground zero on this test; additional heights were attempted, but

lost due to balloon and tether failures. After the test, the cheesecloth was cut from the hoop and retained for a later acid digestion and analysis.

Deposition measurements of particles suspended and dispersed downwind of the shot were made by an array of sticky plates mounted at a height of 0.35 m. Each plate was a square, light aluminum plate with an inner area (0.093 m^2) lightly coated with white petroleum jelly and a 2.5-cm margin to border the sticky area. The sticky plates were placed in a graduated grid density, starting at 25-m spacing near the ground zero and increasing to 50-m spacing and to 100-m spacing outward from the ground zero. Eighty-five of these plates were positioned from 150 m upwind to 850 m downwind of the shot and out to 250 m on either side of the downwind centerline. After exposure, the sticky plates were placed in a box with slots to prevent them from touching each other and the walls of the box. After transporting the box to the laboratory, the greased surface was washed with toluene in a hood with the aid of a disposable brush. The toluene solvent and trapped particles were eluted into a large plastic funnel and through a filter assembly (0.3-μm cellulose fiber filter, 47-mm diameter) while the toluene and dissolved grease were drawn into a collection flask under low vacuum. These filters were air dried and dissolved in a mixture of nitric and perchloric acid prior to analysis.

Chemical analysis was performed on air filters, cheesecloth, and deposition samples by infrared fluorimetry with a lower limit of detection of 0.05 μg Be and 0.25 μg DU in a certified commercial laboratory (Radiation Detection Company, Sunnyvale, California). Quality assurance for Be and DU was provided by adding blanks and samples spiked with NBS-traceable standards to the samples submitted by LLNL for analysis.

Meteorological measurements were made to determine the trajectory of the smoke and dust plume through the array of instruments and to determine the wind speed and atmospheric stability. Two automatic weather stations were placed at 150 m and 300 m downwind of the ground zero. These stations continuously recorded on cassette magnetic tape the averages of wind speed, wind direction, air temperature, and temperature gradient. A meteorological sounding system, consisting of a tethered, 5.6-m^3, airfoil balloon carrying an instrument package, was raised to nearly 300 m and lowered prior to and

immediately following the shot. The instrument package measured and transmitted the wind speed, wind direction, temperature, and pressure altitude. This sounding system was located about 1600 m upwind of the shot at the firing control point and was used to start the shot countdown when the wind direction aligned with the downwind array centerline.

B. Results

1. Meteorological Conditions

The shot was detonated at 10:36 a.m. local time on August 6, 1987, at Tonopah Test Range. The average wind speed was 5.7 m/s (12.8 mph) at a height of 3 m during the test. The wind speed and direction were persistent during the test; from a half-hour preceding to a half-hour following detonation the mean speed remained within 20% of the test average 95% of the time and wind direction remained within 25° of the centerline 95% of the time. The observed meteorological data and derived scaling parameters at the 3-m height and 150 to 300 m downwind of ground zero are given in Table 5.

Table 5. Meteorological Data at the 3-m Height During the Test

Wind speed	5.7 m/s
Wind direction	194°
Temperature	28°C
Friction velocity (u*)	0.7 m/s
T* (dT/d $\log_e$ Z)	-1.0°C
Temperature gradient	-0.33°C/m
Monin-Obukhov length	-53 ± 15m
Stability class	D

Analysis of the sounding data showed that windspeed was decreasing with height to a minimum near the 50-m height and then increasing with height; however, the speeds were generally close to the mean observed on the surface (Table 5). Wind direction was turning clockwise with height until about the 50-m height. Temperature profiles showed that surface heating was producing a strong lapse rate and a major increase in layer temperatures between soundings (3°C). See Table 6.

Table 6. Meteorological Sounding Data Before and After the Tests

Pre-Shot (0855-0930 LT) Average Height (m)	Average Temp (°C)	Average WS (m/s)	Average WD (true)
7.6	26.8	5.8	172.8
32.9	25.7	4.0	222.0
81.0	25.2	6.1	169.1
105.1	24.7	6.5	167.7
139.4	24.4	5.9	172.8
169.2	23.9	8.2	158.6
Post-Shot (1105-1145 LT) Average Height (m)	**Average Temp (°C)**	**Average WS (m/s)**	**Average WD (true)**
5.5	30.1	5.3	179.2
12.6	30.0	5.3	174.8
34.1	28.3	4.7	189.2
62.4	27.9	6.7	227.1
97.7	27.1	6.1	236.4
126.6	26.9	5.7	233.6
145.5	27.1	4.9	266.3
168.4	26.6	6.6	190.4
213.9	26.1	5.4	218.0
233.6	25.9	5.5	200.3
262.0	26.0	6.1	177.8
281.2	25.5	6.9	230.0

2. Be and DU Concentrations Downwind

Particle concentrations were measured in the plume centerline, since by good fortune, the passive cheesecloth samplers and deposition plates all showed that the maximum values were very near to the locations of the high-volume air samplers and cascade impactors at 150 m and 300 m downwind of the ground zero. The nephelometers indicated that the dust and smoke was present for 99 s at 150 m from ground zero, and 33 s at 300 m from ground zero (Figure 15). The peak concentration indicated for the nephelometer at 300 m was obtained by comparison with the high-volume air sampler, assuming that the measured bulk scattering coefficient was proportional to DU concentration.

The maximum Be and DU integrated exposures and 15-min average concentrations are shown in Table 7. Because the smoke and dust cloud was present for only a brief period of time, the peak concentrations (e.g., Figure 15) integrate to very small exposures. The 15-min maximum concentrations of Be should be compared to the 30-min TLV of 25 $\mu g/m^3$ and the 8-h guideline of 2 $\mu g/m^3$; the values observed are orders of magnitude below these Be standards. Extrapolation shows that the Be concentration averaged for 15-min would be lower than the 0.01 $\mu g/m^3$ emission standard (30-day average) at a distance about 2 km from the detonation. Hence, it would take 2880 identical tests per month (30 days x 24 h/day x 4 periods/h) to exceed this standard at 2 km from ground zero.

Table 7. Maximum (Centerline) Integrated Exposures and 15-min Average Concentrations of Be and DU Observed Downwind at Tonopah Test Range at 1.2-m Height

Downwind Distance m	Plume Presence s	DU Exposure μg-min/m^3	DU (15 min) Concentration $\mu g/m^3$	Be Exposure μg-min/m^3	Be (15 min) Concentration $\mu g/m^3$
150	99	168	11	2.0	0.13
300	33	265	18	1.0*	0.066*

*Values given were observed at 12-m height, none found at 1.2 m.

The 15-min maximum concentrations of DU should be compared to the 15-min TLV of 600 $\mu g/m^3$; the values observed are more than 30 times lower than this standard. It is not possible to extrapolate downwind from the observed DU data, but at a distance of 2 km the 15-min concentration would be lower than 6.9 $\mu g/m^3$ (3 x 10^{-6} $\mu Ci/m^3$), the more stringent air quality standard.

We have high confidence in the data (Table 7) because the high-volume and cascade impactor air samplers agreed with each other, with the deposition patterns, and with the cheesecloth samplers placed on stakes. The vertical

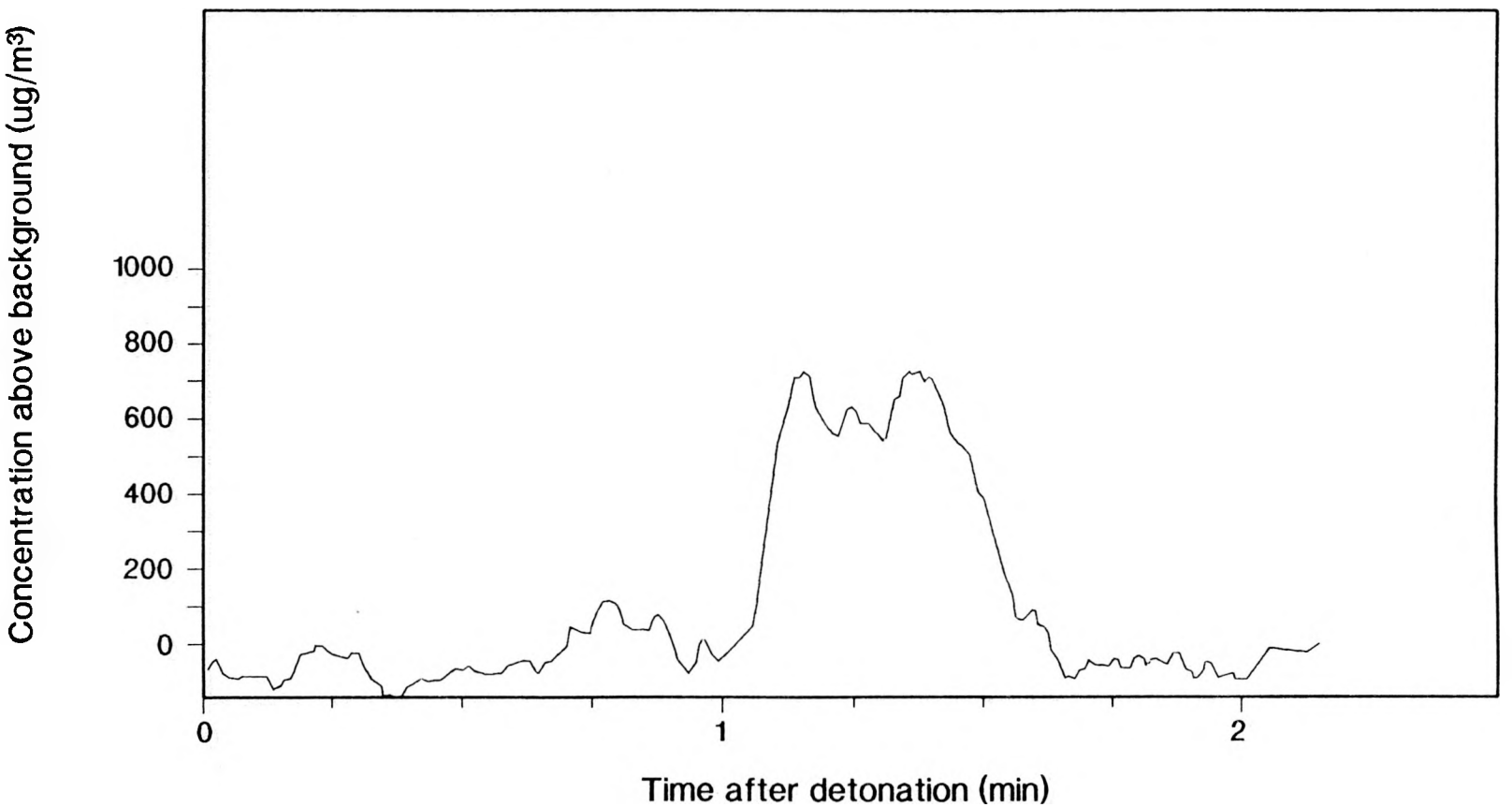

Figure 15. Atmospheric concentration of DU observed 300 m downwind on centerline at TTR.

distribution of material collected on the passive cheesecloth samplers at 300 m downwind showed that the concentration was a maximum near 12 m and decreased with height above that (Table 8). These data are converted to concentration by comparison with high-volume samplers; this is strictly an approximation.

Table 8. Vertical Distribution of 15-m Concentrations in the Plume Centerline at 300-m Downwind

Height m	DU (15 min) Concentration $\mu g/m^3$	Be (15 min) Concentration $\mu g/m^3$
1.2	18	ND
2	2.9	0.044
12	9.7	0.066
22	1.7	ND
32	1.2	ND

3. Deposition Downwind

The deposition data from the sticky plates had a better yield for DU than for Be; hence, it is more useful to discuss the results for DU. The maximum DU deposition (greater than 300 $\mu g/m^2$) occurred in a small area 50 to 100 m downwind of the ground zero; see Figure 16. A smoothed and interpreted distribution of the DU deposition data is shown in Figure 17. The deposition of Be is estimated to have the same pattern. We investigated the ratio of DU/Be and found that it had no systematic variation with distance downwind in the center of the array. The geometric mean of the ratio DU/Be was 109 with 70% of the samples within a factor of two of this ratio.

The area that was downwind of the ground zero and that encompassed deposition greater than 300 $\mu g/m^2$ DU and greater than 3 $\mu g/m^2$ Be was approximately 2000 m^2 (0.5 acre); see Figure 17. The maximum deposition observed was 1400 $\mu g/m^2$ DU and approximately 13 $\mu g/m^2$ Be. These maximum values are not significantly large to warrant remedial action. If we use a resuspension factor of $10^{-5}/m$, appropriate for (worst case) vehicle traffic over the site, the estimated air concentrations would be 0.014 $\mu g/m^3$ DU and

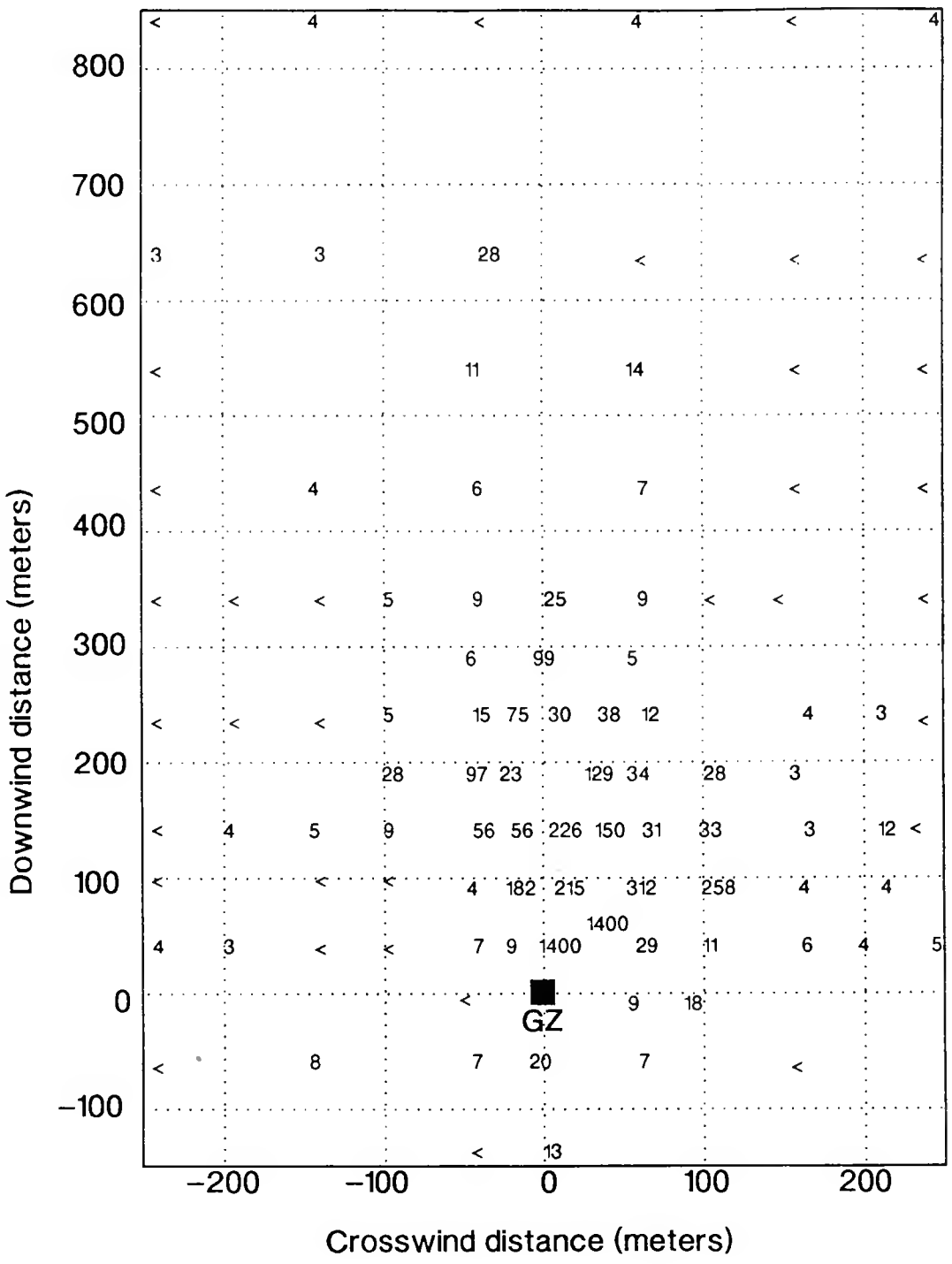

Figure 16. DU deposition ($\mu g/m^2$) in area downwind of ground zero. The "less than" symbol indicates a sample collected was below the detection limit.

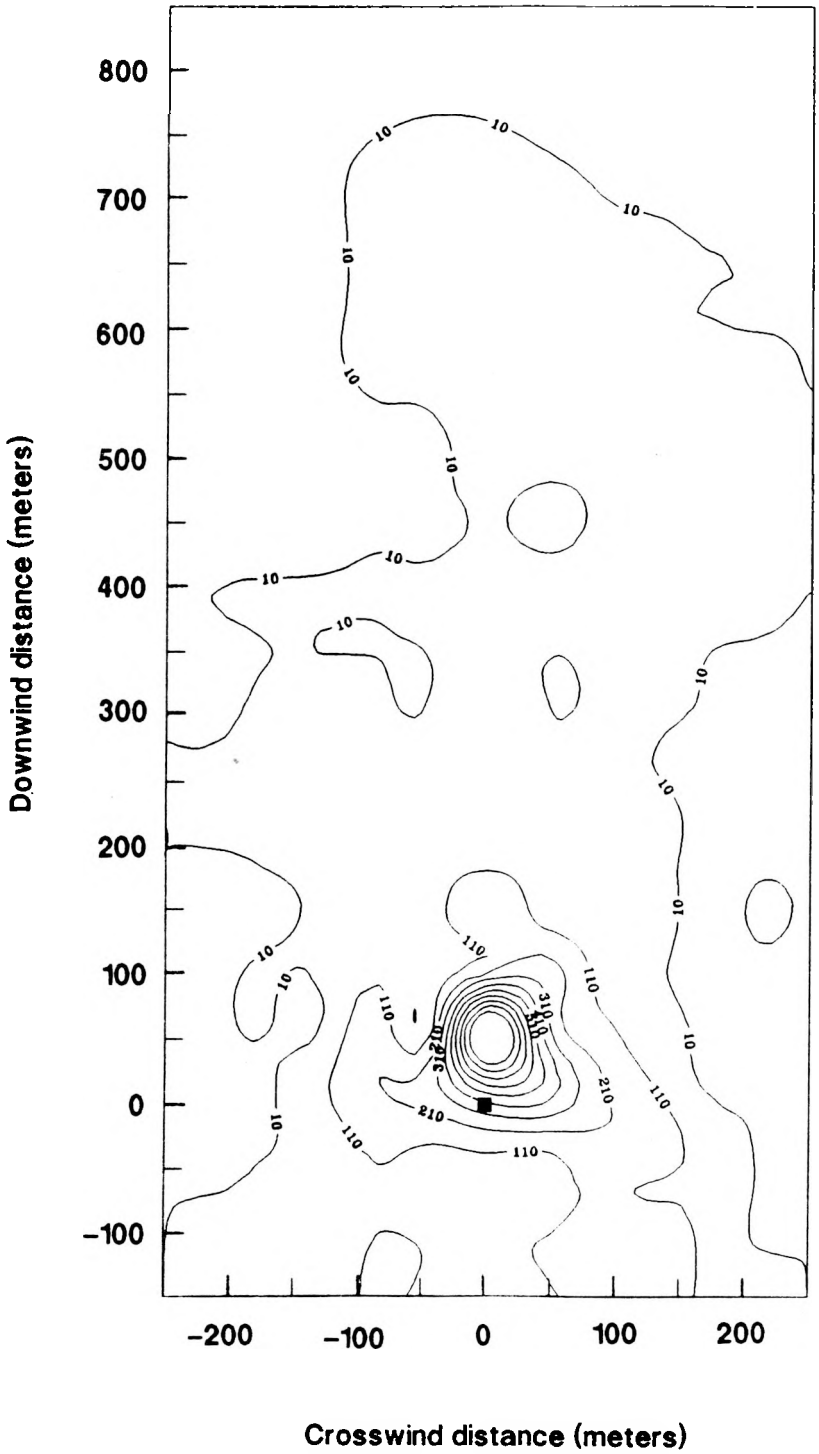

Figure 17. Contours showing interpolated DU deposition ($\mu g/m^2$) downwind at TTR.

0.00013 $\mu g/m^3$ Be due to traffic. These are not significant values as potential long-term sources of pollutants compared to the stringent air quality standards of 6.9 $\mu g/m^3$ DU and 0.01 $\mu g/m^3$ Be.

4. Particle-Size Distributions

The cascade impactor data were analyzed to obtain size distributions of suspended DU and dust. Not enough Be was collected to quantify. We found that the median aerodynamic diameter of DU particles was 3.8 μm at 150 m downwind and 2.0 μm at 300 m downwind. The distribution of DU was quite broad with a geometric standard deviation of 5.3 in both cases. This means that 68% of the DU was contained within an approximate range of 0.5 μm to 15 μm and 84% was less than 15 μm. The dust briefly generated by the detonation was 6300 $\mu g/m^3$ for 99 s at 150 m and 500 $\mu g/m^3$ for 33 s at 300 m. The background dust concentration was 56 $\mu g/m^3$. This dust had a median aerodynamic diameter of 2.6 μm at 150 m and 4.6 μm at 300 m. The dust had a relatively narrow size distribution, however, with a geometric standard deviation of 1.5 in both cases. This means that 68% of the dust was contained within an approximate range of 2 μm to 5μm.

5. Comparison of Observations with Model Predictions

Our model for computation of downwind deposition and air concentration is dependent upon apportionment of the Be and DU into recovered parts (large fragments), ejecta (small fragments), smoke, and dust (lofted in the cloud stem). The apportionment data come from the tests with XM753 and XM785 at Yuma Proving Ground. The comparison between recovered parts at Tonopah Test Range and the previous experiments shows that this test was unusual. In this test, 85% of the Be and 18% of the DU were recovered as large fragments. That was about 260% of the expected recovery for Be and 27% of the expected recovery for DU. We estimate that only about 20 grams of Be may have been aerosolized.

Without changing the usual model parameterization for this test, we found that our model overpredicts the deposition and the downwind concentration of DU and Be.

The deposition of DU along the centerline of the cloud was smoothed by averaging the two largest, observed deposition values in each cross section of the plume path (Figure 16). Our model predicted values that were 330 times greater than those observed for DU deposition and 100 times greater than those observed for Be deposition. The predicted rate of change of deposition with distance, however, was about the same as observed.

Our model overpredicted the downwind concentrations also. The ratio of predicted to observed concentration was about 3 for DU and about 100 for Be. Although the model did better for DU than Be concentration, we can only speculate why. Perhaps most of the unrecovered DU was in the form of small fragments (ejecta) rather than in the smoke and dust, compensating for the misapportionment.

Because the model did well when compared against tests at Yuma Proving Ground, and the fraction of recovered metal parts was different from previous experience, there is no reason to change the model. It served to make estimates that were conservative in this case, in terms of protecting human health and the environment.

C. Summary of XM785 Test at Tonopah

This single test provided an additional set of data for comparison, but proved to produce very little aerosolized Be. The detectable smoke and dust cloud persisted for 99 s at 150 m and even less time further downwind in this test where the wind speed was 5.7 m/s. The 15-min average concentrations in the plume centerline were far below the recommended short-term exposure limits for Be and DU. We observed 11 to 18 $\mu g/m^3$ (15-min average) DU at 150 to 300 m downwind, which should be compared with 600 $\mu g/m^3$ TLV; we also observed 0.13 to 0.066 $\mu g/m^3$ (15-min average) of Be at 150 to 300 m downwind, which should be compared with the 25 $\mu g/m^3$ recommended 30-min short-term exposure guideline. We determined that at 2 km downwind, the concentrations for 15 min did not exceed the stringent guidelines of 6.9 $\mu g/m^3$ DU and 0.01 $\mu g/m^3$ Be.

The maximum deposition observed was 1400 $\mu g/m^2$ for DU and 13 $\mu g/m^2$ for Be at a distance of 50 m from the detonation point. These are levels far below the criteria for remedial action based on vehicular resuspension considerations. The major deposition, greater than 300 $\mu g/m^2$ DU and 3 $\mu g/m^2$ Be, occurred within a small area of approximately 2000 m^2 (0.5 acre).

The test was unusual for XM785 tests in that most of the Be was recovered as metal parts, and the apportionment of metal into the components of parts, ejecta, smoke, and dust was different from our normal experience. We estimated that only about 20 grams of Be might have been aerosolized. Consequently, our model predictions of downwind concentrations and deposition were far higher than normal, but the error is on the conservative side in terms of effects on human health and the environment.

V. Environmental Consequences

A. Soil

1. Background Concentrations

The background levels of Be and DU in YPG soils are similar to those normally found elsewhere. Luna et al. (1983) gave values for Be of 0.96 μg/g and for DU of 2.27 μg/g. Examination of the soil-sample data shows the ratio of the standard deviation to the mean for these determinations is about 0.4. Thus, the range 0.6 to 1.4 μg/g for Be, and 1.4 to 3.2 μg/g for DU, would be expected for 68% of the cases. In an example, Luna et al. show that at ground zero on the XM753 tests, there was a post-test increase of 0.4 μg/g of Be and 2.7 μg/g of DU. Considering natural variability, these are not statistically significant increases due to deposition from one round. If it is true that even in the worst case at ground zero there will not be major increases in Be and DU, it is doubtful that the use of any practical soil-sampling strategy could result in detection of significant changes from one round. The purpose of soil monitoring would be to look for cumulative and regional effects. Because the KOFA Firing Range at YPG is used heavily (50,000 rounds were impacted in 1974 alone), it is safe to assume that it is already environmentally affected by many other contaminants and that a higher degree of ecological protection is not warranted.

For comparison purposes, the background soil levels at other Be/DU explosive-testing facilities, i.e., at Livermore, California and Moronvilliers, France, are the same within normal variation of values reported for YPG. Long-term air monitoring at Livermore shows that the background average air concentration is 5×10^{-5} μg/m^3 DU and 2×10^{-5} μg/m^3 Be (Griggs et al., 1985). These values can be entirely explained by the process of wind-driven resuspension.

Extensive air monitoring at Moronvilliers, France shows that the Be background air concentration varies over a range of 5×10^{-6} μg/m^3 to 5×10^{-4} μg/m^3; this variation is attributed to increased farming activities during certain seasons of the year (Gros, R., 1987, personal communication, French Atomic Energy Commission, Bruyeres-Le-Chatel, France). Seasonal variation at Livermore shows a minimum in the wet season (April) and a maximum

in the dry season (June to July). Because varying human activities and wind erosion can cause different levels of dust aerosol, the same factors would affect the background air concentrations of Be and DU at YPG.

2. Surface Soil Deposition

Luna et al. (1983) were concerned about the environmental effects of XM753 tests at YPG. Their best estimate of maximum total ground deposition of respirable particles (less than 10-μm aerodynamic diameter) for a groundburst was 8 x 10^3 μg/m^2 Be and 4 x 10^4 μg/m^2 DU; the maximum deposition would occur in the immediate vicinity of the crater at ground zero. These values agree very well with our model results for concentrations less than 80 m from the ground zero (see Figure 11). These depositions are well below the criteria that are derived later for maximum deposition allowable to protect against wind-driven resuspension. Vehicular traffic should be avoided near the crater.

Measurements of Be at Moronvilliers, France (Gros, 1987, ibid.) and Livermore, California (Shinn, 1987) for groundbursts of similar magnitude to those evaluated by Luna et al. (1983) for YPG, show that the deposition of respirable Be decreased to less than 10 μg/m^2 within 4000 m of ground zero. The observed values agree with those calculated by Luna et al. (1983) for the XM753. Our model results for the XM753 at Yuma Proving Ground show that deposition is about 3 μg/m^2 Be and about 100 μg/m^2 DU at 4000 m (see Figure 11). This gross agreement gives confidence that any single round will not produce major changes in the soil concentration. The deposited contaminants will be weathered into the soil by rainfall. The precipitation of 8.5 cm/year for YPG is not large, but over the years will have a slow leaching effect. At a desert site in Nevada, the weathering-in of surface deposited radionuclides has occurred to a depth of about 5 cm in 20 to 30 years. This effect has reduced the resuspension factor in proportion to $1/t^{1/2}$ according to Anspaugh et al. (1975).

Airburst shots present a different perspective because the maximum deposition occurs at some distance downwind of ground zero. Luna et al. (1983) estimated maximum deposition of respirable Be of 9 μg/m^2 at 4000 m and respirable DU of 46 μg/m^2 at 400 m for an airburst of the XM753.

SAI found that for static tests (groundbursts), the highest concentration of Be was found close to the crater, as would be expected. "Large" particles (undefined by SAI) were found at 25 m, but not at 50 m. "Large" pieces (again undefined) were found up to 180 m from the blast. Fallout was detected at 1000 m downwind of the blast. At this distance, the mass median diameter of the particle was >50 μm, but it was undetermined if these were particles of Be or if they were particles of Be agglomerated to soil particles. Recoverable material depends somewhat on the type of soil in which the projectile detonates. The JB3 series was shot in rocky soil, leading to a smaller recovery rate, whereas subsequent testing was done in "soft" targets (Table 9; Mishuck and Hartung, 1981). Missing material is estimated to consist of particles from 1 mm to 1 cm in diameter; this component is that designated as ejecta in Tables 1 and 2.

Table 9. Recovery of Materials

	Mean	Std. Dev.
JB3 Series	33.1%	15.15%
JFF4 Series	48.3%	33.9%

SAI found that of the 360 pre- and post-shot soil samples taken for the JB3 series and JFF4 series, only 8 pre- and 16 post-shot samples and 6 pre- and 2 post-shot samples, respectively, exceeded 3 μg Be/g soil. No data were presented on samples containing less than 3 μg Be/g soil.

Recoverable parts and ejecta make up approximately 80% of the Be according to the normalized mass balance conducted by SAI. Missing material was thought to be close to the crater, so additional sampling was conducted. A large percentage of material was found in this inner zone with the particles being in the size range of 0.1 to 2 mm (Table 10). As we have seen from Section IV, uncertainty in apportionment of material into recovered parts and ejecta directly causes the largest uncertainty in modeling.

Table 10. Be Found in Inner Soil Samples by Screen Size

Beryllium (μg/g)	Screen Size (mm)
666	2
362	1
26.7	0.6
7.1	0.3
7.0	0.15
2.7	0.075

A well thought-out implementation plan will need to be followed to prevent buildup to hazardous levels of Be in an area. If all shots are confined to a limited area, reclamation would need to be conducted if levels become excessive. SAI recommends that cleanup should occur when DU concentrations reach 500 μg/g soil. If this is not a viable option, then new target zones would need to be used each time. Criteria for reclamation can be derived from considerations of resuspension of respirable particles.

3. Resuspension

The aerosol concentration ($\mu g/m^3$) due to the resuspension of previously deposited contaminants can be estimated by means of the resuspension factor, k (m^{-1}), multiplied by the deposition amount ($\mu g/m^2$). In a previous study of the YPG dispersal problems by Luna et al. (1983), a thorough application of the resuspension concept was made. The sources of resuspension factor data are the numerous experiments conducted by the Department of Energy and its predecessor agencies. These have been reviewed extensively by Healy (1980). The concept also was applied to establish criteria for remedial action for contaminated soil by Anspaugh et al. (1975).

Practical scenarios of the physical processes are (1) wind-driven resuspension, (2) forced resuspension, such as by vehicular traffic, and (3) detonation resuspension. Luna et al. (1983) chose representative resuspension factor values of (1) 10^{-7}/m for wind-driven resuspension at YPG, and (2) 5×10^{-6}/m for early vehicular traffic. These values are reasonable considering the conditions of the proposed tests and values given in Healy (1980). For detonations on the same site of previous deposition, it is

estimated that resuspension is insignificant based on the small quantity of DU and Be available. These are expected values of the resuspension factors chosen from reasonably similar conditions to YPG. The weathering of deposited material into the surface soil produces a well-mixed layer on the top due to thermal and hydrological cycles. In desert soils, 95% or more of these refractory materials will be in the top 5 cm of soil after twenty years (Anspaugh et al., 1975). Given the most stringent air quality standards of 0.01 $\mu g/m^3$ for Be and 6.9 $\mu g/m^3$ (3 x 10^{-6} $\mu Ci/m^3$) for DU, we estimated the Maximum Deposition Allowable (see Table 11).

Table 11. Criteria for Maximum Deposition Allowable

Wind-driven resuspension	
$(0.01\ \mu g/m^3)/(10^{-7}/m) = 1 \times 10^5\ \mu g/m^2$	Be
$(6.9\ \mu g/m^3)/(10^{-7}/m) = 6.9 \times 10^7\ \mu g/m^2$	DU
Early vehicular traffic	
$(0.01\ \mu g/m^3)/(5 \times 10^{-6}/m) = 2 \times 10^3\ \mu g/m^2$	Be
$(6.9\ \mu g/m^3)/(5 \times 10^{-6}/m) = 1.4 \times 10^6\ \mu g/m^2$	DU

The maximum predicted Be deposition of 10^4 $\mu g/m^2$, if allowed to weather-in to 5 cm, would cause an increase of Be much less than the background concentration, which is approximately 1 $\mu g/g$, using a bulk density 1.5 g/cm^3:

$$(10^4\ \mu g/m^2)[1/(5\ cm)](cm^3/1.5\ g)(m^2/10^4\ cm^2) = 0.13\ \mu g/g.$$

Thus, the deposition of 10^4 $\mu g/m^2$, which only occurs near ground zero, is not a major change in Be soil level, and the deposition decreasing to 10 $\mu g/m^2$ at 4000 m is inconsequential. Maximum predicted deposition of 10^5 $\mu g/m^2$ of DU, likewise would increase the soil level near ground zero by 1.3 $\mu g/g$, which is also less than the background concentration, 2.3 $\mu g/g$, for uranium. Deposition decreasing to 100 $\mu g/m^2$ at 4000 m is thus inconsequential for DU also.

In other words, if vehicular traffic is prohibited, remedial action should not be necessary unless the cumulative deposition from overlapping fallout of many shots exceeds 0.1 g/m^2 of Be or 69 g/m^2 of DU.

B. Air Quality

The air concentration of Be and DU will be elevated for only a brief period following each shot. Air concentrations decline exponentially with time. For example, in the elevated cloud, the air concentrations have decreased two orders of magnitude within 5 min for the typical XM753 detonation shown in Figure 14. The maximum respirable air concentrations (cloud centerline) at breathing height are 2 $\mu g/m^3$ for Be and 50 $\mu g/m^3$ for DU as predicted by the model and averaged over the 5 min of cloud passage; see Figures 12 and 13. These concentrations would occur only on the centerline and within 100 m of the ground zero for this brief period of time. Furthermore, these values are more than an order of magnitude lower than the guidelines for short-term exposures: 30-min TLV of 25 $\mu g/m^3$ for Be (OSHA) and 15-min TLV of 600 $\mu g/m^3$ for DU (ACGIH). Exposures of consequence are avoided by the effects of both the factor of ten lower concentration and the shorter time of cloud presence.

At LLNL's Site 300, where tests release Be to the environment, certain standards have been established to maintain safe levels for on-site personnel. Soil levels as high as 500 μg of Be/g of soil are permissible in areas where Be operations are in progress, provided average airborne concentrations do not exceed 2.0 $\mu g/m^3$ and short-term (30 min) concentrations do not exceed 25 $\mu g/m^3$. For areas where no Be operations are being conducted, a level of 100 μg Be/g of soil is permissible as long as average airborne concentrations do not exceed 0.2 $\mu g/m^3$ and short-term concentrations do not exceed 25 $\mu g/m^3$ (Johnson, 1980).

Within 100 m of the crater, surface density levels are increased by deposition and debris fallout. The highest level of deposition recorded at 500 m for any of the SAI tests was 100 times less than the background level. Safety precautions should be followed within 100 m. Limited excavation in the vicinity of 200 m can be conducted without special precautions. Vehicular traffic is acceptable up to 100 m from the crater.

Unrecovered toxic materials remain on 200 acres per round (Mishuck and Hartung, 1981), although at levels that are not hazardous.

1. Workplace Environment

Managers of sites where Be and DU testing is conducted may want to establish conservative guidelines. LLNL's Site 300 is governed by a more conservative work-place standard than that established in 1949 by the AEC for permissible airborne concentration of Be in in-plant air (Johnson, 1980):

- Normal operating levels: when the average concentration of Be is consistently less than 0.2 $\mu g/m^3$ during an 8-h workday, the control measures are considered adequate.

- Warning levels: when the average concentration of Be exceeds 0.2 $\mu g/m^3$, but is not over 2 $\mu g/m^3$ for 3 consecutive working days, the operation is studied and the problem corrected. If concentrations exceeding 1 to 2 $\mu g/m^3$ persist for three consecutive weeks, the operation is stopped until the problem is corrected.

- Emergency action levels: when the average concentration exceeds 2 $\mu g/m^3$, but is below 5 $\mu g/m^3$ for 2 consecutive 8-h days, operation is stopped until corrective measures are taken.

Protective clothing and footwear are required for personnel who enter the firing table after a shot. Half-mask respirators and gloves are worn when the soil is disturbed by means other than walking (Johnson, 1980).

2. Off-Site Be Concentrations

Low, nonhazardous levels of Be will leave the KOFA Firing Range boundary if the present sites are used. Smoke and dust fallout account for about 200 g of Be per shot (Table 4). SAI calculated that the average amount of material leaving the boundary per XM753 shot would be 141.3 g of Be with an average concentration of 0.052 $\mu g/m^3$, well within standards set by OSHA (NIOSH, 1986). The average Be concentration based on 20 rounds in one month

(historical maximum) would be 0.000296 $\mu g/m^3$. This would occur only in the highly unlikely event that each shot passes over the same location. Under the worst-case condition of six shots fired consecutively, with the plume passing over the same off-site position, the exposure would be at a level of 0.052 $\mu g/m^3$ for 1.21 h. This concentration is below the OSHA standard of 2 $\mu g/m^3$ for 8 h; it also is below the average allowable concentration for exposure of the public, which is an average concentration per month of 0.01 $\mu g/m^3$.

The air quality limit for Be of 0.01 $\mu g/m^3$ is most stringent because it represents an ambient concentration, averaged over 30 days in the vicinity of a source, as an emission criteria. Luna et al. (1983) point out that this concentration (but not the criteria) is exceeded in virtually all shots even at great distances. However, direct exposures are brief for an individual shot. Downwind air concentration of Be, not to exceed 0.01 $\mu g/m^3$ for 30 days, is equivalent to 432 $\mu g\text{-}min/m^3$. If tests do not result in concentrations in excess of 0.1 $\mu g/m^3$ at the site boundary (Figure 12) then 4320 min of tests or 864 5-min tests would not exceed this guideline for offsite emissions. A more appropriate standard for exposure during a shot would be the workplace standard of 25 $\mu g/m^3$ over 30 min. For a groundburst of the XM753, we calculated a short-term, respirable, maximum, off-site (5280 m) concentration of 0.05 $\mu g/m^3$, which is far less than the more appropriate 25 $\mu g/m^3$ standard.

3. On-Site Be Concentrations

A safety zone of 500 m has been established for previous test areas. Ground-based air samples showed that no quantity greater than 0.2 μg Be per filter (or a dose of 0.005 μg Be) was detected between 500 and 2000 m from the crater. SAI estimated that the Be standard was exceeded only within 100 m of the blast for less than 50 s after detonation. At 1200 m downwind, the average exposure level would be about 0.2 $\mu g/m^3$. A mask would not be required at this distance if the exposure is less than 15 min and personnel are not exposed more than once per day.

Results from helicopter sampling have shown that more than half the mass of the smoke cloud is in the respirable range: 6.6 μm is the median diameter.

C. Human Health Effects

DU, as either a heavy metal or a radioactive substance, is not nearly as toxic as Be.

Beryllium and its compounds are toxic in both soluble and insoluble forms, and the symptoms of the toxicity resulting from Be exposure are manifested as a body-wide systemic disease. Both the acute and chronic forms of the disease can cause severe respiratory damage. The acute form can result in a chemical pneumonitis with inflammation of the mucosa of the respiratory tract. Chronic Be poisoning, which has a different clinical manifestation than does the acute form, can result in pulmonary granulomas, which may result in permanent respiratory and cardiac impairment and, ultimately, damage to the entire body.

To determine possible hazards resulting from exposure to Be, we must know the exact state of the Be in the smoke cloud. While beryllium oxide (BeO) produced at temperatures below 500°C is highly toxic, the same compound produced at much higher temperatures, such as those that occur during a high explosive (HE) detonation, is relatively less soluble and less hazardous (Fishbein, 1976; Durocher, 1969). Research has shown that the aerosol cloud will contain beryllium carbide (Be_2C) and beryllium nitride (Be_3N_2) as well as Be metal and high-fired BeO (Hercules Powder Co., 1965). The Be_2C and Be_3N_2 are slightly soluble in water but not to the extent that the solubility would cause an increase in the health risk. The Be_2C, however, will react with water to form BeO (Sax, 1979; Stokinger, 1966). Since this BeO, formed from the Be_2C, will not be high fired, it will be highly toxic. Because of this, it will be important to minimize the exposure of personnel to high surface concentrations during periods of precipitation.

The major factors that appear to influence the toxicity of Be compounds include the solubility, particle size, and the amount of Be in the compound. Although the BeO produced during an HE detonation is relatively insoluble, from 58% to 80% of the mass of particles in the smoke cloud are 7 μm or less (Mishuck and Hartung, 1981), which is within the respirable range.

1. Acute Respiratory Effects

The eventual fates of inhaled particles are governed by the mechanics of respiration and the respiratory system. The deposition of inhaled particles is dependent almost entirely upon the size of the particle (Kanapilly et al., 1982). According to Luna et al. (1983), 5% of the particles of Be aerosolized in an HE detonation will have a particle-size distribution between 0.5 μm and 5 μm. The data from SAI show that up to 33% by weight of the particles of Be have a mass median effective diameter of 10 μm or more, approximately 30% of the Be particles in the smoke cloud will have a particle-size distribution of between 5 and 10 μm, and less than 2% of the particles in the smoke cloud have a diameter of 1 μm or less.

The respiratory tract is divided into three sections - the nasopharynx, the tracheobronchial region, and the pulmonary region. Cells in these three areas can defend against inhaled particles through the processes of filtration, removal, inactivation, or destruction.

The nasopharynx includes the nose, pharynx, and upper bronchi. The mucous and hair-like cilia that line the nasopharynx serve to remove inhaled particles that are larger than 10 μm. The sneeze reflex assists in this process. These particles are not considered respirable because they are too large to reach the lower portions of the respiratory tract.

In the tracheobronchial area, muscular contractions of the bronchial tree and the cough reflex help to remove inhaled particles. Particles that are between 5 and 10 μm in diameter will be removed almost entirely by the removal mechanisms of the nasopharynx and the tracheobronchial tree.

In the pulmonary region, the distribution of gases and ultrafine particles with a diameter of 0.5 μm or less is governed by Brownian motion. In continuous motion, they will move about randomly in the air space of the lungs by colliding with the molecules of the surrounding air. Most of them will be exhaled.

Approximately 32% of the ultrafine particles that have a particle-size distribution of 0.5 μm to 5 μm will be deposited at the alveolar walls

(Kanapilly et al., 1982). As Luna et al. (1983) has stated, 5% of the cloud will have Be particles with a diameter of between 0.5 and 5 μm. This means that it is possible for 1.6% of the aerosolized Be particles to be deposited in the pulmonary region (0.32 x 0.05). Insoluble particles of BeO may penetrate the alveolar ducts and lodge in the interstitial tissue or enter the lymphatic system and, eventually, the bloodstream. Research has shown that a large percentage of the high-fired insoluble BeO particles of 1 μm or less may remain in the alveoli for many years (Rhoads and Sanders, 1985). Moreover, an additional percentage of particles that are resuspended as a result of post-shot vehicular movement may adhere to dust particles and, therefore, may not be in the 5 μm or smaller range.

The acute respiratory effects of Be that are clinically manifested in humans range from a mild inflammation of the mucous membranes of the nose and throat (Be rhinitis), to inflammation of the bronchioles (tracheobronchitis), and to an acute chemical pneumonitis (US Deparment of Health, Education and Welfare, 1972).

Be rhinitis displays symptoms similar to the symptoms of the common cold and is, therefore, difficult to diagnose. The condition is frequently accompanied by mild nosebleeds. Fluid and blood accumulate in the mucous membranes, and ulcerations may occur in the nasal passages.

The symptoms of acute Be tracheobronchitis do not appear to be related to exposure to Be and, because of this, the disease is difficult to accurately diagnose. As with any case of bronchitis, there will be a cough, chest discomfort and tightness, and moderate difficulty breathing upon mild physical exertion. Body temperature will be normal, but there will be a decreased vital capacity and varying degrees of breathing difficulty.

Acute chemical pneumonitis may develop with exposure to brief but massive exposures to Be. Fortunately, this condition is rare. The symptoms are similar to those of bronchitis but are more severe. There is a great amount of chest pain, an exhausting cough, and cyanosis. An X ray shows a diffused haziness and the appearance of nodules in both lungs.

2. Acute Effects on Skin and Eyes

Acute health effects that occur as a result of exposure of the skin and eyes to Be are contact dermatitis, Be ulcers, and conjunctivitis.

Contact dermatitis will be characterized by itching and reddened, elevated, or fluid-accumulated lesions that appear on the exposed surfaces of the skin such as the face, neck, arms, and hands (US Department of Health, Education and Welfare, 1972). The eruptions will usually disappear within one to two weeks after the cessation of exposure.

Beryllium ulcers will result when Be crystals become implanted in previous cuts or abrasions on the skin. Recovery will take place in one to two weeks and depends on the complete removal of the Be material.

Effects to the eye will occur as inflammation of the conjunctiva of the eye. The inflammation is known as "splash burn" and is similar to burns that are produced by acids or alkalis.

3. Chronic Health Effects

a. Noncarcinogenic

Beryllium disease is considered chronic if the symptoms last for more than a year (US Department of Health, Education and Welfare, 1972). The diagnosis of chronic Be disease depends on several factors: the disease will generally have a latent period with symptoms appearing years after exposure, it will be prolonged in duration without evidence of a permanent cure, and it will become progressively severe, eventually becoming a disease that affects all parts of the body.

Characteristic chronic Be disease will often include a pneumonitis with cough, chest pain, and general weakness. Because of the pulmonary dysfunction, there may also be right-heart enlargement and resultant cardiac damage. This heart damage is ultimately more dangerous than the Be disease itself. There may also be enlargement of the liver and spleen, cyanosis,

"clubbing" of the fingers and toes, and the appearance of kidney stones. The delayed onset of symptoms will quite often be precipitated by some form of stress, such as surgery or another unrelated illness (US Department of Health, Education and Welfare, 1972). Attempts to rid the body of Be have not been successful in animal experiments (Schubert and Rosenthal, 1959).

As with the acute chemical pneumonitis resulting from Be exposure, the chronic form of the disease will show a diffuse haziness on a lung X ray. Quite often, the Be particles that have been deposited in the lungs will invade the cells of the lungs, and granulomatous nodules will be formed. Research has shown that the degree of granulomatous formation in the lungs can be used to predict the severity of the disease (Freiman and Hardy, 1970).

b. Carcinogenic

The Committee of Toxicology of the National Academy of Sciences, National Research Council (1966) stated, "While certain beryllium salts and oxides have been productive of osteogenic sarcomas in rabbits following intravenous administration and primary lung tumors on rats and monkeys following inhalation, there is no evidence that community or industrial exposure to beryllium compounds is associated with an increase in the incidence of cancer in humans" (Committee on Toxicology, 1966). Since that time, however, the International Agency for Research on Cancer (IARC) and the National Institute for Occupational Safety and Health (NIOSH) have recommended that Be and its compounds be handled in the work place as if they were carcinogens (NIOSH, 1986).

In a recent epidemiological study, it was found that, based on lung cancer mortality experience, lung cancer mortality among Be-exposed workers was significantly greater than that in comparable workers in the viscose rayon industry who had similar employment patterns (Mancuso, 1980). In the same study, it was stated, "It is reasonable to believe that the amount of exposure to beryllium to establish a biological effective dose for the induction of lung cancer can occur within a few months of exposure regardless of whether the individual is employed for less than 1 year or employed for 10 years. The additional amount of exposure to beryllium may not be required, once the biological effective carcinogenic dose has occurred in a worker" (Mancuso, 1980).

D. Biological Resources

1. Wildlife

Studies have been conducted on the effects of Be on animals. Unfortunately, few data are available on the effects of Be on animals such as those found in the area surrounding Yuma Proving Ground. However, many of the effects would be similar, although the severity of the injury may be different.

a. Toxicity of Be to Wildlife of YPG

The major factors that will determine the toxicity of Be to animals are, as with humans, the solubility, particle size, and the percentage of Be in the compound.

The high-fired BeO that is deposited after the detonation of the XM785 is a compound that will, compared to other Be compounds, induce a minimal amount of cellular reactions and fewer adenocarcinomas (Spencer et al., 1968).

Previously deposited BeO, which may have been converted to other compounds due to environmental factors of the area, has the potential to be significantly more toxic to the animal population at YPG than the freshly deposited compound. Beryllium hydroxide ($BeOH_2$), for example, is acutely toxic through all routes of administration. Chemical pneumonia can be produced after a single exposure (US Department of Health, Education and Welfare, 1972). The high stability and insolubility of the high-fired BeO, however, would likely prevent desert complexing factors from converting it to other compounds.

In animals, Be does not appear to localize in the lung, but rather appears to be transported to all tissues of the body. There is a significant difference between oral toxicity and toxicity through other routes of exposure. Ingested Be compounds appear to pass through the gut with very little absorption (Reeves, 1965).

There are a number of effects of Be that appear to be peculiar to animals:

- Beryllium oxide, hydroxide, sulfate, and fluoride will produce primary pulmonary cancer (Schepers, 1961).

- Beryllium oxide and zinc beryllium silicate will induce osteosarcoma (Kelley et al., 1961). This particular study was conducted on rabbits, and because black-tail jackrabbits and cottontails are found at YPG, these results may merit special consideration. Again, however, the high-fired BeO is considerably less toxic than other forms of BeO.

- Young animals exposed to 0.5% and 2% beryllium carbonate compounds developed rickets (Guyadt et al., 1933).

- Animals exposed to Be compounds have developed a toxic macrocytic anemia (Stokinger, 1953).

b. Paths of Toxic Effects to Wildlife

Wildlife of the area could take in Be through a number of different routes: ingestion of plants or other animals that contain traces of Be, inhalation of debris from a nearby test, inhalation of resuspended soil containing Be particles, and by drinking water with traces of Be.

Beryllium can be found on vegetation or on the skin of an animal as a result of deposition from either a recent test or resuspension of contaminated soils. Because of the insolubility of Be, its presence in water should not be a hazard since ingestion is not a hazardous route of exposure. Beryllium passes through the digestive system and is not absorbed (Reeves, 1965).

Inhalation of Be particles in the air as a result of a recent test presents a hazard to any animal that would be found close to ground zero and in the debris cloud's path. It is difficult to quantify what kind of dose the animal will receive, because it may be close to the ground and the height and density of the cloud varies depending on meteorological conditions and distance.

The concentration in air of resuspended Be from contaminated soil will, of course, be lower than the original cloud concentration at a given distance. As a result, resuspended Be will be less of a hazard, although it can be an important factor in determining the effects of the testing program on wildlife of the area. Fresh deposits of Be in soil may be resuspended by a factor of 10^{-4}/m with a decline to a factor of 10^{-9}/m after a period of 20 years (Anspaugh et al., 1975).

Testing of warheads has been conducted in the KOFA Range for years, and animal populations are probably somewhat small due to the human activity in the area, destruction of habitat, and noise. Any animals that inhabit the area are either somewhat tolerant of the noise generated from the testing or are transient with respect to the area.

2. Vegetation

The vegetation at YPG is characteristic of a transition desert area. The species found there are mainly small-leaved desert scrub and succulents. This area contains a mix of species from the Mohave and Colorado Deserts of California, the Arizona Succulent Desert, and some Mexican Desert flora. Plant types include drought deciduous and evergreen species, trees, shrubs, herbs and grasses, cacti, and ocotillo. For more detailed information on the species found at YPG, consult the site environmental assessment (US Army, 1978).

Protection of vegetation of the area is covered by state and federal laws that must be followed. Arizona's Native Plant Law makes it unlawful to destroy or mutilate any living plant of a protected species on state or public land. Also, any plant listed as threatened or endangered by the federal government should be protected from adverse impact.

Only soluble compounds of Be have negative impacts on vegetation. Beryllium carbide (Be_2C) and beryllium nitride (Be_3N_2) are slightly soluble compounds that are found following the detonation of a shot. The highest percentages of Be_2C and Be_3N_2 found by analysis of the Be compounds resulting from a shot was 13.5 and 36.5%, respectively, of the total

Be (Hercules Powder Company, 1965). All other values of Be_2C were found to be less than 9% of the total Be. For Be_2C, assuming that it is uniformly distributed among the Be compounds, a factor of 0.135 (13.5%) multiplied times the deposition value (0.56 µg/g soil from Luna et al., 1983) will result in a soil concentration of available soluble Be. Therefore, under worst-case conditions of having all the Be_2C available to plants in soluble form, the maximum percentage of Be_2C detected by past analysis, and the vegetation being near the crater, the concentration of maximum available soluble Be was calculated as follows:

$$0.135 \times 0.56 \text{ µg/g} = 0.075 \text{ µg/g or } 0.075 \text{ ppm.}$$

The same method can be used to determine the maximum amount of soluble Be_3N_2 found in the soil:

$$0.365 \times 0.56 \text{ µg/g} = 0.204 \text{ µg/g or } 0.204 \text{ ppm.}$$

These values are likely overstated because these compounds are only slightly soluble and the highest percentages of Be_2C and Be_3N_2 are found as larger pieces of Be compounds, >590 µm and >840 µm, respectively (Hercules Powder Company, 1965). Also, these values are calculated for the amount of the compound; the amount of the Be alone would, therefore, be even less. In addition, these compounds were only found in 22 of 79 sampled test shots.

The following data demonstrate that, at this level of contamination and with the pH of the soil greater than 7, significant impacts would not occur from a single shot. However, if many shots' footprints overlay a small area, soil levels could build to levels that may adversely affect vegetation. The following text will help to explain the phytotoxic effects of Be in soil and the corresponding levels at which they occur.

A few studies have been conducted on the toxic effects of Be on a variety of species. Kosak-Channing (1986) studied Be in solution on tobacco plants; Holst et al. (1980) studied the effects on barley seedlings; Bohn and Seekamp (1979) looked at the effects of liming and time on potato and oats; Encina and

Becerra (1986) analyzed Be's effect on cytokinesis; and Romney and Childress (1965) studied the effects on alfalfa, barley, lettuce, peas, bean, wheat, and ladino clover. The pH of all the soils in these studies was between 5 and 6. Unfortunately, no data were found on the effects of Be on desert vegetation - studies were conducted on species of commercial value.

a. Uptake

Beryllium uptake by plants is influenced by many factors. The most important factor is the state of the Be in the soil. Vegetation will not be affected by Be in an insoluble form. Soluble forms of Be must be present in the soil for toxic effects to occur (Romney and Childress, 1965). Other factors include availability of other nutrients, soil pH, season, and temperature.

When other nutrients are available, Be can often be excluded from uptake by the roots of the plant. This depends on the amount of other nutrients in the soil in relation to Be. Beryllium uptake increases with higher Be soil content (Bohn and Seekamp, 1979). Beryllium either competes with calcium (Ca) and magnesium (Mg) root-absorption sites or disrupts the normal influx by damaging or clogging the absorption sites (Romney and Childress, 1965). An increase in the amount of Be in the soil decreases the amount of uptake of Ca; however, if enough Ca is available, it occupies its binding sites (Encina and Becerra, 1986). Zinc (Zn) and aluminum (Al) depress Be uptake, with Al having a stronger effect; alternatively, the amount of Be entering is not influenced by addition of a like amount of Mg. This may show that there is a chemical component to absorption because there is a chemical similarity of Be to Zn and Al and a dissimilarity to other earth metals such as Mg. However, the evidence indicates that the primary mechanism is physical, not chemical, and that uptake is limited by a metabolically active barrier (Holst et al., 1980). Unlike the other elements, uptake of phosphorus (P) increases with increased levels of Be in the soil (Romney and Childress, 1965).

Temperature and soil pH also have an effect on Be uptake. Holst et al. (1980) showed that colder temperatures (1^{o}C) slowed Be absorption, though not dramatically. Beryllium uptake proceeds more rapidly at higher pH levels

with a marked increase in uptake above a pH of 5.5. However, Romney and Childress (1965) found that at a pH over 6.0, growth-inhibition effects are less severe, and Williams and Le Riche (1968) found that greater growth can occur in soil with pH above 7.

b. Translocation and Accumulation

As much as 95% of the total plant accumulation of Be occurs in the roots (Holst et al., 1980; Romney and Childress, 1965), and although roots can concentrate high levels of Be, it is not readily translocated to other plant parts (Romney and Childress, 1965). Leaves appear to concentrate higher levels of Be than does the stem (Romney and Childress, 1965), although Kosak-Channing (1986) found the opposite to be true in tobacco. The concentrations in tobacco leaves and stems are similar, while the root concentrations are markedly higher (Kosak-Channing, 1986). Beryllium was found in orchard leaves at 0.026 ppm and Australian tobacco at 0.25 ppm of ash weight (Kosak-Channing, 1986). Vegetation near a phosphate-processing plant contained 1 to 2 ppm by ash weight (Severson and Gough, 1976). The Be content of plant tops increased with the soil Be content or with increased soil acidity (Bohn and Seekamp, 1979).

Food crops can accumulate Be in consumable plant parts, although at low levels. Potatoes and tomatoes contained 0.08 to 0.24 ppm dry weight (Kosak-Channing, 1986). Bohn and Seekamp (1979) found potatoes with less than 5 ppm of Be on a dry weight basis. However, they feel that plant uptake is an unlikely source of Be in the food chain, because relatively large amounts (>100 ppm) were necessary to produce much Be uptake, even in very acidic soil.

c. Phytotoxicity

The toxic effects of Be in plants are reduced yield with gross root stunting, poor germination, cytokinesis inhibition (although with low efficiency), and upper-part darkening, which usually occur at levels in excess of 1 ppm Be in soil (Holst et al., 1980; Romney and Childress, 1965; Bohn and Seekamp, 1979; Encina and Becerra, 1986; Williams and Le Riche, 1968). Nondeciduous perennials are probably more susceptible to Be toxicity because of year-round accumulation (Severson and Gough, 1976). Romney and Childress (1965) and Horovitz and Petrescu (1964) found that low levels of Be appear to

stimulate growth, but this effect disappears as the plants developed. The greatest effect of Be uptake is a decrease in yield with a threshold about 2 ppm (Romney and Childress, 1965). However, at a level of 4 ppm, visual damage appeared in seedlings after just one week. The roots turned brown, had a swollen appearance, and ceased normal elongation. In addition, roots and foliage became stunted, and the foliage turned a darker blue-green. Bohn and Seekamp (1979) found that the few seeds that germinated in high-Be treatments died soon after emergence.

Toxic effects appear to occur at certain threshold levels, depending on soil type. At levels of 0.5 ppm Be in soil, Williams and Le Riche (1968) found that plants achieved their highest yields. Kosak-Channing (1986) found that absolute Be concentration in each tissue type was not significantly different for plants grown in 0.3 and 1.0 mg/L of Be. There is a marked increase between the 1.0 and 3.0 mg/L treatment for all tissue (Table 6). Chlorosis, leaf mottling, and growth reduction are not apparent at 0.3 and 1.0 mg/L of Be. However, at 3.0 mg/L, stems were smaller and yellowing of leaves occurred. Romney and Childress (1965) did not observe chlorosis or mottling of foliage up to 16 ppm Be; the only effect observed was earlier flowering and senescence induced in legumes treated with Be. Williams and Le Riche (1968) observed necrotic specks on the leaves of plants receiving 5 to 10 ppm Be after the first seven days. By the end of the experiment, all plants given more than 2 ppm had stunted growth, but the leaves were free of lesions. In summary, concentrations of Be in the soil that exceed 2 μg/g are potentially able to cause reduced yields.

Bohn and Seekamp (1979) observed that liming, which increases the alkalinity, and time can mitigate Be toxicity in successive harvests. This technique also increases the percentage of germination. Liming reduces uptake (due to available Ca) and, therefore, toxicity of Be into the plant. Successive harvests also showed greater yield and an increase in soil pH.

E. Water Resources

SAI thought the XM753 tests had no significant adverse effect on water quality due to the water transport distance, low solubility of test material, and intermittent flow of water. There is no additional new information to warrant modification of this statement.

VI. Summary and Recommendations

This report has reviewed the information available since the previous review of YPG Environmental Assessment by Los Alamos National Laboratory (Rodgers et al., 1984). A model that we have used to predict Be deposition at Livermore, California, was applied to assess the downwind deposition and air concentrations of Be and DU during actual tests with the XM753 and XM785 at YPG. In addition, one single, static test of an XM785 was conducted at Tonopah Test Range.

Our findings were that a model developed by Sandia National Laboratory (Luna et al., 1983) and our model agree very well. Both models predict the downwind deposition and air concentrations within a factor of ten for a majority of the tests. This precision was generally obtained 50% of the time in complex terrain and 90% of the time in rolling terrain (Dickerson and Lange, 1986). This model precision depends, however, on an estimate of the apportionment of metal into the components of recoverable parts, ejecta, smoke, and dust. In the test at Tonopah Test Range, the amount of recovered parts differed substantially from the norm observed at YPG for the XM753 and XM785 tests. Consequently, our model overpredicted the downwind deposition and air concentrations.

Given the normal deposition and air concentrations predicted for the XM785 (and even the amounts observed at Tonopah Test Range), we drew several conclusions about the environmental and health effects at Yuma Proving Ground. First, the aerosolized Be and DU will produce exposures usually less than 5-min long. The maximum depositions and air concentrations will be directly downwind in the plume centerline. The maximum deposition will occur in a small area of approximately 50-m radius (2000 m^2 or 0.5 acre) and will be less than 1×10^4 $\mu g/m^2$ Be and 1×10^6 $\mu g/m^2$ DU. These values compare with an estimated maximum deposition allowable of 1×10^5 $\mu g/m^2$ Be and 6.9×10^7 $\mu g/m^2$ DU. The maximum deposition allowable was derived from consideration of wind-blown resuspension to produce long-term concentrations equal to the most stringent air quality standards. Remedial actions should not be necessary for depositions less than these maximum amounts. The increase in soil concentrations of Be, if allowed to weather-in, would be

undetectable from the background concentration of Be considering the normal variability of the background concentration, even if deposition is the maximum expected (1×10^4 $\mu g/m^2$).

The maximum air concentrations of Be averaged over 15 min will be a factor of ten or more below the 30-min exposure guideline of 25 $\mu g/m^3$, and usually below the 8-h exposure guideline (for workplace environments) of 2 $\mu g/m^3$. Measurements of particle size show these concentrations are partly respirable. At 5280 m downwind, the estimated distance to a site boundary, the airborne Be concentrations will not normally exceed the emission criterion, 0.01 $\mu g/m^3$ averaged over 30 days, unless many, many tests were conducted in the same wind direction. For example, we estimated (for stability category D and 5 m/s wind speed) that 4320 min of tests or 864 5-min tests, would not exceed this criterion (assuming testing would not normally be done during worst-case meteorological conditions).

The maximum air concentrations of DU averaged over 15 min will be a factor of ten or more below the 15-min TLV of 600 $\mu g/m^3$. Offsite concentrations of DU would be trivial and undetectable following the same arguments as for Be above.

Enough information is available from both the observed tests (both XM753 and XM785) and the modeled interpretations of tests to conclude that there will be no potential health effects in the immediate vicinity of the tests and that no air quality criteria will be exceeded anywhere downwind. To make these conclusions, we assumed that reasonable judgment is maintained in exclusion of personnel from 1000 m downwind of the tests for 15-min following the test and that vehicle traffic and construction activity is excluded in areas where deposition has taken place in the past, to avoid resuspension of deposited metal aerosols. At ground zero, near the crater, precautions should be taken to recover metal fragments, to protect workers from respiratory exposure, and to secure the area from inadvertent traffic or potential accidental exposure.

Potential ecological effects can be assessed on the basis of the deposition and air concentration patterns observed and predicted. Deposition and debris close to ground zero would contribute to elevated soil levels in an

area estimated to be on the order of 2000 m^2 (0.5 acre) in the immediate vicinity of ground zero and extending downwind. It would be difficult to detect the increased soil Be and DU concentrations by soil sampling outside of this area because of normal variations in background values. The potential effects of Be and DU on plants and animals should be insignificant outside this area because of the low solubility of the likely decomposition products. Within the area of maximum deposition, there may be increased uptake of Be by plants, but not necessarily at a level toxic to sensitive species, unless the soil levels exceed 2 μg/g of soluble Be compounds. The potential effects on animals eating (grazing) or breathing (respirable dust) would be insignificant. The potential effects of the blast and debris in the immediate vicinity of the ground zero would likely be a more serious disturbance to ecosystems than any potential toxicity of Be and DU.

Recommendations for mitigation and environmental management are to (1) exclude personnel during the tests, (2) protect personnel from exposure during post-test operations near ground zero, (3) recover parts and debris as much as reasonably prudent near ground zero, (4) maintain a long-term exclusionary control over any vehicular or personnel access to deposition areas in excess of 1×10^4 $\mu g/m^2$ Be and 1×10^6 $\mu g/m^2$ DU, (5) conduct soil sampling in the vicinity of the ground zero as delineated by estimates of downwind deposition patterns, or when opportunities arise, by measurements of downwind deposition during tests using sticky plates, (6) conduct tests only under normal meteorological conditions when the atmospheric stability, wind speed, and wind direction are favorable, (7) avoid a series of test shots so numerous that downwind deposition patterns could overlap to cause significant cumulative effects, (8) maintain records and long-term monitoring as appropriate to document the tests for regulatory agencies, and (9) utilize a dedicated dispersion code to estimate downwind concentrations and deposition patterns for management purposes.

There is sufficient information available at present to perform an adequate environmental assessment of continued testing at Yuma Proving Ground of groundburst projectiles containing Be and DU. The potential effects of continued testing of devices such as the XM753 and XM785 appear to be insignificant, providing that prudent mitigations and environmental management practices such as those recommended are carried out.

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References

Anspaugh, L.R., J.H. Shinn, P.L. Phelps, and N.C. Kennedy, 1975. Resuspension and Redistribution of Plutonium in Soils. Health Physics 29: 571-582.

Bohn, H.L., and G. Seekamp, 1979. Beryllium Effects on Potatoes and Oats in Acid Soil. Water, Air and Soil Pollution 11(3): 319-322.

Committee on Toxicology and the Advisory Center on Toxicology, 1966. Air Quality Criteria for Beryllium and Its Compounds. National Academy of Science, National Research Council, Washington, D.C.

Dickerson, M.H., and R. Lange, 1986. An Example of Emergency Response Model Evaluation Studies Using the MATHEW/ADPIC Models, Lawrence Livermore National Laboratory, Preprint UCRL-94388.

Durocher, Norman L., 1969. Preliminary Air Pollution Survey of Beryllium and Its Compounds: A Literature Review. Contract No. PH 22-68-25. U.S. Department of Health, Education and Welfare, Public Health Service, Raleigh NC.

Encina, C.L., and J. Becerra, 1986. Inhibition of Plant Cytokinesis by Beryllium and Its Reversion by Calcium. Environmental and Experimental Botany 26(1): 75-80.

Fishbein, L., 1976. Environmental Metallic Carcinogens: An Overview of Exposure Levels. Journal of Toxicology and Environmental Health 2: 77-109.

Freiman, S.G., and H.L. Hardy, 1970. Beryllium Disease - The Relation of Pulmonary Pathology to Clinical Course and Prognosis Based on a Study of 130 Cases from the U.S. Beryllium Case Registry. Human Pathology 1: 25-44.

Griggs, K.S., D.S. Myers, and R.W. Buddemeier, 1985. Environmental Monitoring at the Lawrence Livermore National Laboratory 1984 Annual Report. Lawrence Livermore National Laboratory, Livermore, California. UCRL-50027-84.

Guyadt, B.L., H.D. Kay, and H.D. Branion, 1933. Beryllium Rickets. Journal of Nutrition 6: 313-324.

Healy, J.W., 1980. Review of Resuspension Models in Transuranic Elements in the Atmosphere, W.C. Hanson, Ed., Department of Energy Report, DOE/TIC-2280.

Hercules Powder Company, Magna, Utah, 1965. High Energy Propellant Combustion Hazards Particle Size Study, Volume II. Air Force Flight Test Center, Air Force Systems Command, Edwards Air Force Base, California. Technical Documentary Report No. AFRPL-TR-65-81, 1965.

Holst, R.W., W.E. Schmid, and J.H. Yopp, 1980. Beryllium Uptake by Excised Barley Roots. Plant and Cell Physiology 21(5): 737-743.

Horovitz, C.T., and O. Petrescu, 1964. The Roles of Beryllium and of Magnesium in Plant Metabolism. Transcript of the 8th International Congress Soil Science 4: 1205-1213.

Johnson, J.S., 1980. Lawrence Livermore Laboratory's Beryllium Control Program for High-Explosive Test Firing Bunkers and Tables. Lawrence Livermore National Laboratory, Livermore, CA. UCID-18006-Rev. 1.

Kanapilly, G.M., R.K. Wolff, P.B. DeNee, and R.O. McClellan, 1982. Generation, Characterization and Inhalation Deposition of Ultrafine Aggregate Aerosols. Annals of Occupational Hygiene 26(1-4): 77-91.

Kelley, P.J., J.M. Janes, and L.F.A. Peterson, 1961. The Effect of Beryllium on Bone. Journal of Bone Joint Surgery 43A: 829-844.

Kosak-Channing, L., 1986. Beryllium Distribution in Hydroponically-Grown Tobacco Plants. Plant Science 46(3): 175-180.

Lange, R., 1978. ADPIC - A Three-Dimensional Particle-In-Cell Model for the Dispersal of Atmospheric Pollutants and Its Comparison to Regional Tracer Studies, Journal Applied Meteorology 17, 320-329.

Langham, W.H., 1971. Plutonium Distribution as a Problem in Environmental Science, LA-4756, E.B. Fowler, R.W. Henderson, and M.F. Milligan, Eds., Proceedings of Environmental Plutonium Symposium, August 4-5, 1971, Los Alamos, New Mexico. National Technical Information Service, Springfield, Virginia.

Luna, R.E., D.R. Parker, and J.M. Taylor, 1983. Final Report: YPG Dispersal Evaluations. Sandia National Laboratory, Albuquerque, NM. SAND83-0876.

Mancuso, T., 1980. Mortality Study of Beryllium Industry Workers Occupational Lung Cancer. Environmental Research 21: 48-55.

McMahon, T.A., and P.J. Denison, 1979. Empirical Atmospheric Deposition Parameters - A Survey. Atmosphere Environment 13: 571-585.

Mishuck, E., and W.H. Hartung, 1981. Final Engineering Report: XM753 Environmental Data Analysis Services, Science Applications, Inc., San Diego, CA, Report No. SAI-132-81-378LJ.

National Institute for Occupational Safety and Health, 1986. Recommendations for Occupational Safety and Health Standards. U.S. Department of Health and Human Services, Public Health Service, Atlanta, Georgia.

Reeves, A.L., 1965. Absorption of Beryllium from the Gastrointestinal Tract. American Medical Association Archives of Environmental Health 11: 209-218.

Rhoads, K., and C.L. Sanders, 1985. Lung Clearance, Translocation, and Acute Toxicity of Arsenic, Beryllium, Cadmium, Cobalt, Lead, Selenium, Vanadium, and Ytterbium Oxides Following Deposition in Rat Lung. Environmental Research 36: 359-378.

Rodgers, J.C., D.R. Dreesen, E.H. Essington, G.C. White, E.J. Cokal, 1984. Review of the Environmental Quality Aspects of the US Army Test and Evaluation Command Depleted Uranium Munitions Program. Los Alamos National Laboratory, Los Alamos, New Mexico. LA-UR-84-3929.

Romney, E.M., and J.D. Childress, 1965. Effects of Beryllium in Plants and Soil. Soil Science 100(3): 210-217.

Sasaki, Y., 1970. Some Basic Formalisms in Numerical Variational Analysis, Mon. Weather Review 98, 875.

Sax, Irving N., 1979. Dangerous Properties of Industrial Materials. Van Nostrand Reinhold Co., New York.

Schepers, G.W.G., 1961. Neoplasia Experimentally Induced by Beryllium Compounds. Progress in Experimental Tumor Research 2: 203-244.

Schubert, J., and M.W. Rosenthal, 1959. Chemical Approaches to the Treatment of Beryllium Poisoning. American Medical Association Archives of Industrial Health 19: 169-178.

Sehmel, G.A., 1980. Particle Resuspension: A Review. Environment International 4: 107-127.

Severson, R.C. and L.P. Gough, 1976. Concentration and Distribution of Elements in Plants and Soils near Phosphate Processing Factories, Pocatello, Idaho. Journal of Environmental Quality 5(4): 476-482.

Sherman, C. A., 1978. MATHEW: A Mass-Consistent Model for Wind Fields over Complex Terrain, Journal of Applied Meteorology 17, 312-319.

Shinn, J.H., 1987. Beryllium Dispersal Studies at Lawrence Livermore National Laboratory, presentation to the French Atomic Energy Commission (Commissariat A L'Energie Atomique, CEA), Scientific Exchange for High Energy Explosive Test Facilities, Moronvilliers, France, February 4-5, 1987.

Spencer, H.C., R.H. Hook, J.A. Blumenshine, S.B. McAllister, S.E. Sadek, and J.C. Jones, 1968. Toxicological Studies on Beryllium Oxides and Beryllium-Containing Exhaust Products. Aerospace Medical Research Laboratories, Air Force Systems Command, Wright-Patterson AFB, Ohio. AMRL-TR-68-148.

Stokinger, H.E., 1953. The Effect of Various Pathologic Conditions on In-Vivo Hemoglobin Synthesis. Biochemical Biophysical Acta 12: 439-443.

Stokinger, H.E., Ed., 1966. Beryllium: Its Industrial Hygiene Aspects. Academic Press, New York.

United States Army, 1978. Installation Environmental Impact Assessment for United States Army Yuma Proving Ground, Yuma Arizona. US Army, Materiel Development and Readiness Command, Test and Evaluation Command.

United States Department of Health, Education and Welfare, 1972. Occupational Exposure to Beryllium: Criteria for a Recommended Standard. National Institute for Occupational Safety and Health.

United States Department of Health, Education and Welfare, 1969. Preliminary air pollution survey of beryllium and its compounds. National Air Pollution Control Administration Publication No. APTD 69-29.

Williams, R.J.B., and H.H. LeRiche, 1968. The effect of traces of Beryllium on the Growth of Kale, Grass, and Mustard. Plant and Soil 29: 317-326.