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by

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Executive Summary

The Risk Assessment Corporation (RAC) report reviewed here was intended to evaluate the potential health risks associated with the burning of a part of Los Alamos National Laboratory (LANL) during a wildfire in May of 2000. The authors presented a methodical approach that divided the problem into several components; contaminants were assessed according to their hazardous characteristics (radiological or chemical) and according to the two major pathways of exposure (air and water). Risks were estimated using measured concentrations of contamination where it was judged to be reasonable and using modeled concentrations of contamination where it was judged to be necessary. My general conclusions regarding the report are as follows; more detail is provided in subsequent sections.

1) Estimated risks based on measured concentrations of contamination suggest that the additional risk from burning Los Alamos property was lower than the EPA level of concern for each contaminant assessed. The RAC authors observe that measured data were insufficient for a meaningful risk assessment; the estimates derived from actual data are very uncertain.

2) Modeling was also associated with a great deal of uncertainty; the conservative approach used here, although it is relatively efficient, tends to obscure this uncertainty. Conservatism in modeling may also give the impression that the authors intend to reassure the affected community that risks were insignificant; this impression, real or imagined, may in fact foster distrust.

3) In most cases model predictions could not be validated because of the conservative assumptions used in the models (predictions tend toward ‘upper-bound’ estimates). Although predictions should be overestimates, in several cases observed concentrations of contamination were greater than modeled predictions. This suggests that the modeling could not adequately simulate the dynamics of the Los Alamos area during the fire and/or that substantial elements of the system were not accounted for.

4) Where the model could be validated, specifically in predictions of particulate matter, the model did not produce realistic predictions until some rough and questionable adjustments were made to input parameters. Following this adjustment the model was reasonably accurate for areas close to the fire and less accurate for areas further away.

5) Both the measured concentrations of contamination and the modeled estimates of the RAC report are unreliable. Although there is not evidence here to suggest any substantial risk from the Cerro Grande fire, the report is not capable of providing much reassurance that the risk can be accurately estimated. This is mainly due to insufficient data characterizing the area and insufficient monitoring data for the burn period and immediate post-burn period.

6) The inability to quantify meaningful estimates of risk from the Cerro Grande fire indicates that site characterization and regional monitoring need to be greatly improved. If Los Alamos is to be a credible risk manager in the eyes of surrounding communities there should be real community involvement in the design, oversight, and possibly operation of an improved monitoring infrastructure.

7) In general, the proper role of modeling in the management of LANL risks should be carefully reconsidered. Retrospective estimates of risk that are too uncertain to be meaningful may not be worth the high costs associated with modeling. Modeling could be useful prospectively in targeting data gaps and potentially significant exposures.

Introduction

The Cerro Grande fire began in Bandelier National Monument on May 4, 2000 and burned for the next two weeks. Approximately 7500 acres of Los Alamos National Laboratory (LANL) were burned. New Mexico Environment Department (NMED) contracted with RAC to evaluate incremental health risks from radionuclides and chemicals released during the fire. The report was published in 2002[2].

The discussion presented below was written by the Community-Based Hazard Management Program at Clark University as part of a more general program of collaboration between academia and communities affected by radiological hazards. The following discussion covers the three subsections of the report, releases to air (task 1.7), releases to water (task 2.7), and conclusions and recommendations (task 3)), in order.

1. Releases to Air (Task 1.7)

The approach to assessing risks from airborne contaminants was limited by very poor monitoring data. Since risks could not be meaningfully estimated using real data the authors modeled the dispersion of contamination within a modeling domain. The modeling domain was limited by data storage requirements and computer runtimes. The final modeling domain included 3300 km^2 with a grid spacing of 500 m. Taos, a community that experienced smoke exposure and potentially associated pollutants exposure, fell outside of this domain and was assessed with an alternative methodology.

1.1 Measured data

The RAC report relies mainly on modeling of atmospheric dispersion because real monitoring data were insufficient; this is despite data collection by LANL, NMED, EPA, and the DOE Radiological Assistance Program. Limited airborne chemical information was collected by EPA only. There were several sources of radionuclide data for the period of the burn but this was also limited[3]. Soil and biota measurements were made after the fire by LANL and NMED. Particulate matter (PM) data were collected by LANL, NMED and EPA and used primarily for calibrating the model used by RAC.

Gross radioactivity was elevated during the fire; the nuclides involved were mainly short-lived and could have been largely of natural origin.

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[2] Risk Assessment Corporation (RAC) 2002. Analysis of Exposure and Risks to the Public from Radionuclides and Chemicals Released by the Cerro Grande Fire at Los Alamos. RAC Report No. 5-NMED-2002-FINAL.

[3] For example, LANL air monitors collect airborne particles on a 2-week cycle and each sample is analyzed for alpha and beta activity. Composites of individual samples are analyzed for gamma activity quarterly. During the fire the filters were changed more frequently, at intervals of one to several days, because of dust accumulation on the filters. The dust accumulation reduced the sensitivity of the monitors and increased the uncertainty of measurements.

Radionuclide-specific air monitoring data were mainly unreliable (not statistically positive) due to analytical uncertainty. There were only a few positive readings-

- ^{241}Am in three onsite samples, reported to be normal at that location, and in two offsite samples from the eastern boundary of LANL.

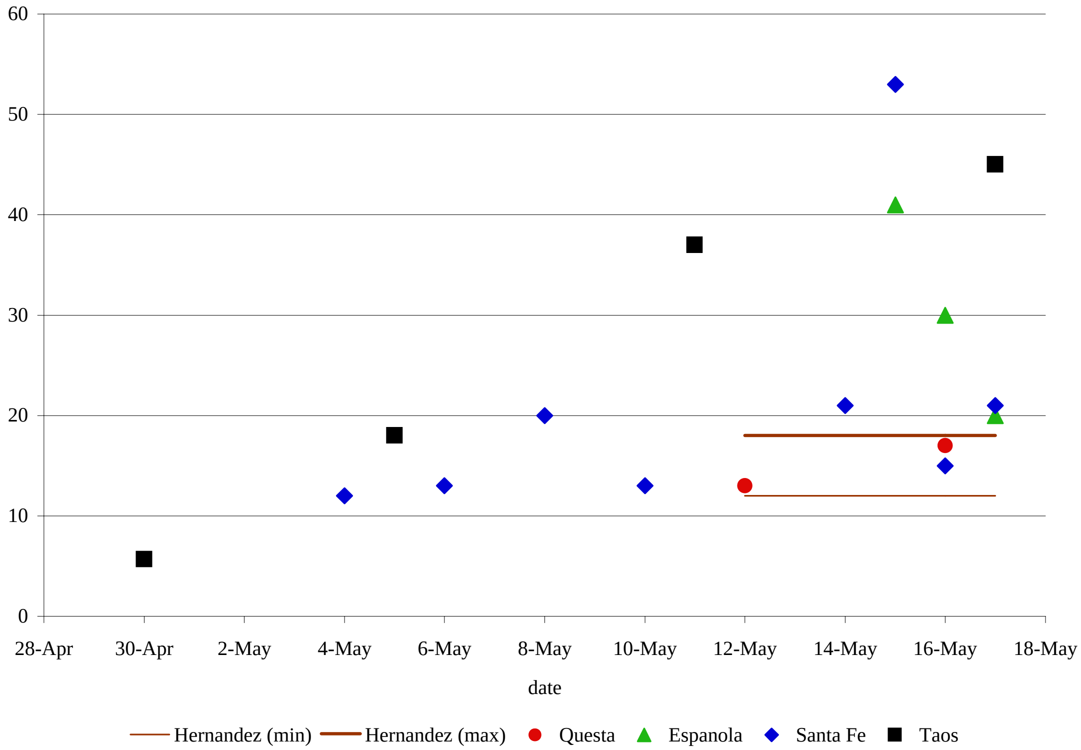
- ^{239,240}Pu in two Los Alamos town samples, one county landfill sample, and one sample from TA-21. A particularly high reading, 1,000 times higher than any previously reported offsite reading, was made at the Tsankawi National Monument.

- Two onsite measurements of elevated depleted uranium (^{238}U).

Soil measurements were made by LANL at 12 locations onsite, 10 locations on the site boundary, and at 4 locations offsite (after the fire). Additional measurements were made at farms in the path of the plume by LANL and NMED. Pre- and post-fire values are reported to be consistent with each other indicating no change due to the fire; lower values are reported for farms, and these were further from points of release.

Chemical data analyzed by the EPA included several detectable compounds that are all reported to be below public safety standards. RAC was unable to locate pre-fire background data for comparison.

Offsite samples indicate increased PM during the fire. Data points mentioned in the text of the RAC report are shown below. PM measurements onsite peaked on May 13-14.



RAC did not use the radionuclide or chemical monitoring data in their final risk estimates. However, it is reported that under extreme assumptions[4] the elevated concentrations of radionuclides noted above would have resulted in cancer risks below 10^{-7} for LANL-derived isotopes and below 10^{-6} for ^{210}Po, a naturally occurring isotope.

1.2 Model assumptions and uncertainties

Since monitoring data were insufficient RAC modeled releases to air using dispersion models and estimates of contamination at LANL Potential Release Sites (PRS). These estimates were based on characterizations developed during 1993-1997.

An initial round of screening narrowed the list of contaminants that were considered for more involved modeling; priority was placed on those contaminants most likely to present a risk. Potential contaminants of concern were screened individually (cumulative risks from multiple contaminants were not estimated) and at a lifetime risk level of 10^{-5}; this level was chosen over a more protective level of 10^{-6} presumably because of a series of conservative assumptions in the development of screening risk estimates. A less protective level of 10^{-4} was not used because contaminants were screened individually[5].

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[4] The screening calculations assumed that a person would be exposed to the maximum reported concentration for 14 days, 24 hours a day.

[5] The risk level of 10^{-4} to 10^{-6} (1 in 10,000 to 1 in 1,000,000) is considered by the EPA to be a level of concern.

Noncarcinogenic chemicals were screened using reference concentrations; if a given chemical concentration was less than the reference concentration then it was rejected from further consideration.

Although different sets of assumptions were used for screening and for the full model, in both cases the authors chose to pursue a conservative approach. Conservative assumptions include:

- entire inventory of a PRS is released (screening)
- stability class F (minimum dispersion; screening)
- exposure distance 100 m downwind of release site (screening)
- for noncarcinogens the reference concentrations are ideally conservative Additional conservatism in chronic (up to seven years) exposure assumptions in derivation of reference concentrations as applied to two-week exposures assumed in the case of the Cerro Grande fire
- measurements below the detection limit were not used in calculating average soil concentrations of contaminants (averages would be skewed toward higher values[6])

The mean soil temperature was assumed to be 1832° F. This was intended to be conservative, and indeed it is probably much too high for several reasons[7]. This assumption leads to an unrealistically high estimate of volatilization of chemical contaminants (radionuclides, as metals, are not expected to volatilize). In fact, all volatile chemicals were assumed to be completely volatilized in the fire (p 3-28).

Some assumptions were not conservative. A few key assumptions were reasonable based on the very limited available information, but were just as likely to underestimate the true value as they were to overestimate the true value:

- mean concentration value of multiple samples for a PRS used to describe PRS
- single measurements used to describe one or more PRSs
- homogeneous background concentrations across LANL

There was additional and substantial uncertainty in the spatial extent of the PRSs. LANL Environmental Remediation personnel redefined 223 PRS boundaries, apparently based on sampling data, and in most cases the redefined areas were larger than indicated by the

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[6] This might not be conservative in all cases; see the note on samples below detection in the discussion on the water report below.

[7] Maximum temperatures in high intensity fires rarely exceed 1832° and the peak observed surface temperature mentioned in the text is 1320° in a California fire (p 3-18). Most of the burned area in the Cerro Grande area was classified as low or moderate severity. Fires burning with the wind, high wind speeds, and fires in tall forests (all characteristics of the Cerro Grande fire) cause peak temperatures to occur at greater heights (resulting lower surface temperatures) (p 3-21). There were hydrophobic soils in some burned areas indicating temperatures greater than 482° F but less than 572° F (p 3-20). Soil temperatures at depths of 1 inch rarely exceed 212° F (p 3-20) and temperature decreases with depth. The average depth of PRS characterization data was 6 inches and the maximum depth was 18 inches (p 3-21).

original GIS-based information, by an average of twofold[8]. It is not clear what fraction of the original GIS-based PRS boundaries were not redefined or what fraction might be incorrectly defined.

Soil to a depth of 1 cm was considered to be available for resuspension of particles; however, concentrations of contaminants measured at depths of up to 18 inches were used to define the assumed concentration within this top layer. It is not clear how representative the sampling data are, or if this is a conservative assumption. It is also not clear where the resuspension depth of 1 cm comes from (it is claimed to be very conservative; p 3-29).

No chemical-specific properties influencing atmospheric reactions were taken into account. This means that decay and transformation into less toxic or more toxic compounds are both ignored.

Although measured concentrations were possibly elevated after the fire, perhaps due to continuing smoldering, releases of contaminants were only estimated for the burn period. The report states that modifications were made to the Emissions Production Model shifting a fraction of the release from the active burn period to the smoldering period, leaving the total amount of released contamination the same. This point is unclear, however; for example, on page 4-40 it is stated that "releases from PRSs were limited to three days from May 11 to 13" and on page 4-1 it is stated that releases after the burn were not modeled. This is despite several references to post-fire releases. Mean PM10 concentrations were slightly higher after the fire (p 4-2), and it is suspected that this came from continuing smoldering and spot fires (p 4-36). The authors of the RAC report cite an estimate that 70% of total PM emissions occurring during the smoldering stage (p 4-27).

There was evidence of convective dispersion of soil particles (crustal alkali and alkaline elements) although this type of dispersion is not included in the Emissions Production Model.

1.3 Model predictions

Original model iterations underpredicted measured PM10 concentrations and the fuel loading inputs were changed from the central estimates to the 97.5 percentiles of fuel load measurement distributions. This was justified by assuming that live biomass was not included in the original inventory. Such a rough and arbitrary adjustment suggests that the model was not appropriate for these conditions.

Final model predictions were notably lower than measured PM10 concentrations for May 16 and it is suggested that this is because of ongoing smoldering and spot fires. Overall predictions underestimated observations by an average of 13% (Table 4-10).

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[8] Table 3-2 of the RAC report (p 3-17) indicates that the median ratio of original to redefined area was 0.52; the minimum was 1.3×10^{-6}. This means that the average PRS was twice as large when it was redefined; the maximum increase in size was over 700,000-fold. 73 out 223 PRSs were redefined as being at least 10 times larger than originally thought.

^{239}Pu predictions were orders of magnitude lower than the handful of positive, reliable measured concentrations[9]. This difference is explained by a precise interpretation of the relevant source of contamination and time of release: on page 4-50 it is proposed that plutonium was released from unburned or previously burned PRSs during the high wind generated by the fires. RAC was responsible for assessing the releases from burning PRSs and restricted emissions analysis to a three-day period.

The alternative modeling for Taos (p 4-58) was very limited. The peak concentration within the plume at the boundary of the model domain was 28 ug/m3 (24 hour average) and it is stated that this is an extreme upper bound on the possible concentrations further downwind due to dispersion and dilution. Yet the 24-average concentration reported in Taos during the fire had a mean of 41.1 ug/m3 (Table 4-1, p 4-3). The background concentration in Taos is most likely in the area of 14 ug/m3 (same table) leaving 27 ug/m3 from the fire. Since the modeled estimate should be higher than the observed concentration in Taos (due to dispersion and the low probability of Taos being at the most concentrated height of the plume), this alternative model is apparently underpredicting concentrations in Taos.

1.4 Discussion of the air pathway assessment

Modeling. As observed above the precise scope of the RAC analysis may have left out the majority of radioactive releases (and possibly some of the chemical releases) from LANL associated with the fire, those that were released from unburned or previously burned PRSs. The assignment given to RAC by NMED may have been inappropriate since contamination from unburned or previously burned PRSs appears to have contributed a substantial fraction of offsite contamination and since the LANL is responsible for these releases as much as it is responsible for releases from active burning. Several additional major sources of uncertainty in the model, including a lack of data (PRS characterizations, background concentrations, measurements during the fire), call into question the utility of the predictions generated by the model. The adjustment of fuel loading inputs from central estimate values to 97.5 percentile values suggests a substantial inadequacy in the model.

The evaluation of exposure potential in Taos is also not very informative; since the overall assessment uses PM10 as a tracer for other nonvolatile contaminants the high PM10 concentrations reported in Taos imply a relatively high exposure potential. This potential was not effectively explored in the RAC report. In general this suggests that exposures outside of a practical modeling domain, although potentially significant, may be very difficult or impossible to model. In these cases direct measurements are critical.

Modeling, even of an upper bound risk estimate, is shown in the RAC report to be quite problematic; if modeling is to be carried out it can be dramatically improved by better

[9] In the extreme case there was an EPA measurement of 8800 aCi/m3 at Tsanakawi National Monument on May 15. The three-day average (May 11-13) concentration predicted at TA-54 was 0.0012 aCi/m3. These differ by a factor of over 1,000,000.

background characterization data, PRS characterization data, and fire characterization (temperatures, fuel loads, active burn behavior vs. smoldering behavior, etc.). It would also be greatly improved with better validation data provided by an improved monitoring system. The modeling approach is expensive and these improvements would increase the cost. In light of this constraint the proper role for modeling should be carefully considered. This is discussed more below.

Risk. Section 2.5 contains the best available bounding estimates of risk. Here it is reported that the maximum measured concentrations of carcinogens were associated with risks below 10^{-7} (assuming that the maximum concentrations were present for 14 days). We have verified that the maximum reported ^{239}Pu concentration, 8800 aCi/m3 at Tsankawi National Monument, would have resulted in a lifetime cancer incidence risk of approximately 1×10^{-8}. The conclusions that the cancer risk from burning LANL PRSs was very low, below the EPA allowable risk range, appears to be a reasonable claim. This is also likely to be true for unburned or previously burned PRSs since measured contaminant concentrations are not segregated by source. Concluding remarks by the RAC authors, however, do not always convey the appropriate degree of uncertainty associated with the assessment.

2. Releases to Air (Task 2.7)

As in Task 1.7, this report uses a series of conservative assumptions to make an upper-bound estimate of risk, in this case for exposure to contaminants in surface water, storm water, and sediment[10]. Conservative assumptions include, for example, non-depleting sources of contamination and soil-to-water partition coefficients that are biased toward dissolution of contaminants. Other assumptions are listed on page 4-77. Many of the main comments made in reference to the Task 1.7 (air) report apply to this report as well, particularly relating to the poor characterization of PRSs. The major difference between the air and water pathways appears to be that there is a better set of environmental monitoring data available with which to validate the water pathway model.

2.1 Measured data

PRS characterization data is presumably the same for both the air and the water pathway reports; comments relating to the high degree of uncertainty in PRS data apply to both reports. The water report provides additional light on this problem;

- “Furthermore, in a number of instances the data providers gave us data that they knew to either be incorrect or have some sort of associated bias without indicating these issues” (p2-10).

- “Documented inventories of contamination for many areas and canyons do not exist” (p2-16).

- “The majority of the Cs-137 and Pu-239/240 inventories in Los Alamos and Pueblo Canyons… resides in stretches of canyon that are not characterized by actual sampling data”. The combined Los Alamos, Pueblo, Rendija and Guaje watershed is ranked highest in concern in Table 3-1; the fact that the amount of contamination in these canyons is unknown cripples any modeling.

Other critical data sets were also inadequate; most prominently there were apparently no credible post-fire chemical monitoring data (p2-20).

Assessing the data sets available to the RAC authors was challenging in many cases because of my unfamiliarity with collection techniques. Some of the puzzles I encountered, however, may reflect substantial data limitations:

- There appears to be a difference between “nondetect” and “below detection limit”. On page 2-2 it is stated that “we generally considered only positive results reported as ‘detected’ because the detected values represent an estimate of the true concentration as opposed to the upper bound value represented by a ‘nondetect’ value”. On page 2-17 it is stated that “Sample results were below the detection

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[10] Exposure to sediment is assumed to include external exposure, skin contact, and inadvertent ingestion. The soil ingestion rate was assumed to be 0.1 to 0.2 grams per day depending on age and activity.

limits for many of the naturally occurring radionuclides in surface water measured after the fire. For other radionuclides like ^{137}Cs, about 50% were not detected (nondetects) in surface water samples and 70% of storm water samples". Finally, on page 2-19 we read that average background water concentrations of U-238 and tritium were higher for nondetects than for detectable concentrations. I found all of this very confusing[11].

- Table 2-1 provides a summary of ESH-18[12] water monitoring data for radionuclides, pre- and post-fire. Each data point is accompanied by the total number of samples and the number of samples with detectable concentrations of a given radionuclide. These can be converted quickly to the percent of samples below detection limit. In 8 out of 12 pre-fire estimates there were no samples below the detection limit. For example, 1,030 out 1,030 Pu-239 measurements were detectable. The average fraction of samples below detection for all pre-fire data is approximately 1.5%. For post-fire data, however, the average fraction of sample below detection is 56%. These data suggest that pre-fire sampling almost always yielded a detectable concentration while post-fire sampling only yielded a detectable concentration about half the time. There are several possible explanations for this inconsistency, but in any event there appear to have been very different sampling protocols before and after 2000, and this should have been explained and accounted for. Certainly any analysis of temporal trends would be impacted by this change.

- Figure 2-4 shows Cs-137 concentrations in background surface water samples and samples from the Rio Grande just below LANL and below Cochiti. There was apparently a great deal of variability in the estimated concentration over the 1973-1993 period; this was true of both background and Rio Grande samples. However, for both samples there was little or no apparent variability after 1993. It is hard to imagine an explanation for this but again, this would impact any analysis of temporal trends.

Presumably based on puzzles like these, the RAC authors make the reasonable statement that "changes in analytical and/or sampling methods over time complicated drawing definitive conclusions based on these comparisons" (p 2-23); this point could be made more prominent.

Monitoring data are presented in a series of tables for both surface water and storm water. Table 2-6 shows the average concentrations of four radionuclides in storm water. The concentrations are segregated into pre- and post-fire measurements made above or below LANL. These data show that post-fire measurements below LANL were higher than pre-fire measurements for Pu-239/240, for Am-241, and for Sr-90. Pu-239/240 also increased after the fire above LANL. Cs-137 in storm water below LANL was roughly the same

[11] In fairness I should point out that RAC also found this confusing: "We do not have an explanation for why the average for values reported as nondetects would be higher than the average for detectable concentrations, and a detailed investigation into this issue is beyond the scope of this work" (p2-19).

[12] LANL Environmental Safety and Health division.

after the fire. The concentrations above LANL, however, were much lower after the fire, suggesting previous contamination before the fire. This might indicate that the Cs-137 concentrations below LANL were higher than they would have been without the impact of the fire.

The report suggests that there was no apparent increase in waterborne radionuclides after the fire (p2-25), although in a concluding section considering more data the authors admit that the data "suggest the possibility of fire-related and/or LANL impacts" (p2-51).

2.3 Model predictions

The approach was based on a series of conservative assumptions yet the model underpredicted concentrations in a number of cases. For example, the predicted:observed ratios of sediment contamination at Point of Exposure (POE) 2.1R (Rio Grande at Cochiti) were less than 1 (underpredicted) for Arsenic, Barium, Chromium, Copper, Lead, Mercury, Pu-238, Th-228, Th-230, and all three uranium isotopes. These represent all of the chemicals and half of the radionuclides assessed in this category (chemicals and radionuclides with background values; Table 4-39). The chemical N-Nitrosodimethylamine (NDMA) was underpredicted in sediment (Tables 4-40 and 4-41) and in water (Tables 4-44 and 4-45). The underprediction in this case was huge, ranging from 10^{-6} to 10^{-9} (600,000 to 100,000,000-fold). This is explained as "likely the result of a small source term and predicted concentrations well below the detection limit" (p4-85). Although I presume that this problem involves the reporting of nondetect values or values below the detection limit, admittedly beyond my comprehension, this explanation was not clear and the magnitude of underprediction warrants a more extensive discussion.

3. Risk communication (Task 3)

The Task 3 report presents an honest assessment of the limitations of the risk assessment process given the present state of data collection. The authors observe that “it is generally preferable to use measurement data rather than modeled or predicted concentrations to reduce the significant uncertainty often associated with predicted concentration” and comment on current deficiencies in this area. Specifically, monitoring data collected for regulatory compliance purposes (air monitoring at site boundary, PRS characterization data) are insufficient for the purposes of estimating offsite contamination or adequately characterizing PRSs as source terms. The authors allude to the fact that the proper role for modeling in the management of LANL risks is in the prediction of future risks; the ability to accomplish this is clearly dependent on an improved assessment of PRSs. Additional data needs include a better assessment of background conditions and a better understanding of short-term fluctuations in suspended particles or pollutants. The creation of an improved knowledge base would enable credible predictive modeling to be used for the design of improved emergency (and routine) monitoring systems.

The authors also bring up the idea that all raw data could be made publicly available. This would be a very convenient way to allow downwind communities to keep track of the best available information and would be compatible with data interpretation at the community level through, for example, the use of GIS technology.

4. Concluding remarks

It would be reasonable, from the perspective of community concern, to reject the modeling approach used here. Although the cancer risks from burned LANL property were unlikely to have been equal to or greater than EPA acceptable risk levels, this is best demonstrated with the limited available monitoring data. The modeling effort suggests that the range of risks was likely to be, on average, lower than risks estimated from measurements. This was accomplished, however, with a large effort on the part of RAC and it revealed several fundamental problems. The conservative approach used by RAC can be useful. If we are interested in some kind of upper-bound estimate of risk so that we can make precautionary policy decisions, a conservative estimate of risk might be appropriate. However, as applied here these conservative methods are not helpful for the following reasons.

- A risk assessor decides which assumptions will be made conservatively and how conservative they will be. This process is subjective and not always transparent. It is very hard for an outside stakeholder to objectively evaluate or interpret the significance of the assumptions or the quality of the outcome. No one, including the risk assessor, can even begin to estimate how conservative the resulting risk estimate really is.

- Predictions based on conservative assumptions are thought to be overestimates, but the degree of overestimation is unknown. This makes validation of modeled predictions with real measurements impossible. When the conservative predictions *underestimate* observed concentrations, the whole approach should be seriously questioned.

Although the RAC authors may be overconfident, they acknowledge that the modeling approach is limited and advocate for a monitoring database sufficient to make measurement-based risk estimates[13].

Modeling, as a tool, should be carefully considered in light of its potential uses, its costs, and its substantial limitations. Retrospective estimates of risk that are too uncertain to be meaningful may not be worth the cost; on the other hand, prospective modeling could help identify data gaps and potential risks and be a useful tool in targeting prospective risk mitigation strategies (such as cleanup and monitoring).

The best opportunity for increased community awareness and control in the case of future fires, and generally, is an improved monitoring infrastructure that has appreciable community involvement in design, oversight and interpretation. In particular, areas downwind and outside of a realistic modeling domain, such as Taos, should be priority locations for such an infrastructure.

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[13] For example, Task 2.7, page 2-49: "If the goal is to understand possible risks based on current conditions and concentrations, environmental monitoring data are preferred because they are not impacted by the many uncertainties inherent in environmental transport modeling"