A Critical Review of ATSDR Public Health Assessment for Lawrence Livermore National Laboratory (public comment release May 24, 2002)

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Disclosures:

We have attempted to provide as full and fair review of the ATSDR Health Assessment as we could, given the time and resources we could devote to the task. As part of our attempt at fairness, we feel obliged to notice for ourselves and disclose to our readers potential sources of influence on our judgment. Our sponsors, TriValley CAREs (Communities Against a Radioactive Environment), the San Francisco Bay Area Physicians for Social Responsibility (PSR) and Western States Legal Foundation (WSLF), have all had long-standing concerns about Livermore operations, and they felt significant dissatisfaction with both the content of the report and the process ATSDR followed in preparing it. We have an ongoing ATSDR sponsored community education project in Western Shoshone and Southern Paiute communities affected by nuclear weapons testing and we feel a strong commitment to it. We have also worked with ATSDR as technical support to a community advisory panel and received ATSDR support for health studies around the Massachusetts Military Reservation.

Table of Contents

1.0 Introduction and Executive Summary

Every Superfund site represents a tragedy. The causes may have been a lack of knowledge at an earlier time, carelessness or neglect, or failures in oversight. Whatever the reasons, the legacies are vulnerable communities and lingering distrust of American industry and government. The Superfund legislation of the 1980s was designed to correct these legacies. An important component of the legislation was the creation of the Agency for Toxic Substances and Disease Registry (ATSDR). ATSDR was established because Congress believed that vulnerable communities and the state and federal agencies managing the clean up would need trustworthy information and advice about environmental health threats. ATSDR was thus intended from the beginning to have a community health focus, including a mandate to work collaboratively with affected communities.

Public Health Assessments are one of the tools that ATSDR uses to help inform communities and to develop recommendations to parties with concerns about a site. The ATSDR Public Health Assessment for Community Exposures to the 1965 and 1970 Accidental Tritium Releases from the Lawrence Livermore National Laboratory (LLNL) (ATSDR 2002) was released in draft form for public comment May 24, 2002. It has had an unusual history. Tritium, the radioactive form of the element hydrogen, was not on the list of substances for which LLNL was declared a superfund site. Tritium has a twelve year half life and is found most often in mobile form as water and water vapor; it thus poses rather different problems for cleanup than plutonium and many chemical contaminants. However, operations at LLNL have released significant amounts of tritium every year for more than 40 years (ATSDR 2001b, p. 55) including the two major accidental releases that are the subject of the draft Public Health Assessment: approximately 350,00 curies of tritium were released in 1965 and another 300,000 curies were released in 1970.

These releases have quite naturally led to public concerns about their health implications. Because of such concerns, and because LLNL appeared to have no information about organically bound tritium (OBT) in the environment (the concern was that OBT may be more persistent and may lead to unanticipated radiation doses), community members requested that ATSDR conduct measurements of OBT in the LLNL region. Rather than make such measurements or recommend that LLNL or other entities carry out such measurements, ATSDR chose to conduct a series of consultations and assessments to decide whether such measurements could be justified based on public health concerns.

The first of these efforts was the convening of an expert panel to “evaluate site specific tritium monitoring and evaluation programs and determine whether adequate data and dose evaluations currently exist for assessing the public health implications of tritium exposure and uptake at these sites.” ATSDR presented a summary of the panel’s deliberations and ATSDR’s conclusions from the summary in a draft Health Consultation (ATSDR 2001b). Very shortly thereafter, ATSDR presented a second draft Health Consultation specifically considering the health implications of the 1965 and 1970 accidents (ATSDR 2001a). It is not totally clear why ATSDR chose to focus on the two accidents; one reason may be that in later years annual releases have been summarized and interpreted in LLNL annual reports: also the accidents constitute the largest releases of tritium from the facility. Whatever the reasons for ATSDR attention, these accidents were major events for which there should be a public record and assessment. After receiving a variety of public comments and criticisms on the health consultation, ATSDR created the Public Health Assessment for those accidents (ATSDR 2002); that assessment is the main subject of this review. There is one further ATSDR document which bears on the tritium concerns, and that is the ATSDR Toxicological Profile for Ionizing Radiation (ATSDR 1999).

As we shall discuss later there are very substantial inconsistencies between these various documents and ATSDR has made no attempt to explain or rationalize the differences. However, the overall message appears to convey the following reasoning: 1) radiation doses from tritium releases in ongoing operations at LLNL to a hypothetical maximally exposed individual are “below public health concern”; 2) this conclusion is extremely unlikely to be altered by the unknown component of OBT in the LLNL environment; 3) therefore an effort to measure OBT would not be justifiable; 4) the major accidents also gave doses to the maximally exposed individual which are “below public health concern”; 5) therefore ATSDR will not make any further recommendations to itself or other entities.

Among the public comments on the health consultation (ATSDR 2001a) was a request by community members for funds to support an independent review of the forthcoming Public Health Assessment. This request was denied, although ATSDR indicated that it would arrange for some sort of peer review of the document. Subsequently a coalition of community groups, TriValley CAREs (Communities Against a Radioactive Environment), the San Francisco Bay Area Physicians for Social Responsibility (PSR) and Western States Legal Foundation (WSLF), received funds from the Citizens’ Monitoring and Technical Assessment Fund to support our review.

Our primary task was to review the Public Health Assessment (ATSDR 2002). We also spent time reading through the earlier iterations and some related documents as listed in our bibliography. With one exception we have not had access to any other comments or reviews of the ATSDR documents or any further work performed by ATSDR. After completion of our draft review, we did receive a copy of a LLNL report (Peterson et al. 2002) that was provided to ATSDR last summer, reviewing the still classified accident report for the 1965 accident and reproducing one appendix to it; this report provides information on the release and on meteorological conditions which was not available to ATSDR when the draft assessment was prepared. We discuss the implications of this report in Appendix B.

We have not attempted to reconstruct or run the two models used by ATSDR to quantify the transport of tritium in the form of hydrogen, its deposition to the ground and its re-emission and transport in the form of water vapor. However we have performed calculations to test the plausibility of the modeling results. We have reconstructed the Monte Carlo simulation model from which ATSDR infers doses and uncertainties in their dose estimates and we have developed a revised version of it. In the course of creating the revised model, we have made a systematic discussion of uncertainties in the estimates of doses and risks from the accidents. We have also made an effort to draw some inferences from evidence within the documents themselves about the process which ATSDR followed in creating the health assessment and the suitability and effectiveness of their effort in informing the public.

Our primary findings are as follows:

- **Process problems.** The health assessment process has been marked by a lack of responsiveness to community concerns, a series of contradictory documents, and very limited attention to establishing a record of what happened in the accidents and to informing the public in a detailed and understandable way about what happened. ATSDR has lost its opportunity to serve as an honest broker on these issues, and thus departed from its defined public health mission.

- **Treatment of uncertainties**. These calculations involve a large degree of uncertainty, in part due to the unfortunate lack of information about the conditions around the accidents. ATSDR has made no evaluation of the reliability of the information derived from LLNL records on the release. Other key factors for which uncertainty was underestimated include meteorological conditions and the rate of tritium deposition. The treatment of uncertainty in the retention time of tritium in the human body is incoherent. There has been no attempt to explain in ordinary language the reasons for and the implications of the uncertainties in the modeling effort.

- **Calculation and presentation of dose estimates**. The health assessment makes mistakes in presenting its results. In particular, the models predict higher rather than lower doses for the 1965 accident contrary to assertions in the text. A significant factor in calculating dose, the dose and dose-rate effect factor, was misused. We present revised dose estimates that correct these errors; our estimates are 3-4 times higher than those presented in the health assessment. Population dose estimates should have been made and we make a rough attempt at doing so here.

- **Discussion of health risks.** In its treatment of risks from radiation exposure the authors of the health assessment contradict standard practice as described in the National Academy of Sciences BEIR V report (NRC 1995), in international commissions (ICRP 1991, UNSCEAR 2000), and the ATSDR Toxicological Profile for Ionizing Radiation (ATSDR 1999). The authors give no indication that their assumption of a threshold for radiation induced cancer is at variance with standard risk assessment practice or that there has been a very substantial scientific and policy debate on the issue. In contrast, using standard methods we find that within the range of uncertainty there was potential for cancer mortality risks that are considered ‘significant’ in common regulatory practice- that is, the risks using both ATSDR’s and our estimates for a maximally exposed individual are in the vicinity of 1 in 10,000; in some uncertainty ranges, the risks exceed 1 in 1,000.

- **Irresponsible conclusions.** The assessment and the consultations use the term "below levels of public health concern" in a number of places, including in its conclusions about potential risks. There are serious problems with this usage. The term is nowhere defined, nor is there any indication of what the authors would consider to be a level of public health concern. Risks calculated using standard practice from the radiation doses presented in the assessment are at levels that are generally taken to be significant by the agencies supervising Superfund clean ups. Most disturbingly, the inferences drawn by ATSDR directly subvert the ALARA principle (as low as reasonably achievable), a cornerstone of the social compact for managing radiological hazards. The impression left by the ATSDR documents is indifference to releases of 300,000 Ci of tritium (in the form of hydrogen, or 10,000Ci in the form of water vapor) in a highly populated area.

These last two points in particular constitute serious flaws in the draft and if left unaltered would represent a significant departure by ATSDR from its public health mission. Both should be corrected in the final version of the assessment. The potential risks from major releases close to a highly populated area can be described without exaggeration but in a way that respects the ability of the public to digest quantitative information and that supports an ALARA approach. Failures in process and loss of trust are not easily corrected; however, steps in the right direction would be well received. These would include correcting the above two deficiencies and making efforts, perhaps through the regional ATSDR office, to assure that the full story of the accidents becomes available and that appropriate lessons are drawn for future operations and monitoring. If instead ATSDR persists in avoiding its public health mission, the likely consequences are worsened community relations with LLNL, greater distrust of government, and unnecessary community fears.

2.0 Our expectations for an ATSDR Health Consultation or Health Assessment

For a critical review of a document to be fair and helpful it must take into account the goals of the effort being documented. The review should assess how well the goals have been achieved and it can also consider whether the goals were appropriate. In the case of the ATSDR health assessment (ATSDR 2002), the principal objectives have not been clearly stated in the report; we have thus felt it important to define at the beginning what we expect of an ATSDR health assessment. These expectations considerably influence our review.

The two accidents that are the subject of the assessment were, by any measure, major radiological events. 300,000 Ci of a relatively long lived radioactive isotope like tritium is a very substantial accidental release of radioactivity and fully warrants public attention and concern. We think that a couple of caricatures of what an assessment of these accidents might turn out to be will help explain what we believe an assessment should aim for.

The assessment could have as its primary goal the formulating of a decision by ATSDR; it might then become a largely private agency document, even though presented to the public for review, because it uses technical criteria (that members of the public might or might not understand and/or accept) and a technical analysis (that also might or might not be effectively communicated) to enable ATSDR to decide whether it wanted to make certain further recommendations.

The assessment could instead have as its primary goal the reassurance of a public in order to convince it not to blame or make demands on the party responsible for the accident or any of the regulatory agencies. In pursuing this goal the assessment might then take a minimalist approach – 'be reassured because we have done a complicated technical analysis and, as we keep emphasizing, our models say there is nothing to be concerned about'. We rather doubt that such an approach to reassurance works in practice, and in any event we consider it inappropriate.

In contrast, our view is that the primary objective of an ATSDR assessment should be to inform the various stakeholders, including the public, about what happened in the accidents and about possible health implications. These stakeholders will have diverse perspectives and understandings about risks to health and other accident implications; the assessment should be helpful in informing a diverse audience. A secondary, but still potentially important objective, is to provide a basis for ATSDR to decide (in consultation with stakeholders) what if any further recommendations are appropriate for ATSDR, other government agencies, and other entities.

In pursuing both of these purposes gaps in knowledge and other uncertainties require careful attention. The parties need to know both what is known and what isn't, and what might be learned with a reasonable amount of additional effort. Different people weigh uncertain information differently, so part of the task of informing stakeholders is finding alternative appropriate ways of describing uncertainties. In this context, the use of "conservative" assumptions or calculation methods can prove unhelpful. There are numerous situations in which it is easier to estimate an upper ("conservative") bound than to create a best estimate with uncertainty, and sometimes it is appropriate to do so. However, information about how well you know the quantity you are calculating gets lost and communication becomes more difficult. This is discussed further in Section 6.

The original idea of Congress in establishing ATSDR in the Superfund legislation was that communities in the vicinity of sources of hazardous contamination would have various needs for public health information and other public health resources. In these situations in which the public trust had already been abused, it was particularly important that communities could call for such information and resources from an agency that they could communicate directly with and that they could trust to be directly concerned about them, more perhaps than those responsible for the contamination or the agencies which had failed to protect them.

Public understanding is thus critical to the mission of ATSDR and the creation of a health assessment requires attention to effective communication. And effective communication requires responsiveness - paying attention in the formulation of a study to what community members and other stakeholders want to know and how they will best be able to understand it. The National Academy of Sciences report *Understanding Risk* (NRC 1996) provides a very helpful road map for collaborative efforts by analysts and interested parties to frame important questions and develop approaches to answering them, described as an "analytic and deliberative process". Such collaborative work can improve interactions among stakeholders and agencies and ultimately build trust. But the parties have to believe that the ATSDR analysts will treat all parties fairly and that they haven't predetermined the questions and the modes of analysis to produce predetermined answers.

Building trust also requires persistence in working together and continuity in the effort. As we describe next, there are a variety of ATSDR documents which pertain to the accident: there is the health assessment (ATSDR 2002) that is the subject of our review. One year before it, ATSDR prepared a health consultation (ATSDR 2001a) on the accidents. There is another health consultation and panel report on Tritium Releases and Potential Offsite Exposures (ATSDR 2001b) also from a year ago, and an ATSDR toxicological profile on ionizing radiation (ATSDR 1999). While certainly an agency can change its approach and there can be important changes in scientific understanding, it is vital that inconsistencies between documents be understood and effectively explained. When the agency chooses to change its approach to particular problems or issues, there should be both explanation and justification for the change, and it should be made clear how the change affects the findings.

3.0 Context of the May 24, 2002 Assessment

3.1 Livermore Lab

What is now Lawrence Livermore National Laboratory (LLNL) was first occupied by the Atomic Energy Commission in 1950 and in 1952 it became a separate part of the University of California Radiation Laboratory. LLNL currently occupies 821 acres of land approximately 40 miles east of San Francisco and 3 miles east of central Livermore.

A variety of activities occur at the laboratory, with special emphasis on nuclear weapons research and development. In addition to potential radiological health hazards LLNL poses chemical hazards for which it was placed on the Superfund list in 1987.

One structure of particular concern is Building 331, the Tritium Facility. This unit has operated since 1956 and the amount of tritium in the building averaged 200-300 grams (2-3 million Ci) through 1990, after which the inventory was brought under 5 grams (50,000 Ci). Both accidents being considered here occurred in Building 331.

3.2 The two accidents

There is very little information about the accidents available to us. Apparently there are some accident reports that are still classified. We find it difficult to understand what nearly 40 year old information about an accidental release still requires classification or, presuming that there still are some concerns about secrecy, why a declassified version could not be created. We know that on January 20, 1965 at 3:30 pm roughly 350,000 curies were released from a stack in building 331. We are told that this was the result of human error and that most of the tritium was in gas form ($^3H_2$). Nothing is known about meteorological conditions at the time and apparently no local environmental sampling was done following the release[1].

On August 6, 1970 at 6:14 am another 300,000 curies were released from the same stack when a component of a pressurized gas system failed. We have some information about the meteorological conditions on that day (wind from the southwest at 2.2-4.5 mph). It appears from maps generated after the accident that the plume moved east and then southeast down the San Joaquin Valley. Sampling was done after this accident and we have a small amount of information on air, water, vegetation, milk and urine concentrations over the next four days.

3.3 Other (ongoing) activities at Livermore which release radiation

Based on LLNL annual reports and other documents, we observe that tritium has been released to the local environment in sanitary sewage effluent and from an onsite

[1] This paragraph was written before we were aware that one of these classified documents had been reviewed by LLNL staff and that this review was made available to ATSDR last summer in response to the Public Health Assessment (ATSDR 2002). We have very recently received a copy of this review. Our preliminary comments on its implications will be made in Appendix B.

radioactive waste facility (Site 300) in addition to stack releases from Building 331 and other buildings (212). Numerous other radionuclides have also been released from the Laboratory (AEC 1961, Olsen 1973). Based on the information we have seen it is plausible that 80% of the total tritium releases from LLNL were during the 1965 and 1970 accidents, as stated in the assessment. It is important to note, however, that while the releases from building 331 during the two accidents were reported to be in the form of tritium gas, any releases of the water vapor form (HTO) would pose a more substantial health threat and should be weighed heavier. Much of the yearly emissions other than the two major accidents has been in the form of water vapor.

3.4 Other documents incorporated in this Health Assessment

Two earlier versions of this document were made available to us. One (ATSDR 2001a) was a very brief document that made assumptions about exposure, and drew conclusions about potential doses received, that were different from those used in the more recent version; this is discussed in more detail below. The other was a document that had a much broader focus, including routine releases from LLNL and the Savannah River Site. This document included a final report from an expert review panel convened by the ATSDR; the expert review panel report contained a fair amount of useful information about dosimetry and risk assessment that was very selectively applied by the authors of the current public comment draft health assessment (ATSDR 2002).

3.5 The public comment and review process

The public comment and review process as described in the health assessment began with the mandatory ATSDR assessment that was triggered by the addition of LLNL to the Superfund list in 1987. Although the Superfund listing was in response to volatile organic compounds in local wells, public concerns about tritium releases were noted. The broad document for LLNL and the Savannah River Site, mentioned above, did not explore in detail the two large accidents at LLNL and the health consultation (ATSDR 2001a) was intended to focus specifically on these accidents. This consultation was presented at a public meeting in November 2001 and received a distinctly negative review- meeting attendants found serious problems in approach, methodology and presentation of results. Comments were received and the report has been modified in response. In addition, a coalition of local organizations solicited funds from the Citizens' Monitoring and Technical Assessment Fund to hire technical consultants (us) that would review the document.

4.0 How we prepared this report

Following initial discussions with our sponsors – TriValley CAREs (Communities Against a Radioactive Environment), the San Francisco Bay Area Physicians for Social Responsibility (PSR) and Western States Legal Foundation (WSLF) – we reviewed the three ATSDR documents described above (ATSDR 2001a, ATSDR 2001b, ATSDR 2002), together with some further background material (see the reference list attached). In this review we asked:

- Is the picture presented in the reports, particularly the health assessment (ATSDR 2002) basically correct in its main features? in the details? Could more surprises be anticipated which would have the sort of magnitude found in the differences between the first draft consultation (ATSDR 2001a) and the second assessment (ATSDR 2002)?
- How well do the reports meet the charge of informing stakeholders, addressing their concerns, identifying gaps in information?
- Is the recommendation to do no further public health activities related to the accidents justified by the report?

In attempting to answer these questions we have organized the rest of the report as follows: in the next section (section 5) we present what we hope is a readable summary of the interpretation of the two accidents as made by ATSDR (ATSDR 2002) and we contrast it with their original interpretation (ATSDR 2001a). We were surprised by the discrepancies between the two. In Section 6 we discuss the uncertainties we have noted in that picture and describe the ATSDR treatment of those uncertainties. The Public Health Assessment presents its own, non-standard treatment of the literature on radiation-induced health effects. We review this presentation in section 7. In section 8, we present a summary of the results of our effort to recreate a portion of the ATSDR dosimetry model as part of our test of the ATSDR findings. Despite public requests, ATSDR made no attempt to estimate population doses, so in section 9, we provide some crude illustrative examples for population doses in the region and globally. We summarize our conclusions about the ATSDR effort in section 10. Technical details for sections 8 are put into an Appendix. After preparation of our draft of this report we obtained some further information about the 1965 accident (Peterson et al. 2002), and we provide some additional discussion of accident conditions in a second Appendix.

5.0 The story of the accidents and its interpretation as told in the ATSDR consultation and assessment

What we are told about the accidents in the ATSDR reports is that there were two accidental releases of tritium, at approximately 3:30 PM on January 21, 1965 and at approximately 6:30 AM on August 6, 1970. As described in the ATSDR health assessment (ATSDR 2002) the releases were entirely in the form of hydrogen gas. The earlier health consultation (ATSDR 2001a) in its calculation effectively assumes that 99% of each release was in the form of hydrogen gas (HT) and 1% was in the form of tritiated water vapor (HTO); the health consultation labels this a "conservative" assumption consistent with the accident report by Myers et al. (1971) and comments that there is evidence for conversion of HT to HTO through deposition to soil. We know little about what caused the accidents or how they were dealt with - the ATSDR assessment states that relevant DOE accident reports are not available or are still classified and we are left with two draft environmental impact statements that say the following about the two accidents:

> January 21, 1965 - 350,000 Ci of tritium were released from building 331 and vented to the atmosphere through a 30 m stack. The accident was the result of human error while working on a system containing ^3H gas under pressure. Most of the ^3H remained as $^3H_2$ gas rather than oxidizing to tritiated water. Thus, no significant exposures or deposition occurred, either on-site or off-site.
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> August 6, 1970- The failure of a component in a pressurized gas system containing tritium resulted in the loss of 300,000 Ci of $^3H_2$ to the atmosphere through the 30 m stack at building 331 (DOE 1978, 1986).

For the 1970 accident an article by Myers et al. (1973) describes radiation measurements taken in the immediate aftermath of the accident. It states that 289,000 Ci were released and gives meteorological information, some of which was used by ATSDR in the consultation and assessment (ATSDR 2001a, 2002). The health assessment (ATSDR 2002) concludes that "There are insufficient historical environmental sample data available to adequately evaluate these past exposures. Consequently this evaluation will use modeled data to determine past exposure concentrations." (ATSDR 2002 p. iii) There appear to be no available radiation measurements or meteorological information available for the 1965 accident. The health consultation (ATSDR 2001a) uses default values in their copy of the RASCAL model; this effectively applies a single, arbitrary weather scenario to the accident. The health assessment (ATSDR 2002) identifies a most common condition from a survey of local 1990-94 late January afternoon meteorological data and uses that wind speed, direction, and atmospheric stability to develop dose estimates.

The health consultation (ATSDR 2001a) and assessment (ATSDR 2002) now diverge in their description of how people became exposed. The health consultation considers the direct air exposure pathway most important. The idea is that after tritium was released in the accident the resulting radioactive cloud was carried and dispersed by the wind. People could have inhaled the tritium over a 30-minute time frame (until the plume blew away). Because water vapor is retained in the body, while hydrogen is immediately exhaled, inhaled HTO gives radiation doses about 10,000 times the doses from the same amount of HT. Consequently, the 1% of water vapor that is assumed by the consultation will contribute practically all of the radiation dose. The consultation estimates air concentrations of tritium at what it identifies as a maximum exposure location, at the site boundary .5 mi downwind from the release, and it calculates doses for an adult who inhales at standard rates that correspond to activity levels ranging from resting to strenuous exercise. So calculated, the doses for the 1970 accident range from 1 to 8 mrem (.01 - .08 mSv). Some uncertainty in these calculations is estimated quantitatively in the consultation and the authors present 95th%ile values that range from 3 to 19 mrem (.03-.19 mSv). The consultation asserts that similar calculations, using a hypothetical weather default which it characterizes as "health protective", give values of 1-4 mrem (.01-.04 mSv) for the 1965 accident. The consultation does not present risks corresponding to those doses, nor does it attempt to calculate doses to a population. It asserts that "Based on current peer-reviewed scientific literature, the one-time exposure to tritium resulting in a committed effective dose of 19 mrem (.019 mSv) is not considered a public health hazard nor are any adverse effects expected".

The health assessment (ATSDR 2002) uses a different, more complicated pathway for exposure. It assumes that all of the tritium was released as hydrogen gas and that a small fraction of that was captured by soil and vegetation, converted to water vapor, and released in that form back into the atmosphere. A second model was used to calculate doses resulting from a person inhaling these soil releases assuming this time an average mix of types of activities.

In the Health Assessment results are presented for doses to maximally exposed individuals at distances of one mile and two miles downwind from the release. The adult dose estimates for this pathway are 19 mrem (.19 mSv) at one mile and 8 mrem (.08 mSv) at two miles. Again some of the uncertainty is estimated quantitatively giving 95th%ile values of 50 mrem (.5 mSv) at one mile and 20 mrem (.2 mSv) at two miles. The health assessment also provides estimates for doses to a child (approximately 5 years old). Children's lungs are smaller than adults, but they breathe more rapidly and their bodies are smaller, so any amount of tritium inhaled is more concentrated in the body. The combined effect of these differences is higher doses for the child when both the child and the adult are in the same radioactive cloud. The child dose estimates from the assessment are 46 mrem (.46 mSv) at one mile and 20 mrem (.2 mSv) at two miles; the 95th%ile confindence values are 130 and 50 mrem (1.3 and .5 mSv) respectively. The health assessment also presents dose calculations for direct inhalation of the tritium as hydrogen, a very small dose, and doses from ingestion of tritium contaminated food items which they find to be approximately 1% of the inhaled water vapor doses.

For the 1965 accident, the health assessment treats the pathways for exposure in the same way that it does the 1970 releases: as noted above, it assumes a representative set of weather conditions drawn from sampling some 1990-94 weather data. The assessment asserts that maximum doses from the 1965 release are expected to be lower than for the 1970 release. As with the health consultation, no quantitative estimates are made of doses to populations, or of potential risks to individuals or populations. Both the consultation and the assessment assert that the measurements reported by Myers et al (1973) support the interpretation that the dose estimates are “conservative”.

Unlike the health consultation, the assessment devotes some attention to evidence of tritium-caused health effects. Their review does not follow standard practice in assessing the risks from exposures to low levels of radiation (as described, for instance, in the ATSDR Toxicological Profile for Ionizing Radiation (ATSDR 1999), and the National Academy of Sciences BEIR V report (NRC 1990)). They conclude:

> All of the adverse health effects from exposures to tritium (or low-energy external gamma radiation or x-rays) that we found in the medical literature occurred at much higher levels than the exposure levels we estimated for people living near the LLNL facility at the time of the accidental releases. Therefore, we conclude that inhalation and ingestion of tritium from those releases plus any chronic or long-term exposures were not public health hazards. Specifically these releases are considered to be no apparent health hazards which means that while some exposure probably did occur, those exposures are not likely to produce adverse health effects and are below levels of public health concern.

There is no interpretation or discussion in the health assessment of the differences between the consultation and the assessment in assumptions, modeling approaches, or the magnitude of the findings: The estimated doses to a maximally exposed individual in the assessment are almost an order of magnitude higher than the estimated doses in the consultation yet the consultation claimed that those dose estimates were “conservative”, and they were for an individual only half as far from the release. One technical puzzle in this comparison is that only about 0.2-.8% of the original amount of tritium released appears in the re-released water vapor for the dose to people at one and two miles; this is less than the 1% assumed to be water vapor in the consultation and seems hard to reconcile with a much greater dose. The explanation is that the deposition and re-release as water vapor keeps the HTO near the ground for more of its time in the close-in region and thus it is much more available for breathing. But, the explanation is not immediately obvious and should serve as a warning against assuming too quickly that a calculation is “conservative”.

The concluding recommendations provided in both the consultation and the assessment are based on the following argument: doses to a maximally exposed individual are small compared to the doses seen in studies that observe health effects from radiation, doses to other individuals are even smaller, and therefore there is nothing further to recommend in the public health arena.

6.0 Identification and discussion of the key uncertainties and missing topics in the studies and how they were addressed by ATSDR

There are uncertainties in almost every step of the progression of events that led to human exposures to tritium from the two accidents. It is often the case that a full list of uncertainties gives the impression that it is impossible to know 'the answer', even roughly. For this reason we would like to point out some of the limits within which we are operating: The lower limit to the question at hand is that no one experienced any health effects. This could be because no one inhaled or ingested any tritium or because the random nature of the potential diseases resulted in no effect. We find it extremely unlikely that no exposures occurred, but it is the least possible outcome. On the other hand it could be that there were modest exposures and, because of randomness, no actual health effects occurred. At the other extreme is the upper limit - the maximum exposure scenario. This is what the authors claimed was presented in the exposure assessment, but it should be noted that the extreme upper limit was not assessed[2]. One of the largest uncertainties that we discuss below is the chemical form of the released tritium; if all of the tritium coming out of the stack were tritiated water (HTO) the dose estimates would be much higher than they are under the current assumption of mostly tritium gas (HT) with some HTO evolving from soil oxidation[3].

Within this broad range of possible outcomes we believe that there is a narrower range of more likely outcomes; this range is defined by the uncertainties in the contributing variables. We list them chronologically, give our perspective on them and briefly discuss how the ATSDR dealt with each one.

- **Causes of the accidents**. As described in section 4.0 we know very little about the series of events leading up to the accidents or how the accidents were managed. These details are important to understanding the chemical form of the released tritium (third bullet). They may also have implications for monitoring and emergency response. To date there has been no publicly available independent review of LLNL reports on the accidents by ATSDR or another credible source.

- **Amount of tritium released**. Releases are stated in terms of kCi, 1000 Ci, indicating that the actual amounts released are not known precisely. ATSDR does not address this uncertainty (Myers et al (1973) state that 289 kCi were released in 1970 and ATSDR rounds this number up to 300 kCi, the number reported by the DOE). Without fuller accident reports, we have no particular reason to believe that the uncertainty is more than a few percent.

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[2] John Gofman, the founder and former head of the LLNL biomedical department, has given an estimate that the two accidents could have caused 60 deaths (Several 1987).

[3] It seems very probable, in our view, that most or all of the tritium coming out of the stack was in the form of hydrogen gas (HT). It should be noted, though, that standard practice for LLNL staff in the case of an accidental release is to estimate doses to offsite populations with the assumption that tritium releases are pure HTO, the extreme maximum exposure scenario.

- **Chemical form of the released tritium**. Tritium as a gas (HT) is less dangerous than tritiated water vapor (HTO) because water is more likely to be retained by the body once inhaled. The ICRP estimates that inhaled HTO is ~10,000 times more damaging per unit of exposure than inhaled HT, as noted in the consultation[4]. The short description of the 1965 event states that "most of the ^3H remained as $^3H_2$ gas rather than oxidizing to tritiated water" (DOE 1978), indicating that some of the tritium in the plume may have been in the form of tritiated water. The assessment (ATSDR 2002) assumes that all of the released tritium was $^3H_2$ gas, while the consultation (ATSDR 2001a) assumes that 1% was water vapor. Dilute hydrogen oxidizes quite slowly in the atmosphere (Brown et al. 1988). However, the immediate release conditions which involve a tritiated hydrogen bubble mixed with air and significant amounts of energy and ions available from radioactive decay are different and the possibility of more rapid oxidation or even ignition should be assessed by ATSDR. As a back-of-the-envelope estimate of how sensitive the model is to this uncertainty, consider the possibility that 1% of the tritium plume was HTO: The assessment, assuming 100% HT, estimates a dose of 0.02 mrem for an adult 1 mile from the point of release exposed to the plume for 30 minutes. This dose would be 2.02 mrem with 1% HTO, one hundred times the original estimate and ~10% of the estimated 12-day dose from inhaled HTO re-emitted by the soil. Clearly this is an important consideration and one with monitoring and emergency response implications[5].

- **Atmospheric dispersion of the HT plume.** This category includes both uncertainty in weather conditions and uncertainties in the model used to predict plume dispersion from these conditions (RASCAL 3.0). Meteorological information was available for 1970 but not 1965. Probable conditions for 1965 were drawn from 1990-1994 weather patterns for afternoons in late January. There was an asymmetry in the times the weather was sampled since the accident occurred closer to sunset than the midpoint of the sampling time. More importantly, no information on the uncertainty in possible 1965 weather patterns was developed since only one set of conditions (wind speed 2.4 m/s from 208^o) was used in the analysis. It should be possible to capture this uncertainty, for example, defining the proximate endpoint as 'number of houses in the close-in plume' and estimating the range of possible endpoints based on all observed wind conditions for late January afternoons 1990-1994. From this a 95% confidence interval of the number of exposed houses could be established. Variability in wind patterns would not affect the 'most exposed individual' approach but would affect population risk calculations. It is important to at least make an attempt at answering the obvious question "how often does the wind blow over the most

[4] The ICRP may have placed this factor of difference at 25,000 in the past (Okada and Momoshima 1993) and the current estimate of this factor is 10,000 (ICRP 2001).

[5] It is worth noting that the LLNL staff conduct preliminary offsite dose calculations for nonroutine (accidental) tritium releases with the assumption that all of the tritium is in the oxide form (HTO) (DOE 1991). Under this extreme assumption the 30-minute plume inhalation dose would be 200 mrem and there would be a further significant dose from deposition and re-emission.

densely populated area on late January afternoons?" In contrast, variability in stability can strongly affect the exposure to the most exposed individual since that will determine how much dispersion there is in the plume.

The significance of uncertainties in stability class can be seen in the sometimes contradictory discussions found in the two reports (ATSDR 2001a, ATSDR 2002). The default release conditions with neutral stability (Pasquill class D) are described in the consultation as "health protective" defaults "which will result in maximum transport with minimum dispersion". Yet the concentrations and doses are lower than those calculated for the 1970 accident which had a slightly lower release and was assigned stability class B. The explanation is that at the close-in distance of .5 mi the plume was so little dispersed that most of it remained well above ground. At greater distances, the modeled exposures would be larger and substantially larger than those calculated for the 1970 release; stability differences can contribute as much as an order of magnitude to differences in concentration even at distances of one to two miles. The assessment compounds the confusion by making both a mistake in its calculation and in its explanation. It asserts that the concentration calculated at .5 mi for the 1965 release is lower than that at 1 mi for the 1970 release (when, in fact the calculation gives a concentration in air about 3 times greater); and it goes on to explain this effect by asserting that there is more rather than less dispersion for the stability class C assumed for the 1965 release compared to the class B assumed for the 1970 release.

Even with some knowledge of the meteorology for the 1970 release there is good reason to consider uncertainty in stability. The release occurred early morning when there was a known inversion and relatively light winds. Although the EPA classification used by ATSDR assigns a class B to those conditions (because of the light winds), it is not unlikely that there was more stability, perhaps even moderately stable conditions (class E) left over from night time. In that case the plume would have held together for a substantial distance, the maximum exposed location could have been further away, and the radiation measurements described by Myers might have missed the early high concentration locations. This set of possibilities should be investigated with an eye to better describing the uncertainty in the dose evaluation.

In addition to uncertainties in the input parameters to a model like the RASCAL 3.0 model used by ATSDR to model initial plume dispersal, there is a remaining uncertainty which is how well the model represents the transport on a particular occasion. Experimental work that we have done at Clark University (Yersel et al. 1983) indicates that differences in peak concentrations at short distances by a factor of two is common, even for essentially identical meteorological conditions.

- **Deposition of the plume on soil or other substrates.** HT deposition velocity, in units of meters per second, is used to convert cumulative air concentrations to soil concentrations. The consultation uses a normal distribution with a mean value of 0.000802 m/s, a 10th percentile of 0.0004 and a 90th percentile of 0.0012, based on the mean, minimum and maximum from experiments conducted at Chalk River, Ontario (Brown et al 1988). We looked at both sources of data (Brown et al 1990, Ogram et al 1988) and determined that the data fit a lognormal distribution slightly better than a normal distribution, and this is what one would expect on theoretical grounds as well. This has little effect on the fitted distribution - we found a geometric mean of 0.0007 m/s, a 10^{th} percentile of 0.0004 and a 90^{th} percentile of 0.0011. We should note, however, that deposition velocities are notoriously uncertain and variable from location to location, so extrapolation from Ontario to California should be assumed to introduce additional uncertainty and we use a broader range in our analysis.

- **Oxidation in soil and emission as HTO.** The rate of HTO loss from soil was set at 1%/hr in the consultation. Again we looked at the data in Brown et al 1990 and Ogram et al 1988 and found sufficient data (22 data points) to create an uncertainty distribution around this point (a geometric mean of 0.45%/hr and a GSD of 1.6, or 10^{th}, 50^{th} and 90^{th} percentiles of 0.24%, 0.51% and 0.76%, respectively). This would produce lower estimates of inhalation doses at any given time but would not substantially alter cumulative inhalation doses. This being the case, we recognize that the difference between a soil loss rate of 1% and a rate of 0.5% is insignificant and accept this parameter choice.

- **Atmospheric behavior of HTO.** We have the same concerns about this part of the model that we had about the 1965 weather conditions. The behavior of HTO in the breathing zone was defined by another submodel, the Industrial Source Complex Short-term air dispersion model. Like the RASCAL model, it uses meteorological inputs to estimate air concentrations. Again the ATSDR staff used weather data from 1990-1994 to establish the model inputs, apparently using August data for both accidents. In this case they chose a 12 day scenario from a particular year which maximized risk (produced the highest air concentrations). Since only five years were sampled this gives a limited idea of the uncertainty that could be expected for a dispersion in a 12 day period of another year. A fuller uncertainty analysis would have been informative.

- **Inhalation of tritium.** The consultation considers two exposure routes, inhalation and ingestion. Inhalation of HTO is the dominant route, but inhalation of HT provides an additional dose, as does ingestion of contaminated food. The inhalation rate is calculated for an adult assuming a mix of activities. We believe that the adult breathing rate and its associated variability are appropriately described. This breathing rate was then applied to 5-yr olds; we feel that more age-appropriate data should be used, and this would produce a lower dose estimate for children. We also feel that the inhalation rate of a neonate should be evaluated for a neonatal dose estimate. However, the major problem with the ATSDR model treatment is that variability in breathing rate is combined with uncertainty in other parameters. The result is that there is confusion between having a confidence level that everyone's dose would have been lower than some value, and describing a situation in which a certain percentage of people will be known to have higher doses.

- **Ingestion of tritium.** The consultation uses published measurements of tritium in milk and produce following the 1970 accident, along with standard reference values of ingestion rates, to estimate exposure through this route. We agree with this approach especially since ingestion contributes a relatively small amount to the dose (however, uncertainty still should be distinguished from known differences between people.)

- **Other exposure routes.** We feel that drinking water and groundwater should have been specifically evaluated, perhaps in a very rough way, to demonstrate that these routes are negligible. More importantly, we feel that exposure routes relevant to very young populations should have been considered. These include breast milk (for newborns and infants) and fetal exposures. A fetus would be exposed to tritium that was inhaled or ingested by the mother; since tritium is assumed to rapidly equilibrate in the body (as water), the fetal dose would be based on the same concentration as the maternal dose. This is discussed further in the next bullet and in section 6.

- **Dosimetry of inhaled tritium.** There are a series of uncertainties associated with dosimetry that can be folded into the 'dose coefficient', a value that translates the amount of radioactive substance in the body (in units of curies or becquerels) into a dose (in units of Sieverts or rem). We propose an alternative, slightly different, dose model that has similar uncertainty bounds to the dose model used in the consultation, though with somewhat different implications. This also is an important arena in which one should distinguish between known variability and uncertainty as in the case of inhalation coefficients Even more importantly, it is not the case that there is tremendous uncertainty in the residence time for tritium in the body: what is known is that there is both a short-lived and a long-lived component, and both should be included for each person's exposure. We mention above that fetal exposures should be considered explicitly; the National Radiological Protection Board of the UK estimates that the fetal dose can be assumed to be 1.4 times the adult male dose for a given exposure scenario (NRPB 1995). This is discussed further in section six.

- **Health risks of tritium exposures.** There is a great deal of uncertainty about the health effects of radiation even though it is one of the most-studied and best-understood health hazards that we know of. A consensus scientific opinion has emerged, however, that any amount of radiation, no matter how small, can lead to adverse health effects. Uncertainty remains about the shape of the dose-response curves for various endpoints. This section of the consultation ignored both the majority of scientific evidence and the well-established norms in the field of radiation protection by attempting to define thresholds for harmful effects including cancer. Uncertainty in the dose-response relationship was of course avoided along with any discussion of such a relationship.

Finally, we should comment on the use of conservative assumptions in the assessment. The idea of making a conservative assumption is very natural: when in doubt, give the benefit of the doubt to the health protective side of the analysis. There are numerous situations in which it is easier to estimate an upper ("conservative") bound than to create a best estimate with uncertainty, and sometimes it is appropriate to do so. However, information about how well you know the quantity you are calculating gets lost and communication becomes more difficult. The real dangers appear when you start swapping uncertainties: you have been "conservative" for this part of the calculation so now you feel that you don't need to be too careful with another part. At this point it is very hard for others to evaluate the calculation or to have any idea of how "conservative" it is. At worst, being "conservative" can make one overconfident and miss some important alternative. The doses in the health consultation (ATSDR 2001a) were presented as "conservative" yet they proved to be significantly lower than the dose estimates in the assessment (ATSDR 2002) because they used a different set of assumptions about the pathway. The best solution is to make a thorough effort to assess and keep track of uncertainties, to limit the use of conservatisms to situations when they make a real simplification, and to exert care not to make excessive claims about the degree of conservatism.

7.0 The ATSDR assessment discussion of health risks

The amount of evidence concerning radiation health effects is large and it substantially supports the idea that there is no threshold for carcinogenesis from exposure, meaning that any amount of radiation can be harmful. This is the 'establishment' position on the issue and every major standard-setting body and research committee, including the ATSDR, has agreed on this principle (NRC 1990, ICRP 1991, ATSDR 1999, UNSCEAR 2000). Our central critique of the risk section of this consultation is that it ignores this principle by attempting to define thresholds for harmful effects including cancer. The general problem with such an approach is that any major deviation from a standard approach requires explanation so that the deviation is compared with the standard approach, and justification with good reasons why the agency chose to take this particular path. Both explanation and justification are lacking in the assessment. Our specific observations below are that the authors overlooked a large amount of meaningful, relevant literature on the issue. We think that this very important section should be addressed methodically, so we react to one paragraph at a time:

Paragraphs one and two. We agree that animal studies of tritium effects and human and animal studies of the effects of exposure to gamma radiation and x-rays are the only sources (and good sources) of data bearing on the effects of tritium in humans. Of course, we also believe that where human data are available they should be the principle evidence considered.

Paragraph three. The statement that "the lowest tritium dose capable of causing adverse health effects is 3 rad" is contrary to the no threshold approach and an irresponsible claim. Even if we consider this to be a dose below which no health effects have been observed we find that it is simply not true (see below). Furthermore, this value is based on mice exposed *in utero* when there are several studies available that address *in utero* exposures to low-dose, low-LET radiation in humans.

Paragraphs four and five. In these paragraphs the authors describe a linear no-threshold dose-response for negative effects on intelligence after *in utero* exposures. As stated above, we consider this to be the standard model for cancer and can easily conceive of such a relationship existing for other health effects as well. Unfortunately the authors draw an erroneous conclusion. At doses being considered here, intelligence would only be changed by a fraction of a point on the scale used to evaluate the A-bomb survivors. The consultation states that since this would not be measurable there would be no effect. However, even effects that can't be measured can still be present. In this case, if the model being cited were correct, there would be a small effect on intelligence. At low doses such an effect would not be measurable at the individual level but if sufficient numbers of people were exposed, it could manifest itself as a change in the distribution of intelligence based on a similar scale (more students requiring remedial education or fewer 'honors' student). We should also mention that if you consider lost intelligence to be a health effect than this admission of a no-threshold dose-response contradicts the threshold mentioned in paragraph three.

Paragraphs six and seven. These paragraphs deal with cancer following adult and *in utero* exposures, respectively. The threshold for leukemia after adult exposures is placed at 100 rad and the threshold for cancer after fetal exposures is placed at 10 rad. Lower doses of 20 rad (adult) and 1 rad (*in utero*) are mentioned but disregarded.

Again we would like to point out that the consensus about radiation carcinogenesis is that there is no threshold for cancer. That said, the field of epidemiology has provided numerous examples of cancers attributable to radiation doses well below 20 rad. These are plentiful enough to have been reviewed several times (for example Ron 1998, Schbauer-Berigan and Wenzl 2001). In one of the most comprehensive analyses of existing literature that has been undertaken Wilkinson and Dreyer (1991) pooled the data from seven nuclear worker cohorts in the US and the UK and detected an excess of leukemia in the group exposed to between 1 and 5 rem. The A-bomb survivorship have also shown excess cancers at low doses; Pierce and Preston (2000) point out that most of the survivors had doses less than 20 rem. They emphasize that “ there is direct, statistically significant evidence of risk in the dose range of approximately 0-0.10 Sv (0-10 rem)”. Clearly a 100-rad threshold for cancer following adult exposures is unjustifiable.

Cancer following *in utero* exposures has also been extensively studied and the body of evidence reviewed. The pioneering work of Alice Stewart with prenatal x-rays (Stewart et al 1956) grew into the Oxford Survey of Childhood Cancers and its results were quickly reproduced in the US (MacMahon 1962). Much more evidence has piled up supporting sensitivity to radiation during fetal development and one of the recent reviews concludes that “radiation doses on the order of 10 mGy (1 rad) received by the fetus *in utero* produce a consequent increase in the risk of childhood cancer” (Doll and Wakeford 1997).

Paragraph eight. This paragraph suggests that low doses of radiation might increase the lifespan of exposed individuals and that the doses received downwind of LLNL would not shorten lifespan. We acknowledge that there are proponents of this life-increasing hypothesis; this is a controversial suggestion, evidence is not coherent on this issue, and it is in any case a distraction.

Paragraph nine. This introduces the idea that women and children are more vulnerable to the effects of radiation exposures than the reference man. We agree that children, exposed to the same air concentrations as an adult, will receive a higher dose, and this probably true of fetal doses as well. This is explained in the dosimetry section of Appendix A. It may also be the case that a woman will receive a higher dose than a man based on a smaller average body weight (Richardson et al 2001a).

Higher doses are half of the problem. The other half of the problem is increased sensitivity to a given dose. Human exposures early in life (*in utero* or as a young child) and late in life (over age 50) have been associated with higher risks of cancer, although this is complicated by the fact that the pattern of risk seems to be different for different cancer sites (Richardson et al 2001b, Ritz et al 1999). There is also animal evidence for this effect (Walinder and Sjoden 1973, Sasaki et al 1978, Di Majo et al 1990, Benjamin et al 1991, Sasaki 1991, Van Den Heuvel et al 1995). Another possible sensitivity was described by Nomura (1984) looking at mice. This study observed that fetal x-ray exposures made mice hypersensitive to chemical-induced lung cancer later in life, an effect not observed with postnatal x-ray exposures. Despite the extensive literature on this topic, the increased sensitivity of young and old individuals to radiation was not explored in the consultation.

Paragraphs 10 through 13. These paragraphs mention genetic effects in a muddled way. Genetic effects, also called heritable effects, arise from mutations in male or female reproductive cells and are expressed in the offspring of those exposed. Sperm count reduction and effects of *in utero* exposures are described although they are not genetic effects. This consultation does not in fact mention any of the evidence for genetic effects, much less discuss the magnitude of genetic risks, despite an enormous amount of available information (Chapter 2 of BEIR V is devoted to the topic). Animal data is of course the bulk of the evidence, including studies that show heritable cancers (Nomura 1982, Mohr et al 1999). There are also a few human epidemiological studies, including a controversial study of leukemia in the offspring of exposed nuclear workers (Gardner et al 1990) and a robust pair of experiments looking at germline mutations around Chernobyl and the Semipalatinsk nuclear test site (Dubrova et al 1996, 2002). Much of this evidence is difficult to translate into a risk estimate, but Straume (1993) methodically laid out the categories of genetic risk and estimates a comprehensive genetic risk based on the data in BEIR V. This number could have been easily used in the consultation to present a ballpark genetic risk estimate for the accidents.

The final paragraph concludes that “the doses to all members of the Livermore community are not at levels of public health concern”. We find this statement to be as ill-grounded as the discussion that precedes it; furthermore, the doses and risks in question may have been relatively low, but to dismiss community concerns so trivially is nothing less than an insult.

8.0 Our reconstruction of the ATSDR Monte Carlo model for a maximally exposed individual

8.1 Exposures

The exposure models run in the Assessment were beyond the scope of our work, but we did note a couple of interesting inconsistencies in the model output tables and the corresponding dose estimates. On page 10 of the Assessment we read that:

> This evaluation focuses on the potential tritium exposure to the hypothetical maximally exposed individuals for each release. These individuals are assumed to be 0.5 miles from the LLNL tritium facility for the 1965 accident and 1 mile for the 1970 accident.

We later read that 1965 doses were lower than 1970 doses at all locations and this is the justification for using the 1970 doses as the maximum of the two exposures. However, if 1965 doses at 0.5 miles were higher than 1970 doses at 1 mile then the earlier doses should be used as the maximum. This is in fact the case; we see from Tables A-2 and A-3 that the cumulative HT concentration at 0.5 miles on the centerline of the 1965 plume was 9.27 Ci-sec/m3, roughly three times greater than the concentration at 1 mile on the centerline of the 1970 plume (3.27 Ci-sec/m3). We therefore included a multiplying factor in our dose model of 2.83, the ratio of the 1965 0.5-mile air concentration to the 1970 1-mile air concentration. We also added an uncertainty distribution around this point (90% confidence interval of 2.0 to 4.0); this was added as a minimum estimate of meteorological uncertainty- it has been observed that the same weather conditions can produce behavior that varies by a factor of two (Yersel et al. 1983)

8.2 Dose estimates

Although we were unable to exactly recreate the atmospheric parts of the ATSDR assessment model (exposure) we were able to recreate the dose estimation part of the model. Our approach was to deconstruct the dose equation presented in the assessment and substitute our own parameters where we felt it was necessary. Appendix A more fully describes the dose model and parameter choices.

Our results suggest that the dose estimate for the most exposed adult should be four times higher than the value reported in the Assessment. The neonatal dose estimate would be roughly equal to the adult estimate; neonates have a lower breathing rate (in terms of cubic meters per day) and a faster elimination of tritium from the body, but they also have a much lower body weight, and in this case the differences cancel out. It should be kept in mind, however, that neonates would presumably receive additional exposure through breast milk. The dose estimate for the most exposed 5-yr old should be higher than the adult estimate; our value for 5-yr olds is three times higher than the ATSDR estimate. Our dose estimate suggestions and those reported in the Assessment are shown in Table 1.

Table 1: Estimated 12-day HTO inhalation doses for the maximum exposure scenario.

Exposure age	Reported in ATSDR Public Health Assessment for 1970 tritium plume (dose in mrem, 95th %ile)	Our estimates for maximum exposure for 1970 tritium plume (dose in mrem, 90% CI)	Our estimates for maximum exposure for 1965 tritium plume (dose in mrem, 90% CI)
Adult	19 (50)	29 (7, 115)	82 (20, 333)
5-yr old	46 (130)	47 (12, 177)	134 (34, 559)
3-month old		29 (8, 110)	83 (21, 326)[6]
Fetus		41 (very uncertain)	115 (very uncertain)

These dose estimates may still underestimate the true maximally exposed individual dose in a couple of ways. One follows from the discussion of the RBE factor in Straume and Carsten (1993); they suggest that the RBE estimates may underestimate the true RBE for tritium since the studies used in assessing this value were obtained using relatively high doses and dose rates. This possibility means that tritium beta radiation may be slightly more damaging than we assume and therefore that the doses received may be slightly higher. This cannot be quantified but should be kept in mind.

8.3 Risks and Health Impacts

As discussed above the ATSDR consultation failed to present a meaningful picture of the health risks of radiation. Given that any amount of radiation is expected to have the potential for harm it should be expected that the public will want to know how much harm, however uncertain the answer may be. Risks were not quantitatively evaluated in the ATSDR report despite the fact that the expert review panel spelled out a way to perform these calculations[7].

We made rough risk estimates for a specific scenario- risks from the inhalation of tritium in the form of tritiated water (HTO) at the ATSDR-defined maximum public exposure point of 0.5 miles from the release directly in the center of 1965 plume passage. Our risk estimates do not include the risks from ingestion of HTO or organically-bound tritium (OBT) or the risks from inhalation of tritium gas (HT). We present these estimates as reference points to a) demonstrate the feasibility of making such calculations, and b) indicate the general range of expected individual risk based on the primary exposure route. The inputs to our risk calculations are discussed in Appendix A.

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[6] We have left out the dose that a newborn would receive from its mother through breast milk; this could be significant but we did not feel that we had enough information to make this estimate.
[7] Osborne et al 2001. Straume (1993), one of the expert panel, had previously published a separate illustration of tritium risk assessment that we found very useful in addition to the expert panel report.

In our risk equation we used the BEIR V risk coefficients for leukemia and for other cancers. We calculated risks of non-leukemia cancers two ways, using the DREF and leaving it out. Our risk results for the most exposed individual are shown in Table 2.

Table 2: Risks of leukemia and other cancers for the 'most exposed individual' scenario (a neonate, 5-yr old or adult at the LLNL fence directly in the path of the 1965 tritium plume), assuming the only dose received is from inhalation of tritiated water vapor (HTO).

Expected mortality per 100,000 exposed (90% CI)				
Age at exposure	Leukemia	Other cancers assuming DREF	Other cancers assuming no DREF	Leukemia + other cancers (with DREF)
Fetus				(roughly 6 fatal lifetime cancers)
neonate[8]	2.0 (0.24, 17)	11 (2.0, 60)	24 (5.0, 110)	13 (2.2, 77)
5-yr old	3.2 (0.37, 28)	19 (3.2, 100)	39 (8.0, 190)	22 (3.6, 130)
adult	0.56 (0.08, 4.0)	8.9 (1.5, 47)	19 (3.7, 120)	9.5 (1.6, 51)

We would like to point out several things about these results:

- These risk estimates are slightly lower than they would be if we considered all exposure routes including ingestion of water and contaminated food. If the ATSDR report is correct as far as the proprtional contribution of exposure routes then the doses and the risks would be 1-2% higher.

- Any individual exposed to tritum from both the 1965 and the 1970 releases would of course have higher risk than an individual exposed only once.

- We have left out ingestion exposures here, and this omission is particularly sensitive for neonates who would be exposed to a significant amount of tritium in breast milk. In addition, the neonatal risk estimate is based on the risk coefficient for 5-yr olds, the youngest group for which this coefficient was reported, and it is likely that neonates are more susceptible to radiation-induced damage than five-year olds. The actual risk to neonates are therefore underestimated here.

- In the case of leukemia, BEIR V reports that older adults may be at greater risk than younger adults. We have not included these risks in our table, but the risk coefficients for older adults (55+) are roughly twice the values for younger adults (25-45).

[8] Neonatal risk based on the BEIR V coefficient for 5-yr olds, the youngest group tabulated.

- For regulatory purposes the EPA defines the upper bound of acceptable risk as 10^{-5}, or 1 death out of 100,000 exposed. The best estimates of risk presented in our table are above this threshold and we cannot with 95% certainty rule out risks in the 10^{-3} range, a level of risk that is of concern to regulatory agencies and communities.

9.0 Population Dose Estimates

In general, estimates of maximal individual doses and estimates of population doses provide complementary information for understanding an environmental hazard. The individual dose estimate provides an answer to a question about what was the maximum threat to a person. A population dose estimate answers a question about the overall burden on a community. In an area of low population, even moderately high individual doses can coexist with relatively small population doses; conversely, small individual doses applied to a large population can make up a substantial population dose. Despite community requests, the Public Health Assessment (ATSDR 2002) does not make an attempt to estimate doses to a population. Within the limited scope of our review and with our lack of immediate access to site-specific atmospheric modeling capabilities, it has not been possible for us to correct this omission. However, we provide some very crude estimates for population doses for three geographical regions – within 50 miles of LLNL, within the U.S. for the first pass of the tritium cloud, and throughout the Northern Hemisphere for the life of the tritium in the atmosphere. We provide these because even very crude numbers provide a context for interpreting the effects of the accidents. In addition, by making these calculations we show that it would be possible to do a more refined version of the analysis. Also of interest would be estimates of population doses to the vicinity of Livermore – say within 5 miles of LLNL. We have chosen not to attempt such close in estimates, partly because the results would be very sensitive to model choice, and, even more importantly, because the uncertainties in meteorology are particularly significant for close in doses. As noted previously, the conditions during the release for the 1965 accident are unknown[9] and the different possibilities for stability class, class B versus class E, for the 1970 accident can have a very substantial impact on the close in doses from that accident and also on where the doses would have been received.

For population doses within 50 miles of LLNL, our approach has three steps: 1) to estimate the amount of tritium deposited on the ground and re-emitted as water vapor within 50 miles for the 1970 accident; 2) to use typical LLNL model calculations for tritiated water vapor releases from LLNL as surrogates for population dose estimates from the water vapor emissions which are spread out across the fifty mile region; 3) to consider the limitations in such an estimate. Based on the range of deposition velocities, described in section 6, we expect that roughly .7 – 7 % of the tritium was deposited within 50 miles and we expect that most of it was subsequently re-emitted in the form of water vapor. This corresponds to 2,000 – 20,000 Ci of tritiated water vapor released for the 1970 accident; we double these values to include the 1965 accident. Each year LLNL models the contribution of tritiated water vapor releases from LLNL to a population radiation dose. These values depend on the weather conditions for the particular releases and different years give different relationships between amounts released and population dose: in 2001, an 18 Ci release gave a population dose of .16 person rem, while in 2000 a 36 Ci release gave a population dose of .47 person rem. Within the very large uncertainties that attend these calculations, it is reasonable to assume a rough

[9] Except see the discussion of new information in Appendix B.

correspondence: 100 Ci lead to 1 person rem in dose, for present day releases. The population 30 years ago, was, however, substantially smaller. We thus estimate for the two accidents combined a population dose in the range 15 – 300 person rem. Probably the biggest uncertainty in this estimate is the rate of deposition (characterized by a deposition velocity), but there are many other important uncertainties: the key aspects of the weather that relate to deposition are not known for the 1965 accident and are uncertain for 1970; different weather is used for modeling dose estimates from the amounts of released tritiated water vapor; furthermore we have not evaluated the validity of using the LLNL model for local releases as a surrogate for estimating the impacts of the widely dispersed re-emitted water vapor.

For estimates of population doses in the North American continent during the initial pass of the tritium cloud, we begin by estimating that the amount of deposited tritium should be 10 or more times the amount deposited in the initial 50 miles. Population densities for much of the travel time of the cloud will be appreciably lower. We thus estimate population doses of 70 – 1500 person rem.

Our approach in estimating population doses throughout the Northern Hemisphere over the lifetime of the released tritium has a similar spirit. While direct conversion of dilute hydrogen gas t water vapor is quite slow, deposition to soil, conversion to water vapor, and subsequent release occurs on a time scale of 0.5-1% per hour. Consequently an appreciable fraction of the tritium becomes water vapor even on its first pass across North America. Most of the tritium in the lower atmosphere goes through this cycle many times over its radiological life. Again we use surrogate estimates. In this case we take estimates summarized by Eisenbud (Eisenbud 1987 p. 406) for the annual doses in the Northern Hemisphere resulting from 3600 million Ci released in nuclear weapons testing and from the steady state reservoir of about 13 million Ci (Northern Hemisphere) from natural (cosmic ray) tritium production. These annual dose estimates are 2 mrem/yr and .001 mrem/yr respectively. Note that there is a seven-fold difference in the dose/Ci estimates. To calculate the total doses attributable to the presence of a Ci in the Northern atmosphere, we should multiply by the average lifetime of tritium, a factor of 17 years (corresponding to the half-life of 12 years) and by the population of the Northern hemisphere in the period around 1970. The resulting estimate is a population dose range of 3000 – 20,000 person rem.

The BEIR V (NRC 1990) risk relations discussed in section 8.3 provide a basis for estimating the numbers of cancers that can be expected, allowing for randomness from these dose estimates. For the 15-300 person rem estimated to have been received within 50 miles, 0 or at most one fatal cancer would be expected. For the 70 – 1500 person rem estimated to have been received in North America during the initial pass of the cloud, 0 to 2 cancers can be expected. For the global estimated exposures of 3000 – 20,000 person rem, 3 to 25 cancers is the mostly likely range of expected numbers of cancers, but 0 or more than 25 cancers cannot be excluded.

10.0 Our Findings and Recommendations

Our observations about the ATSDR assessment (ATSDR 2002) fall naturally into three categories: observations about exposure and dose estimates, observations about health and risk implications, and observations about the process of creation of the assessment and its usefulness.

10.1 Observations about estimates of exposures and doses from the two accidents

- The ATSDR assessment (ATSDR 2002) appears to have identified the most significant pathway for exposure from releases of tritium in the form of hydrogen; it is the inhalation of tritiated water vapor released after capture of the tritiated hydrogen by soil and vegetation. This is an important finding and merits emphasis. It contradicts the approach used in the earlier health consultation (ATSDR 2001a). The assessment dose estimates for nearby exposures are surprisingly large compared to those in the consultation, but they are plausible. The discussion of uncertainties in the dose estimates, however, is limited and misleading.

- The unavailability of detailed information about both accidents is very unfortunate. No information has appeared about the 1965 accident[10] so we do not even know what direction the wind was blowing, whether LLNL made any radiological measurements, or had any sort of emergency response. One article after the 1970 accident (Myers et al.) describes the meteorological conditions and a significant effort at monitoring as a follow up to the accident. No detailed work has been done to combine modeling with the monitoring report to reconstruct a better picture of what happened after the release.

- The assessment gives an inadequate treatment of key uncertainties in the modeling effort. The most important of these are: 1) a failure to consider the implications of uncertainties in the actual meteorology at the time of the accidents: a particular wind speed, direction, and stability assumption is used for the 1965 accident without considering what doses might be expected for alternative conditions; uncertainties in stability conditions could be significant for modeling the 1970 accident as well; 2) deposition velocities are more uncertain than the three-fold factor considered by ATSDR and they directly affect the availability of tritiated water vapor for inhalation; 3) the treatment of retention time in the dosimetry is incoherent: it is not the case that we don't know whether retention times are 1 day or 40 days; rather we know that there is a short and long component to retention times, that there is some variation between people in the relative importance of those components, and we have some uncertainty about times and relative importance within this picture; 4) similar incoherence appears in the treatment of other dose-affecting parameters like breathing rates, activity

[10] Except see Appendix B

patterns, and body weight; there is variability in these, but not much uncertainty and the modeling should reflect this.

- The health assessment gives a description of "conservative" or "health protective" assumptions that clouds the interpretation of its findings, especially since there is no complete or coherent treatment of uncertainty and no assessment of the magnitude of the conservatisms.

- The health assessment makes mistakes in presenting its results. In particular, the models predict higher rather than lower doses for the 1965 accident contrary to the assertions in the text and tables.

- The assessment only calculates doses to "maximally exposed individuals" at various distances close to the accident. We have made some very crude sample estimates of doses to populations at the regional and global spatial scale. These are 15-300 person Rem for within 50 miles, 70 -1500 person Rem within the U.S. and, very roughly, 3000 – 20,000 person Rem in the Northern Hemisphere (including the U.S.) in later months and years.

10.2 Findings regarding risks and health effects

- The assessment provides its own interpretation of the literature on radiation-related health effects. This interpretation is inconsistent with standard practice, exemplified by the ATSDR toxicological profile on Ionizing Radiation (ATSDR 1999) and the National Academy's BEIR V report (NRC 1995). The assessment discussion does not relate its approach to standard practice, nor does it consider other alternative interpretations of the literature.

- The assessment and the consultations use the term "below levels of public health concern" in a number of places, including in its conclusions about potential risks. There are three serious problems with this usage:

    1) The term is nowhere defined, nor is there any indication of what would be a level of public health concern.
    2) Risks calculated using standard practice from the radiation doses presented in the assessment are at levels that are generally taken to be significant by the agencies supervising Superfund clean ups.
    3) Most disturbingly, the inferences drawn by ATSDR directly subvert the ALARA principle (as low as reasonably achievable), a cornerstone of the social compact for managing radiological hazards.

  The impression left by the ATSDR documents is indifference to releases of 300,000 Ci of tritium (in the form of hydrogen, or 10,000Ci in the form of water vapor) in a highly populated area.

- The estimated risks associated with these accidents were arguably significant. The assessment does not calculate risks from its dose estimates; based on its interpretation of the health literature, it asserts that there were not likely to have been health consequences from the releases. Using standard methods, we have calculated risks to the maximally exposed individuals and estimated the potential for health impacts in the population. We find that within the uncertainty bands, there was the potential for risks that are considered “significant” in common regulatory practice – that is cancer mortality risks to “maximally exposed individuals” that may exceed 1 in 10,000. We also find that there were potential health consequences in a broad population: using standard estimates and allowing for some uncertainty in the standard approach, most likely there were 0 to 1 cancers caused by the accidents within 50 miles, 0 to several cancers in the U.S. and 0 to 50 in the Northern Hemisphere. These risks and likely health consequences are relatively modest in comparison with other human encounters with natural and man-made radiation (such as that from weapons testing) and could not be explicitly linked to the accidents in a health study.

10.3 Findings regarding the process of creating the assessment

We have not attempted any study of the history behind the creation of the assessment (and other documents). However, we do have concerns about process that we take from the content of the documents.

- To begin with, we find no clear statement of what questions are being answered within the assessment, why these answers address stakeholders’ need for information, or why the calculations are done the way they were.

- As we have noted above, continuity among the documents is lacking. New documents that represent ongoing agency analysis should explicitly address their relationship with previous work. When new approaches are used or there are new findings, the reasons for the changes should be explained and placed in a context of ongoing work. In the absence of such explanations the agency creates confusion and undermines trust.

- There is very little evidence within the assessment of the “give and take” in problem definition and interpretation of results that one would expect of an analytic and deliberative process involving stakeholders.

10.4 Discussion and Recommendations

The first two points in the findings regarding health risks constitute serious flaws in the draft; if they are left unaltered they would represent a significant departure by ATSDR from its public health mission. Both should be corrected in the final version of the assessment. The potential risks from major releases close to a highly populated area can be described without exaggeration but in a way that respects the ability of the public to digest quantitative information and that supports an ALARA approach.

Failures in process and loss of trust are not easily corrected; however, there is still an opportunity to take steps in the right direction in correcting the draft assessment; such steps would be well received. These would include correcting the above two deficiencies and making efforts, perhaps through the regional ATSDR office, to assure that the full story of the accidents becomes available and that appropriate lessons are drawn for future operations and monitoring. If instead ATSDR persists in avoiding its public health mission, the likely consequences are worsened community relations with LLNL, greater distrust of government, and unnecessary community fears.

An appropriate set of recommendations might be based on the following observations.

- Recommendations should be developed through a collaborative process involving stakeholders.

- The stakeholders are entitled to a more complete story of the accidents with some outside review of the information provided by LLNL. ATSDR could facilitate this.

- A health study that would directly relate health effects specifically to the accident releases is not feasible

- There are lessons that could and perhaps should be drawn from the accidents regarding emergency planning, environmental monitoring, and, conceivably, health monitoring. .

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Appendix A: Dose and Risk modeling.

In evaluating the dosimetry used in the assessment our approach was to:

a) recreate the Monte Carlo dose coefficient simulation reported in the assessment,
b) recreate an alternative simulation reported by D. M. Hamby (1999),
c) construct our own dose coefficient model for Monte Carlo simulation, and
d) make alternative dose estimates based on the ratio of our dose coefficient to the dose coefficients described in the Public Health Assessment.

What is a Monte Carlo Analysis?
Before addressing these steps a description of Monte Carlo analysis should be laid out. A Monte Carlo analysis is used to estimate the uncertainty in a value, for example cancer risk from tritium, by using the uncertainty in all of the factors contributing to that value. In the case of tritium exposures some of the factors contributing to risk are breathing rate, the energy of a tritium beta particle, and the cancer response seen in atomic bomb survivors. The factors can be listed in an equation (for example, risk = $a * b * c$) and the equation can be solved by using the most likely value of each factor; this would produce a single value for an answer. This single value would be misleading because it would carry with it the implication that the answer is known with a lot of certainty.

In fact, the answer is very uncertain because some or all of the contributing factors are uncertain. A Monte Carlo analysis takes this into account by solving the equation many times over, each time drawing from a distribution of possible values for each factor. The distributions for the factors in our tritium model are shown in figure A1. The first picture in that figure is of the distribution for a tritium quality factor (this is a number used to adjust the dose upward to account for the fact that one rad of tritium beta particles is thought to be more destructive than one rad of gamma or x-rays). The Monte Carlo software would sample this distribution randomly and repeatedly, usually picking a value close to 2.25 but occasionally picking something as low as 1 or as high as 4. You can see that below the picture of the distribution there are numbers given for a 2.5 percentile (1) and 97.5 percentile (3.5). These describe the 95% confidence interval- 95% of the time the software will generate a number between 1 and 3.5.

In the end, after 5,000 or 10,000 times solving the equation, the software will have generated 5 or 10 thousand solutions, and these will also be distributed around a most likely value as most of the contributing factors are. When we report a 90% confidence interval we are in effect saying that out of these solutions, 90% were between value *x* and value *y*. This implies that we are 90% confident that the true answer is within the stated range.

It is important to point out the distinction between variability and uncertainty. Uncertain parameters are those which we don’t know very well. For instance, we are not sure how many excess cancers were caused by atomic bomb radiation or about the shape of the dose-response curve because cancer is a random process that occurs naturally. Variable factors, such as body weight, are different- we actually know reasonably well how much the average person weighs and how often the extremes in weight occur. We could run the model for a fat person or a skinny person, and the answers would be different, and each would be embedded in a range of uncertainty from other factors. What is often done, and what is done here, is to run a model in which uncertain and variable factors are all treated the same. The reported best estimate of the answer is therefore the most likely dose or risk, evaluated for an average person, and the uncertainty distribution is a tossed salad of potential answers for a mixed population including a series of truly uncertain factors like a-bomb survivor risk or the tritium quality factor.

Some of the most significant variability is teased out of the mix by evaluating doses and risks separately for adults and for children- these two groups have obviously different body weights, breathing rates, and sensitivity to radiation. What we include in this case is the variability of these factors within each group, for example the variability in weight among five-year olds.

There are several types of distributions discussed below. One is the normal distribution, the familiar bell curve. In this case the best guess is the mean in the center of the curve and the variability is described by a standard deviation- there is a 90% chance that the true value is between (mean minus 1.65 standard deviations) and (mean plus 1.65 standard deviations). Another distribution is the lognormal, where the logarithms of the values are distributed normally and the values themselves are distributed in a curve that leans to the left. In this case the best guess is the geometric mean or median, and the width of the curve is described by a geometric standard deviation (GSD)- there is a 90% chance that the true value is between (geometric mean / GSD^1.65) and (geometric mean * GSD^1.65). A more simple distributional assumption is the triangular. In this case the minimum, central, and maximum possibilities describe a trriangle. The most simple distribution is the uniform, a straight line, where all values between the minimum and the maximum are equally likely.

Dose Parameters

This section briefly describe the parameters used to estimate doses received. The equation as shown on page 40 of the ATSDR Assessment is:

> Dose (Sv) = Tritium concentration (Bq) x Energy of Tritium beta decay x J/MeV x sec/day x DDREF x wt factor/body mass)/lambda;
>
> Where DDREF is a dose and dose-rate effect factor[11], the wt factor serves to account for relative biological effectiveness (RBE)[12] and lambda is equal to the ln(2) divided by the biological half-life of tritium.

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[11] The DDREF, often described with different acronyms, is used to adjust risk estimates downward to account for the idea that low doses of radiation delivered at low dose rate are less damaging than the radiation typically used in establishing risk (a-bomb radiations).

[12] RBE stands for relative biological effectiveness and is used to compare the effects of different radiations. The standard, with an RBE of one, is usually gamma rays or x-rays. Beta particles may cause

The equation assumes a certain amount of radioactivity and calculates the energy absorbed with each radioactive disintegration. It also calculates how many disintegrations will happen before all of the tritium has left the body. Dose is calculated by dividing the energy delivered by all of these doses by the body weight, leaving a certain amount of energy per kilogram of body tissue. This dose is adjusted by certain factors, in this case DREF and RBE.

We looked at an alternative tritium dose calculation model presented by D. M. Hamby, one of ATSDR's expert panel. Hamby's equation was written as:

> Dose (Sv) = $(f_1 * QF * \tilde{A} * E * M_s^{-1} * T_B)/\ln(2)$;
>
> where DCF is the dose conversion factor, f_1 is fractional absorption, QF is quality factor (or RBE), $\tilde{A}$ is the integrated activity intake, E is average beta energy per disintegration, M_s is soft tissue mass, and T_B is biological half-life.

Parameters for the ATSDR model and the Hamby model are described in Table 1. The major difference in the two equations is the choice of adjustment factors; Hamby adjusts the dose upward with a tritium QF (RBE) of between 1 and 3.5. The ATSDR equation uses a factor (the wt factor, used in the same way as RBE) of 1.3. The ATSDR equation also adjusts the estimate downward with a DDREF of 0.4. We take issue with this use of the DDREF. The expert panel report used the DDREF in a risk equation, and we agree that this is where such a factor belongs. DDREF adjustments will be considered in more detail in the risk estimation section below.

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slightly more damage per rad, and alpha particles are generally assigned an RBE of 20 due to their substantially more damaging effect. Another name for RBE is quality factor (QF).

Table A-1: parameters used in dosimetry models

Parameter	Public Health Assessment	Hamby 1999	Our model
Fractional absorption	-	Uniform distribution (0.9, 1)	Uniform distribution (0.9, 1)
Dose and dose-rate reduction factor	(ddrf) Triangular distribution (0.1, 0.4, 1)	-	-
Tritium quality factor	(wt factor) Triangular distribution (1, 1.3, 3)	Two custom distributions in the range of (1, 3.5)[13]	Normal distribution with 95% interval of (1, 3.5)
Average beta energy (keV)	Uniform distribution (5.90, 5.92)	Triangular distribution (5.63, 5.69, 5.75)	Uniform distribution (5.90, 5.92)
Body mass (kg)	Adult mean of 70 kg, standard deviation of 7 kg. Child mean of 16 kg.	Adult: median soft tissue mass of 70 kg GSD of 1.15	Adult, child, and neonate median masses of 71.9, 20.1 and 6.8 kg; GSD of 1.26
Biological half-life (days)	Triangular distribution (1, 10, 40)	Median 8.7 days GSD 1.31	Adult, child, and neonate median half-lives of 8.7, 4.6 and 3 days; GSD of 1.31

We set up an alternative adult dose model and also created similar models for 5-yr olds (as in the ATSDR report) and for neonates, who may be especially sensitive. For our model we chose to use Hamby's parameter definitions of fractional absorption, quality factor, average beta energy and adult biological half-life. Our Parameters for body weight were all drawn from the most recent releases of the National Health and Nutrition Exam Survey (NHANES 1999-2000). Our breathing rates for children were drawn from International Commission on Radiological Protection (ICRP) publication 66 (1994) on respiratory tract modeling. Our data on the biological half-life of tritium were also drawn from the ICRP (2001).

For illustrative purposes we also considered possible *in utero* exposures. The fetus, sharing water with the mother, can be expected to experience the same internal concentrations of tritium. The fetal body, however, is composed of relatively more water and less solid tissue. The National Radiological Protection Board of the UK assumes that, based on a body water content of 88% versus 60% in an adult male, a mid-term fetus would experience an effective dose 40% higher than a reference adult male (NRPB

[13] Hamby evaluated the model using two separate Quality Factors (the same as relative biological effectiveness (RBE)). Using orthovoltage x-rays as the standard the distribution is skewed to the right. Using gamma rays as the standard the distribution is approximately normal.

1995). We therefore calculate the fetal dose as the adult dose x 1.4, or 115 mrem. Our full model is described in table A-2.

Estimating Risks
Risks can be calculated by multiplying the dose estimates (described above) by one or more factors:

- **Risk coefficients.** There are various sources of reference coefficients that can be multiplied by the dose to estimate the expected number of cancers, cancer deaths, or other endpoints. The standard source for cancer and other endpoints is the National Resources Council Committee on the Biological Effects of Ionizing Radiations (BEIR V, NRC 1990). BEIR V lists coefficients for leukemia separately from other cancers; this is because leukemia risks are different from other cancer risks in the pattern of incidence following exposure, in the pattern of risk with age at exposure, and in the dose and dose-rate effects. Table 4D-4 of BEIR V lists risk coefficients for different ages at exposure along with 90% confidence intervals. For example the number of expected leukemia deaths among 100,000 males exposed to 10 rem at 5 years of age is 111. The 90% CI is (20, 455). This number can be converted to a risk coefficient of (111/100,000) = 0.000111 per rem or 0.0111 per Sv. The 90% CI for this coefficient is (0.002, 0.0455). Alternative risk estimates are available; for example, both the United Nations and the ICRP use a figure of 0.05% per cGy for fatal lifetime cancers following *in utero* exposures (UNSCEAR 1993, ICRP 1996). Slightly higher numbers are also suggested in the literature (Stovall et al 1995), but we chose to use the UNSCEAR/ICRP value for our ballpark estimate.

- **DREF.** BEIR V uses an additional factor to account for the fact that doses delivered at low dose rates may have a smaller effect per unit dose than those delivered at high dose rates (dose rate effect factor, or DREF). This factor is largely based on observations of a non-linear dose-response curve in the A-bomb survivors. In the case of leukemia, the BEIR V committee effectively included the DREF in their linear-quadratic model, and no further adjustments to risk estimates are warranted. The committee used linear models for other cancers, however, and suggested that the risk estimates be modified by a DREF. We would like to suggest that the use of this factor may not be appropriate. As noted by Straume (1993) the dose rate effect diminishes to insignificance at low total doses. More importantly, recent analysis of the A-bomb survivorship indicate that the dose-response curve for solid cancers is best described by a linear model without any adjustments (Nussbaum and Kohnlein 1992, Nussbaum 1998, Pierce and Preston 2000, Little and Muirhead 2000). We therefore calculated the risks of other cancers twice, once with a DREF and once without. Our DREF factor was the same as that used in the consultation, with a triangular distribution having a low end of 0.1, a central value of 0.4 and a high end of 1. We calculated leukemia risks using the BEIR V risk coefficients.

- **Multiplying factor for A-bomb radiation differences.** The expert review panel recommend an additional factor to account for the abnormally high energy of the reference radiation (A-bomb exposures), and we use this factor, which has a central value of 2 and a range of 1-5.

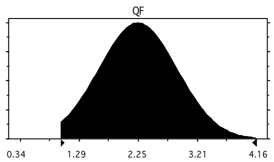
Dose factors	Value	Source/ notes
Numerator		
ATSDR reported 12-day HTO dose (Sv) for maximally exposed adult case	0.00019	Table 2 of ATSDR 2002
Energy of tritium beta decay (MeV)	Uniform distribution (0.00590, 0.00592)	ATSDR 2002
Joules / MeV	1.6E-13	ATSDR 2002
Seconds / day	86,400	
Tritium quality factor	Normal distribution; 95% interval 1, 3.5	Hamby 1999
Biological half-life of tritium (d)	Adult, child, and neonate medians of 8.7, 4.6 and 3; GSD of 1.31	ICRP 2001 (variability observation from Hamby 1999)
Factor adjusting for using the 1965 accident as the maximum exposure scenario	2.83	Ratio derived from ATSDR 2002
Factor for meteorological uncertainty	Geometric mean of 1, GSD of 1.23	Yersel et al 1983
Factor for deposition velocity uncertainty	Geometric mean of 1, GSD of 2	GSD expanded from ~1.5 in ATSDR 2002 to capture additional uncertainty
Ratio of child or neonate breathing rate (m^3/d) to adult rate	0.88 (child), 0.019 (neonate)	Adult breathing rate from ATSDR 2002, child and neonate rates from ICRP 1994
Denominator		
Adult dose coefficient from Health Assessment	2.06E-11 Sv/Bq	Derived from ATSDR 2002
Body weight	Adult, child and neonate median masses of 71.9, 20.1, and 6.8 kg; GSD of 1.31	NHANES3
Ln(2)		
Risk factors	**Value**	**Source/ notes**
Dose (Sv)	-	From dose equation above
Factor for A-bomb gamma energy	Triangular distribution (1, 2, 5)	ATSDR 2001b
DDREF	Triangular distribution (0.1, 0.4, 1)	ATSDR 2002
Cancer mortality risk coefficients	Several used	BEIR V (NRC 1990), Table 4D-4

Figure A1: Assumptions used in monte carlo dose and risk model.

Assumption: Tritium Quality Factor (QF, RBE) Cell: G14

Normal distribution with parameters:
2.5% - tile 1.00
97.5% - tile 3.50

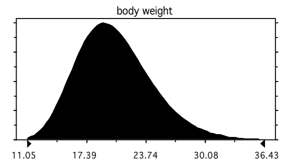
Selected range is from 1.00 to ∞
Mean value in simulation was 2.29



Assumption: 5-yr old body weight (kg) Cell: K28

Lognormal distribution with parameters:
Geometric Mean 20.06
Geometric Std. Dev. 1.22

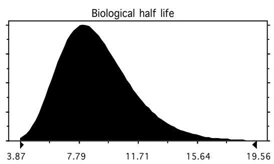
Selected range is from 0.00 to ∞
Mean value in simulation was 20.40



Assumption: Biological half life, Adult (days) Cell: G24

Lognormal distribution with parameters:
Geometric Mean 8.70
Geometric Std. D 1.31

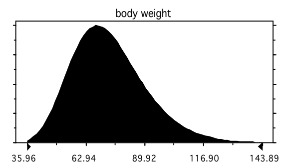
Selected range is from 0.00 to ∞
Mean value in simulation was 9.01



Assumption: body weight, Adult (kg) Cell: G28

Lognormal distribution with parameters:
Geometric Mean 71.93
Geometric Std. Dev. 1.26

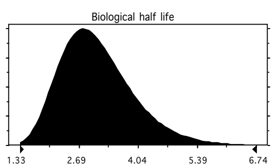
Selected range is from 0.00 to ∞
Mean value in simulation was 73.90



Assumption: Biological half life, neonate (days) Cell: I24

Lognormal distribution with parameters:
Geometric Mean 3.00
Geometric Std. D 1.31

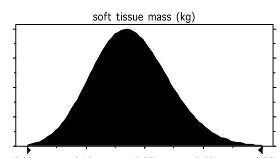
Selected range is from 0.00 to ∞
Mean value in simulation was 3.11



Assumption: body weight, neonate (kg) Cell: I28

Lognormal distribution with parameters:
Geometric Mean 6.84
Geometric Std. Dev. 1.09

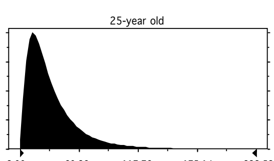
Selected range is from 0.00 to ∞
Mean value in simulation was 6.86



Assumption: 25-year old leukemia risk (deaths/100,000/0.1 Sv) Cell: G47

Lognormal distribution with parameters:
Geometric Mean 26.38
Geometric Std. D 2.07

Selected range is from 0.00 to ∞
Mean value in simulation was 34.78



Assumption: 5-yr old old leukemia risk (deaths/100,000/0.1 Sv) Cell: I47

Lognormal distribution with parameters:
5% - tile 20.00
95% - tile 455.00

Selected range is from 0.00 to ∞
Mean value in simulation was 149.12

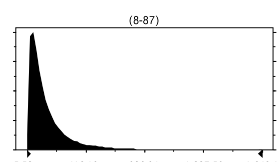


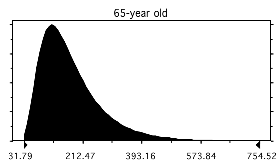
Figure A1: Assumptions used in monte carlo dose and risk model (continued).

Assumption: 65-year old leukemia risk (deaths/100,000/0.1 Sv) **Cell: G50**

Lognormal distribution with parameters:

5% - tile	65.00
95% - tile	369.00

Selected range is from 0.00 to ∞
Mean value in simulation was 179.94

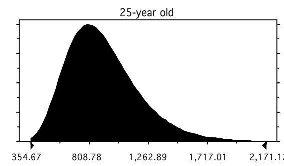


Assumption: 25-year old nonleukemia cancer risk (deaths/100,000/0.1 Sv) **Cell: G55**

Lognormal distribution with parameters:

5% - tile	534.00
95% - tile	1,442.00

Selected range is from 0.00 to ∞
Mean value in simulation was 926.83

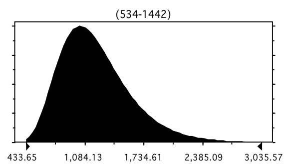


Assumption: 5-year old nonleukemia cancer risk (deaths/100,000/0.1 Sv) **Cell: I55**

Lognormal distribution with parameters:

5% - tile	673.00
95% - tile	1,956.00

Selected range is from 0.00 to ∞
Mean value in simulation was 1,206.41

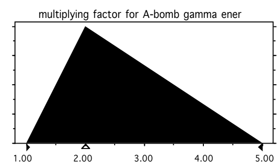


Assumption: multiplying factor for A-bomb gamma energy **Cell: G59**

Triangular distribution with parameters:

Minimum	1.00
Likeliest	2.00
Maximum	5.00

Selected range is from 1.00 to 5.00
Mean value in simulation was 2.65

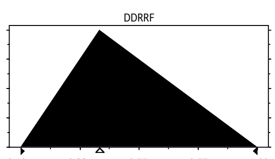


Assumption: Dose rate factor (DDREF, DREF) **Cell: G61**

Triangular distribution with parameters:

Minimum	0.10
Likeliest	0.40
Maximum	1.00

Selected range is from 0.10 to 1.00
Mean value in simulation was 0.50

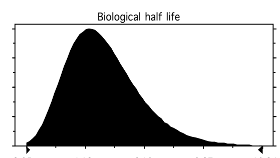


Assumption: 5-yr old biological half life (d) **Cell: K24**

Lognormal distribution with parameters:

Geometric Mean	4.60
Geometric Std. Dev.	1.31

Selected range is from 0.00 to ∞
Mean value in simulation was 4.79

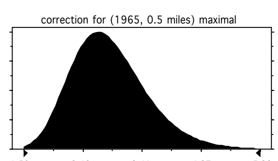


Assumption: correction factor for maximum exposure error in ATSDR report **Cell: G18**

Lognormal distribution with parameters:

Geometric Mean	2.83
Geometric Std. D	1.23

Selected range is from 0.00 to ∞
Mean value in simulation was 2.90

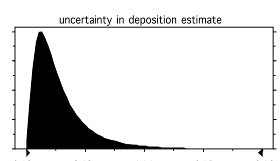


Assumption: factor for uncertainty in deposition estimate **Cell: G16**

Lognormal distribution with parameters:

Geometric Mean	1.00
Geometric Std. Dev.	2.01

Selected range is from 0.00 to ∞
Mean value in simulation was 1.28



End of Assumptions

Appendix B: Implications of newly available information

(to be supplied later)