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Confederated Tribes and Bands
of the Yakama Nation

Established by the
Treaty of June 9, 1855

February 28, 2011

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Re: Comments on *DOE/RL-2007-21 Volume II Draft C: River Corridor Baseline Risk Assessment, Human Health Risk Assessment,* December 2010

Dear Mr. McCormick, Mr. Faulk and Ms. Hedges:

The Yakama Nation has reviewed the U.S. Department of Energy's document DOE/RL-2007-21 Volume II Draft C: River Corridor Baseline Risk Assessment (RCBRA), Human Health Risk Assessment (HHRA). Yakama Nation comments reflect our best effort considering the limited review period. Please find attached the following items:

- General comments on RCBRA Vol. II Draft C
- Specific comments on RCBRA Vol. II Draft C
- Attachment 1 – *Supporting comments on RCBRA Vol. II Draft C by IEER*
- Attachment 2 – *Supplemental comments on RCBRA Draft A*

The Yakama Nation is concerned that this document, while titled a baseline risk assessment, does not provide a complete assessment of the cumulative risks that a Tribal member or other members of the public would encounter on the site. Such information is critical for making future management decisions. Although the Yakama Nation has engaged in the process to fully understand the potentials risks to our people and treaty resources from Hanford, the majority of our previous comments on Draft A and our basic concerns regarding the risk assessment process have not been addressed. We also did not receive Draft B for review and comment.

Our major recommendations for the RCBRA Volume II Draft C, which are discussed further in the attached comments, include:

baseline HHRA suggests that doses estimated from soil concentrations measured in background samples will be excluded from total radiation dose used to calculate risk, which is not appropriate for a baseline risk assessment. All exposures (and associated doses) measured at the site contribute to baseline risk and should be included.

Under MTCA, each carcinogenic hazardous substance is limited to a concentration corresponding to a 1×10^{-6} lifetime cancer risk. When more than one hazardous substance is present, the combined lifetime cancer risk limit for chemicals and radiation is 1×10^{-5}. While MTCA has been generally interpreted as applying to chemicals only at Hanford, this interpretation is too limited and should also consider Hanford's extensive radionuclide contamination. Since MTCA explicitly defines radionuclides as hazardous substances, the combined limit for radionuclides and chemicals should correspond to a lifetime cancer risk of 1×10^{-5} or less. The risk assessment should adopt agency-refined and accepted MTCA and CERCLA risk thresholds, rather than 15 mrem/year and 100 mrem/year dose limits, using age-specific dose conversion factors to derive dose guides for cleanup. Individual drinking water maximum contaminant levels (MCLs) for man-made radionuclides should also be developed and adopted for anthropogenic radionuclides corresponding to a 10^{-6} lifetime cancer incidence.

- **Provide additional characterization and more transparency.** Due to the scope and complexity of the River Corridor, additional characterization data are needed to fully assess baseline conditions. The assessment has excluded certain radionuclides from consideration without adequate justification. For example, the reason for the exclusion of thorium-232 and its decay products, thorium-228 and radium-228, is not adequately supported. It is also unclear why certain sites were selected over others, and why certain contaminants were analyzed at some sites and not others. The requirement that a contaminant needs to be reported at one-third of the wastes sites to be considered a COPC is not protective, as this screen potentially eliminates relatively unique waste sites, as well as adds to the problem of not including all contaminants in the risk evaluation. In many cases, it is unclear whether enough or appropriate data were available for calculating risks because assumptions were not transparent or justified. Additional plant samples (all tissues) should be collected to more accurately quantify concentrations and obtain site-specific contaminant transfer factors to help to assess dietary and non-dietary exposures to these resources from consumption, production, and use. Including reference and background data that have been collected from, and adjacent to, the Hanford Site is inappropriate, as these areas have most likely been influenced by releases from the Hanford Site in the form of airborne contamination and abiotic and biotic movement of contaminants.

Yakama Nation uses of the Hanford area will result in unique contaminant pathways and exposure rates. High level, transuranic, low-level and mixed radioactive wastes, nuclear facilities, proposed waste treatment operations, contaminated biota, and polluted water pose threats to the Yakama Nation, the health of our people, and the vitality of our traditional subsistence lifeways. To protect Yakama Nation uses, all contaminant sources and hazards should be identified and assessed comprehensively to support appropriate cleanup decisions.

We expect that the Department of Energy will consider the total risk to Yakama members and analyze all exposure routes, including potential groundwater consumption, to evaluate cleanup

Yakama Nation General Comments
DOE/RL-2007-21 RCBRA Volume II Draft C

The Yakama Nation has completed a preliminary review of the U.S. Department of Energy's DOE/RL-2007-21 Volume II Draft C: River Corridor Baseline Risk Assessment (RCBRA), Human Health Risk Assessment (HHRA), and has a number of general concerns with DOE's approach to assessing baseline risk to tribal members exposed to Hanford contaminants. See Attachment 1 for supporting commentary on some of the general issues presented below. Most of the Yakama Nation's concerns and original comments on the RCBRA Draft A (DOE, 2007) have not been addressed, and many of the supplemental comments on Draft A (submitted to DOE in December 2010) still apply; these supplemental comments are provided again in Attachment 2. Although the Yakama Nation has engaged in a process to fully understand the risk to its people and treaty resources posed by the Hanford Site, to date DOE responses to comments on basic concerns regarding the risk assessment process have generally been dismissive.

The release of RCBRA drafts and the assessment itself continues to be disjointed and potentially biased. A detailed table or diagram of the timeframe for actions (documents, data collection, decisions, etc.) would help the reader understand the chronology of events as well as the relationship to the current and future conditions at the site. Overall, this baseline HHRA is biased because it assumes anticipated land use and institutional controls. According to DOE guidance, "EPA directed that exposures that are limited by institutional controls may not be factored into a baseline risk assessment for a CERCLA RI/FS" (DOE, 1992). Language in the baseline HHRA indicates that DOE is not considering cleanup to unrestricted use and is striving toward a less stringent cleanup based on the Comprehensive Land-Use Plan. Baseline risk should drive cleanup decisions that allow for unrestricted and multiple uses consistent with the conclusions of the Hanford Site Future Uses Working Group, which emphasized that cleanup allow multiple uses of the site once remedial actions are complete. Assuming that contaminants remain in place implies a Long-Term Stewardship Program Plan must be implemented that will remain effective longer than any human institution has ever existed. An explicit acknowledgement of this challenge should be carried forward into the baseline HHRA. The Yakama Nation has the following general comments:

1. **Acknowledge the Treaty of 1855 as an ARAR.**

   The Yakama Nation developed an exposure scenario and requested that it be correctly incorporated into the RCBRA, assuming site-wide, unrestricted, residential use. The Yakama Nation's consideration of this document and all other similar documents at Hanford is governed in the first instance by compliance with the Treaty of 1855 (12 Stat. 951), which should be considered as an applicable or relevant and appropriate requirement (ARAR).

   The Treaty of 1855 between the Yakama Nation and the United States of America reserved specific rights and resources. These rights listed in Article 3 of 12 Stat. 951 include "...the right of taking fish at all usual and accustomed places...together with the privilege of hunting, gathering roots and berries, and pasturing their horses and cattle upon and unclaimed land." The U.S. Constitution in Article VI states, "...all Treaties made, or which shall be made, under the Authority of the United States, shall be the supreme Law of the Land..." The U.S. government has a fiduciary responsibility to the Yakama Nation to protect our Treaty rights and resources, our culture, health, and welfare. The Hanford Site is a portion of the Yakama Nation's homeland ("front yard"). In light of these facts, 12 Stat. 951

must, at a minimum, be identified as an ARAR in the CERCLA Remedial Investigation/Feasibility Study cleanup process (40 CFR 300.430(b)(9) and at (d)(3)). The Treaty has not yet been recognized as such in this effort or under other CERCLA actions undertaken at the Hanford Site. A full analysis of the risks to Yakama Treaty resources and peoples' health has yet to be performed. The risk assessment is deficient without this complete analysis.

2. **Adopt CERCLA and MTCA risk levels.**

It is incorrect to consider only incremental risk "above background levels." The baseline HHRA cites the EPA OSWER Directive "Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination" as the "origin" of a cleanup threshold of "15 mrem/yr above background." However, the referenced EPA document only refers to a value of 15 mrem/yr, not 15 mrem/yr *above background* (EPA, 1997). This statement suggests that doses estimated from soil concentrations measured in background samples collected on-site will be excluded from total radiation dose used to calculate site risk, which is not appropriate for a baseline risk assessment. All exposures (and associated doses) measured at the site contribute to baseline risk and should be included. If IAROD's included a cleanup level of 15 mrem/yr above background, residual risks could be higher than the 3 x 10^{-4} probability indicated in the baseline HHRA.

EPA guidance equates a 15 mrem/yr dose limit to a lifetime cancer risk of 3 x 10^{-4} (based on a specific risk coefficient), which is three times the maximum allowable value under CERCLA. Moreover, if the EPA's own risk coefficients for radiation are used, it equates to a fatal cancer risk of more than 5 x 10^{-4} and a cancer incidence risk of 1 x 10^{-3} (see Attachment 1), which is well outside the CERCLA range of 10^{-6} to 10^{-4}. The CERCLA limit for managing hazardous waste cleanup is referred to in the National Contingency Plan and EPA's directive 9355.0-30 as a target risk range of 10^{-4} to 10^{-6}. It is important to consider this *range* when arriving at "acceptable" risk limits for all peoples who may reside on or live near the Hanford site. The upper-bound risk level of 1 x 10^{-4} can be determined unacceptable (i.e., not protective enough) based on site-specific conditions, particularly when there are uncertainties in the assessment results, as in this baseline HHRA.

Under MTCA, each carcinogenic hazardous substance is limited to a concentration corresponding to a 1 x 10^{-6} lifetime cancer risk. When more than one hazardous substance is present, the combined lifetime cancer risk limit for chemicals and radiation is 1 x 10^{-5}. While MTCA has been generally interpreted as applying to chemicals only at Hanford, this interpretation is too limited and should also consider Hanford's extensive radionuclide contamination. Since MTCA explicitly defines radionuclides as hazardous substances, the combined limit for radionuclides and chemicals should correspond to a lifetime cancer risk of 1 x 10^{-5} or less.

The risk assessment should adopt agency-refined and accepted MTCA and CERCLA risk thresholds (10^{-5} or 10^{-6}), rather than 15 mrem/year and 100 mrem/year dose limits, using age-specific dose conversion factors to derive dose guides for cleanup. Individual drinking water maximum contaminant levels (MCLs) for man-made radionuclides should also be developed and adopted for anthropogenic radionuclides corresponding to a 10^{-6} lifetime cancer incidence.

3. **Conduct a comprehensive baseline risk assessment.**

EPA guidance (EPA 540-R-01-003, Appendix B [EPA, 2001a]) states specifically that all substances present at a site that exceed risk threshold concentrations should be included in the baseline HHRA. A baseline risk assessment considers all site risks, including those from naturally occurring and ubiquitous contaminants. Data should *not* be selective (e.g., excluding waste sites or contaminants) but should include all data sources applicable to evaluating current and future conditions at all upland, riparian, and nearshore operational and non-operational areas. Without full characterization and evaluation, it should be assumed that the nonoperational areas or areas in between the operational areas have been impacted by Hanford Site releases and therefore pose a risk. A holistic approach would ensure that protective decisions are made for the site in its entirety. Comparisons to background concentrations should only be considered during the feasibility study to support risk management decisions and select appropriate cleanup actions.

While cleanup decisions may ultimately be defined by management boundaries, the risk assessment should be based upon actual human behaviors. It is particularly inappropriate to infer institutional controls in a baseline risk assessment. In the discussion of the methodology (Section 3), the authors state that the "risks associated with yet-to-be remediated waste sites are not a focus of the remainder of this report…." This suggests that this risk assessment is incomplete and will be finished later. Please clarify when such areas will become a focus and included in assessing baseline risks at the site. The current piecemeal approach to assessing separate decision units and select waste sites does not sufficiently evaluate risk to human health and the environment posed by Hanford contaminants. Please include more discussion in the uncertainty section regarding this disparity between exposures to humans and ROD decision units. Risk management decisions and assumptions about what areas to remediate are being made prematurely.

As noted above, *no* institutional controls should be assumed in conducting the baseline risk assessment. Residual radioactive and hazardous contaminants may remain in and around the River Corridor for a period of time extending hundreds to thousands of years into the future. The "River Corridor" may change in this time because of geologic events or flooding. To be consistent with other EPA criteria in CFR Part 191 for establishing a lower limit on the probability of events and processes that need to be considered for the protection of human health, criteria stating that the events and processes that have a probability of greater than 1 in 10,000 of occurring within 10,000 years should be specified. These events and processes should include consideration of intruder scenarios within and without the River Corridor boundary that would increase the exposure of Yakama Nation residents to harmful contaminants. The assessment criteria should include elicitation of an expert panel to establish the probability of such events and processes. Such experts should be independent from the DOE and its risk assessment contractor. The Yakama Nation should be given the opportunity to provide other qualified experts to present information to supplement the panel's information to aid in the panel's evaluations and determination of appropriate events and processes.

4. Conduct a cumulative risk assessment for a tribal resident.

All contaminated media and sources within the geographic boundaries of the site must be considered and summed together in a cumulative risk assessment. The conceptual site model presented in the baseline HHRA is insufficient because it fails to accurately and completely identify all sources of contamination, describe transport mechanisms through various environmental media, and evaluate potential risks to tribal members. Several pathways and sources have been omitted from consideration in the risk assessment, such as contaminants in the vadose zone and air releases from past operations. The Public Health Assessment for Hanford (ATSDR, 2006) also identified several exposure pathways that were not included in the baseline HHRA. These pathways, including consumption of milk from cows raised on or near site contaminants and the consumption of yarrow and mulberries, should also be included in the assessment. A cumulative risk assessment for a Tribal Resident scenario should include the following media, exposure pathways, and receptors:

- Soil (all depths)
- Surface water
- Sediment
- Porewater
- Seeps
- Groundwater (migration, irrigation)
- Vapor intrusion
- Plants (including roots)
- Game animals
- Fish ingestion
- Milk ingestion
- Sweat lodge (including children)
- Breast-fed infant
- Embryo/fetus

Concentrations of contaminants in foodstuffs should be accurately quantified and site-specific contaminant transfer factors should be calculated to help assess dietary and non-dietary exposures to these resources. Models alone are insufficient. Tribal members use wild plant species, for example, that have not been characterized, and which have different qualities from the garden plants that have been monitored and modeled. Whole plant tissue, as well as leaf and root tissue separately, should be analyzed to adequately reflect realistic exposures based on tribal uses. Measured concentrations of plant (and game) tissue collected from the site should then be compared to modeled values to validate modeled results.

Although the authors state that the baseline HHRA does not include a complete assessment of the fish ingestion pathway, it is misleading to present the limited data at this point since they are inadequate and not applicable to a final assessment of baseline risk. Restricting the evaluation of aquatic species to clams, crayfish, and sculpin is not sufficient. Other species should be evaluated, such as salmon, sturgeon, lamprey, sucker, bass, and other species more representative of a tribal diet (i.e., Columbia River data should be included). Since the characterization of this pathway is currently incomplete, it is unclear how decision-makers can use this information in the remedial RI/FS process.

5. Include contaminants from the Central Plateau and in the Columbia River.

The goal of protecting the Columbia River and of protecting groundwater requires restoration of groundwater to meet drinking water standards throughout the site. The baseline risk assessment should include current and future conditions, which includes contaminant concentrations in groundwater migrating from the Central Plateau to the River Corridor. Many radiological contaminants will pose a risk far into the future, and the effect of contamination migrating to the River Corridor and the Columbia River, including effects from irrigation, should be assessed. See Attachment 1.

The separation of exposures from water and sediments in the Columbia River presents a challenge for the individual or community that may someday inhabit this site. Their exposures are not defined by the current decision boundaries identified for cleanup. It is not possible for an individual or a community to reside in this area without encountering the river as well as upland and riparian habitats. Restricting the evaluation of the river to just the nearshore areas bordering the Hanford Site to a water depth of 6 feet is not sufficient. To be fully protective and provide a true picture of the risks at the site, the entire river system must be evaluated. Contamination has been found, for example, in the main river channel during recent upwelling studies in sloughs along the Hanford Reach, on the 100-D Island, in and around the remaining effluent pipes, and in abiotic and biotic samples throughout the Hanford Reach and beyond (Tiller, et al., 2009).

6. The groundwater assessment is incomplete.

As discussed above, the baseline HHRA should consider concentrations of groundwater contaminants that are predicted in the future through migration from the Central Plateau to the River Corridor. Migration of elevated concentrations of contaminants is not only occurring today, but has been estimated to be even greater in the future, as shown in the Draft Tank Closure / Waste Management (TC/WM) Environmental Impact Statement (EIS) calculations (DOE, 2009). Estimation of risks for future use should not be restricted to the years 2075 and 2150. While there is some discussion of exposure in those futures years in terms of radioactive decay, there should also be a discussion included regarding other future conditions that include contaminant transport from the Central Plateau.

Both the methodology and data used to conduct the screening-level groundwater risk assessment presented in the baseline HHRA raise a number of concerns. Because of these concerns, the overall applicability of the screening analysis results for characterizing risks in the River Corridor is unclear. The first concern is related to conducting risk calculations separately for each Operable Unit (OU). This especially concerns the OUs in the 100 Area, which are located very close to each other. This approach does not account for cumulative risks. The contaminant concentrations in groundwater used in the screening analysis represent the average values for an OU calculated using the available data over a 10 year period (1998 through 2008). The following concerns relate to the approach for calculating contaminant concentrations:

- The contaminant concentrations calculated using this approach may not represent the groundwater exposure concentrations for the residential scenarios. A key assumption in any residential scenario is that a resident will install a groundwater well and pump groundwater for residential uses. Pumping water will inevitably result in capturing multiple contaminant plumes from the different OUs and mixing these contaminants in the well water. The contaminant concentrations in the well water will depend on the well locations with regard to the different plumes, well pumping rate, well productivity interval, aquifer flow and transport parameters, and contaminant-specific properties. Contaminant concentrations should be calculated based on the simulation of these processes, not by averaging the observed concentrations over an OU.
- Using a 10-year period to calculate the average contaminant concentration may not be appropriate because the contaminant plumes can either move in or move out of the OU during this period. For example, if the contaminant plume reached the OU in 2005, then using data from 1998 to 2004 to calculate the average concentration of this contaminant is insufficient.
- Average concentrations over a 10-year period do not represent the possible dynamic evolution of concentrations with time. This evolution can only be determined if all the contaminant sources within the OUs and outside the OUs are defined and the future concentrations are predicted based on the contaminant transport from these sources to the location of the potential groundwater well.
- The simulations of the contaminant plumes considered in the Draft TC/WM EIS indicated that a number of contaminant plumes will reach the River Corridor in the future. The results of these simulations should have been considered in developing representative concentrations.

Section 6.1 of the baseline HHRA provides a list of activities that DOE is planning to implement to reduce the uncertainties, update the conclusions of the screening-level groundwater risk assessment, and ensure that no contaminants were inadvertently overlooked, based on the use of the existing data set. However, the planned activities do not include the most important actions required to reduce the major uncertainties. The additional activities should identify and characterize the major sources of contaminants within and outside the OUs and collect the data needed to predict the contaminant concentration evolution with time.

The irrigation scenario should also be considered in the screening analysis. A potential significance of this pathway is discussed in the uncertainty analysis of the local-area risk assessment in Section 5.0. The potential impacts, which should be considered in this discussion, are the recycling of contaminants through irrigation and secondary soil contamination. Irrigation recycling may occur when contaminated groundwater is used for irrigation. The contaminants will be returned to the aquifer (recycled) through recharge of irrigation water not transpired by plants. Through this process, the contaminants may accumulate in the groundwater and the groundwater concentrations may increase with time. The sorbed contaminants in the irrigation water will also contaminate the surface soil in addition to contributing to the irrigation recycling. This secondary contamination of soils constitutes one of the potential inter-relations between the groundwater and soil related risks, which was not considered in any of the risk calculations.

7. More characterization and transparency in data analysis are needed.

To support coherent and protective cleanup decisions, CERCLA RI/FS guidance calls for fully characterizing the nature and extent of contamination. The baseline HHRA is incomplete due to the lack of characterization of the nature and extent of contamination, including characterization for unremediated waste sites and the land in between these sites. Historical documents indicate widespread contamination throughout operational areas (Gerber, 1992). More complete characterization of environmental conditions is required to allow a more spatially robust evaluation and to reduce the current level of uncertainty. Also, clarify how 156 waste sites were selected, particularly when 164 were identified elsewhere in the document, and others exist.

There is no cohesive presentation of the criteria or decision tree for representative data selection and/or disqualification. In characterizing risk, the baseline HHRA improperly excludes contaminants without sufficient justification and, therefore, potentially underestimates risk estimates. For example, no data were provided at certain waste sites for specific contaminants that were either found at other waste sites or that would have been expected at that waste site (e.g., decay products). It is unclear whether results for certain COPCs were simply not reported, or whether they were not analyzed at select waste sites. It would helpful to reorganize the document to include a better presentation of the data selection and use in generating the exposure point concentrations.

The requirement that a contaminant needs to be reported at one-third of the wastes sites to be considered a COPC is not protective. This screen potentially eliminates relatively unique waste sites, as well as adds to the problem of not including all contaminants in the risk evaluation. Because different reactors had different auxiliary missions, such as the production of special nuclear materials, this methodology allows for removing COPCs from consideration that may be present at significant concentrations at only a few sites (e.g., COPCs present in the K-reactor fuel basins or the 618 burial grounds). The COPC selection process should be revised and the list of accepted COPCs in the baseline HHRA should include contaminants of this nature so that unique site contamination and associated risks are not overlooked.

There are statements throughout the baseline HHRA that cleanup verification soil data from interim actions are "protectively biased." However, the sampling and analytical criteria (e.g., number of samples, detection limits) that were used for the interim actions may be too limited to allow such a determination to be made. It is not appropriate to assume that the sampling and analysis for interim decisions can be applied to final actions. In certain instances in the baseline HHRA, various contaminant concentrations (e.g., uranium and chromium) were eliminated from the risk estimates because of sampling and/or analytical method. Although laboratory sample preparation and analysis were not considered a major uncertainty, they did affect whether certain samples were excluded (e.g., numerous incidences of high PCB detection limits). This suggests that laboratory analyses are indeed a major source of uncertainty. Discuss how uncertainty and data gaps will be addressed. Also, clarify how censored results and high detection limits affect risk estimates in the uncertainty section.

8. Select appropriate reference sites and background sample locations.

Including reference and background data that have been collected from, and adjacent to, the Hanford Site is inappropriate. These areas, due to their location either onsite or proximal to the site, have most likely been influenced by releases from the Hanford Site in the form of airborne contamination and abiotic and biotic movement of contaminants. Background soil samples should have the same basic characteristics as Hanford soils, but should be collected from areas not influenced by releases from the Hanford Site. Include a cohesive presentation of the criteria or decision tree for selection and/or qualification of an area to act as a reference site. A reference site evaluation conducted in 2007 concluded that "additional off-site reference locations should be identified if a comparison is being made to pre-Hanford conditions," i.e., conditions without Hanford contamination (Hart Crowser, 2007). The two off-site locations sampled near Beverly, Washington are not sufficient.

The EPA human health risk assessment guidance (EPA, 1989) is cited as the basis for selecting reference site data. However, the cited document only discusses using statistics to identify site-derived versus non-site-derived substances. This guidance document as well as much newer guidance are clear that COPCs are all substances posing risk, whether site-related or not. EPA offers that reference "targets" for contamination *may* be derived from an evaluation of the contaminant gradient on a site based on the lowest concentrations, but no such gradient analysis was conducted to select the lowest concentrations from the reference data set (EPA, 2001a).

9. Review of GiSdT data used in the RCBRA questions "reference" site selection.

There is concern regarding "reference" site selection based on our limited evaluation of americium, thorium, and uranium data. Soil samples collected outside of decision areas contained concentrations of americium (Am)-241, an anthropogenic radionuclide, and concentrations of thorium that exceeded concentrations measured in waste site soils for these radionuclides. Am-241 is a decay product of plutonium (Pu)-241, which is a radionuclide that is also produced during the production of Pu-239 (of primary interest during Hanford's active operations). Am-241 contamination was likely released from Central Plateau stacks as well as in liquid wastes associated with plutonium reclamation. Argonne National Laboratory has stated that "at DOE sites such as Hanford, americium can be present in areas that contain waste from the processing of irradiated fuel" (Peterson, et al., 2007). Furthermore, the Yakama Nation noted in supplemental comments provided on the RCBRA Draft A (Attachment 2) that data presented in Draft A showed high concentrations of Am-241 in soil and fish tissue – neither of which were derived from identified waste sites. Review of the data provided in the Guided Interactive Statistics Decision Tools (GiSdT) database for the RCBRA also identified Am-241 in samples considered "reference." The presence of Am-241 in samples from outside decision areas indicates that the sites selected as reference have been contaminated by Hanford Site activities.

The technical rationale for excluding thorium as a COPC is unclear, and a citation is needed for the Tri-Party Agreement exclusion that was noted in the baseline HHRA. Thorium (Th)-232 was handled in large quantities at the Hanford Site, particularly during the period of uranium (U)-233 production. Resulting Th-232 contamination was known to be widespread and common, particularly in the 300

Area of the River Corridor (EPA, 2001b; Gerber, 1992; Kubiak, et al., 2009; Zachara, et al., 2007). Quantile plots can be constructed (per EPA 2002 guidance) of usable, non-zero, unflagged thorium soil measurements from the GiSdT that demonstrate "reference" site concentrations of Th-232 were greater than those observed in cleanup verification samples from waste sites (consistent in the upper 70% of the data range). While Th-232 does occur in soil naturally, it normally occurs at concentrations of approximately 10 mg/kg (Peterson, et al., 2007), which is equivalent to 0.1 pCi/g of activity. Measured concentrations of Th-232 at "reference" sites ranged from 0.5 pCi/g to 1.5 pCi/g (and waste site data ranged from 0.1 to 0.3 pCi/g below reference values). Not only are the reference values as much as 15 times higher than the natural abundance of Th-232, they are also up to three times greater than those observed in waste site cleanup verification packages. Because background soil samples were not collected from areas uninfluenced by Hanford contaminant releases, it is incorrect to state that site concentrations of thorium are "not different from background."

Inadequate justification was provided for rejected U-238 measurements. Because of its exceptionally long half-life, high quality factor (i.e., causing biological damage), and the large amounts handled at Hanford, uranium is of particular importance and interest when evaluating site data. Evaluation of the uranium data from the usable and unusable data sets in the GiSdT revealed that a large number of relatively high U-238 measurements (from approximately 2 pCi/g upward) were rejected because they were collected using gamma spectrometry. Since these data, if included in the risk assessment, could indicate much higher potential risks, these data should not be rejected as a matter of convenience based on the analytical method alone; there should be some demonstration that the measurements made are not comparable to other methods considered acceptable, and associated data gaps should be filled.

10. Review of GiSdT data used in the RCBRA raises data quality concerns.

A limited review of the baseline HHRA data quality identified concerns related to uranium measurements. Uranium measurement data are not consistent, suggesting data quality issues with the GiSdT data used for the RCBRA. Previous review of uranium measurements used in the RCBRA Draft A identified several problems with uranium measurements used as part of the study (Attachment 2). Review of 1,040 unflagged, non-zero, usable soil samples in the GiSdT database that have measurements for U-234, U-235, and U-238 indicates that similar problems remain. Uranium at the Hanford Site should have an isotopic signature similar to that observed in natural uranium. As such, the ratios between uranium isotopes in the same sample should follow consistent trends wherein the relative percent of total radioactivity in a sample is attributed to the following proportions:

- U-234 = 48.9 percent
- U-235 = 2.2 percent
- U-238 = 48.9 percent

Examination of the usable GiSdT data revealed that 202 of the 1,040 values selected – nearly 20 percent – had a percentage of the total radioactivity attributable to U-235 equal to or greater than five percent. Such a percentage is not possible with natural uranium. In fact, only about one-half of the samples (524 of 1,040 total) had U-235 activity ratios that were in the more reasonable range of 2 to 4 percent of the total radioactivity. Additional inspection of the ratios for U-234 and U-238 activity to total activity revealed similar data quality concerns. As noted above, each of these radionuclides should account for approximately 50 percent of the total radioactivity in the sample. However, one quarter (255 out of 1,040) of the samples inspected had U-238 to total radioactivity ratios less than 46 percent. This bias was particularly evident for samples with higher total activity measurements. The systematic underestimation of U-238 was relatively consistent with a similar bias of U-234 activity ratios that were too high. Again, the trend was particularly evident in samples with high total activity.

Another method of evaluating the data was also conducted. Ten samples with complete uranium isotope data also had chemical measurements of inorganic uranium. Using the specific activity of uranium, each sample's mass could be calculated based on measured radioactivity and compared with the direct measurement of the inorganic metal. While seven of the ten samples available showed good agreement between the two measurement methods, the remaining three points have chemical measurements that range from 120 to 150 percent of the uranium mass calculated from radioactivity measurements. Measurements continue to *not* meet expected trends, finding more uranium in chemical measurements than in radioactivity measurements.

Given uranium's history on the Hanford site, these data should be treated with special diligence due to the known widespread contamination and significant associated risks posed by the contamination. Yet this analysis has identified serious problems with a significant portion of the usable data and, therefore, the criteria used for its selection. Due to the serious nature of these concerns, additional review and evaluation of the data and analyses performed are necessary before final comments and proposed revisions can be provided.

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No.	Section	Page, Figure/Table	Comment
Forward – Hanford Site Cleanup Completion Framework			
1	Goals for Cleanup	Forward-2, Figure F-1	The Forward states that Cleanup Goals carry forward key values captured in forums such as the Hanford Future Site Uses Working Group. *The Future for Hanford: Uses and Cleanup, The Final Report of the Hanford Future Site Uses Working Group* (1992) states, "Following completion of waste management activities, the Working Group desires that the Central Plateau be suitable for other general uses 100 years from decommissioning of waste management facilities and closure of waste disposal areas." Define the necessary steps to ensure remaining contamination will be minimized to allow for general uses, and for the Yakama Nation to exercise treaty rights.
2	Goals for Cleanup	Forward-2, Figure F-1	The Forward states that Cleanup Goals carry forward key values captured in Hanford Advisory Board advice letters. However, Hanford Advisory Board Advice #226 recommends "An estimate of the cumulative risk of the sum of DOE's future actions should be an integral part of the cleanup planning process." Re-evaluate Cleanup Goals that fail to address the need for a comprehensive cumulative risk assessment.
3	Goals for Cleanup	Forward-2, Figure F-1, Goal 1	A set of criteria have not been developed to meet "Goal 1, Protect the Columbia River," and it is unclear how cleanup of the Columbia River will be incorporated into the CERCLA Remedial Investigation/Feasibility Study (RI/FS) process. Clarify how the scheduled ecological and human health risk assessments for the River Component Remedial Investigation will be incorporated into the RCBRA.
4	Goals for Cleanup	Forward-2, Figure F-1, Goal 7	Goal 7 is contradictory to Goals 1 and 2. Institutional controls will fail to protect the environment, particularly groundwater, from long-lived radionuclides. As such, institutional controls will not restore the land to allow full and safe use by the Yakama Nation. The goal should be a cleanup that protects human health and the environment without the need for long-term institutional controls.
5	River Corridor Cleanup	Forward-5	Concerns previously expressed by the Yakama Nation regarding the Comprehensive Land Use Plan still apply, including an over-reliance of land use designations and institutional controls to minimize exposures and limit cleanup that are not consistent with tribal treaty rights and the federal trust responsibility.
6	Central Plateau Cleanup	Forward-6	Federal regulations (40 CFR 191.14(a)) indicate that active institutional controls cannot be relied on for environmental protection for more than 100 years. Therefore, passive institutional controls such as covers, markers, and public records would be the only mechanisms to inhibit intrusion onto the Hanford Site and into waste sites (sacrifice zones) after 100 years; such controls have been shown to fail over time.
7	Long-Term Stewardship	Forward-8	The *Hanford Long-Term Stewardship Program Plan, Preliminary Draft, Revision C,* released February 25, 2010, acknowledges that long-term stewardship: 1) must address significant challenges to demonstrate long-term fiscal viability and the minimization of intergenerational liability; and 2) could be required for many generations and longer than nearly any other human institution has survived intact. It has also not yet been determined how Hanford's long-term stewardship program will "ensure continued protectiveness of cleanup remedies" or "protection of natural resources, the environment, and human health" as stated in this section of the RCBRA. Revise the text to more accurately reflect the unreliability of long-term institutional controls and that the *Hanford Long-Term Stewardship Program Plan* is in an early stage of development.

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Executive Summary, Glossary			
8	Completion of Cleanup Actions	ES-4	The determination of cleanup actions (e.g., risk management decisions) cannot be made at this time for areas of the site. Revise the last sentence of this section to state that there are areas where cleanup decisions have not been made, rather than "cleanup actions are not anticipated."
9	Current Conditions in the River Corridor	ES-4	The first sentence of this paragraph should describe site characterization as "limited." Page ES-3 establishes that the characterization is limited and should be described as such here and elsewhere in the document. Revise to be consistent throughout the baseline HHRA.
10	Current Conditions in the River Corridor	ES-5	The determination of adverse impacts cannot be made at this time for areas of the site, including non-operational areas. Revise the description of non-operational areas in the last sentence to reflect that impacts are largely unknown because of lack of characterization. Stating that these areas are "not anticipated to be adversely affected by releases" is incorrect given the mobility of contaminants through biological or abiotic events.
11	Assessment of Interim Actions	ES-6, Text Box	Particular site-specific conditions that would justify the acceptability of a risk estimate "around" 1×10^{-4} are not defined. OSWER Directive 9355.0-30 states that a risk manager may decide that a baseline risk level less than 1×10^{-4} is unacceptable (i.e., still not protective enough) due to site-specific reasons and that remedial action is warranted where, for example, there are uncertainties in the risk assessment results. The text box language should be revised to more accurately reflect the full range of alternatives put forth by the OSWER directive.
12	Assessment of Interim Actions	ES-6 to E-7	A cumulative risk assessment is defined by EPA as "an analysis, characterization, and possible quantification of the combined risks to health or the environment from multiple agents or stressors" (EPA Cumulative Risk Assessment Framework, 2003). It is misleading to refer to "cumulative" cancer risks only for chemicals and only from remediated waste sites, as used in this baseline HHRA.
13	Assessment of Interim Actions	ES-7, Text Box	This section, as well as section 3.6.4, references the EPA OSWER Directive "Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination" as the "origin" of a cleanup threshold of "15 mrem/yr above background." However, the referenced EPA document only refers to a value of 15 mrem/yr, not 15 mrem/yr *above background.* DOE's statement suggests that doses estimated from soil concentrations measured in background samples collected on-site will be excluded from total radiation dose used to calculate site risk – this does not include cosmic and other natural radiation dose – and is not appropriate for a baseline risk assessment. All exposures (and associated doses) measured at the site contribute to baseline risk and should be included. If IARODs included a cleanup level of 15 mrem/yr *above background*, residual risks could be higher than the 3×10^{-4} probability indicated in this section. It should also be noted that a 15 mrem/yr dose produces a cancer risk far greater than allowed under CERCLA and MTCA; EPA admits that the lifetime risk is 3×10^{-4}, which is three times the maximum allowed under CERCLA. Additionally, if EPA's own risk factors (published as public information) are considered, the *fatal* cancer risk is 5×10^{-4} to 6×10^{-4} and the cancer *incidence* risk as estimated by the National Academies is about 1.1×10^{-3} (see Attachment 1). The maximum allowable dose from residual radioactivity from all pathways should be reduced to conform to CERCLA and MTCA as described in the general comments.

14	Assessment of Interim Actions	ES-7	Uncertainty associated with using cleanup verification data to estimate risk should not be described as possibly overestimated because of backfill. Although perhaps not representative of surface soil concentrations, risk at these waste sites may also be considered underestimated since confirmation samples may not 1) reflect additional contamination at depth or horizontally, and 2) may be located at depths accessible by an individual (e.g., excavating for dwellings, wells, or native plants). Revise the description accordingly here, as well as in ES-17, 2-43, and other sections of the baseline HHRA.
15	Assessment of Groundwater	ES-8	The 331 wells used in the evaluation represent a very small fraction of available wells. It is unclear why so few wells (of the thousands of active wells) were used for the evaluation. Clarify the selection of limited wells (and hence data) for the groundwater assessment.
16	Broad-Area and Local-Area Risk Assessments	ES-9	It is misleading to consider the samples associated with 20 remediated waste sites "a conservative representation of average contaminant concentrations," since it is unknown if all waste sites have been identified. Revise the sentence to delete "conservative" and read "average *known* contaminant concentrations..."
17	Broad-Area and Local-Area Risk Assessments	ES-10	It is stated that arsenic concentrations in upland and riparian site soils are not significantly different from background. Are background arsenic concentrations derived from locations that are impacted as a result of historic pesticide use? If so, then different, uninfluenced background locations should be selected to assess the level of impact and related risk from arsenic at the Hanford site.
18	Broad-Area and Local-Area Risk Assessments	ES-11	It is appropriate that the considerable uncertainty regarding edible plant contaminant concentrations and site-specific soil-to-plant uptake factors are noted. Concentrations of site contaminants in these materials, however, is a critical data gap in this risk assessment. Collection and analysis of site-specific plants should be mandatory, not just "considered" as part of the RI/FS process.
19	Broad-Area and Local-Area Risk Assessments	ES-12	It is incorrect to state that the three species of fish analyzed are not plausible food sources for chronic human exposure, as they are consumed by tribal members. This statement should be removed. There is, however, too much uncertainty from the limited species and limited analytes evaluated. Revise the assessment to include data from multiple species of Columbia River fish, and/or clarify how results from the Columbia River Component assessment will be combined with these results to obtain a complete assessment of risk.
20	Final Recommend-ations	ES-19	Only three scenarios were used to develop PRGs, none of which were based on residential scenarios. Revise the statement to reflect that only a *limited* set of the scenarios were used to develop PRGs, and explain when the PRG development process will include residential scenarios.
21	Glossary	xix	Reference site: the definition should not include "comparatively uncontaminated site." This is misleading. While the EPA definition allows for the possibility of "least affected or altogether unaffected" it is clear that EPA/540/F-94/012 also states that the reference site should be "unaffected by site contamination."
22	Glossary	xx	Uncertainty analysis: This definition should include statistical comparisons of variability as well as qualitative statements regarding lack of knowledge.
1.0 Introduction			
23	1.1	1-3	The first statement about characterizing "current and potential future risks" should be qualified by adding to the statement that they are posed by "current, known" releases. In the case of Hanford, where many contaminants are long-lived, decay into other hazardous substances, and are migrating from the Central Plateau to the River Corridor and into the Columbia River, transport should be considered and modeled peak concentrations throughout the site should be used to assess future risks.

24	1.1	1-5	We disagree with the statement that an overarching goal is to "minimize the cleanup footprint." This is inconsistent with a comprehensive and complete cleanup of Hanford contaminants.
25	1.2	1-6	This statement "Nonoperational areas include large portions of the River Corridor that are outside of the operation areas and are not anticipated to be impacted by Hanford Site releases" is not correct. Mobility of contaminants through biological or abiotic events may transfer contaminants to areas beyond the "operational areas." Available wind-rose data indicates that a large portion of the site, state, and Columbia River basin has been affected by Hanford air releases. Revise the last sentence of the second bullet to state that non-operational areas may be impacted, although the impacts are unknown because of lack of characterization.
26	1.3.2	1-7	A baseline risk assessment should not rely on land use restrictions or institutional controls. Therefore, we do not believe that "The scope of the human health and ecological risk assessment processes depend on site-specific factors such as reasonably anticipated future land use and anticipated beneficial uses of groundwater and surface water."
27	1.3.4	1-8	This section includes the statement "Certain protectiveness standards for WAC 173-340 are pertinent to the baseline risk assessment effort," but does not indicate what those protectiveness standards include. Section 702(10) of MTCA (WAC 173-340) states that "When evaluating cleanup actions performed under the federal cleanup law, the department shall consider WAC 173-340-350, 173-340-355, 173-340-357, 173-340-360, 173-340-410, 173-340-420, 173-340-440, 173-340-450, 173-340-700 through 173-340-760, and 173-340-830 to be legally applicable requirements under Section 121(d) of the Federal Cleanup Law." All of these requirements should be included as applicable requirements for CERCLA actions, including the maximum allowable risk thresholds of 1×10^{-6} for individual carcinogens and 1×10^{-5} for multiple carcinogens and multiple pathways. As radionuclides are considered hazardous substances under MTCA (WAC 173-340-200), they should be subject to the same risk thresholds as all other carcinogens, including the total site cancer risk threshold of 1×10^{-5}.
28	1.3.5	1-8	In addition to the MTCA requirements identified in this section, groundwater discharges to surface water at the Hanford site must also meet requirements included in WAC 173-340-720 (8)(d)(i), which allows for a "conditional point of compliance that is located within the surface water… where ground water flows into the surface water."
29	1.4.1	1-9	Clarify up front how the calculated areas for the 100 and 300 area decision units are adequate to conduct both broad-area and local-risk assessment (e.g., are any areas of the site not included?).
30	1.4.2	1-10	It is unclear how the Columbia River Comprehensive Impact Assessment (CRCIA) was incorporated into the RCBRA. Clarify both the aspects of the CRCIA that were used and how they were used in the baseline HHRA.
31	1.5	1-14	Release of RCBRA drafts has been incongruent. Draft A was released in 2007. Draft B was never released to the Yakama Nation. Draft C was released in 2010, after a 15-month delay. With the release of Draft C during year-end holidays (and a limited review period), Volume II (Human Health) was released and comments due to DOE before the release of Volume I (Ecological). Lastly, the risk assessment has been conducted before all remedial investigation activities, such as adequate site characterization and the availability of data requisite to assessing cumulative site risk.

2.0 Site Background and Cleanup Activities			
32	2.1.3	2-2	The accuracy of the statement "there is no longer significant artificial recharge due to operations in the 100 and 300 Areas, as disposal of liquid waste to ground has ceased" is questionable depending on the definition of the "100 and 300 Areas." At times, it is used specifically to refer to the 100 and 300 Areas, while in other instances it is used to include all river corridor ROD decision areas (see paragraph 2 of Section 3.3, Pg 3-39 as an example). There is evidence that artificial recharge may be occurring at Energy Northwest (ENW). Both mounds and depressions can be found in close, if not direct proximity to the 618-11 burial ground. There are two known outfalls for waste water at ENW, and mounding is possible. However, the largest mound occurs directly under the cooling structures. Either this statement should be qualified with respect to the ENW site, or a consistent definition of the "100 and 300 Areas" should be used throughout the document.
33	2.4.4.1	2-20	First paragraph: "Some of these high-priority waste sites are included in this ecological risk assessment." Please review the entire document for inadvertent text from the Ecological Risk Assessment (Volume I).
34	2.5.1	2-24	The text misleadingly states that the methods used to initially collect waste disposal information were "exhaustive." A significant number of waste sites have been identified since the initial discovery effort, and it is expected that additional waste sites have yet to be discovered by the orphan waste site identification and evaluation process. Delete the word "exhaustive" from the text.
35	2.5.2	2-24	According to the Tri-Party Agreement Appendix C, waste/release site listings are intended to be updated according to the official list of sites requiring remedial investigation/action under CERCLA §120. The current version of Appendix C, dated December 8, 2010, does not accurately reflect all of the CERCLA waste sites, or even all the sites used for RCBRA input data. Revise the text to indicate that the Hanford Site Waste Management Units Report contains a more accurate listing and status of CERCLA waste sites than does Tri-Party Agreement Appendix C.
36	2.5.3	2-24 to 2-25	Although the Hanford Past-Practice Strategy (HPPS) allows for limited field investigations (LFIs), focused feasibility studies (FFSs), and qualitative risk assessments (QRAs), these streamlined approaches are intended to support the RI/FS process, not substitute for it as the text incorrectly implies. Due to the scope and complexity of the River Corridor aggregate area, additional investigation and characterization is necessary to provide sufficient information for a cumulative risk assessment. Revise the text to clarify that while LFIs, FFSs, and QRAs supported the Interim Remedial Measures, they are not sufficiently comprehensive to support a final ROD.
37	2.5.4.4	2-26	Explain how the criteria were used to identify high-priority sites recommended for remedial action, in particular "insufficient information for conceptual model" through the Qualitative Risk Assessment process.
38	2.7.1	2-34	The rationale for the baseline HHRA not considering intruders into cocooned reactor buildings and structures in the 100 Area is not explained. Clarify the rationale for not considering intruders into cocooned 100 Area reactor buildings and how exposure to these sites will otherwise be addressed.
39	2.8	2-39 to 2-44	This screening-level assessment of residual risks at remediated waste sites seems out of place and the purpose is unclear. Although this section utilizes previous models (scenarios and parameters) that might relate to cleanup activities, it presents calculated risk results that precede an explanation of the methodology (Section 3). Consider creating a separate results section (similar to Sections 4, 5, and 6), and clarify the purpose of this section (e.g., to present past results, compare methodologies, or support additional remediation decisions).

40	2.8	2-40	Clarify what the IAROD Rural Residential Scenario for defining cancer risk and noncancer hazards from radionuclides and chemicals entails.
41	2.8	2-41	The reader is referred to Section 7.5 for a detailed explanation of the calculations, but Section 7.5 only provides a summary of key conclusions. Revise the paragraph accordingly and reference the correct location of the calculations.
42	2.8.4	2-43	It is incomplete to summarize risk calculation results for residual contamination at waste sites by listing only results without arsenic. Although it is appropriate to note that arsenic is also a naturally-occurring compound, its presence at the site (natural and anthropogenic) contributes to total baseline risk similar to other naturally-occurring compounds, such as uranium. Revise the assessment to include all contaminant contributions to accurately reflect baseline risk conditions.
3.0 Human Health Risk Assessment Approach			
43	3.0	All	The discussions of data in Section 3 should clarify that not all contaminants were measured in all samples. In the summary tables, the total sample counts do not necessarily reflect the same number of data records for each analyte.
44	3.1	3-2	The approach should also consider <u>future</u> conditions within the upland, riparian, and near shore environments.
45	3.1.1.1	3-3	Risks associated with the yet-to-be remediated waste sites are noted as not being a focus of the report, indicating that while unacceptable risks at these waste sites are acknowledged, they are not added to all other baseline risks to provide a complete picture of cumulative site risk. Unremediated waste sites should be included for the assessment to be complete.
46	3.1.1.1	3-3	Waste sites remediated in accordance with requirements in the IARODs may not meet the cleanup requirements of the final RODs. Please revise the baseline HHRA as appropriate to acknowledge that additional remediation may be necessary to meet the requirements of the final RODs.
47	3.1.1.2	3-4	The revised baseline HHRA and remedial investigation reports should include complete integration of all media and exposure pathways, including groundwater transport of residual contamination from waste sites.
48	3.1.2	3-4	Examples are provided of non-CERCLA activities that may be useful for evaluating the non-operational areas, such as data collected as part of the Environmental Monitoring Program; however it is unclear what data from these sources are included in the baseline HHRA. Please clarify and use all appropriate data.
49	3.1.2	3-4	The extent to which non-operational areas are undisturbed is not well documented. Please revise the text to acknowledge that past practices at Hanford likely resulted in the disposal of unusual or particularly toxic waste outside of normal operational units in shallow undocumented waste sites. Accordingly, there may be many undocumented waste sites in the inter-operational areas that have not yet been discovered by the orphan waste site identification and evaluation process.
50	3.1.2	3-5	Past releases from operational areas likely contaminated surficial soils and plants in non-operational areas.
51	3.1.2	3-5	Please revise the discussion of aerial surveys to acknowledge and discuss that aerial radiological surveys are not able to detect and reliably quantify alpha radiation, which is emitted by uranium and transuranic elements. Revise the text to explain specifically how the aerial survey information was incorporated into the baseline HHRA, or specify that it was not used, if that is the case.

52	3.1.2	3-5 to 3-6	Risks associated with the non-operational areas are noted as not being a focus of this baseline HHRA, indicating that both the known and the unidentified risks potentially associated with these areas remain as a data gap. Evaluation of non-operational area data should be addressed in the baseline HHRA. At the very least, remedial investigation reports should include complete integration of operational and non-operational areas.
53	3.1.3	3-6	The discussion of the framework for assessing the riparian environment is very limited, focusing only on the 100-D island. Revise this section to include discussion of the overall methodology used for evaluating riparian areas potentially affected by contaminants.
54	3.1.3	3-7	Although Co-60 may not be detected in sediment downstream of the 100-D island at elevated concentrations, there is no mention of any other contaminants of potential concern. A search of the GiSdT database shows many more contaminants were detected in these samples. Revise the data summary to include risks estimated from potential exposure to other contaminants.
55	3.1.4	3-7 to 3-8	The discussion of the framework for assessing the nearshore environment is too limited, focusing only on the effluent pipelines. Revise this section to describe the overall methodology used for evaluating the nearshore areas potentially affected by contaminants, including groundwater seeps and aquatic biota (such as fish).
56	3.1.4.1	3-10	Risks associated with the nearshore pipelines are noted as not being a focus of the report (based on previous investigations), indicating that these data are not included in the baseline HHRA. Revise the baseline HHRA to include all available nearshore data, including those associated with pipelines, to estimate total baseline risk.
57	3.1.4.1	3-10	The baseline HHRA indicates that if portions of river effluent pipelines become dislodged and wash ashore, there may be elevated human health risk. However, the nature of the elevated human health risk is not mentioned. Expand the text to more fully explain the nature and magnitude of the associated risks under this scenario.
58	3.1.4.1	3-10	The baseline HHRA incorrectly indicates that the river effluent pipelines will be discussed again in Section 7.0 (Conclusions and Recommendations); however, river effluent pipelines are not mentioned again in the remainder of the document. Please revise the appropriate sections to include discussion the river effluent pipelines.
59	3.2.1	3-11	Examination of uranium-238 data provided in the GiSdT indicates a very large percentage of uranium data that was collected after 1998 that is >2 pCi/g was rejected from use in the RCBRA on the basis of the type of analytical method used. Further examination reveals that of the 2,596 *unusable* results, 1,690 were >1 pCi/g, while of 2,517 *usable* results, only 172 were >1 pCi/g. This evaluation was performed across all environments and sample categories using DOE provided data. The results suggest a strong bias in the uranium data that was ultimately used in the baseline HHRA. Please see our general comment on this topic. Review, rescreen, and revise the data used in the baseline HHRA using criteria that provides unbiased data reflective of the observed site conditions.

60	3.2.1.1	3-11	In reference to the statement "It is the incremental risk above background levels that is of primary concern...," it should be noted that comparisons to background should only be considered during the feasibility study when selecting appropriate cleanup actions and making risk management decisions. EPA guidance, cited in the document, (EPA 540-R-01-003, Appendix B) states specifically that all substances present at a site that exceed risk thresholds concentrations should be included in the risk assessment. Although it is true that naturally-occurring compounds can contribute to site risk, no distinction should be made from Hanford-related contaminants in a baseline risk assessment. As defined by EPA, "baseline risks are risks that might exist if no remediation or institutional controls were applied at a site" regardless of source (EPA 540/1-89/002), and it is not correct to consider only "incremental risk above background levels" to assess baseline conditions. Revise the baseline HHRA to consider all sources of risk to estimate baseline risks. Risks from man-made and Hanford-origin contaminants should be identified and evaluated in this context. Background or reference concentrations can be considered more specifically in the risk management part of the cleanup process.
61	3.2.1.1	3-11	EPA/540/1-89/002 is cited as the basis for using reference data to select COPCs for the site. However, the document and section only discuss using statistics to identify site-related versus non-site related substances. This guidance document as well as much newer guidance are clear that COPCs are all substances posing risk, whether site-related or not.
62	3.2.1.1	3-14 to 3-15, Figure 3-2	It is inappropriate to consider samples collected from the Hanford Site as "background" or "reference" as the term is used in a baseline risk assessment because no area of the site can be considered as "absent contamination" (considering air, ground, or biota dispersion). The background or reference site should not be within the Hanford Site boundaries or downwind of predominant winds. Revise the baseline HHRA to consider background samples as only those collected off the Hanford Site and outside of the influence of Hanford-derived releases. Background or reference concentrations can be considered more specifically in the risk management part of the cleanup process.
63	3.2.1.1	3-15	The number of reference sites, particularly for the 300 area, and the proximity of the reference sites to contaminated sites seems significantly inadequate to provide appropriate, and uninfluenced data.
64	3.2.1.1	3-16, Table 3-3	Upland reference sites have likely been impacted by emissions from operational areas, including long-lived radionuclides.
65	3.2.1.1	3-18	In reference to the statement "...have heterogeneous, or patchy, contamination...," the inclusion of contaminated references sites in the assessment is not protective and should not be used.
66	3.2.1.1	3-20	In reference to the statement "each location was characterized with a single sample of sediment and surface water," a single sample is not sufficient to characterize these sites. Please revise the data analysis to include a larger data set.
67	3.2.1.1	3-22	The Yakama Nation has previously raised concerns regarding large radioactive exposure doses at reference sites in the RCBRA Draft A. In particular, significant concentrations of americium-241 (a man-made radionuclide known to be of Hanford origin) were found in soil and biota at reference locations that could result in large doses of radiation to individuals based on the Yakama Nation exposure scenario. Review of available reference data indicates that americium remains present at several reference locations, indicating anthropogenic contamination at these sites. Revise the baseline HHRA to include reference sites that are not influenced by Hanford-derived contamination.

68	3.2.1.4	3-25	This section indicates characterization of groundwater exposures is "currently under development." Please revise the baseline HHRA to include this important and potentially significant source of additional exposure.
69	3.2.1.4 and 3.3.1	3-25 and 3-44	Using only present day groundwater data does not account for future migration of groundwater contamination from the Central Plateau. The migration of this contamination has already been observed and is expected to continue into the foreseeable future. An analysis of Central Plateau contamination movement was performed as part of the Tank Closure / Waste Management (TC/WM) Environmental Impact Statement (EIS) and should be acknowledged and incorporated into the baseline HHRA for the River Corridor. Even if, as DOE contends, CERCLA actions in the Central Plateau will be protective of groundwater, a baseline HHRA should reflect potential risks absent remediation. Please revise the text to either include groundwater exposure and dosage or to acknowledge this important data gap and the difference between the generally accepted purpose of a baseline risk assessment and what has been presented here.
70	3.2.1.4	3-25	It is unclear what criteria are used to define "representative" samples. The process for selecting the "representative" wells should be discussed. For example, what specific criteria were used; what data from which wells were not used and why; were data from multiple samplings of a well combined and how; etc.?
71	3.2.1.5	3-25	Although current and appropriate, not all Environmental Monitoring Program and Surface Environmental Surveillance Program (SESP) data appear to be included as data sources, and it is unclear why. Please revise the baseline HHRA to include all available and relevant data.
72	3.2.2	3-29	The opening paragraph of this section is another example of where the authors misunderstand or misrepresent the purpose of a risk assessment. The baseline HHRA needs to identify the sum total of all risks from all substances. A risk management document or a feasibility study is the correct place to determine the contributions to the overall risk from natural or anthropogenic background concentrations, which may or may not need to be remediated to meet clean-up goals.
73	3.2.2	3-29	The document cites DOE/RL-2005-42 as the accepted guidance for selecting COPCs. The process described in this paragraph was not found in the cited document. The citation stated only that "indicator contaminants" had been identified as those exceeding interim clean-up goals. A companion document cited in 2005-42, BHI-01757 (dealing primarily with the ecological risk assessments), states that for human health, all contaminants contributing "substantially" to human health risks would be included as COPCS. No process for "refinement" of the human health contaminants was found in either document. The process and agreements indicated on this page for the selection of COPC is poorly documented and requires better justification.
74	3.2.2	3-29	The statement "…comparing mean concentrations at study sites to background or reference…" is not acceptable. The comparison should use a range of concentrations with a statistical measure of uncertainty.
75	3.2.2	3-29	Despite previous workshops and discussions, it is inappropriate to selectively exclude data or COPCs in a baseline risk assessment. Revise the baseline HHRA to include all sources of risk, including substances that are naturally-occurring, ubiquitous, or otherwise considered background, to accurately reflect baseline risk conditions.
76	3.2.2.1	3-30	A citation to the Tri-Party agreement excluding some contaminants from consideration in the baseline HHRA should be provided, as should adequate justification.

77	3.2.2.1	3-30	Thorium-232 should not be excluded from the COPC list without further consideration and more detailed justification. See general comments and Attachment 1 for more details. Thorium-232 was known to be handled in very large quantities at the Hanford Site during periods of uranium-233 production, which utilized a thorium-232 reactor target. However, very few data points were identified in the GiSdT. We are concerned that the failure to find any residual Th-232 represents a failure to adequately sample potential source areas. Revise the COPC list to incorporate thorium-232 and other naturally occurring radionuclides used in large quantities at the site.
78	3.2.2.2	3-30, Tables 3-7 to 3-8	Revise the COPCs to include americium-241. Currently this radionuclide is neither included nor excluded from the baseline HHRA. This man-made radionuclide has a half life of 430 years and a relatively high specific activity for an alpha emitting isotope, making it particularly dangerous. Furthermore, americium-241 has been identified in high concentrations at many locations within the Hanford site, including several background and reference sampling sites, artificially implying that remediated waste sites are actually cleaner than uncontaminated locations. Finally, americium-241 is present in waste sites not considered in the baseline HHRA, for example the 116-N-1 site, in concentrations that exceed those found in waste sites considered in the baseline HHRA.
79	3.2.2.2	3-30	A "meaningful and effective regulatory document" requires a holistic evaluation of the total risks to human receptors at the waste sites and in all areas of the Hanford Site.
80	3.2.2.2	3-30	The COPC selection criteria eliminate compounds not found in at least one third of the waste sites in the 100 Area. Because different reactors had different auxiliary missions, such as the production of special nuclear materials, this methodology allows for removing COPCs from consideration that may be present in large quantities at only a few sites (e.g., COPCs present in the K-reactor fuel basins or the 618 burial grounds). Revise the COPC selection process and the list of accepted COPCs in the baseline HHRA to include contaminants of this nature so that these unique sites are not overlooked. This concern would also be corrected by including all of the substances posing risk, as is appropriate for a baseline risk assessment.
81	3.2.2.2	3-31	The requirement that a contaminant needs to be reported at one-third of the wastes sites is not protective. This screen potentially eliminates relatively unique waste sites, as well as adds to the problem of not including all contaminants in the risk evaluation. Similarly, the 300 Area sites should be screened on their own merit. The activities in the two operational areas were not the same. No justification is provided for not attempting to identify the "worst-case" sites, rather than artificially generating some sort of "representative" exposure.
82	3.2.2.4	3-32	Provide justification for the decision to use Hanford Site background data in preference to Washington State Yakima Basin Region background data when performing statistical evaluations for whether an inorganic analyte should be included as a COPC. Both have similar geologic histories, and where possible, preference should be given to data collected from sites outside the area of influence of Hanford-derived contaminants and away from the Hanford Site.
83	3.2.2.4	3-37	This section indicates no background or reference data are available for groundwater. Explain why, and identify and incorporate into the baseline HHRA a site which will allow groundwater to be sampled, analyzed and evaluated to establish background and reference data for selection of COPCs.

84	3.3	3-39	The text alludes to "many waste sites" as having been remediated. The baseline HHRA should be explicit in identifying whether data are available from other remediated waste sites not included in this assessment, and, if so, how the sites are used and selected, and why other sites (including non-remediated sites) were not used.
85	3.3	3-39	Similar to Draft A, the exposure scenarios evaluated are again referred to as "hypothetical," suggesting that DOE does not intend to clean up to unrestricted use. Revise the text to replace "hypothetical" with "potential future" exposure scenarios.
86	3.3	3-41, Figure 3-13	This conceptual site model of contaminant sources demonstrates contaminant transport from the Central Plateau and other upland areas to the River Corridor and into the river itself (including the pipelines). The conceptual model should also depict contaminant pathways from the vadose zone and subsurface surrounding or under the reactors to groundwater and the river. Revise the document to include a combined evaluation of all of these areas that will provide a complete picture of baseline risks.
87	3.3	3-41, Figure 3-13	The cross-section cuts Gable Mountain without acknowledging the underlying basalt layer. Redraw the cross-section line to accurately account for the basalt layer and/or acknowledge its presence in the text.
88	3.3	3-41, Figure 3-13	This figure lacks depiction of bioturbation (e.g., rabbits burrowing, moles digging, ants making colonies) as potential contaminant transfer pathways. These animals are an important part of the food chain and potential vectors of contaminants.
89	3.3.1	3-43	It is true that present day workers are under surveillance and are managed under health and safety plans. However, accidents happen and workers may be contaminated with residual chemicals. "*Because potential exposures and associated risks are monitored for these workers, they are not considered potential receptors for the HHRA.*" There is no way to ensure that a worker will not be contaminated. It must be assumed that an accidental exposure could occur. The purpose of the risk assessment is to calculate potential risks from contaminant exposure to people, including workers, without institutional controls, surveillance or monitoring. The current beryllium program, for example, shows that workers can still be exposed. Please revise to include this scenario for workers.
90	3.3.1.3	3-44	The statement "...cancer risk and radiation dose will be calculated using present-day radionuclide activities in soil and with radionuclide activities in soil decayed to the years 2075 and 2150" is not adequate. Consider not only decayed concentrations, but future estimated concentrations due to migration.
91	3.3.2	3-45	As applied in the RCBRA, the Yakama Resident scenario inappropriately assumes that an individual contacts soil only within a limited area surrounding a home. This does not necessarily provide the most conservative assumption for contaminant exposure. While exposure may exist within a limited area (e.g., residing on a former waste site), exposure should also include other pathways, e.g., hunting, gathering, fishing and consuming the resulting foodstuffs; contacting seeps, springs, sediment and surface water in the Columbia River to determine total risk.
92	3.3.2	3-45	Please clarify these statements "...likely to be exposed over much broader areas...:" "... a residential component that pertains to localized exposure..."
93	3.3.2.1	3-46	Regarding the second paragraph in this section, it is unreasonable to assume that activities by humans and small animals will be limited to a depth of 6 inches. For instance, bioturbation from insects, worms, trees, mammals and other biota may mix soil, and digging for wild plant roots may occur at depths of approximately 6 feet below ground surface. Please change this assumption as it will change the input data for the baseline HHRA.

94	3.3.2.1	3-46	The Resident Monument Worker scenario should be revised to include exposure to upland and riparian contaminants through plants and animals, or provide adequate justification for why these exposure pathways are not included.
95	3.3.2.1	3-47	The statement "*Because the volume of drill cuttings will be very small relative to the volume of soil encountered by receptors when averaging exposure over many years, the potential contribution of drill cuttings to chronic health risks from soil exposures is likewise small*" may not be true. If a resident or worker handles contaminated cuttings, exposure to elevated concentrations may occur. The assumption cannot be made that chronic health risks would be small via this exposure pathway. Please include this pathway in the baseline HHRA or provide better justification for its exclusion.
96	3.3.2.1	3-47	The protectiveness of using the backfilled soil data depends on the assumption that the exposure to the side wall concentrations is small and does not extend beyond the remediated footprint of the site. More importantly, there are numerous realistic scenarios of future activities at the site wherein natural or anthropogenic activities would expose those currently buried soils. In addition, if such an event occurred, the exposure area of contaminated surface soils could be much greater that the waste-site footprint.
97	3.3.2.1	3-48	If, as stated in the last paragraph, the problem with using measured concentrations in upland vegetation is only related to organic contaminants, then at least use the data for inorganic substances and radionuclides.
98	3.3.2.2	3-49	In the first paragraph: "...soil data because residual contaminant concentrations are generally higher than in the sediment data..." Is this based on theory or empirical evidence of variation in upland soils from the Columbia River Basin and sediments from the riverbed?
99	3.3.2.2	3-49	The explanations provided for excluding Surface Environmental Surveillance Program (SESP) sediment data are not adequate (e.g., simply because other data are available?). Are the results comparable to RCBRA data? Clarify the data quality issues that compromise this data set.
100	3.3.2.2 and 3.3.3.3	3-49 and 3-59	It is incomplete to assume that only the recreational and nonresident tribal scenarios have potentially complete exposure pathways to surface water and sediment. A tribal resident would certainly use and contact surface water from the Columbia River to drink, swim, fish, and sweat, while also contacting and inadvertently ingesting sediment. Please revise the baseline HHRA to include potential exposure to surface water and sediment as part of the Yakama Nation resident scenario.
101	3.3.2.2	3-49	It is incorrect to assume that chronic exposure to seep water is unlikely because of seasonal flows. A Tribal resident or non-resident could access a nearshore area with seeps over a lifetime. It is also incorrect to assume that porewater is not a potential human health medium, as a Tribal member could contact sediment and therefore porewater. Revise the baseline HHRA to include estimating risks from exposure to seeps and porewater for all Tribal scenarios.
102	3.3.2.2	3-49	Explain how the Resident Monument Worker and Industrial Worker scenarios interface with the Recreational Use scenarios. It would be very likely that the Resident Monument Worker, in particular, would also be exposed via pathways similar to the Recreational User.
103	3.3.2.3	3-49	The statement that "Groundwater in the River Corridor areas...flows in the direction of the Columbia River" is incomplete. Groundwater also flows inland locally during periods of high water in the River.

104	3.3.2.3	3-50 and 3-52, Figure 3-15	In assessing future risks, it is appropriate to evaluate future groundwater concentrations due to leaching of soil contaminants and migration from up gradient sources. Revise the assessment to include groundwater as a complete pathway for all scenarios, and to include future groundwater concentrations (for example, those estimated in the Draft TC/WM EIS). Also, current and future groundwater concentrations should be evaluated via the vapor intrusion and irrigation pathways, as these are plausible future uses.
105	3.3.2.3 and 3.3.3.2	3-50 and 3-56	The baseline HHRA fails to assess risk from groundwater to Industrial Workers. Justify this omission, including what water source industrial workers will use for drinking, washing, and industrial operations in this desert environment each work day. Include risks of exposure through ingestion, absorption, and inhalation of potential contaminants.
106	3.3.2.3	3-50	The baseline HHRA omits the exposure pathways of irrigating a garden, and it is unclear if risk is assessed for providing water to livestock. These pathways are present in the exposure scenario and should be included in the risk assessment.
107	3.3.3.1	3-51 to 3-53, Tables 3-11 to 3-13	These three tables do not provide a sufficient view of actual exposure pathway scenarios that may be encountered at the site. For example, given the nature of activities that children participate in, a casual user child would be expected to have a complete exposure pathway to soil. Also, groundwater could be used for residential purposes, including dermal exposure and inhalation during showering. These exposure scenarios should be revisited to ensure that they represent realistic behaviors and pathways.
108	3.3.3.1	3-51 to 3-53, Table 3-11	The exposure of young children to contaminants via the sweat lodge pathway should be evaluated. Revise the assessment to assume child exposure of at least one hour per day.
109	3.3.3.1	3-51 to 3-53, Table 3-11	There is no evaluation of exposure and doses to the embryo/fetus and to young children from the breast milk pathway. Revise the assessment to include potential exposures to the embryo/fetus and to young children from the breast milk pathway for the Tribal resident and other scenarios to ensure that the most vulnerable members are adequately protected.
110	3.3.3.1	3-54, Table 3-11	It is incomplete to assume that the nonresident Tribal scenario has a potentially complete soil exposure pathway only to the top 6 inches. An intermittent Tribal site user could access deeper soil to dig roots, build a ground oven or temporary shelter. Revise the nonresident Tribal scenario to include deeper soil, e.g., waste site samples collected below ground surface similar to the resident scenarios.
111	3.3.3.1	3-54, Table 3-11	Upland surface soil (0-6 inches) is not used for casual user (adult or child) or avid angler (adult or child). Users would be exposed to more categories than described here. For example, children that crawl, play on the ground, and inadvertently eat dirt. Please consider these exposure pathways.
112	3.3.3.1	3-54	Tribal children should be considered in a consistent manner with non-Tribal children, starting at the age of 1 year, not 2 years old.
113	3.3.3.3	3-57, Figure 3-17	The Yakama Resident residential scenario should include and show complete pathways for sediments (inadvertent ingestion, dermal absorption, and external irradiation); river water (ingestion); wild plants and wild game (ingestion, traditional uses).
114	3.3.3.3	3-58	It is unlikely that a Resident Monument Worker would never engage in gardening, raising livestock, or fishing from the Columbia River. Please consider these exposure pathways.

115	3.3.3.3	3-59	It is incorrect to assume that Native American residents would only use plants and animals raised domestically. Similar to fishing in the nearby river, a Yakama resident (living on a waste site) would also hunt in the nearby upland and riparian areas. Although it may be appropriate to assume that wild plants and game have taken up contaminants from a waste site, revise the statement from "assumed to be domestically raises" to more accurately reflect the collection of wild plants and game. Concentrations should not only be modeled from remediated waste site soils, but also modeled from unremediated waste site soils as well as measured directly.
116	3.3.4	3-59	It is not only "desirable" but correct methodology to calculate cumulative risks for all exposure pathways for each scenario. By dividing the assessment into spatial scales, exposure assumptions associated with the Yakama scenario have been only selectively applied. For example, some exposure pathways that are applied to the non-resident are not applied to the resident and vice versa. This does not provide a complete picture of cumulative exposures to a Tribal member. Revise the baseline HHRA to consider a more complete "broad-area" evaluation of risk to Yakama residents and other scenarios that reflect potential exposures site-wide, including soil (surface and subsurface), groundwater (seeps, porewater, and future migration), surface water, sediment, and upland, riparian, and aquatic biota.
117	3.3.4	3-60	Fish ingestion risk estimates should be directly summed with risks from other exposure pathways. The limited fish data have resulted in an incomplete fish consumption analysis. Risk estimates for ingesting these surrogate fish species (which actually are consumed by Tribal members) and other species should be included in a cumulative risk assessment.
118	3.4.1	3-61	The EPA guidance "Methods for Evaluating the Attainment of Cleanup Standards" does not describe the Reasonable Maximum Exposure (RME), but presents sampling and analysis methods to verify remediation activities using *average* concentrations. Please revise the statement, and assessment as needed, to accurately reflect the intent of the RME in risk assessment, which is to estimate a conservative exposure case (i.e., well above the average case).
119	3.4.3.2	3-70	Regarding the statements "...Migration of gas phase VOCs upward through vadose zone soil and into a residential..." and "...risks related to this exposure pathway are not quantified..." Explain how this affects risk estimates in the uncertainty section.
120	3.4.3.4	3-72	The failure to collect edible plant samples for analysis is a data gap that introduces significant uncertainty into the data set. The baseline HHRA should address edible plant measured data and site-specific uptake factors.
121	3.4.3.5	3-73	"...chicken feed is store-bought..." explain in uncertainty section how store bought chicken feed versus on-site harvested chicken feed affects the risk estimates.
122	3.4.3.7	3-74	"Riparian plant tissue EPCs are calculated for each individual ROD decision area because there are adequate riparian soil data in each area..." The sites were pre-selected and do not represent exposures to individuals or communities. The statement that it may bias risk estimates is correct. The effect of this bias should be more thoroughly discussed in the uncertainty section.
123	3.4.4	3-75 to 3-76	"The inclusion of both RME and CTE...semiquantitative measure..." What does semiquantitative mean in this context?
124	3.4.4.4	3-80, Table 3-18	The nonresidential Tribal scenario exposure parameters do not accurately represent a Tribal unrestricted use scenario. Please revise the broad-area assessment to include the Yakama resident scenario, reflecting an individual residing on the site (possibly on a former waste site) and using all of the resources available on a broad scale: hunting game and collecting plants found in different areas of the site, harvesting fish from the river, sweating and participating in cultural activities in different areas. Accordingly, revise the exposure parameters to reflect a Tribal resident, including collecting 100% of wild plants from onsite.

125	3.4.4.4	3-80, Table 3-19	Some of the Yakama Resident exposure parameter values are not appropriate. Activity-specific soil adhesion factors (AF), for example, used for an adult farmer and child playing in wet soil are based on EPA's geometric mean rather than the 95th percentile (per RAGS, Part E). Also, the sweat lodge exposure duration only represents adults, and should be at least 70 years to include child exposure. Revise the exposure parameters to represent all upper-bound estimates.
126	3.4.4.6	3-82, Equations 3-12 and 3-24, Tables 3-19 and 3-30	The equations provided for calculating external radiation dose used in conjunction with the values provided in Table 3-19, Table 3-30, Appendix D, and Waste Site CVP confirmation sampling do not yield equivalent dose rates to those provided in the text of the baseline HHRA. Independently-calculated values are not even of the same order of magnitude. More information is required to replicate these calculations, in particular the residual levels of radionuclides, since it is not clear which values are taken from the CVP for each site. Please revise the baseline HHRA to present the calculation of present radiation doses in a format that is readable, repeatable, and consistent with the text of the document, and provide example calculations for review.
127	3.5.2	3-84	The definition noted for chronic RfD (referenced from EPA/540/1-89/002) is not accurately reflected here. Include the full definition, such as: "including sensitive subpopulations" (e.g., the Yakama Nation) and "Chronic RfDs are specifically developed to be protective for long-term exposure to a compound (as a Superfund program guideline, seven years to lifetime)."
128	3.5.2	3-84	It appears that subchronic and developmental RfDs were not addressed by this baseline HHRA. Subchronic covers an exposure time period of two weeks to seven years. Developmental covers a single exposure event. Provide clear rational for not including these calculations, particularly for nonresidential tribal scenarios.
129	3.5.3	3-87	Paragraphs 2 and 3 are inconsistent. Both refer to PAHs listed in EPA/630/R-03/003F. However, they appear to be inconsistent as to the COPCs. Paragraph 2 indicated that 4 chemicals identified as mutagens in the EPA document may be COPCs. Paragraph 3 indicates that just 2 of these same chemicals are risk assessment COPCs. Revise to provide better clarification.
130	3.5.6	3-89	The equations presented for calculating dermal toxicity do not take into account the mode of absorption (water or soil) for which the Dermal Risk Assessment Guidance for Superfund document provides. If the appropriate information was not available as the text suggests, then specify exactly what values were not available and why the water absorption and soil absorption values could not be calculated. Also, it does not appear that age increments were calculated.
131	3.5.7	3-90	The first full paragraph suggests that surrogate chemicals may be used for determining toxicity criteria. Identify clearly both in a table and text, what chemical surrogates were used, for what calculations, what values were used, and provide peer reviewed justification for their use.
132	3.5.9	3-93	The use of Aroclor data rather than a full PCB congener analysis will affect the risk estimates. This should be discussed in the uncertainty section.
133	3.5.9	3-93	The text lists the TEFs as referenced from WHO 2003, however, Table 3-32 lists 12 PCB congener TEFs from the 2005 re-evaluation. Please correct the text.
134	3.5.10	3-94, Table 3-7	Calcium-41 has been excluded from the COPC list and is designated a detected analyte not included in the quantitative risk assessment. However, the 1993 Technical Baseline Report for the 100-D Area (cited in Table 2-1 as WHC-SD-EN-TI-181) states on page 2-6 that sources of contamination at the 100-D Area include Calcium-41, an activated element in reactor cooling water, which is called a "notable exception" to the other short-lived radionuclides, with a half-life of 103,000 years. Please address potential calcium-41 contamination and risk, and differentiate calcium-41 from calcium the essential nutrient.

135	3.5.10	3-95	Lithium data were not included in the risk assessment, but this section states that lithium was detected in cleanup verification shallow zone soil data from 100-D/100-H and 100-B/C areas. Please provide and evaluate these data.
136	3.6	3-98 to 3-102	Despite the fact that determining synergistic or antagonistic effects is difficult, it should be attempted when such effects between certain compounds are known. For example, consider the potential for synergistic interactions between radiation and certain types of hormonally-active agents and heavy metals.
137	3.6.4	3-101	The statement included in this section "The origin of this threshold was in guidelines published by the EPA for establishing cleanup levels for radionuclides under CERCLA that stated that 15 mrem/yr above background levels should generally be the maximum dose limit for humans" is attributed to OSWER 9200.4-18, "Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination", but is inaccurate and should be deleted. This cited guidance indicates that 15 mrem/yr, and not 15 mrem/yr above background, is a "minimally acceptable dose limit", and further states that EPA has "explicitly rejected levels above 15 mrem/yr as being not sufficiently protective."
138	3.7	3-103	This section indicates that PRGs for radionuclides were developed based on a target cancer risk level of 1 x 10^{-4}. This risk level is inconsistent with EPA Risk Assessment Guidance for Superfund Part B, Chapter 4, titled "Risk-Based PRGs for Radioactive Contaminants," which states "calculate risk-based PRGs for each carcinogen corresponding to a pre-specified target cancer risk level of 10^{-6}." To be consistent with EPA guidance and with the risk requirements of MTCA (WAC 173-340), PRGs for all carcinogens, including radionuclides, should be developed based on a target risk level of 1 x 10^{-6}.
4.0 Broad-Area Risk Assessment Results			
139	4.1	4-1	From the first few sections of Chapter 4, it is unclear exactly what the COPCs are for any area of the Hanford site. Clearly state, at the beginning of the chapter, what COPCs are applicable to what areas.
140	4.1	4-1	After reviewing the chapter, it does not appear that the following two questions have been adequately answered: 1) Are residual conditions for cleanup actions under the IARODs protective of human health and the environment? 2) What are the uncertainties associated with the risk results and conclusions? Provide clear and justified answers.
141	4.1	4-2 to 4-3	It is inappropriate that exposure to contaminants from groundwater seeps and fish consumption are evaluated separate from exposure to soils, river water and sediment. Revise the baseline HHRA to combine all exposure media, not just limited to the 100-K Area, to determine total risk, including deeper soils, groundwater, seeps, and other fish species throughout the reach.
142	4.1	4-2	For the Avid Hunter and Nonresident Tribal exposure scenarios, it appears no game species such as deer, elk, or duck were sampled and analyzed for this risk assessment. It appears that "deer" and "elk" contaminant concentrations were modeled from soil data. Provide adequate sampling for a quality risk assessment, for both broad and local areas.
143	4.1.1	4-2	Nonresident Tribal risk should not be based solely on modeled plant concentrations. Wild plants, particularly roots and other plant tissues harvested by tribal members, should be better characterized.
144	4.1.1	4-2	This section includes many generalizations and speculative statements that do not appear to be supported by data or references. For example, the text states that exposure scenarios "are generally protective of human health." Provide justification or reference data calculations supporting such statements.

145	4.1.1	4-3	Since this section is a "broad area" risk assessment, explain why so much emphasis is placed on many specific decision units, particularly since Chapter 5 already addresses individual decision units. This chapter should focus more thoroughly on addressing the risk *throughout* the entire Hanford River Corridor site, pulling in (and identifying) where local data are used.
146	4.2	4-5	No clear description or explanation is given as to why more representative sampling was not conducted throughout the River Corridor region. Provide statistically sufficient data to eliminate this data gap.
147	4.2	4-6, Figure 4-1	This figure is inadequate to show specific sampling locations for the Broad Area Risk Assessment. Sampling areas should be clearly identified as to location, media sampled, and parameters analyzed. Also provide a table summarizing this information. A statistically defensible number of samples that adequately represents the site should be used for the broad area.
148	4.2.1.1	4-7	Soil sampling collection is inadequate. Multi-increment sampling is better to characterize a relatively small area for known contamination, as it may miss "hot spots." An area 2.47 acres with only 50 samples is not adequate to characterize the area. This is only one sample per 2100 square feet. The depth of sampling is also inadequate at 0-15 cm. Please provide additional soil characterization.
149	4.2.1.1	4-7	Water level fluctuations between riparian area multi-increment sampling events result in sampling grid dimension variations and introduce uncertainty into the MIS analytical data. Address uncertainty in riparian area MIS analytical data introduced by variations in sampling grid dimensions between sampling events.
150	4.2.1.1	4-7	How many river sediment samples were taken in 2007 to replace unusable data from 2006? Were 35 new samples taken from the same 17 locations? Explain this additional data and show how the replacement data adequately addresses the gap.
151	4.2.1.1	4-8, Table	The unlabelled table on this page indicates that plants were sampled, but only the leaves of upland and riparian vegetation. Roots are vegetation and an important food source for tribal members and must also be sampled. Additionally, no game animals were sampled. No rabbits or other small mammals other than mice were sampled. It is not specified if the mice sampled were native species or European-introduced *Mus musculus*. Please fill these data gaps.
152	4.2.1.1	4-9	The third paragraph states the reason for not sampling more fish or plant tissue was that it was simply too difficult and soil sampling was easier. This reasoning does not justify why critical and substantial sampling and analyses were not conducted. Please collect, analyze, and include the necessary samples.
153	4.2.1.2	4-10	It is unclear if additional sampling for the broad area analysis was conducted at the five remediated waste sites or the same data was used as for the local area risk assessment. Please provide clarification.
154	4.2.1.2	4-11	The number of samples, in many cases, is insufficient to provide adequate certainty of analysis. For example, sampling for the 100-B/C pilot project lists only two sediment samples. Of the 5 to 8 clam tissue samples, only one was analyzed for beta- and gamma-emitting radionuclides. Of the 5 to 8 sculpin tissue samples, only one was analyzed for beta- and gamma-emitting radionuclides. This also raises another concern: why a definitive number of samples is not listed for clam or sculpin. Please sample, analyze, and provide statistically defensible data.
155	4.2.1.3 to 4.2.2	4-11 to 4-15	Much of Chapter 4 is vague regarding where samples were collected, what contaminants were analyzed where, and expectations of contaminant behavior. For example, the text states that "processes also affect concentrations of groundwater-related contaminants in the seep water." Please clarify what and how these processes will affect seeps and associated contamination. Also, please provide more details (or citations to appropriate references) regarding the COPC refinement process in general.

156	4.2.2	4-12	On this page, and several others, statements are made regarding the use of statistical tests to analyze River Corridor data. However, no discussion is provided describing what statistical methods or parameters were employed or exactly what data were used. Additionally, no statistical results are provided. When providing definitive statements, provide either peer reviewed justification or statistically defensible data to back up the statement.
157	4.2.2.2	4-14, Table 4-7	The broad area data tables list Aroclor 1262 as being found in all five of five samples, but no information was provided in the main report as to where that substance was detected. Please provide clarification regarding this inconsistency.
158	4.4	4-27	It is unclear why 50% of wild plants are considered from upland and 50% from riparian environments; a distinction should not be necessary. Revise the baseline HHRA to utilize all available plant data from all habitat areas to determine the exposure point concentrations and reasonable maximum exposures for a Tribal member consuming wild plants.
159	4.4	4-27	It is inappropriate to only present child hazard index (HI) results. Although the adult HI results may be different (in this case "generally" lower than the child), they are equally important to present in the baseline HHRA. Revise the reporting to include chemical hazard results for both the child and adult receptors.
160	4.4	4-27	It is incomplete to present risk results from fish ingestion for only limited species. Cleanup decisions in the River Corridor cannot be made without a complete understanding of potential risks to Tribal residents who will be fishing for all species from the river. Please revise the baseline HHRA to include Columbia River Component data, and sum the risks from fish ingestion with other exposures.
161	4.4	4-31	It should be noted that radiation dose results that are based on RESRAD modeling will underestimate the risk to a Tribal member because exposure assumptions in the RESRAD model do not account for a Tribal subsistence lifestyle.
162	4.4	4-32	The hazard index (HI) approach was developed to "assess the overall potential for noncarcinogenic effects posed by more than one chemical" (EPA 540/1-89/002). It assumes simultaneous exposure over a comparable timeframe (e.g., chronic) and a proportional magnitude of adverse effect, but not necessarily similar target organs. Segregating hazard indices by effect and mechanism of action, such as was done with arsenic and cadmium, requires a very complex toxicological analysis to identify all of the major effects and target organs / mechanisms of action. If not done carefully, segregating hazard indices can underestimate the true risk. Revise the non-cancer hazard index analysis of arsenic and cadmium by removing the statement about being "biased high by approximately 10%."
163	4.4	4-32	It is not appropriate to make comparisons of "representative" sample concentrations to those considered "reference areas" when the reference samples were collected onsite.
164	4.5	4-33	The fourth paragraph states that calculating chronic health risks from intermittent seeps for the Avid Angler and Nonresident Tribal scenarios is not feasible, yet the scenarios presumes exposure "will occur between 30 and 60 days per year for many years." Please clarify the statement, and calculate the risk for chronic exposure to seeps for these scenarios.
165	4.5	4-34	In addition to the "six key contaminants," list any other contaminants that have been identified in groundwater seeps. Determine if any of these exceed any ARARs and include them in the baseline HHRA.
166	4.5.1	4-35	The third paragraph in this section states that seep water samples were "analyzed for either total chromium or hexavalent chromium." Describe why a sample was designated for one or the other of these analyses. Please clarify whether total chromium is assumed to be a surrogate for hexavalent chromium, or vice versa (e.g., for comparison to the drinking water standard), and include a statistically sufficient number of data points for analyzing the risk to human health.

167	4.5.1	4-36 to 4-38, Figures 4-2 to 4-7	The number of sites varies from box plot to box plot. Some box plots have sites that others do not. Please clarify the number of samples associated with each box plot and explain how sites were chosen.
168	4.5.1	4-39, Figure 4-8	Only five seep water samples are represented in the figure. The text states that "There is a decreasing trend shown in these data." Clearly this is not enough data to show any type of trend, particularly considering the short time frame represented and the effects of river stage on seep concentrations. Please include all available data to determine if a trend exists. If sufficient data are not available, this is a data gap and a trend cannot be determined.
169	4.5.2	4-41	The first sentence in the section states "For the majority of the shoreline springs for which data have been made available, there is negligible risk related to exposure to key groundwater contaminants being released to the Columbia River at these locations." The quoted statement is too vague to accurately convey results of a risk calculation. Define what "negligible risk" means. Clearly, some seeps show contaminants above drinking water criteria, particularly for tritium, total uranium, total chromium, and strontium-90. Determine the chronic risk of these seeps to all of the exposure scenarios, particularly avid angler and nonresident tribal.
170	4.5.2	4-42	No risk calculations appear to have been conducted for shoreline springs or seeps. In the summary, the text states "one may conclude there is minimal risk from occasional use of the water, particularly for adults." Please estimate the risk from potential exposure to shoreline springs for adults and children using, at a minimum, avid angler and nonresident tribal scenarios. "One may conclude..." is not an appropriate or defensible method of quantifying risk. Also, the summary continues, "caution is appropriate if young children might be exposed..." This provides no information to protect young children from contaminants at these locations. Define "caution" and "young children" and appropriate measures to take.
171	4.5.2	4-43	Summary point 5 states "it is possible that short-term risks may exist for uranium exposures at the Spring 42-2." By conducting a thorough risk assessment, risks for uranium exposures should be much better understood, assisting in a safer and more thorough cleanup.
172	4.7	4-50	This section is vaguely written with no or little justification given for most assumptions made. In addition, it does not include an analysis of the uncertainties one would expect to see in a risk assessment, such as uncertainty associated with each type of sample, analyte, or media, but instead focuses on "third-party" uncertainty, such as from reference dose calculations and EPA's designation of mutagenic or nonmutagenic carcinogens. Please include more transparent information regarding assumptions (primarily from the actual sampling and analyses conducted and exposure scenario parameters) and provide reasonable justification for those assumptions.
173	4.7.1.2	4-53	Detection limits for toxaphene, 2,4,6-trichlorophenol, and pentachlorophenol should have been specified in the DQO process at concentrations below any ARARs. Relying on the conceptual site model to conclude that these chemicals pose no threat to groundwater is inappropriate. The uncertainty analysis section is for discussing uncertainties. Conclusions about groundwater contamination should be moved to another section with analytical data provided to justify why these chemicals should not be a threat to groundwater.
174	4.7.1.3	4-54	The first paragraph discusses background reference samples. It states "the other reference site selection criteria ensure that these sites are applicable as reference areas..." State what the criteria are in the guidance, what was used to determine the reference or background sites in this baseline HHRA, and how the sites met or did not meet these criteria, including a table with the discussion.

175	4.7.1.3	4-54	The statement "Because the COPC identification process was systematic, it is unlikely that Hanford site related analytes could contribute…" is misleading. While the methodology may have been systematic, it is not clear how it was applied to individual chemicals nor does it seem to be protective since many chemicals were eliminated based on limited data. Stating that "it is unlikely that Hanford Site-related analytes that could contribute to significant health risks were eliminated in this process" is a significant overstatement. Provide a clear description of how each COPC was determined, including all data considered and a thorough discussion of decisions.
176	4.7.1.3	4-55	When and where were these orchards in operation? Please define the aerial extent of the orchards relative to the 100 Area, and explain how to propose separating site related contaminants from past practices.
177	4.7.1.3	4-55	The first two paragraphs on this page discuss arsenic and its source on the Hanford site. Please explain how this information is used appropriately in the uncertainty analysis (and consider moving it to the appropriate uncertainty section).
178	4.7.1.4	4-55	Using less than five samples (actually, using less than 30 samples) is not statistically robust. Additional sampling should be conducted for plants and tissues. When less than five samples are used in representative calculations, always provide the full range of values in addition to the average.
179	4.7.1.4	4-56	Nowhere in chapter 4 is there a description of statistics used for any section or calculation. Please provide detailed descriptions, assumptions, and examples of statistics used; at the very least, cite the appropriate documentation where that information is provided.
180	4.7.2.2	4-57	The uncertainty analysis should include identifying all data gaps for foods such as plants (for example, no roots appear to have been sampled or modeled) and game animals. The GiSdT database does show concentration data for some game animals, so it is unclear why these data were not used. More data should be obtained to fill data gaps and provide a basis for calculating uptake factors.
181	4.7.2.2	4-58	The third paragraph states, "Because nonvolatile contaminants have no vapor pressure, this equation is physically implausible..." However, the vapor pressure of a solution of a non-volatile solute is equal to the vapor pressure of the pure solvent at that temperature multiplied by its mole fraction. Try recalculating using Raoult's Law.
182	4.7.2.2	4-59	The baseline HHRA does not include an analysis of risk for nonresident tribal use of Columbia River water for sweat lodges. Water would be used for producing steam as well as for drinking, usually over several hours. The sweat lodge scenario must be re-evaluated using appropriate numbers, and a risk analysis of Columbia River surface water and groundwater must be conducted.
183	4.7.3	4-59	While uncertainty is definitely associated with dose extrapolation, modeling, cancer slope factor calculations, and reference dose calculations, the emphasis given on these items seems out of proportion with what should have been addressed in this chapter. The purpose of the uncertainty section is to identify those uncertainty issues *specific to the particular assessment*, not discuss at length uncertainty theory inherent to any risk assessment.
184	4.7.3.1	4-59	The statement regarding cadmium that "…the three fold change in the PPRTV will not affect the results…" is misleading. Since risks are summed for systemic chemical affects, the change in one may result in a hazard quotient that exceeds 1.0. Please remove the misleading statement.

5.0 Local-Area Risk Assessment Results			
185	5.1	5-1	It is incomplete to assess risks to a resident living only a single remediated waste site. There is no rational provided for how a subset of remediated waste sites was selected for evaluation. It is not appropriate to calculate risks based on a select few remediated waste sites (or select depths), when the risk from other waste sites, other depths, and other areas may be greater. The assessment should include data from all of these areas of the site to obtain a complete understanding of baseline conditions and potential risk absent remediation.
186	5.1	5-1, Table 3-13	The Local Area Risk Assessment omits risk from COPCs in soil beyond 15 feet below ground surface (bgs), despite known contamination in the vadose zone. Soil contaminant characterization and assessment of risks below 15 feet bgs is omitted from the Broad Area and Groundwater Risk Assessments as well, which fails to provide a comprehensive and cumulative risk assessment in this baseline HHRA. Also, no consideration is give to migration of contamination in the vadose zone to groundwater, which will result in an increase in risk via exposure to groundwater contaminants. Furthermore, it is speculative to assume institutional controls will prevent excavation beyond 15 feet bgs (such as to install a drinking water well) in a residential scenario.
187	5.1	5-1	The Resident Monument Worker scenario is for adults only. What provision would prevent the worker from sharing residence with their family and children, and thus potentially exposing children in this scenario? Please include children in this scenario.
188	5.1.1.1	5-6	Explain further and provide specific details on the differences between the calculation methods for determining representative concentrations in soil during the cleanup verification process versus the RCBRA process.
189	5.1.1.2	5-6	The uncertainty presented regarding residual subsurface contamination does not overestimate risk (e.g. present a conservative bias) as stated for the scenarios considered in this baseline HHRA. Although there is uncertainty associated with the CVP samples, in some cases they may not reflect more contaminated areas in the deep zone of the remediated waste site. Contaminated soil from the shallow and deep vadose zone may easily be brought to the surface through any number of natural or human activities, including construction of basements or foundations, burrowing animal transport, drilling wells, surface erosion or collection of borrow material. Revise the text of this section and elsewhere to acknowledge that the proposed characterization may also underestimate the risk posed by residual contamination, particularly with even modest erosion or activities such as gravel mining or other resource extraction activities that may occur after institutional controls are no longer effective.
190	5.2	5-7, Table 5-8	In the 100-B/C local area summary table, the maximum detected value for gross beta in soils is listed as 33.7 pCi/g; however, Ni-63, a beta emitter, has a maximum detected value of 78.9 pCi/g. Please provide clarification regarding this inconsistency.
191	5.2	5-7	Revise the baseline HHRA to specifically include criteria for inclusion or exclusion of identified waste sites in each decision unit. Examination of CVPs from adjacent waste sites in the 100-N area found that waste sites, such as 116-N-1, were excluded from consideration despite having similar levels of contamination as other sites, such as 116-N-3, that were included. Furthermore, known sites with elevated levels of contamination, such as the 618 burial grounds, do not appear to have been included in this HHRA. Regardless of future cleanup plans, some residual contamination will remain at these sites, as the CVPs for 116-N-1 and 116-N-3 demonstrate. Revise the baseline HHRA to include these sites, and others with similar levels of contamination.

192	5.2	5-8 to 5-16, Figures 5-1 to 5-8	Revise the baseline HHRA to include complete exposure pathways to the reactor cores and associated contamination left in interim safe storage. Loss of institutional knowledge or failure of institutional controls makes direct exposure to the cores a real possibility within the time periods evaluated by this HHRA. The cores are housed in large, high profile buildings that could provide obvious shelter or other utility to people unaware of the contamination risk inside.
193	5.2.1	5-8 to 5-14, Figures 5-1 to 5-8	It is not appropriate that only remediated waste sites be included in the assessment. For example, contaminated wastes such as the 318-10 and 318-11 burial grounds, which would contribute significantly to overall site risk, are not included. Revise the baseline HHRA to include unremediated sites as well.
194	5.2.2.1	5-17, Table 5-21	Justify omitting the important Hanford contaminant uranium-235 from the COPC list for 100-K Shallow Zone soil, other than percentage of censored data (see general comments), and clarify the process for handling non-detected values.
195	5.2.2.1	5-17, Table 5-24	Justify omitting the important Hanford contaminant uranium-235 from the COPC list for 100-F/100-IU-2/100-IU-6 Shallow Zone soil, other than percentage of censored data (see general comments), and clarify the process for handling non-detected values.
196	5.2.2.2	5-17	Revise the baseline HHRA to include complete exposure pathways to contamination in the deep vadose zone that is beyond the diffusion of VOCs to the surface. This HHRA acknowledges that the "applicability of any specific exposure scenario to future conditions" is a significant uncertainty in the exposure assessment (Table 5-141), and scenarios such as mining are considered possible at the site. It is entirely possible that significant erosion or human intrusion will result in direct contact with the contaminated media below a 15-foot depth. Additionally, such a depth is not a particularly large obstacle for transport to the surface under natural conditions where deep penetrating roots and biota may cause bioturbation and subsequent exposure to humans. Contamination in the deep zone of the 116-N-3 trench includes 4,900 pCi/g cesium-137, 1,460 pCi/g strontium-90, and 5,580 pCi/g cobalt-60 as well as other radionuclides (CVP-2002-00002). Similar or higher levels of contamination were observed at the 116-N-1 trench (CVP-2001-00021), which was not included in this baseline HHRA.
197	5.2.2.2	5-18	Previous concerns have been raised by the Oregon Department of Energy in 2009 regarding the use of the single partition coefficient (K_d value) for modeling contaminant leaching and transport in the subsurface (Niles, 2009). K_d-based models have frequently demonstrated unreasonable results (100-Area RI/FS Work Plan Addendum 5 discussion of K_d values). The value of K_d is thought to change with a variety of environmental variables including temperature, pH and geochemistry. Revise the discussion to include a greater explanation regarding the K_d values assigned and the implications of each value for overall exposure that is presented. As noted previously, the results of groundwater exposure should be included as part of the total dose considered.
198	5.2.2.2	5-18	Tetrachloroethane (TCE) is omitted from the Deep Zone COPC list without explanation. This is an important contaminant that should be better characterized, including addressing the problem of well screening depths not being adequate to measure TCE in groundwater.
199	5.3.1.1	5-23	The statement that europium-152, europium-154, and cobalt-60 have half lives of 13.5 years or less and thus would not pose excessive risk in 2075 is misleading regarding the risk to site users today and before 2075 – what is the magnitude of risk in the near future?

200	5.3.2	5-24	How is risk calculated for a Resident Monument Worker who would eat fish from the Columbia River? It is unreasonable to assume that these workers would not ingest fish recreationally. Please include this exposure pathway in the risk assessment scenario.
201	5.3.5, 5.4.5, 5.5.5, 5.6.5, 5.7.5, and 5.8.5	5-40, 5-62, 5-74, 5-94, 5-119, and 5-143	The Yakama Nation developed an exposure scenario and requested that it be correctly incorporated into the RCBRA, assuming broad-area, site-wide residential use. The Yakama Nation's consideration of this document and all other similar documents at Hanford is governed in the first instance by compliance with the Treaty of 1855 (12 Stat. 951), which should be considered as an applicable or relevant and appropriate requirements (ARAR). The Treaty of 1855 between the Yakama Nation and the United States of America reserved specific rights and resources. These rights listed in Article 3 of 12 Stat. 951 include "...the right of taking fish at all usual and accustomed places...together with the privilege of hunting, gathering roots and berries, and pasturing their horses and cattle upon and unclaimed land." The U.S. Constitution in Article VI states, "...all Treaties made, or which shall be made, under the Authority of the United States, shall be the supreme Law of the Land..." The U.S. government has a fiduciary responsibility to the Yakama Nation to protect our Treaty rights and resources, our culture, health, and welfare. The Hanford Site is a portion of the Yakama Nation's homeland ("front yard"). In light of these facts, 12 Stat. 951 must, at a minimum, be identified as an ARAR in the CERCLA Remedial Investigation/Feasibility Study cleanup process (40 CFR 300.430(b)(9) and at (d)(3). It has not been recognized as such in this effort or under other CERCLA actions undertaken at the Hanford Site. A full analysis of the risks to Yakama Treaty resources and our peoples' health has yet to be performed. The risk assessment is deficient without this complete analysis.
202	5.3.5.3	5-45	Similar to the broad-area assessment, the local area assessment inappropriately reports only child hazard index results. Revise the reporting to include both child and adult hazard index results for the Yakama resident (and other scenarios) for every decision unit.
203	5.4.3.3 and 5.4.5.3	5-57 and 5-66	Regarding mercury as a risk driver in the 100-K area, please explain more clearly the difference between the linear and non-linear models (e.g., using sensitivity analysis) to better support the assumption of overestimating risk.
204	5.4.3.3	5-58	Whereas the large difference between the HI values for the Resident Monument Worker scenario and the Subsistence Farmer scenario does indicate the importance of modeling mercury accurately, the fact remains that the models are unreliable and assumptions are being made. Furthermore, it is unreasonable to assume a Resident Monument Worker would never grow or consume farm-raised food from the area.
205	5.4.3.3	5-58	The level of protective bias, or lack thereof, in the 100K area related to not excavating the 116-KE-5 and 116-KW-4 waste sites is not clear – please clarify.
206	5.4.5.1	5-64	The baseline HHRA states that remediated waste sites with Subsistence Farmer RME cancer risks above $1x10^{-4}$ (from the presence of short-lived radionuclides) were generally excavated to a significant depth. Identify those radioactive waste sites that do not fall into this "general" category and were excavated to shallower depths. Explain how this affects the assumed significant protective bias.
207	5.4.5.2	5-65, Table	Please clarify how the modeling for beef ingestion differs from that for wild game, which is a more accurate food source and exposure pathway for the Yakama Nation subsistence lifestyle.
208	5.5	5-67	How many remediated (and unremediated) waste sites in the 100-N area are not included this baseline HHRA? It seems unlikely that there are only two remediated waste sites in the 100-N Area.

209	5.5.2.3	5-72, Table 5-25	1,1,2,2-Tetrachloroethane is deemed not to be a COPC in the deep zone soil based on incomplete analysis. Please clarify the results of "inconclusive" statistical tests for this (and other) compounds in the 300 Area, including options for filling such data gaps.
210	5.5.5.2	5-75	Please clarify why the waste site 116-N-1 was eliminated from consideration, and provide an analyte-by-analyte evaluation for soil and groundwater matrices between accepted and eliminated waste sites in each decision unit for direct comparison.
211	5.8	5-127	Why were the important 618-10 and 618-11 burial grounds in the 300 Area not included as waste sites? Although they are scheduled to be remediated, their current status should be included in the assessment of baseline risk. Similarly, combined risks from chemical and radiological exposure should be evaluated for the multiple 316 remediated waste sites.
212	5.8.5	5-143	Update the scenario to include soil mixing deeper than 6-inches below ground surface. What is the combined cancer risk for both chemical and radionuclides? They are currently only presented individually.
213	5.9.1.1	5-149	Regarding the statement "…because of focused target analyte lists that were used for some waste sites, it is possible that some site related contamination was not captured in the analytical results for the HHRA and therefore total cumulative risks may be underestimated for those sites." This statement should be discussed further, perhaps in a separate uncertainty section of the document, since it contradicts other statements in the report regarding the "protective bias" of the waste site samples.
214	5.9.1.2	5-151	There is not adequate justification for using only "statistical" soil sample results and no "focused" sample results. This approach may not characterize isolated areas of elevated contamination (hot spots), which are not the same as outlying data, despite results of a sensitivity analysis. Revise the assessment to include composite as well as grab sample results.
215	5.9.1.2	5-152	"It is reasonable to assume…MIS samples would be biased low relative to what might be observed using discrete samples." How does this statement support the decision to use composite samples?
216	5.9.1.2	5-152	Similar to the broad-area assessment, samples collected from the site are incorrectly considered reference site data. These samples cannot be assumed to be absent site contamination. Revise the baseline HHRA accordingly.
217	5.9.3.3	5-165	Please add a reference for the table of toxicity uncertainty and modifying factors.
6.0 Screening-Level Groundwater Risk Assessment			
218	6.1	6-1	Please clarify why monitoring well data are limited to between 1998 and 2008, and why the well subset is limited. For example, based on known wells, only 15 to 20% of available wells were used for the baseline HHRA. The last sentence about the data "not adequately representing present-day exposure concentrations" requires further explanation.
219	6.1	6-3	The discussion of the proposed RI/FS work does not explain how additional data results will be integrated with the baseline HHRA to assess overall risk. Please clarify how the HHRA will fully incorporate the additional groundwater evaluations cited as part of the RI/FS reports to evaluate baseline risk and make risk management decisions.
220	6.2.2 and 6.2.3	6-7 and 6-10	Present day COPCs selected for groundwater by decision unit do not consider migration of contaminants from the Central Plateau. Revise the baseline HHRA to include and consider contamination from the Central Plateau in the River Corridor Decision Units. Incorporate the groundwater transport modeling performed as part of the Draft TC/WM EIS.

221	6.2.2.1	6-8	In the explanation of 100-KR-4, aluminum, iron, and manganese are omitted from the COPC list because of the "possibility that their occurrence may be related to well construction and, therefore, not representative of groundwater conditions." This contradicts the definition of assessing baseline conditions, which includes all contaminants despite their origin. These contaminants should not be prematurely excluded from the COPC list.
222	6.2.2.3	6-10	Provide an explanation as to why reference or background data are not available for groundwater. Revise the baseline HHRA to consider a plan for identifying reference groundwater sites for future risk management decisions.
223	6.2.2.3	6-10	This section states that COPCs were identified based on groundwater concentrations with reference to Appendix C-11. Appendix C-11 contains the contaminant concentrations in the shallow and deep soil, not the groundwater concentrations. The correct reference is Appendix C-12.
224	6.2.3	6-10	Protocol for risk assessment is to protect the maximally exposed individual. The selection of 50th and 90th percentile values for representative groundwater concentrations does not represent the reasonable maximum exposure. Provide additional justification for this decision and either demonstrate that it does not artificially reduce the risk calculated for groundwater exposure or select higher percentiles.
225	6.2.3	6-11	This section states that the values for the 50th and 90th percentiles represent general conditions both within and outside groundwater contamination plumes. The average values should not represent the condition outside the contaminant plumes.
226	6.3	6-14, Figure 6-2	The RCBRA used 140 samples to derive nitrate RME and CTE values for the 100-B/C Operable Unit (OU). How many of these samples were from different wells? How were the concentrations averaged in each well? Do all of the wells shown in this figure have nitrate data or only some? What are the contaminant sources of the plumes shown in this figure? This also applies to Figures 6-3 through 6-8.
227	6.3 to 6.7	6-14 to 6-49, Figures 6-2 to 6-8	It is difficult to distinguish the plumes from one another in this graphic. One figure for each contaminant and for each OU should be provided (instead of one figure per OU showing all the contaminants at once). The wells used in delineating each contaminant plume should be clearly identified. The number of samples used in each well and average concentration in the well should be provided. A discussion of sources of these plumes should also be provided in the text. Otherwise, an evaluation of these results is not possible.
228	6.3 to 6.7	6-14 to 6-49, Figures 6-2 to 6-8	The shapes of the contaminant plumes shown in these figures seem to be an artifact of the data used (taken at different times and depth intervals). The mixing and dispersion in the aquifer should have resulted in a more smooth distribution of the contaminant concentrations.
229	6.3.1.1	6-15	This section states that "Although future trends in groundwater concentrations have not been quantified in this assessment, natural radioactive decay of tritium (12.3-year half-life) and strontium-90 (28.8-year half-life) will result in a decrease of risk from these COPCs over time compared to present-day groundwater conditions." This is only the case if there are no other sources of these contaminants either within the OU or outside of the OU upgradient. The DOE's own calculations in the Draft TC/WM EIS show that strontium-90 from non-tank sources will remain at concentrations above drinking water standards (8 picocuries per liter in the absence of any other radionuclide and less if other contaminants are present, which will be the case here) in the River Corridor until about the year 2500. See Figure U-3 Appendix U, Volume 2 of the Draft TC/WM EIS.

230	6.4	6-22, Figure 6-3	The chromium plume shown in this figure was assumed to have a CTE of 47 µg/L and an RME of 97 µg/L. Simulation of the Cr plume (Figure O-11 in the Draft TC/WM EIS) shows the Cr plume in this area with 100 µg/L to 500 µg/L, which is more consistent with the values in the well 199-K-109A (CTE of 117; RME of 544 µg/L). If this is the case, then this well should have been considered in the main risk analysis, not in the supplemental risk analysis.
231	6.5	6-31, Figure 6-4	Sr-90 concentration in the well 199-N-67 (excluded from the main risk analysis) seems to be the center of mass of the plume delineated in this figure. If so, it should have been included in the main risk analysis, not the risk supplemental analysis.
232	6.6	6-40, Figure 6-5	The groundwater flow direction based on the nitrate plume shown in this figure is not consistent with the flow direction of the Cr plume in the north east part of the figure where these two plumes partially overlap each other. Please correct this inconsistency.
233	6.6	6-40, Figure 6-5	Cr and nitrate concentrations in the wells 199-D5-41, 199-D5-99, and 199-D5-104 (excluded from RME and CTE analysis) seem to be the center of mass of the plumes delineated in this figure. If so, they should have been included in the main risk analysis, not the supplemental risk analysis.
234	6.8.2.1 to 6.8.2.2	6-67 to 6-69	The results provided in this section state that inhalation of uranium in sweat lodges is not considered for the Yakama Resident scenario, but is considered in the CTUIR Resident scenario. No justification is given for the elimination of exposure to an identified COPC. Removal of this pathway from one exposure scenario (versus another tribal one) is not appropriate and significantly reduces the total exposure to Yakama Residents and distracts from the serious danger posed by uranium as a result of its long half life and large quality. The Yakama Exposure Scenario did not eliminate any pathways or COPCs as part of the scenario, and the removal of this particular pathway, and exposure to other nonvolatile COPCs should not have occurred on a selective basis in the Yakama Resident scenario for this or any other Decision Unit. Revise the baseline HHRA to include all COPCs in groundwater through the dermal adsorption, inhalation, and ingestion pathways. Specifically, include all radionuclides present in groundwater now and modeled to be present in the future. Identify the risk posed by each contaminant individually as well as the cumulative risk posed by all contaminants present.
235	6.9	6-71	Significant uncertainty regarding the timing, volume, nature, and toxicity of contamination from the Central Plateau reaching the River Corridor should be addressed and included in this section. Use of the RME and CTE parameter values as outlined in previous sections does not factor this contamination into total exposure. Revise the baseline HHRA to include this additional contamination and associated uncertainties and address them both qualitatively and quantitatively.
236	6.9.1.2	6-73	Refer to previous comments regarding the representativeness of the data used to perform this baseline HHRA. The assessment performed does not evaluate the total risks posed by the site, but does perform an analysis of the risks posed by current conditions at selected waste sites. The assertion that radioactive decay will result in ultimately lower concentrations of contamination in the future fails to acknowledge migration of contamination from the Central Plateau which will reach the River Corridor within the period analyzed. Revise the HHRA to evaluate all the risks posed by the site including all waste sites, groundwater contamination in the Central Plateau, and reactor cores over the period of analysis.
237	6.9.3.2	6-79	Revise the baseline HHRA discussion of groundwater cancer risks for the Yakama Resident (and CTUIR Resident) to include alpha radiation emissions such as those produced by uranium and thorium. Further revise this text to explain and justify whether application of linear dose-response factors applied to chronic radiation exposure under-predict, or over-predict the carcinogenic risk.

7.0 Conclusions and Recommendations			
238	7.2.2	7-7	Note that the OSWER Directive 9200.4-18 states, "Guidance that provides for cleanups outside the risk range (in general, cleanup levels exceeding 15 millirem per year which equates to approximately 3 x 10^{-4} increased lifetime risk) is similarly not protective under CERCLA and generally should not be used to establish cleanup levels." This baseline HHRA inconsistently identifies both 15 millirem per year and 15 millirem per year above background as remedial action goals. Not only is this inconsistent, but these represent two very different numbers. Per the guidance, revise the HHRA to state that no dose greater than 15 millirem per year, including doses from background samples, will be the remedial action goal, at a minimum. See comment #13 and general comments for more details.
239	7.3	7-14	Particular site-specific conditions that would justify the acceptability of a risk estimate around 1 x 10^{-4} are not defined. OSWER Directive 9355.0-30 states that a risk manager may decide that a baseline risk level less than 1 x 10^{-4} at a site is unacceptable (i.e., risks below this upper limit are still considered unacceptable and must comply with a more protective limit) due to site-specific reasons and that remedial action is warranted where, for example, there are uncertainties in the risk assessment results. Revise the text box language to more accurately reflect the full range of alternatives from the OSWER directive.
240	7.3.2.1	7-20	The result of the broad-area risk assessment for the nonresident tribal scenario does not include risk from Columbia River water, either ingestion or sweat lodge use. This needs to be calculated and included.
241	7.3.2.2	7-22	It is inaccurate to say that a risk assessment was conducted for seeps. A few data points (with significant data gaps, particularly for chromium) were compared to drinking water criteria. However, no risk assessment using any of the exposure scenarios was described in Chapter 4. The summary statement that "one can assume" is not a calculation of risk.
242	7.4	7-32	In the Introduction (page 1-16), PRGs are described as "levels of contaminants that may remain onsite and still be adequately protective of human health." However, since PRGs were not developed for any tribal scenarios they do not represent levels that are protective of tribal health.
243	7.4	7-33	Based on EPA guidance (Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination and Risk Assessment Guidance for Superfund Part B), there does not appear to be any rationale for using different target risk levels for calculating PRGs for radiological and non-radiological contaminants. Risk-based PRGs for both chemical and radiological carcinogens should be calculated using a target risk of 1 x 10^{-6}.
244	7.5	7-37	The Conclusions and Recommendations chapter summarizes the document's results and uncertainties. However, it does not provide a comprehensive conclusion. While some recommendations are made for each section of the document, there are no comprehensive recommendations or next steps provided for the Hanford Site and human health risk assessment as a whole.
Appendices			
245	Appendix A	A-1	The text incorrectly states that Appendix A contains meeting notes from workshops held between August 2006 and May 2007. Revise the text to indicate that Appendix A contains notes from workshops held between August 2006 and January 2008.
246	Appendix A	A-1	The hyperlink (http://www.washingtonclosure.com/Projects/EndState/risk_library.html#100Area) for the Washington Closure Hanford End States and Final Closure internet web site address (URL) is outdated and cannot be found. Correct the document text with the current web site URL hyperlink.

247	Appendix B	B-4	We have commented previously on the problems with the reference site selection used for this risk assessment. Those comments are still valid and can be summarized by the following concerns: 1) Both EPA and MTCA define a CERCLA "site" as all locations where site-related contamination is present. In this case, due to air releases as well as dust from many operations, the entire Hanford Reservation is the CERCLA site. According to the EPA guidance cited in Appendix B, reference sites are intended to be locations that are similar in habitat but have no site-related contamination. The citation is correct (Page B-5) that one EPA guidance suggests that reference "targets" for contamination can be derived from an evaluation of the contaminant gradient on a site, selecting the lowest concentrations as the suitable reference concentration. DOE did not complete such a gradient analysis to select the lowest concentrations from the reference data set. Besides being located close to non-site specific sources of contamination from human activities, the data presented in this appendix readily demonstrate that relatively high concentrations of many substances are present at some of the reference sties. In addition, this same citation noted that the risk assessment for the Rocky Flats Arsenal used a reference site 50 miles away from the site itself for biota samples, and five miles away for soil samples. 2) The reference site data are used inappropriately to eliminate risk factors present at waste sites before those risks are calculated. The risk assessment should identify the total risk presented by exposure to the waste sites as the first step. If properly selected, the reference and background data have a place in risk management decisions in further steps in the evaluation. Only then can incremental risks posed by sites above and in addition to all the other risks present be evaluated. 3) The RCBRA misinterprets EPA guidance with regard to reference sites as locations for comparing resource use and conditions as an indicator of impacts and the appropriate reference "target" for contamination. EPA guidance is clear that the reference target for contaminants are the lowest concentrations that can reasonably be associated with the general area off the CERCLA site, not the average of all of the concentrations of reference sites selected for the more holistic evaluations.
248	Appendix B	B-20	It would be helpful to include more detailed information regarding the sampling and analytical procedures used in collecting the reference data. As noted in the text, methods can affect the comparability of the results, but the limited presentation does not allow a reader to discern how differences in results may be impacted. In addition, other factors, such as grain size in soils, are also known to affect the concentrations observed. Because of the predominately coarse nature of the soils at Hanford, grain size may be an important factor and should be discussed.
249	Appendix B	B-20	Many substances have no background data for comparison, or use only the Hanford Area Background data for comparison. This lack of data means that the degree of contamination on the site cannot be assessed.
250	Appendix B	B-20	Many substances were noted to have much higher concentrations in at least some of the on-site reference areas compared to other sites or to the background data. As per EPA and MTCA guidance these data are inappropriate for use as representing a reference condition for the risk assessment. If suitable off-site reference data cannot be found, EPA guidance suggests that these data should be screened out of the data set using a gradient analysis.

251	Appendix C	All	Overall it was not possible to determine which of the data presented in the various data trend appendices were chosen for inclusion in the HHRA. It is unclear what data from the data groups were used to create the box plots, the reasonableness of this data selection, and how the data were used after the box plots were created. In many cases neither the sample means nor the number of samples seemed to match between the various data summaries. Appendix C should be drafted to stand alone and provide more examples and transparency on how and what data were ultimately used to create the box plots. The data groups used provide a limited number of contaminants, a limited amount of monitoring data, a limited number of species, and a limited number of scenarios. The use of such data has more than likely biased the results of the risk assessment
252	Appendix C-1	C.1-44	Please provide the algorithm for computing "calculated total uranium."
253	Appendix C-1	C.1-45	The exclusion of data simply because it was collected with a "less-preferred analytical method" is too subjective a reason for rejection. The data should be considered valid unless there is some documented reason to believe they are inaccurate or unacceptably imprecise.
254	Appendix C-3	NA	The H-3 concentrations are one order of magnitude higher in the supplemental risk assessment calculations for 100-F Operable Unit. The number of samples used in the supplemental analysis is more than 10% of the number of samples used in the main analysis. It raises the question of whether this well should have been included in the main analysis.
255	Appendix C-3 to 4	NA	This appendix does not provide the actual data used to derive RME and CTE for each OU. Only the summary of the results for each OU are presented. Without the actual data, the results cannot be evaluated.
256	Appendix C-5	NA	The introduction to Appendix C-5 should make it clear that the CVP data presented in this section include both the shallow and deep soil, or as a potentially better solution, eliminate Appendix C-5 in favor of retaining only Appendix C-11. The high concentrations of some substances noted in Appendix C-5 can cause confusion in determining what data were used in the HHRA.
257	Appendix C-5, C-11	All	It is surprising that fairly high concentrations of Pu-241 were measured in the IU2/IU6 decision unit, while it's decay product, Am-241, was not. The different numbers of samples given for each radionuclide would seem to indicate that these data are from different locations. Sampling the same areas might give different results. In addition, even though the concentrations of Pu-241 were high, Pu-241 was not included as a COPC. Further, no Pu-241 sampling was apparently performed at most of the CVP sites, even though Pu-239/140 was measured. Please clarify and correct these inconsistencies.
258	Appendix C-5, C-11	All	The soils data presented in these appendices show that at least in the 100-N decision unit, very high concentrations of radionuclides, e.g., Pu-239/240 and Sr-90, remain in the deeper soils. The risks of these high concentrations should be discussed.
259	Appendix D-5	Tables	Please add a column for sample size (N) next to exposure point concentrations.
260	Appendix E	E-8	This comment addresses the following statement, "Based on the results of the 1998 risk evaluation, there is no requirement under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) to remediate the river effluent pipelines." The pipelines should be considered for removal because they may pose a risk to humans, the environment, and could expose the population to contamination. Revise and include the pipelines in the assessment and potential removal action.

ATTACHMENT 1

Comments of the Institute for Energy and Environmental Research on Draft C of the *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment*, DOE/RL-2007-21, U.S. Department of Energy, December 2010

February 2010

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Comments of the Institute for Energy and Environmental Research on Draft C of the *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment*, DOE/RL-2007-21, U.S. Department of Energy, December 2010

February 2010

These are comments and recommendations on Draft C of the *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment, DOE/RL-*2007-21, U.S. Department of Energy, Richland, Washington, December 2010, developed on behalf of the Yakama Nation. This document is abbreviated as RCBRA 2010. The 2007 version of this document, Draft A, is abbreviated as RCBRA 2007.

A. ARARs

1. Radiation dose

The radiation dose criterion proposed by DOE is 15 mrem per year, total effective dose equivalent:

> The basis of the IAROD cleanup levels for radionuclides is a radiation dose of 15 mrem/yr above background, and all of the 156 remediated waste sites evaluated meet the cleanup level. In general, a radiation dose of 15 mrem/yr equates to approximately a 3×10^{-4} lifetime cancer risk (OSWER Directive 9200.4-18, "Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination"). Because of this, it is possible that some sites cleaned up to a 15 mrem/yr dose cleanup goal may show cancer risk greater than 1×10^{-4}.[1]

There are two ways in which the maximum dose target is too high. First according to DOE's calculations, DOE acknowledges that 15 mrem per year will produce a life a lifetime cancer risk (over 70 years) of about 3×10^{-4}. This risk is about three times greater than the maximum risks of 10^{-4} allowed under CERCLA.[2] In doing this, the DOE appeals to the EPA guidance on the topic, OSWER Directive 9200.4-18.

[1] RCBRA 2010 v.II pt.1, p. ES-7

[2] EPA 1997 p. 2 EPA 1997, says, in part: "ARARs are often the determining factor in establishing cleanup levels at CERCLA sites. However, where ARARs are not available or are not sufficiently protective, EPA generally sets site-specific remediation levels for: 1) carcinogens at a level that represents an excess upper bound lifetime cancer risk to an individual of between 10^{-4} to 10^{-6}; and for 2) non-carcinogens such that the cumulative risks from exposure will not result in adverse effects to human populations

The EPA guidance in OSWER 9200.4-18 states that 3×10^{-4} is acceptable as a CERCLA lifetime cancer risk limit. This was done in 1997 in the context of impending Nuclear Regulatory Commission (NRC) decommissioning regulations that would have allowed even higher risk levels:[3]

> It is important to note that a new potential ARAR was recently promulgated: NRC's Radiological Criteria for License Termination (See 62 FR 39058, July 21, 1997). We expect that NRC's implementation of the rule for License Termination (decommissioning rule) will result in cleanups within the Superfund risk range at the vast majority of NRC sites. However, EPA has determined that the dose limits established in this rule as promulgated generally will not provide a protective basis for establishing preliminary remediation goals (PRGs) under CERCLA. The NRC rule set an allowable cleanup level of 25 millirem per year (equivalent to approximately 5×10^{-4} increased lifetime risk) as the primary standard with exemptions allowing dose limits of up to 100 millirem per year (equivalent to approximately 2×10^{-3} increased lifetime risk). Accordingly, while the NRC rule standard must be met (or waived) at sites where it is applicable or relevant and appropriate, cleanups at these sites will typically have to be more stringent than required by the NRC dose limits in order to meet the CERCLA and NCP requirement to be protective. Guidance that provides for cleanups outside the risk range (in general, cleanup levels exceeding 15 millirem per year which equates to approximately 3×10^{-4} increased lifetime risk) is similarly not protective under CERCLA and generally should not be used to establish cleanup levels.

But 3×10^{-4} is not 1×10^{-4} – it is three times bigger than allowed by the law. This interpretation of CERCLA, which limits lifetime risk to a range of 10^{-4} to 10^{-6} is not appropriate. The EPA should correct its guidance so that it is in conformity with the law and the DOE should follow what is clearly stated in the law. The Yakama Nation generally recommends that the DOE follow EPA guidance; in this case the EPA guidance conflicts with the risk range specified in CERCLA. Hence the Yakama Nation is recommending that the CERCLA risk limits be respected. Using the same cancer risk coefficient that is implicit in the calculation that 15 mrem per year results in a risk of about 3×10^{-4}, the maximum dose limit should be 5 millirem per year from all pathways.

However, even 5 mrem per year does not actually reflect a lifetime cancer risk of 10^{-4} according to the EPA's own published advice to the public on radiation risk. Specifically, in explaining radiation risks, the EPA states:

> Each radionuclide represents a somewhat different health risk. However, health physicists currently estimate that overall, if each person in a group of 10,000 people exposed to 1 rem of ionizing radiation, in small doses over a life time, we would expect 5 or 6 more people to die of cancer than would otherwise.[4]

One rem each to 10,000 people is a population dose of 10,000 person-rem. Hence, 5 to 6 cancer deaths from this dose equates to a risk factor range of 5×10^{-4} per rem to 6×10^{-4} per rem. This range is also consistent with risk factor for low-level, low-LET radiation of 5.75×10^{-4} provided by the EPA in its latest

(including sensitive sub-populations) that may be exposed during a lifetime or part of a lifetime, incorporating an adequate margin of safety. (See 40 CFR 300.430(e)(2)(i)(A)(2))

[3] EPA 1997 p. 3

[4] EPA 2010

guidance (Federal Guidance Report 13).[5] In contrast, the EPA's calculation that 15 millirem per year over a 70-year lifetime (specified in 1989 EPA guidance[6]) results in a 3×10^{-4} risk equates to a risk factor of only 2.86×10^{-4} per rem. This is only about half the fatal cancer risk factor provided in EPA's latest guidance report FGR 13 and 1.75 to 2.1 times less than the risk factor provided in the public guidance cited above. In other words, using the fatal cancer risk factor in EPA's FGR 13 (5.75×10^{-4}), a lifetime fatal cancer risk of 1×10^{-4} corresponds to a maximum annual dose after cleanup of about 2.5 mrem per year, rather than the 15 mrem per year suggested by the DOE and the EPA.

Finally, EPA's Superfund guidance explicitly refers to determination of "an increase in the incidence of an adverse health effect (e.g. cancer, birth defect)..."[7] as part of hazard assessment and rather than an increase in mortality. Further, the rate of fatal cancers varies according to the state of treatment technology. In view of these two factors, the criterion for deriving dose should be cancer incidence risk not fatal cancer risk. This would further reduce the maximum allowable dose after cleanup by about a factor of two.

In sum, applying the upper end of the cancer risk range of 10^{-4} to 10^{-6} specified in CERCLA, interpreted to mean cancer incidence, which is the most protective of human health, and is the most reasonable interpretation of CERCLA, results in a dose limit of roughly 1 mrem per year (rounded).

When 1 mrem per year is used as the defining ARAR for radiation dose, the number of River Corridor sites that would not meet the cleanup criterion under the Yakama tribal scenario would be much larger. As it is, the DOE calculates that for "the Native American scenarios, total cumulative cancer risks at 111 of the 156 remediated waste sites evaluated exceeded the EPA upper risk management level of 1×10^{-4}."[8] Hence, even under the DOE's own cancer risk factors and dose limits, the cleanup does not meet CERCLA criteria at more than 70 percent of the remediated waste sites. Under a protective interpretation of CERCLA, the cleanup specified by the DOE in RCBRA 2010 would be generally ineffective in meeting even the upper limit of CERCLA risk the in Native American scenarios.

We should note here that the dose conversion factors and lifetime risk factors per unit intake for drinking water and dietary ingestion for individual radionuclides are generally satisfactory as specified in Tables 3-29 and 3-30. For the most part they are taken from the most recent EPA guidance in Federal Guidance Report 13.[9]

2. CERCLA and MTCA

In addition to CERCLA, Hanford is also covered by the Washington State's Model Toxics Control Act (MTCA), which is the state's version of CERCLA. Radionuclides are defined as hazardous substances under MTCA. The concentration limit for each hazardous substance is limited to a level corresponding to a 10^{-6} lifetime risk. When more than one hazardous substance is present, the combined lifetime cancer risk for chemicals and radiation should be 10^{-5} or less. While MTCA is generally interpreted as applying to chemicals only, we believe this interpretation is very partial, especially under the

[5] FGR 13, p. 179.
[6] EPA 1989, p. 6-22 and p. 8-11 for instance.
[7] EPA 1989, p. 1-6.
[8] RCBRA 2010 v.II pt.1, p. ES-15
[9] RCBRA 2010 v.II pt.2, pp. 3-56 and 3-57; FGR 13 and Suppl. We note that we have not done a complete check of all the entries.

circumstances of Hanford, which contains extensive radionuclide contamination. Since MTCA explicitly defines radionuclides as hazardous substances, the combined limit for radionuclides and chemicals should be such that lifetime cancer risk is a maximum of 10^{-5} or 1 in 100,000.[10]

It should be noted that EPA guidance requires the summing of chemical and radiological risk:

> Cancer risk from both radiological and non-radiological contaminants should be summed to provide risk estimates for persons exposed to both types of carcinogenic contaminants. Although these risks initially may be tabulated separately, risk estimates contained in proposed and final site decision documents (e.g., proposed plans, Record of Decisions (RODS), Action Memos, ROD Amendments, Explanation of Significant Differences (ESDs)) should be summed to provide an estimate of the combined risk to individuals presented by **all** carcinogenic contaminants.[11]

Recommendations:

- DOE should use cancer incidence not fatal cancer risk in evaluating compliance with CERCLA.
- DOE should use a fatal cancer risk factor of 5.75×10^{-4} per rem (from FGR 13) and a cancer incidence risk factor of 1.1×10^{-3} per rem[12] in evaluating cancer risks.
- DOE should use an annual all-pathway lifetime risk ARAR of 10^{-6} for individual contaminants and 10^{-5} for all radioactive and non-radioactive contaminants combined.
- For radionuclides, carcinogenic and non-carcinogenic hazardous chemicals, the following criterion should be met:

$$\sum_{i=1}^{N} \frac{MR_i}{MCLR_i} + \sum_{j=1}^{M} \frac{MC_j}{MCLC_j} < 1$$

where

MR_i is the maximum radionuclide concentration for radionuclide and i goes from 1 to N and N is the number of radionuclides,
MC_j is the maximum chemical concentration for chemical j and j goes from 1 to M, and M is the number of chemicals
$MCLR_i$ is the MCL for radionuclide R_i, and
$MCLC_j$ is the MCL for chemical C_j

B. The Central Plateau and the River Corridor

The goal of protecting the Columbia River and of protecting groundwater requires restoration of groundwater to meet drinking water standards throughout the site. This in turn requires compatible levels of soil and vadose zone cleanup, including in the Central Plateau.

[10] WAC MTCA 2007 p. 18 and pp. 94-96
[11] EPA 1997 p. 4. Emphasis in the original.
[12] Average value for males and females estimated in the BEIR VII report, rounded to two significant figures. NAS/NRC 2006, p. 15.

1. Contaminant migration considerations

The protection the Columbia River and the groundwater in the River Corridor cannot be achieved if contamination from outside the corridor returns to the River Corridor after it has been cleaned up. The River Corridor cleanup can be made compatible with shrinking the active cleanup footprint in the Central Plateau, as DOE proposes, only if the latter does not include disposing of quantities of waste that will further endanger and contaminate the groundwater in the Central Plateau. This is because it is clear that in the long-term (and even in the coming decades in some cases) contaminant migration from the Central Plateau will play a major role in determining the level of pollution and the level of risk in the River Corridor.

According to RCBRA 2010:

> Cleanup of the Central Plateau is a highly complex activity because of the large number of waste sites, surplus facilities, active treatment and disposal facilities, and areas of deep soil contamination. Past discharges of more than 450 billion gallons of liquid waste and cooling water to the soil have resulted in about 60 square miles of contaminated groundwater. Today, some plumes extend far beyond the plateau. Containing and remediating these plumes remains a high priority. For areas of groundwater contamination in the Central Plateau, the goal is to restore the aquifer to achieve drinking water standards. In those instances where remediation goals are not achievable in a reasonable time frame, programs will be implemented to contain the plumes, prevent exposure to contaminated groundwater, and evaluate further risk reduction opportunities as new technologies become available. Near-term actions will be taken to control plume migration until remediation goals are achieved.[13]

We support the goal of restoring the Central Plateau groundwater to drinking water standards and near term actions to control plume migration. However, near term and intermediate term actions must not compromise the goals of groundwater restoration to drinking water standards on the Central Plateau and in the River Corridor. These are critical goals in their own right (and also ARARs) and they are also needed to protect the Columbia River from migrating radioactivity. Specifically, using the Central Plateau as the location where wastes from the River Corridor, from cleanup of the Central Plateau itself, and from other DOE sites are disposed of is not compatible with cleaning up either the River Corridor or the Central Plateau.

The RCBRA does not take any considerations relating to migration of radionuclides from the Central Plateau in the future into account. Rather, it limits itself to groundwater contaminant data as collected between 1998 and 2008:

> A key uncertainty with the groundwater assessment is related to the ability of the groundwater data set collected from 1998 to 2008 to represent current baseline conditions and potential exposure within each groundwater OU [i.e., operable unit]. For this reason, the groundwater assessment is presented as a "screening-level" assessment. Additional groundwater data will be collected and evaluations will be presented in the RI/FS [i.e., remedial investigation/feasibility study] reports for the River Corridor ROD decision areas.[14]

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[13] RCBRA 2010 v.II pt.1, p. Forward-6
[14] RCBRA 2010 v.II pt.1, p. ES-8

This approach to groundwater is completely unsatisfactory. Present day pollution levels are not a guide to conditions in the future when, for instance, one may hypothesize a resident farmer being on site and using the groundwater or of Native Americans using groundwater for sweat lodges. At that time the groundwater would be more characterized by migration from areas upstream of the River Corridor, including migration from the Central Plateau and all its waste disposal areas and its entire vadose zone.

The operation of the Environmental Restoration Disposal Facility (ERDF) is making protection of groundwater in the Central Plateau to drinking water standards impossible. Preliminary calculations indicate that the inventory of some of the long-lived radionuclides already disposed of in the ERDF produce doses that would exceed the 1 in 10,000 cancer risk and contamination that would exceed the drinking water standard. This indicates that disposing of long-lived radionuclides in significant amounts in ERDF is compromising the stated goal of restoring the Central Plateau groundwater to the drinking water standard. Waste containing significant amounts of long-lived radionuclides should be managed differently (see below). The same comments would apply to the Integrated Disposal Facility (IDF) planned in the East and West 200 Area for Hanford waste.

Potential Groundwater Contaminants at the ERDF

Constituents	Maximum detected soil concentration	Predicted groundwater concentration	Travel time to ERDF boundary
Radionuclides	**picocuries per gram**	**picocuries per liter**	**Years**
Carbon-14	640	1.3×10^6	520
Technetium-99	1.1	2.3×10^3	520
Total uranium	20034	1.1×10^3	520
Uranium-233/234	2100	5.3×10^2	520
Uranium-235	638.4	2.3×10^1	520
Uranium-238	9143	4.9×10^2	520

(Source: DOE 1994 Table 4-10 (pp. 4T-10c to 4T-10d))

The values in the above table show that technetium-99 exceeds the drinking water limit of 900 picocuries per liter by almost two-and-half times, that of carbon-14 (2,000 picouries per liter) by 650 times and for uranium by 55 times.[15] Moreover, it should be noted that the MCL for each radionuclide is lowered when there is more than one radionuclide present, as is the case here. The concentrations of all radionuclides must conform to the ratio rule (see below). We also note that the dose from C-14 alone exceeds the maximum lifetime cancer risk of 1 in 10,000 from a single carcinogen in the CERCLA framework by about 1,000 times for cancer incidence and by about 700 times for cancer mortality.[16] This is clearly an unacceptable result for the Central Plateau; it also poses risks for the River Corridor and the Columbia River as well, since the half-life of carbon-14 is 5,730 years.

The DOE groundwater calculations show that onsite disposal of significant amounts of long-lived radionuclides in shallow disposal facilities is not consistent with the goals of CERCLA and not protective of human health, even if dose pathways other than drinking water for the resident farmer and for the

[15] 40 CFR 141.66 2009. 40 CFR 141 is the U.S. Environmental Protection Agency's *National Primary Drinking Water Regulation*. It specifies a uranium MCL of 30 micrograms per liter, which translates into 20 picocuries per liter for natural uranium. The technetium and carbon-14 limits are EPA values as cited by the DOE in the Draft Tank Closure & Waste Management EIS 2009, Appendix O, Table O-4, p. O-31.

[16] Rounded values, using a 70-year lifetime and FGR 13 risk factors.

Yakama lifestyle are not taken into account. The results would be even more unacceptable if all relevant long-term pathways are considered.

In addition to the problem of water contamination due to disposal in ERDF, there are also issues related to the proposed Integrated Disposal Facility (IDF) in the East and West 200 Area. The import of "offsite" wastes from facilities other than Hanford as described in the *Draft Tank Closure and Waste Management Environmental Impact Statement for the Hanford Site, Richland,* Washington (Draft TC&WM EIS) would seriously compromise this much more than Hanford waste.[17] Specifically, the problem of water contamination well above federal drinking water standards will be exacerbated by disposal of offsite waste (also known as imported waste).

In the Draft TC&WM EIS Table S-8, reproduced below, it is clear that neither Alternative 2 nor 3 would meet drinking water standards. Since a ratio rule applies, exceedance is much worse than indicated if it occurs in the same time frame. This is, in fact, indicated by the DOE modeling as peak dose times are also provided in the table. In both the cases shown in Table S-8, the calculations assume that imported waste would be disposed of in the IDF.

Table S–8. Maximum Concentrations of Technetium-99 and Iodine-129 in the Peak Year at the IDF-East and IDF-West Barriers

Contaminant	IDF-East (Waste Management Alternative 2)	IDF-West (Waste Management Alternative 3)	Benchmark Concentration
Radionuclide in picocuries per liter			
Technetium-99	1910	20,200	900
	(9005)	(3713)	
Iodine-129	18	173	1
	(8196)	(3797)	

Note: Corresponding calendar years are shown in parentheses.
Key: IDF-East=200-East Area Integrated Disposal Facility; IDF-West=200-West Area Integrated Disposal Facility.
(Source: DOE/EIS-0391 2009 Draft, Summary p. S-100)

Table S-9 of the Draft TC&WM EIS, reproduced below, indicates that almost the entire impact on groundwater in the IDF would come from imported waste. This is clear when we compare the Alternative 3 in Table S-9, which assumes no imported waste is disposed of, with Alternative 3 in Table S-8 above, which includes disposal of imported waste. In the no imported waste case, the drinking water standard is met for Tc-99 and modestly exceeded for I-129. In the case of imported waste, the drinking water standard for Tc-99 is exceeded by more than 20 times for Tc-99 and more than 170 times for I-129.[18] No comparable table is provided in the summary for IDF East, which is "Waste Management Alternative 2." However, we may infer from the IDF-West results that imported waste would also produce the majority of the impact there.)

[17] DOE/EIS-0391 2009 Draft. See, for example, Foreword (found in v.1) pp. 7and 8 and Summary pp. S-100 and S-109.
[18] Note we are using DOE's benchmarks for the drinking water standards in this paragraph, since these are the ones used in the TC&WM EIS. The standards would be stricter if FGR 13 dose conversion factors are used. Hence, the exceedance of the drinking water standards using FGR 13, which is the most recent published Federal Guidance Report, would be greater.

Table S–9. Maximum Concentrations of Technetium-99 and Iodine-129 in the Peak Year at the IDF-East Barrier

Contaminant	Waste Management Alternative 2	Waste Management Alternative 3	Benchmark Concentration
Radionuclide in picocuries per liter			
Technetium-99	1910	471	900
	(9005)	(8991)	
Iodine-129	18	1.4	1
	(8196)	(11,243)	

Note: Corresponding calendar years are shown in parentheses.
Key: IDF-East=200-East Area Integrated Disposal Facility.
(Source: DOE/EIS-0391 2009 Draft, Summary p. S-101)

Further, the Draft TC&WM EIS indicates that groundwater in the River Corridor will remain contaminated well above drinking water standards for hundreds and even thousands of years for individual contaminants due mainly to migration from upland areas. Table U-1 from the Draft TC&WM EIS notes that much or most of the long-lived contaminants of potential concern (i.e., excluding only tritium in the table) released from the Central Plateau and the vadose zone will eventually pollute the groundwater and the Columbia River itself:

Table U–1. Release to the Vadose Zone, Groundwater, and the Columbia River of the COPC Drivers from Non–*TC & WM EIS* Sources

	Radionuclide (curies)				Chemical (kilograms)		
Release to:	**H-3**	**I-129**	**Tc-99**	**U-238**	**Cr**	**NO_3**	**Utot**
Vadose zone	3.43×10^{6}	2.49×10^{1}	7.33×10^{2}	3.13×10^{3}	3.35×10^{5}	7.38×10^{7}	2.53×10^{5}
Groundwater	2.06×10^{6}	2.48×10^{1}	7.12×10^{2}	1.48×10^{2}	3.40×10^{5}	7.42×10^{7}	1.05×10^{5}
Columbia River	1.11×10^{5}	2.46×10^{1}	7.26×10^{2}	1.40×10^{2}	3.51×10^{5}	7.47×10^{7}	9.28×10^{4}

Note: Total amount released over the 10,000-year period of analysis.
Key: COPC = constituent of potential concern; Cr=chromium; H-3=hydrogen-3 (tritium); I=iodine; NO3=nitrate; Tc=technetium; *TC & WM EIS = Tank Closure and Waste Management Environmental Impact Statement for the Hanford Site, Richland, Washington*; U=uranium; Utot=total uranium.
(Source: DOE/EIS-0391 2009 Draft, Appendix U p. U-2)

Table U-2 in the same document shows that the pollution will continue far above drinking water levels for hundreds and in some cases, thousands of years: For instance, the DOE estimates that iodine-129 will peak at more than 9 picocuries per liter (about nine times the drinking water standard) two-and-a-half thousand years from the present and plutonium will peak at 4,250 pCi/L in the year 2983 – a level about 280 times above the drinking water limit.[19]

In this context, using ERDF, IDF, or any other area for permanent disposal of radioactive and chemical hazardous waste disposal from the River Corridor will not only defeat the goal of Central Plateau cleanup but also River Corridor cleanup.

[19] DOE/EIS-0391 2009 Draft, Appendix U Table U-2 (p. U-3)

Recommendations:

- Cleanup RODs should be based on an evaluation of all contaminants migrating into the River Corridor, including those from the Central Plateau, over the time period of evaluation (10,000 years in the present framework).
- Wastes from the River Corridor, including remediation-generated wastes and graphite moderator blocks from the Hanford reactors, should be retrievably stored.
- Many of these wastes, including the graphite blocks, should be disposed of in a deep geologic repository. Preparation of such wastes for deep geologic disposal should be part of River Corridor decommissioning considerations and planning.

C. Exclusions

RCBRA 2010 has incorrectly excluded certain radionuclides from consideration. Specifically, it states:

> Background radionuclides (potassium-40, radium-226, radium-228, thorium-228, thorium-230, and thorium-232): These background radionuclides were identified by consensus of Tri-Party managers as not directly related to Hanford operations or processes.[20]

The reason for the exclusion of thorium-232 and its decay products, thorium-228 and radium-228, is incorrect. Thorium-232 was handled in large quantities at Hanford. Hence its decay products, Th-228 and Ra-228, which build up in a few years were also present. Actually, Th-228 is always present with natural Th-232. Ra-228 gets separated initially when thorium ore is refined but builds up significantly in a few years (with equilibrium achieved in a couple of decades). Thorium dumped at the site would have all three radionuclides in equilibrium and present a long term hazard. It is incorrect to exclude thorium-232 and its decay products.

D. Overall conclusion

Overall, the framework of Draft C of the RCBRA is unsatisfactory. It ignores future migration of contaminants into the River Corridor that will completely negate the cleanup. It does not meet even its own risk criteria for tribal scenarios, much less those that are required under CERCLA and MTCA interpreted to protect public health. In planning to dispose of wastes from River Corridor cleanup as well as the graphite reactor blocks from the 100 Areas in the Central Plateau, DOE will further exacerbate the problem of contaminant migration from the Central Plateau into the River Corridor and the Columbia River itself. In other words, the cleanup goals set forth by the DOE cannot be achieved within the framework set forth in the RCBRA. It should be completely revamped. In particular, retrievable storage of wastes generated by cleanup, rather than their permanent disposal, should be essential in a revised RCBRA document and in all RODs deriving from it.

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[20] RCBRA 2010 v.II pt.1, p. 3-30

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FGR 13 and Suppl.	Keith F. Eckerman, Richard W. Leggett, Christopher B. Nelson, Jerome S. Puskin, and Allan C.B. Richardson. *Cancer Risk Coefficients for Environmental Exposure to Radionuclides.* (EPA 402-R-99-001) (Federal Guidance Report No. 13) CD Supplement. Rev. 1. Oak Ridge, TN: Oak Ridge National Laboratory; Washington, DC: Office of Radiation and Indoor Air, U.S. Environmental Protection Agency, April 2002. Includes original 1999 FGR no. 13, which is also on the Web at http://www.epa.gov/radiation/docs/federal/402-r-99-001.pdf.
NAS/NRC 2006	Richard R. Monson (Chair) et al. *Health Risks from Exposure to Low Levels of Ionizing Radiation: BEIR VII – Phase 2.* Committee to Assess Health Risks from Exposure to Low Levels of Ionizing Radiation, Board on Radiation Effects Research, National Research Council of the National Academies. Washington, DC: National Academies Press, 2006. On the Web at http://darwin.nap.edu/books/030909156X/html.

RCBRA 2010 v.II	U.S. Department of Energy. *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment.* Draft C. (DOE/RL-2007-21) Richland, WA: DOE, December 2010. Links on the Web at http://www.washingtonclosure.com/projects/environmental_protection/mission_completion/project_library. In two parts: text and tables.
WAC MTCA 2007	*Washington Administrative Code*, "Model Toxics Control Act--Cleanup," Chapter 173-340 WAC, Update of October 12, 2007. On the web at http://apps.leg.wa.gov/wac/default.aspx?cite=173-340 and at http://www.ecy.wa.gov/pubs/wac173340.pdf.

ATTACHMENT 2

Supplemental Comments on the
Risk Assessment Report for the 100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment
(DOE/RL-2007-21, Draft A)

December 15, 2010

Confederated Tribes and Bands
of the Yakama Nation

Established by the
Treaty of June 9, 1855

December 15, 2010

Mr. Matthew McCormick
U.S. Department of Energy
Richland Operations Office
P.O. Box 550
Richland, WA 99352

Subject: Supplemental comments on the "Risk Assessment Report for the 100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment" (DOE/RL-2007-21, Draft A)

Dear Mr. McCormick:

Please find attached supplemental comments on DOE's "Risk Assessment Report for the 100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment" (RCBRA; DOE/RL-2007-21, Draft A), released June 2007.

Although we submitted initial comments during the comment period in 2007, we felt that a more thorough technical evaluation of Draft A was necessary to support our upcoming review of the revised Draft B. Since release of Draft B has been postponed for more than a year, we are submitting these supplemental comments for your consideration. Below is a summary of our major recommendations for the RCBRA:

- Adopt MTCA and CERCLA risk levels, and not 15 mrem/year and 100 mrem/year, using age-specific dose conversion factors, to derive dose guides for cleanup. Both of these limits have risks considerably in excess of the largest lifetime cancer risk permitted under CERCLA (10^{-4}). Adopt individual drinking water MCLs for man-made radionuclides corresponding to a 10^{-6} lifetime cancer incidence, and comply with MTCA and CERCLA risk limits.
- Assess the site as a whole to capture the complete risk profile (not just residual contamination at previously remediated waste sites). This requires additional sampling.
- The effect of contamination at other locations, including the 200 Area, on the River Corridor and the irrigation pathway should be assessed and not just that arising from present day conditions.

- Acknowledge the 1855 Treaty with the Yakama Nation as an ARAR for cleanup at Hanford. For instance, the sweathouse exposure pathway should consider temperature and exposure times that represent the practices of all Native Peoples in the Hanford area, additional water intake by children, and release of volatile contaminants from plants used in construction.
- Characterize the transfer of radionuclides and other contaminants to fish by sampling a variety of fish species used by the Yakama Nation and other Native Peoples and do not assume that the contamination of wild animals is the same as that of domestic animals.
- Evaluate doses to the embryo/fetus and to young children from the breast milk pathway to ensure that the most vulnerable members are adequately protected.
- Consider non-dietary exposures to plants, animals, and other natural materials from both production and use, which includes conducting additional sampling to more accurately quantify contaminant transfer factors for such materials.
- A holistic risk assessment, which integrates wellness related to physical, mental, social, and ecologic wellness, is essential when assessing the impacts to Native Peoples. Simply because all of the impacts may not be quantifiable, does not mean they are less important to the process.
- At least begin to assess impacts from combined exposures and potential synergistic effects between and among radiological and chemical contaminants as much as possible, particularly with regard to immune-suppression and metabolism interference from hormonally active agents and heavy metals (particularly uranium).
- Adopt a synergy uncertainty factor of 5 whenever radionuclides are present in combination with potentially significant levels of hormonally active chemicals or heavy metals until a more definitive value can be established.
- Consider future contaminant transport and its impacts on the ecosystem and conduct an ecological risk assessment to complement the human health assessment and identify critical species as part of this assessment.
- Adopt a holistic approach that incorporates interactions between multiple stressors projected over long timescales and over large areas.

The Yakama Nation looks forward to discussing these comments and recommendations.

Sincerely,

Russell Jim, Projects Manager
Yakama Nation ERWM Program

Enclosure

cc: Dennis Faulk, EPA Hanford
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Preliminary Evaluation of the U.S. Department of Energy's *Risk Assessment Report for the 100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment (DOE/RL-2007-21, Draft A)*

Prepared for Ridolfi, Inc. and the Yakama ERWM Program

Brice Smith, Ph.D.

September 28, 2010

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List of Acronyms

ALARA	as low as reasonably achievable
BEIR Committees	The Committee to Assess Health Risks from Exposure to Low Levels of Ionizing Radiation (formerly called the Committee on the Biological Effects of Ionizing Radiation (BEIR)) of the National Research Council of the National Academies
BPA	bisphenol A
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act of 1980

COPC	contaminant of potential concern
COPEC	contaminant of potential ecological concern
CTE	central tendency exposure
CTUIR	Confederated Tribes of the Umatilla Indian Reservation
DCF	dose conversion factor
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
DES	diethylstilbestrol
DOE	U.S. Department of Energy
EPA	U.S. Environmental Protection Agency
FGR 11	Federal Guidance Report 11 containing ingestion and inhalation dose conversion factors for Reference Man
FGR 12	Federal Guidance Report 12 containing external dose rate factors for an average of adult men and women
FGR 13	Federal Guidance Report 13 containing radiation risk factors averaged over age (the 2002 CD Supplement contains risk and dose conversion factors for five age groups ranging from infants to adults)
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
IEER	Institute for Energy and Environmental Research
MCL	maximum contaminant level
MTCA	Model Toxics Control Act
NCRP	National Council on Radiation Protection and Measurements
NOEC	no observable effect concentration
PCB	polychlorinated biphenyl
QAPP	Quality Assurance Project Plan
RCBRA	Risk Assessment Report for the 100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment (DOE/RL-2007-21)
RME	reasonable maximum exposure
UCL	upper confidence level of the mean
Yakama	Confederated Tribes and Bands of the Yakama Nation

Section 1: Introduction

Between its founding in 1943 and the closure of its last production reactor in 1989, the Hanford nuclear site produced plutonium for use by the United States. The site, which covers an area of 586 square miles, is located along the banks of the Columbia River north of the city of Richland, Washington, and approximately 20 miles east of the Yakama Nation Reservation.[1] The Hanford site was divided into different operational areas including the 100 Area where the plutonium production reactors were located, the 200 Area where the plutonium reprocessing facilities were located, and the 300 Area where the uranium fuel processing operations and associated support facilities were located.[2] Following its closure, these three areas, along with the 1100 Area, were added to the National Priorities List identifying them as Superfund cleanup sites under the Comprehensive Environmental Restoration, Compensation, and Liability Act (CERCLA).[3]

The land surrounding the present day Hanford facility is home to four, federally recognized Native American tribes. The present work is being prepared for the Yakama Nation Environmental Restoration and Waste Management Program (ERWM) and, thus, it will focus primarily on the Confederated Tribes and Bands of the Yakama Nation. In addition to the Yakama Nation, the other Native tribes living near the Hanford site include "the Confederated Tribes of the Umatilla Indian Reservation, the Confederated Tribes of the Warm Springs Reservation of Oregon, and the Nez Perce Tribe."[4] Of great significance to our considerations of the risks posed by the past operations at Hanford to these Native Peoples is the fact that, unlike many other Native Peoples in the United States, the lands on which they live are the same lands where their people have lived for thousands of years.[5] This includes, not only the roughly 1.3 million acres of the present Yakama Nation Reservation, but also the nearly 12 million acres of territory (including the lands beneath Hanford), which were ceded by the Yakama in the Treaty of 1855.[6] This is important because the the Yakama Nation retains specific legal rights with respect to the Hanford Site under the 1855 Treaty. Of particular note, the treaty states that

> The exclusive right of taking fish in all the streams, where running through or bordering said reservation, is further secured to said confederated tribes and bands of Indians, as also the right of taking fish at all usual and accustomed places, in common with the citizens of the Territory, and of erecting temporary buildings for curing them; together with the privilege of hunting, gathering roots and berries, and pasturing their horses and cattle upon open and unclaimed land.[7]

Given this history, it is very important that the philosophy underlying the cleanup of Hanford be guided explicitly by the goal of allowing Native Peoples to safely live their lives as they choose with a minimum of land use limitations. This way of thinking will be particularly important

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[1] Ridolfi 2007 p. 3

[2] For a detailed review of the history of the Hanford site and its legacy of chemical and radiological contamination see [Gephart 2003]

[3] Ridolfi 2007 p. 3. The 1100 Area was eventually delisted in 1996, but the other three areas remain on the National Priorities List.

[4] Ridolfi 2007 p. 1

[5] Ridolfi 2007 p. 2

[6] Ridolfi 2007 p. 1 to 2

[7] Treaty with the Yakama 1855

when considering how to incorporate non-quantitative elements into risk assessments such as the spiritual or cultural value of a site. As a guiding principle, all risk assessments should seek to demonstrate compliance with the following, simply stated goal from the *Yakama Nation Exposure Scenario* prepared by Ridolfi Inc. for the Yakama Nation Environmental Restoration and Waste Management Program,

> A safe and healthy subsistence lifestyle should remain an option for the Yakama in their ancestral lands.[8]

The present work is focused on the subset of Hanford sites which were included in the U.S. Department of Energy's *Risk Assessment Report for the 100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment* (DOE/RL-2007-21 Draft A).[9] While excluding areas such as the central plateau region where the far more heavily contaminated 200 Areas are located, the *River Corridor Baseline Risk Assessment* (henceforth the RCBRA) includes the near-surface soils of the 100 and 300 Areas, the riparian zone along the shore of the Columbia, and the waters in the shallow areas near the river banks.[10] We have analyzed "Draft A" of the RCBRA because the final risk assessment had not yet been published by the DOE at the time of writing. Of particular concern with this document is the choice to subdivide the Hanford site, and to look at the risks from each individual area or unit in isolation. This strategy makes it particularly difficult to explore the total, combined risks associated with the contamination on site and is incompatible with the goals of producing a more holistic risk assessment for humans and the rest of the Hanford ecosystem. This overall concern notwithstanding, many of our recommendations relate to specific changes, corrections, or improvements that should be made to the final RCBRA.

As such, we will begin our discussion with a review of the regulatory guidelines and calculation methodologies being used by the Department of Energy for the 100 and 300 Areas at the Hanford site (Section 2). We will find that the principle dose limit used by the DOE to gauge compliance (15 millirem per year for the effective dose) is one that is generally protective of the public and is a reasonable basis for regulatory decision making. However, we will also find that the absence of a separate drinking water standard is a weakness of the report and one that should be corrected in the final risk assessment. We will then turn to an examination of the methodology by which the doses in the RCBRA are calculated. We will find that the use of outdated dose conversion factors prevents the DOE from calculating doses to children and therefore prevents them from being able to ensure that the most vulnerable population is adequately protected. This lack of child specific dose calculations is particularly important given their heightened sensitivity and the fact that the RCBRA makes multiple claims to have considered the impacts on children throughout the document.

We will continue our discussion of the RCBRA with a review of issues relating to the quality and use of contaminant data characterizing the 100 and 300 Areas (Section 3). We will begin by examining the choice in the RCBRA to model only present day conditions at the site and to not to consider the potential impact of future transport of contaminants through the environment. In

[8] Ridolfi 2007 p. 34
[9] DOE 2007
[10] DOE 2007 p. 1-5 to 1-6

particular, the RCBRA focuses exclusively on the residual contamination at previously remediated operational units and, therefore, does not necessarily capture the complete risk profile of the 100 and 300 Areas. Turning to questions about the quality control and other procedures used in collecting the data concerning present day contamination at the site, we will find that there are several lines of evidence relating to the uranium data set that raise potentially important questions about the overall quality of the data used in the RCBRA. We will conclude with recommendations for additional sampling that should be done to help answer the questions we have identified and to fill existing data gaps.

From these considerations, we will turn to a discussion of the risk assessment methodology used in the RCBRA and the challenges presented by seeking to adapt this methodology to the protection of Native Peoples (Section 4). First, we will consider the relatively straightforward issues such as determining how much water a person would consume and how much air they will breathe when practicing a traditional, subsistence lifestyle. Second, we will find that there are a number of unique exposure pathways that are not typically included in risk assessments, but which are amenable to traditional quantitative methodologies such as exposure to waterborne contaminants in sweathouses and the incidental ingestion of plant and animal matter during the manufacture of traditional goods like baskets or digging sticks. Finally, we will find that there are a number of special considerations that must be taken into account that are not easily translated into quantitative risk analysis such as the protection of the spiritual and cultural value of particular plants, animals, and ecosystems. While protection of these values cannot generally be reduced to a simple number or dose limit, they must be treated with the same care and attention as any other element of the risk assessment and, despite their qualitative nature, should be given no less weight in determining cleanup standards.

Related to concerns surrounding the adaption of current risk assessment methodologies to the protection of Native Peoples, the RCBRA evaluates the risks from environmental contaminants under the assumption that each contaminant can be treated in isolation and that risks from multiple combined exposures can be treated simply by adding up the risks associated with each individual contaminant (discussed in Section 5). This assumption, however, ignores the possibility of interactions between the various contaminants that may act to increase the risk relative to the assumption of additivity (typically called a synergistic interaction) or to decrease the risk (typically called an antagonistic interaction). We will find that the kinds of chemical contaminants commonly found on the Hanford site, including hormonally active agents and toxic heavy metals, are of potential concern for synergistic interactions with radiation. Given the complexity of the waste streams at Hanford as well as the fact that contaminants are being transported onsite from locations upriver, it would not be possible, even in principle, to experimentally investigate all of the possible combinations and permutations of exposures to determine whether synergistic or other kinds of interactions will occur. Thus, our recommendation for the interim period will be to adopt synergistic "uncertainty factors" when applying regulatory limits to combined exposures. These uncertainty factors would lower the allowable exposure to account for potential increases in the risks due to interactions among the chemical and radiological contaminants.

Finally, we will conclude the present work with a discussion of the ecological risk assessment included in the RCBRA (Section 6). While this section will focus specifically on issues raised by the contamination at the Hanford site, many of the concerns and recommendations have broader implications for how ecosystem impact assessments should be conducted in general for sites with long-lived and highly complex wastes. We will begin by assessing the hybrid ecological risk assessment framework put forth in the RCBRA and focus in particular on its use of the so-called no observable effects level as one measure for evaluating the impacts of existing contamination on plants and animals as well as its lack of considerations of future contaminant transport and its impacts on the ecosystems of the river corridor. We will then turn to a discussion of issues surrounding both data gaps and data quality in the ecological surveys, and finally, we will turn to the need for the revised RCBRA to include a more holistic approach to ecosystem risk assessment incorporating interactions between multiple environmental stressors projected over long timescales and over large areas.

Given the enormous complexity of the tasks involved with ecological risk assessment we will conclude this report with a proposal for a research plan outlining how to begin moving towards a methodology capable of ensuring adequate protection of the environment consistent with the qualitative goals outlined of ensuring protection of the natural and cultural resources of the Yakama and other Native Peoples. Given the need for long-term data collection and field observation, as well as the enormous complexity and interconnectedness of the effort we are recommending, it is likely that fully implementing this research plan will take considerable effort and time. However, there will, of course, be many interim steps along the way whose findings will provide significant information of use to guiding remediation and cleanup efforts that are protective of both humans and the environment.

Below is a summary of recommendations to apply to the final RCBRA, and for assessing risk at Hanford in general that is adequately protective of the Yakama Nation, other Tribal people with similar ways of life and the general public.

1. **Regulatory compliance:**
   - Replace the use of a 100 mrem/year or a 15 mrem/year dose limit as reference values for cleanup, since they imply lifetime cancer risks considerably greater than the upper CERCLA risk limit of 10^{-4} by a more stringent approach in compliance with CERCLA and MTCA.
   - Adopt individual drinking water MCLs for man-made radionuclides corresponding to a 10^{-6} lifetime cancer incidence.
   - Adopt age-specific dose conversion factors published by EPA and those for the embryo/fetus and breast-fed infant published by ICRP.

2. **Data quality and contaminant transport modeling:**
   - Assess site as a whole to capture the complete risk profile (not just residual contamination at previously remediated waste sites). This requires additional sampling.
   - Assess exposure potential from contamination that is currently subsurface via erosion and groundwater transport pathways for instance, and not just exposure potential from surface contamination.
   - Assess potential impact of future transport of contaminants through the environment, including from the 200 Area into the River Corridor and the irrigation pathway, not just that arising from present day conditions.

- Implement longer-term and more widely distributed sampling of groundwater to characterize the existing contamination.
- Select a reference area that can be definitively shown to be uncontaminated with Hanford contaminants. The use of a "reference" area that is dominated by man-made radionuclides present at Hanford is not appropriate and gives a misleading picture of Hanford-related risks.

3. **Tribal Exposure Scenario implementation:**
   - Acknowledge the 1855 Treaty with the Yakama Nation as an ARAR; provides certain legal rights with respect to Hanford.
   - Characterize the transfer of radionuclides and other contaminants to fish by sampling a variety of fish species used by the Yakama Nation and other Native Peoples.
   - Do not assume that the contamination of wild animals is the same as that of domestic animals; a detailed assessment should be made of relevant differences in both biology and behavior of such species (including uptake factors).
   - Until a value based on data specific to the kind of lifestyle pursued by the Yakama can be obtained a more conservative estimate for the Reasonable Maximum Exposure (RME) level for incidental soil ingestion for children should be in the range of at least 1,000 mg/d, excluding pica children, until a suitable Yakama-Nation-specific value can be determined.
   - The DOE should also include pica children in its risk assessment framework. The RME for soil ingestion for *children* should include a contribution from acute exposures consisting of at least 30 to 40 g/year in addition to the routine exposures.
   - The shielding factor of 0.7 used by RESRAD should be adopted as a more protective estimate.
   - The DOE should evaluate doses to the embryo/fetus and to young children from the breast milk pathway to ensure that the most vulnerable members are adequately protected.
   - The sweathouse exposure pathway should consider temperature and exposure times that represent the practices of all Native Peoples in the Hanford area, additional water intake by children, and release of volatile contaminants from plants used in construction.
   - The DOE should consider non-dietary exposures to plants, animals, and other natural materials from both production and use, which includes conducting additional sampling to more accurately quantify contaminant transfer factors for such materials.
   - A holistic risk assessment, which integrates wellness related to physical, mental, social, and ecologic wellness, is essential when assessing the impacts to Native Peoples. Simply because all of the impacts may not be quantifiable, it does not mean they are less important to the process.

4. **Combine chemical and radiological impacts:**
   - The DOE should at least make a beginning for assessing impacts from combined exposures and potential synergistic effects between and among radiological and chemical contaminants as much as possible, particularly with regard to immune-suppression and metabolism interference from hormonally active agents and heavy metals (particularly uranium).

- Adopt a synergy uncertainty factor of 5 whenever radionuclides are present in combination with potentially significant levels of hormonally active chemicals or heavy metals until a more definitive value can be established.
- Future research should be conducted to build more robust and integrated means of dealing with combined exposures, particularly examining the kinds of waste mixtures prevalent at Hanford, and should focus on the most vulnerable segment of the population.

5. **Ecological risk assessment:**
   - Consider future contaminant transport and its impacts on the ecosystem and conduct an ecological risk assessment to complement the human health assessment.
   - Adopt a holistic approach that incorporates interactions between multiple stressors projected over long timescales and over large areas. A holistic approach includes:
     - Evaluation of the relative biological effectiveness of alpha particles, the concentration factors for the organs of critical species and radionuclides, and the uptake from the environment including food web bioaccumulation.
     - Evaluation of the radiation effects on individual members of species present in the environment, including mortality, morbidity, reduced reproductive capacity, and chromosomal damage in isolation and with other environmental stressors.
     - Doing ecosystem modeling of the affected environs, including determination of effective energy, water, and nutrient flow models of the key ecosystems (taking into account that damage may have already occurred to these areas prior to the creation and validation of these models).
   - Connect the above efforts to identify the critical species from an ecosystem point of view, including species that are culturally or economically important to the Yakama, and to identify what type and severity of effects may be expected to occur at the individual, population, community, and ecosystem levels as a result of the radioactive and chemical contaminants in the Hanford environment (i.e., develop an integrated assessment model).

Section 2: Issues Regarding Regulatory Compliance

We will begin our discussion of the RCBRA with a review of the regulatory guidelines and calculation methodologies being used by the Department of Energy for the 100 and 300 Areas at the Hanford site. We will find that the principle dose limit used by the DOE to gauge compliance (15 millirem per year for the effective dose) is one that is generally protective of the public and is a reasonable basis for regulatory decision making. However, we will also find that the absence of a separate drinking water standard is a weakness of the report and one that should be corrected in the final risk assessment. We will then turn to an examination of the methodology by which the doses in the RCBRA are calculated. We will find that the use of older dose conversion factors first published in the 1980s prevents the DOE from calculating doses to children and therefore prevents it from being able to ensure that the most vulnerable population is adequately protected. This lack of child specific dose calculations is particularly important given the heightened sensitivity of children and the fact that the RCBRA makes multiple claims to have considered the impacts on children throughout the document.

Section 2.1 – Radiation Dose Limits

As mentioned in the introduction to this section, the DOE has adopted a generally protective effective dose limit of 15 millirem per year for the 100 and 300 Areas. As noted by the RCBRA

> Radiation doses for each exposure route and radionuclide are summed to calculate the total annual dose to an individual. The acceptability of a calculated annual dose is evaluated for human receptors in the RCBRA relative to a threshold dose limit of 15 mrem/year. The DOE has published health and safety orders of which DOE Order 5400.5, *Radiation Protection of the Public and the Environment*, is most pertinent to the identification of a radiation dose threshold. DOE Order 5400.5 requires the reduction of all DOE-source radiation doses to a level as low as reasonably achievable (ALARA) below a primary dose limit of 100 mrem/year above background. CERCLA authorizes the EPA to regulate hazardous substances, including radionuclides, released into the environment. EPA has published guidelines for establishing cleanup levels for radionuclides under CERCLA that state that 15 mrem/year above background levels should "generally be the maximum dose limit for humans".[11]

This DOE approach can be seen from two angles – the dose limit adopted and CERCLA-related limits. In regard to dose, the use of a 15 millirem per year effective dose limit is much better than a limit of 100 millirem per year. However, 15 millirem per year is much higher than the CERLCA lifetime risk range for cancer of 10^{-4} to 10^{-6}. Specifically, it is about six times higher than the highest CERCLA risk in that range (10^{-4}). Further, Washington State's law (MTCA) limits combined lifetime cancer risks to 10^{-5}, when there is more than one carcinogen present. This issue is covered in some detail in Section C of Attachment 3 of the Yakama Nation's comments on the Draft Tanks Closure and Waste Management Environmental Impact Statement.

[11] DOE 2007 p. 5-57

[12] We will not go into them here but note as a general matter that cleanup of the 100/300 Area must also conform to MTCA requirements[13] and to all Applicable or Relevant and Appropriate Requirements (ARARs). One of these relates to EPA's drinking water standards for radionuclides. Specifically, the Environmental Protection Agency's National Primary Drinking Water Regulations give the following maximum contaminant levels (MCLs) for radionuclides in water

> (c) *MCL for gross alpha particle activity (excluding radon and uranium).* The maximum contaminant level for gross alpha particle activity (including radium-226 but excluding radon and uranium) is 15 pCi/L.
> (d) *MCL for beta particle and photon radioactivity.* (1) The average annual concentration of beta particle and photon radioactivity from man-made radionuclides in drinking water must not produce an annual dose equivalent to the total body or any internal organ greater than 4 millirem/year (mrem/year).
> ...
> (e) *MCL for uranium.* The maximum contaminant level for uranium is 30 µg/L [i.e., 30 micrograms per liter].[14]

While the drinking water pathway was not a primary focus of the RCBRA, compliance with drinking water limits is necessary for meeting all regulatory requirements as well as for conducting risk assessments that are used to evaluate cleanup goals. This will ensure that appropriate criteria in each particular case will be used to guide the remediation efforts. It is important to note that EPA's drinking water regulations limit the dose to the whole body or to the most exposed organ for the vast majority of radionuclides; the exceptions are those, like strontium-90 or long-lived alpha emitters like plutonium-239 and other similar transuranics, for which the EPA directly specifies MCLs. In the case of the RCBRA, the need to include doses from drinking water is particularly important given the higher than average ingestion of water by the Yakama and other Native peoples due to their active, outdoor life-style and their routine use of sweathouses (see Section 4.2). These drinking water limits generally correspond to a lifetime risk of 10^{-4}.

However, the RCBRA should take into account the fact that drinking water is just one pathway at a site where the overall CERCLA limits are 10^{-4} to 10^{-6} for all pathways and all contaminants. Further, there is a 10^{-6} lifetime cancer risk requirement in MTCA for individual carcinogens.[15] Finally, the DOE itself agreed to surface water limits corresponding to a 10^{-6} risk limit for plutonium and tritium during remediation of the Rocky Flats site. There is no reason for the people of Washington State or the Yakama Nation to be treated with a far more lax standard for Hanford remediation. Thus, in order to be protective of public health, the DOE should proactively adopt individual drinking water MCLs for man-made radionuclides, including transuranic alpha emitters, strontium-90 and tritium corresponding to a 10^{-6} lifetime cancer incidence risk level as part of the cleanup standards for the 100 and 300 Areas.

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[12] Yakama Nation 2010, Attachment 3, Section C.
[13] WAC MTCA
[14] 40 CFR 141.66 2009
[15] Yakama Nation 2010, Attachment 3, Section C, Table 1.

We will now turn from the dose limits to a discussion of the methodology by which the doses are calculated in the RCBRA. In particular, we will focus on the decision by the Department of Energy to exclude consideration of doses to children and the embryo fetus from the RCBRA.

Section 2.2 – Doses to Children and the Embryo/Fetus

In conducting risk assessments such as the RCBRA, one critical step in the process is to predict the radiation dose that could be received by a person who gained access to the site. Once the potential pathways by which the person could be exposed to the contamination (i.e. inhalation of gaseous radionuclides or contaminated dust, ingestion of contaminated food, water, or soil, exposure to external gamma emitters, etc.) are determined, the exposures must be converted into the dose received. In calculating how big a dose an individual will receive from the radionuclides inhaled or ingested, care must be taken to consider such factors as where, if anywhere, the radionuclides may concentrate, how long they will remain in the body, and what kinds of damage they will do before being excreted or exhaled. Currently, all of this information is summarized in a single constant called the dose conversion factor (DCF). The DCF relates the total lifetime dose received by the individual (measured in rem) to the amount of the radionuclide that is ingested or inhaled (measured in curies). A dose conversion factor, therefore, has the units of rem per curie, or its equivalent in other unit systems.[16] As would be expected, given the differences in such things as body size and metabolism, the dose conversion factors for children are, in general, different from those for an adult. For the doses received from external radiation, care must be taken to consider such factors as the energy of the radiation and the size of the individual and their internal organs and thus, as with the dose conversion factors for ingested or inhaled radionuclides, the dose due to external radiation will also, in general, be different for children than adults.

Throughout the RCBRA, numerous statements are made that imply that children are explicitly included in the risk assessment for all scenarios except the "Resident Monument Worker" and the "Industrial/Commercial Worker" scenarios.[17] For example, in the introduction to the seven exposure scenarios considered by the DOE, the RCBRA notes the following

> Future Rural-Residential Scenario. The potentially exposed population for this exposure scenario includes adults and children.[18]
>
> Casual User. The potentially exposed population for this exposure scenario includes adults and children.[19]
>
> Avid Wild Game Hunter. The potentially exposed population engaged in hunting for this exposure scenario includes adults and older children.... The potentially

[16] For example, in the SI unit system the unit of dose is the Sievert (Sv) while the unit of radioactivity is the Becquerel (Bq) giving the dose conversion factor the units of Sievert per Becquerel (Sv/Bq).
[17] See, for example, [DOE 2007 p. 2-26 to 2-29, 5-29, 5-38, 5-39, 5-83, and 5-183]
[18] DOE 2007 p. 2-26
[19] DOE 2007 p. 2-28

> exposed population for ingestion of wild game is not restricted to older children and adults who actively hunt but may also include younger children at home.[20]
>
> Avid Angler. The potentially exposed population engaged in hunting for this exposure scenario includes adults and older children.... As with the Avid Wild Game Hunter scenario, the potentially exposed population for ingestion of fish is not restricted to older children and adults who actively fish but may also include younger children at home.[21]
>
> CTUIR [Confederated Tribes of the Umatilla Indian Reservation] Scenario. Exposure routes and receptors have been defined by the CTUIR for a traditional subsistence lifestyle scenario. A complete lifetime is reflected in this scenario, from infancy through old age.[22]

However, despite these explicit references to the inclusion of children, the RCBRA uses internal and external dose factors from Federal Guidance Reports 11 and 12 published by the EPA in 1988 and 1993 respectively.[23] This is significant because, as noted in the RCBRA itself,

> The DCFs provided in *Federal Guidance Report No. 11* (EPA 520/1-88-020) and *Federal Guidance Report No. 12* (EPA-402-R-93-081) **do not discriminate among various age groups** of receptors in the manner of the radionuclide CSFs [cancer slope factors] from *Federal Guidance Report No. 13* (EPA 402/R-99/001).[24]

In fact, the dose conversion factors in FGR 11 were developed for a model individual known as "Reference Man" which was explicitly defined by the International Commission on Radiological Protection (ICRP) in 1975 as an adult male. More specifically, the ICRP stated that

> Reference man is defined as being between 20-30 years of age, weighing 70 kg [154 pounds], is 170 cm [5 feet 7 inches] in height, and lives in a climate with an average temperature of from10° to 20°C [50 °F to 68 °F]. He is a Caucasian and is Western European or North American in habitat and custom.[25]

While Reference Man remains the basis for many regulatory standards (as noted by the DOE in the RCBRA),[26] much work has been done since the mid-1970s to provide the basis for including children and the embryo/fetus in radiation dose assessments. For example, between 1990 and 1996, the International Commission on Radiological Protection published a series of dose conversion factors for six specific age ranges. These ranges were;

> 0 to 1 years old,

[20] DOE 2007 p. 2-28
[21] DOE 2007 p. 2-28
[22] DOE 2007 p. 2-29
[23] DOE 2007 p. 5-44
[24] DOE 2007 p. 5-47 (bold emphasis added)
[25] ICRP 23 p. 4
[26] DOE 2007 p. 5-47. For a discussion of Reference Man and current regulatory standards see [Makhijani 2009]

1 to 2 years old,
2 to 7 years old,
7 to 12 years old,
12 to 17 years old, and
more than 17 years old.[27]

These age ranges are comparable to those used by the EPA in its 2002 update to Federal Guidance Report No. 13 which included both dose conversion factors and radiation risk factors (called cancer slope factors in RCBRA) for various age groups, including children. Dose conversion factors are provided for infants as well.[28] In addition, the ICRP has also published a collection of dose conversion factors for the embryo/fetus in 2002 and a collection of dose conversion factors for breast fed infants in 2005 allowing doses to these sensitive groups to be calculated as well.[29]

The case of external radiation exposure of children is more complicated, however, given the lack of detailed guidance from either the ICRP or the EPA. The most recent assessment of external exposures, FGR 12, calculated doses for the average of a 59 kilogram, 160 centimeter tall (130 pound, 5 foot 3 inch) woman and a 70 kilogram, 170 centimeter tall (154 pound, 5 foot 7 inch) man.[30] External doses to children would be expected to be higher than those for adults given the same level of contamination. This is due to the smaller size of children, which provides their internal organs less shielding, and due to the fact that they are closer to the ground and, thus, are in closer proximity to the source of the radiation. Taking these factors into account in detail, however, is quite involved since their importance depends upon the energy of the radiation emitted by the contaminants, and thus on the characteristics of the radionuclides present. A rough estimate of the importance of the effects for external radiation can be gained, however, from the recommendations of the National Council on Radiation Protection and Measurements (NCRP). In 1999, the NCRP recommended that, for children up to at least 12 years of age, the estimated external dose for adults should be increased by 20 to 40 percent with a best estimate being a 30 percent increase.[31]

Given the higher dose factors and the higher risk per unit of dose for children and the embryo/fetus, it is important that they be included explicitly in any risk assessment.[32] Interestingly, while the RCBRA explicitly rejects the use of age specific dose conversion factors from FGR 13, it does embrace the use of its updated radiation risk factors. Specifically, the authors note that

> Radionuclide slope factors published by EPA are preferred to the use of risk factors applied as multipliers to calculated radiation dose equivalents. Although such dose equivalents are applicable for comparison to dose-based radiation protection standards, they were derived for application to adults in a workplace

[27] ICRP 56, ICRP 67, ICRP 69, ICRP 71, and ICRP 72. For simplicity the ICRP referred to these age ranges as 3 month old, 1 year old, 5 year old, 10 year old, 15 year old, and Adult. [ICRP 72 p. 11]
[28] FGR 13 CD Supplement
[29] ICRP 88 and ICRP 95
[30] FGR 12 p. 40
[31] NCRP 129 pp. 56-57
[32] For a further discussion see [Makhijani, Smith, and Thorne 2006 p. 35 to 45]

> setting. More recent radionuclide slope factors from *Federal Guidance Report No. 13* (EPA 402/R-99/001), by contrast, were derived to pertain to the general U.S. population and are therefore applicable for use in estimating cancer risks for a general population composed of adults and children. *Federal Guidance Report 13* slope factors are derived using age- and gender-specific values for intake and radionuclide dosimetry.[33]

In other words, the authors recognize that the use of dose conversion factors and risk multipliers for Reference Man are not adequately representative of the general public and support the use of the age-averaged risk factors from FGR 13. In addition, the RCBRA goes on to explicitly note that the use of Reference Man dose factors for the estimation of doses to children potentially underestimates the true impact of the residual contamination.[34] For example, in discussing the uncertainties in their analysis the authors of the RCBRA note that

> With respect to radionuclides, uncertainties may relate to the estimation of radiation dose as well as to the assessment of carcinogenic risk associated with any particular dose. One of the primary distinctions between the toxicity criteria used in this assessment to quantify radiation dose and radionuclide cancer risk is that the former pertain only to adults whereas the latter are applicable for use in estimating cancer risks for a general population composed of adults and children. Therefore, there is less confidence in the estimates of radiation dose for scenarios that include child receptors than in scenarios related strictly to adult exposures. Because infants and young children have proportionally larger organ masses relative to their body size, organ-specific radiation doses may be underestimated for these receptors.[35]

It is important to note here, however, that it is not always young children that will have the highest doses. This means that the RCBRAs typical definition of child as being between one and six years old (or seven to twelve years old for the Avid Angler and Avid Hunter scenarios) may miss the most sensitive time for some contaminants.. For example, the dose conversion factors for ingested strontium-90 published by the EPA in FGR 13 show that the dose to a 15 year old is more than 50 percent higher than that to a child between 1 and 6 years old and more than 90 percent higher than the Reference Man dose used in the RCBRA.[36]

A review of the scientific evidence supporting the need for radiation protection standards and risk assessments to protect the most vulnerable members of society is discussed at length in the IEER report *Science for the Vulnerable: Setting Radiation and Multiple Exposure Environmental Health Standards to Protect Those Most at Risk.*[37] While a review of this report's findings is beyond the scope of the present work, we note that the central recommendation of this work of relevance to the RCBRA is that

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[33] DOE 2007 p. 5-46 to 5-47
[34] DOE 2007 p. 5-44, 5-47, and 5-92 to 5-93
[35] DOE 2007 p. 5-103
[36] FGR 13 CD Supplement and DOE 2007 p. 5-29, 5-83, and 5-234
[37] Makhijani, Smith, and Thorne 2006

> "Reference Man" is clearly an outdated concept that cannot fulfill the needs of environmental health protection in the context of the variety of pollutants and populations to be protected as well as the kinds of health outcomes that are possible. ***One central principle of environmental health protection must be to protect those most at risk for any given pollutant or combination of pollutants.***[38]

Thus, as with our previous recommendation to adopt more stringent drinking water limits for man-made radionuclides in order to be protective of public health, the DOE should proactively adopt the age specific dose conversion factor published by the EPA as well as the dose conversion factors for the embryo/fetus and breast fed infant published by the ICRP.

[38] Makhijani, Smith, and Thorne 2006 p. 79 (emphasis in the original)

Section 3: Issues Regarding Data Quality and Lack of Contaminant Transport Modeling

We will continue our discussion of the RCBRA in this section with a review of issues relating to the quality and use of contaminant data characterizing the 100 and 300 Areas. We will begin by examining the choice in the RCBRA to model only present day conditions at the site and to not consider the potential impact of future transport of those contaminants through the environment. In particular, the RCBRA focuses exclusively on the residual contamination at previously remediated operational units and, therefore, does not necessarily capture the complete risk profile of the 100 and 300 Areas. Turning to questions about the quality control and other procedures used in collecting the data concerning present day contamination at the site, we will find that there are several lines of evidence relating to the uranium data set that raise potentially important questions about the overall quality of the data used in the RCBRA. We will conclude with recommendations for additional sampling that should be done to help answer the questions we have identified and to fill in the other existing data gaps.

Section 3.1 – The Need to Consider Contaminant Transport

The contamination of the 100 and 300 Areas is due to a wide range of historical activities. The principle historical activities of concern included (1) the discharge of liquid waste to retention basins and shallow land disposal facilities such as cribs, ponds, trenches, and French drains; (2) the burning or burial of various kinds of solid wastes including combustible materials; and (3) the discharge of contaminated water into the Columbia River[39] or groundwater. Many of these waste streams began in the 1940s and continued in various forms for decades.[40] Despite the vast array of waste forms and disposal techniques, the long time scale over which some of the contaminants will remain hazardous, and the complex geology and hydrogeology of the Hanford site, the RCBRA makes no effort to project the future contamination profile of the site as a result of contaminant transport. For example, as noted in the executive summary

> The RCBRA is designed to characterize the current and potential threats to human health and the environment that may be posed by residual, post-remediation contaminants under current and a range of hypothetical future site uses. ***This risk assessment evaluates sites as they are now, after cleanup has been completed and approved*** through Waste Information Data System reclassification process, in which the cleanup verification packages (CVP) are generated.[41]

The RCBRA goes on to repeat that the results of the risk assessment it presents are "calculated using ***present-day surface soil COPC [contaminant of potential concern] concentrations*** across the upland portions of the 100 and 300 Areas."[42] Specifically, the RCBRA is based on the

[39] The Yakama name for the Columbia River is Nch'i-wa'na (Big River). For consistency with the RCBRA, however, we will retain the use of the name Columbia River.
[40] Gephart 2003 p. 5.21-5.24, 5.26 to 5.33 and DOE 2007 p. 2-3 to 2-4 and 2-6 to 2-7
[41] DOE 2007 p. ES-1 (emphasis added)
[42] DOE 2007 p. ES-10 (emphasis added)

examination of data collected between October 2005 and December 2006 for 163 sampling sites, 90 percent of which are in the 100 Area, plus 64 ground water wells.[43]

The lack of consideration of contaminant transport has many implications for the validity of the dose projections presented in the RCBRA and is one of the primary weaknesses we have identified in the DOE approach. To begin with, the migration of waste through the subsurface is acknowledged to be an important factor in the historical spread of contamination on site. For example, the RCBRA notes that

> Because groundwater underlying the 100 Area and 300 Area moves towards the Columbia River, it is important to consider the contaminants that have migrated via groundwater to the riparian and near-shore river zones.[44]

However, despite its acknowledgement that past migration "is important to consider", the authors do not extend this recognition to future migration of contaminants despite the fact that the groundwater is only 10 to 15 feet below the surface in some parts of the 100 and 300 Areas.[45] In fact, some fast moving contaminants released into the soil in the 100 and 300 Areas are known to have migrated through the soil in and to the Columbia River within a matter of just a few hours to a few days in some locations within the river corridor.[46]

An additional concern with respect to the lack of consideration of contaminant transport in the RCBRA arises due to the importance of agricultural pathways in setting the ultimate cleanup standards for Hanford (see Section 4). Unlike a risk assessment conducted with programs such as RESRAD which account for the dynamic feedback between contaminants in the water used to irrigate crops or water livestock and the subsequent contamination of those plants and animals, no such pathways were included in the RCBRA scenarios.[47] Without this kind of quantitative consideration of the impacts of irrigation, the RCBRA concludes that "the pathway from groundwater to terrestrial receptors is largely incomplete" and that, as a result, "(with the exception of aquatic foodstuffs) soil is the primary environmental medium harboring contaminants that may migrate to these foodstuffs."[48] The only consideration given to the irrigation pathway in the RCBRA was a qualitative discussion of the uncertainties associated with using contaminated water for agricultural rather than domestic purposes. However, like the rest of the risk assessment, this qualitative discussion only considered present contaminant levels and not those that may be reached in the future due to further contaminant migration.[49]

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[43] DOE 2007 p. ES-3 and 4-1
[44] DOE 2007 p. 2-38
[45] Gephart 2003 p. 5.35
[46] Gephart 2003 p. 5.37
[47] Yu et al. 2001 p. 1-4 and 2-3 to 2-5 and DOE 2007 p. 5-20. RESRAD (short for RESidual RADioactivity) is a computer based simulation program, developed by the Environmental Assessment Division of Argonne National Laboratory and first released in 1989. It is designed to conduct risk assessments for radioactively contaminated sites and has been adopted for use in demonstrating regulatory compliance by a number of federal agencies. [Yu et al. 2001 p. xi and xvii] For more details on RESRAD and how it is used to conduct dose assessments see [Smith 2009].
[48] DOE 2007 p. 2-32 and 5-3
[49] DOE 2007 p. 5-20

In addition to the movement of contaminants through the water, the transport of chemicals and radionuclides via erosion is also likely to be of importance for facilities like Hanford which are located in arid regions. This potential importance arises from two principal considerations. First, erosion of the surface soil can uncover contaminated regions initially underground, resulting in increased exposures through pathways like soil ingestion and the consumption of contaminated plants. Currently, the RCBRA assumes that the only way contaminated soil from the subsurface could come into long-term contact with humans is through the intentional disruption of the soil such as by digging a basement.[50] Thus, the inclusion of erosion would provide additional possibilities for exposure and thus for increased impacts on humans and the environment from subsurface contaminants.

The second mechanism by which erosion may cause important effects is through its role in the migration of contamination into the surface water. For example, at the Los Alamos National Laboratory in New Mexico, one of the most important transport mechanisms for plutonium migrating towards the Rio Grande River is through the erosion of contaminated soil caused by the flooding of the canyons after each rain. For example, William Graf noted in his 1994 book *Plutonium and the Rio Grande* that

> Sediments in Los Alamos Canyon impregnated with plutonium move down the canyon system in a stepwise fashion, with each step taken as a few meters to a few kilometers during each flood event. Each flood stores the sediments as channel or flood-plain deposits, and each subsequent flood remobilizes them until they reach the Rio Grande.[51]

Graf went on to conclude that

> Surface water is the main driving force behind the movement of plutonium through the surface system of northern New Mexico. The energy represented by the water is expended partly by the moving sediments and associated plutonium from one place to another and partly by the mixing and dispersion of contaminants.[52]

To give a sense of the importance of the transport of plutonium via erosion at Los Alamos, Graf noted that "[i]n just one storm at Los Alamos, surface water runoff transported 1 to 2 percent of the entire sediment-bound inventory of plutonium."[53] While Hanford gets somewhat less than half as much rain per year as Los Alamos, the amount of plutonium dumped at Hanford is far larger. Given that the Columbia River runs right through the site and is close to highly contaminated zones in the 100 and 300 Areas, the potential for erosion to transport radionuclides and other contaminants into the riparian zone as well as into the river itself needs to be considered as part of this risk assessment. In that context, it is important to note that programs

[50] DOE 2007 p. 2-24 and 5-7
[51] Graf 1994 p. 127
[52] Graf 1994 p. 235
[53] Graf 1994 p. 10

like RESRAD do not include this transport pathway either and, thus, could not be used to incorporate this pathway in the revised RCBRA.[54]

Adding dramatically to the complexity of the challenge facing any effort to account for the future migration of contaminants in the 100 and 300 Areas is the question of what will happen to the far higher activities of long-lived radionuclides migrating towards the Columbia River from the 200 Area. As of 2003, it was estimated that the groundwater under an area roughly 150 square miles had been contaminated by the activities at Hanford.[55] For comparison, this is an area nearly two and a half times the size of all of Washington, DC. Of this area, it has been estimated that roughly two-thirds was atop groundwater that has been contaminated at a level that would be unacceptable for use as drinking water under current regulatory standards.[56]

While a discussion of the contamination issues in the 200 Area is beyond the scope of this report, it is important to note that the first detection of waste migrating beyond the 200 Area through the groundwater occurred more than half a century ago and that by 1965 the leading edge of a contaminant plume consisting of tritium had already reached the Columbia River. Within another decade, tritium concentrations in groundwater above the drinking water standard had migrated from the 200 East Area to the river.[57] Further, the DOE's own estimates of future migration of contamination in the Draft EIS for tank closure indicate that some contaminants in the river corridor will be present at levels exceeding allowable MCLs by hundreds of times for hundreds of years. For instance, plutonium is projected to peak in the Columbia River nearshore in the year 2983 at 283 times the present drinking water MCL of 15 pCi/liter.[58]

The movement of such large contaminants volumes from the 200 Area into the river corridor greatly complicates the job of performing a risk assessment for the 100 and 300 Areas *since the risks posed by the contaminants in their soil and water cannot be decoupled from the risks posed by the contaminants leaking out of the 200 Area.* This complication was noted several times throughout the RCBRA. For example

> Groundwater contaminant plumes from 200 Area waste sites have migrated southeast toward the 300 Area. These plumes were driven east and southeast by the natural groundwater gradient across the Hanford Site and the large-volume discharges of cooling water to ponds and ditches in the Central Plateau.[59]

> Groundwater contamination is known to occur within the water table underlying the Hanford Site. Key contaminant plumes affecting groundwater in the 100 and 300 Areas include hexavalent chromium at the 100-D, 100-H, and 100-K Areas, strontium-90 at the 100-N Area, and uranium at the 300 Area. Additional contaminants originating from the Central Plateau are migrating through the aquifer, towards the Columbia River.[60]

[54] Yu et al. 2001 p. E-26
[55] Gephart 2003 p. 5.34
[56] Gephart 2003 p. 5.34
[57] Gephart 2003 p. 3.9, 5.35, and 5.37
[58] DOE 2009 Vol. 2, p. U-3
[59] DOE 2007 p. 2-6
[60] DOE 2007 p. 2-10

> Groundwater well data may in some cases reflect contamination from a particular liquid waste disposal site, and other times reflect contamination from multiple 100 Area, 300 Area, and/or 200 Area sources.[61]
>
> It is also significant that residual groundwater contamination in the investigation areas has been impacted by releases outside of these areas, and that groundwater contaminant concentrations are dynamic and have not necessarily peaked for all combinations of contaminants and locations in the 100 Area and 300 Area.[62]

As a result, the RCBRA actually chooses not to sum the doses from drinking water with those from other exposure pathways such as soil ingestion or the ingestion of contaminated plants "because there is no tractable way at this time to correlate existing groundwater data with potential future groundwater exposure concentrations for the individual waste sites."[63] Thus, not only is there no application of a sub-limit for the drinking water pathway as discussed in Section 2.1, the drinking water dose is not even included by the RCBRA in the total dose compared against the 15 millirem per year limit for each operational unit. The fact that drinking water doses cannot be easily disaggregated should not serve as a rationale for removing them from considerations of the effective dose and should instead serve as a powerful motivating factor for considering the risks posed by the Hanford site as a whole and for seeking to determine the peak dose received regardless of where the contributing contaminants began their journey.

Before turning to the question of data quality in the sampling of the 100 and 300 Areas, we will conclude this discussion of contaminant transport with a brief word of caution regarding the modeling of radionuclide migration in a geologic environment as complicated as Hanford. In a previous review of the transport of radium and plutonium through the environment, IEER noted the difficulties that the DOE has had in the past regarding the prediction of contaminant transport. Our review of that history is worth quoting at length here

> Given the inhomogeneous and highly complex chemical, biological, and physical properties of soil, rocks, groundwater, and surface water, it has been found that predicting the mobility of radionuclides is far from simple. For example, when many of the sites within the U.S. nuclear weapons complex were founded, it was believed that their arid climate and thick unsaturated zones would help to protect the groundwater beneath the sites for hundreds to thousands of years. However, investigations of contaminant mobility at these sites have revealed these early assumptions to be in substantial error. For example, the travel time estimated by the DOE for radionuclides to reach the Snake River aquifer under the Idaho National Engineering and Environmental Laboratory (now the Idaho National Laboratory) has fallen from tens of thousands of years in their predictions from the mid-1960s to just a few tens of years today. This thousand fold increase in the contaminant's estimated mobility was prompted by the discovery that plutonium

[61] DOE 2007 p. 5-3
[62] DOE 2007 p. 5-4
[63] DOE 2007 p. 5-59

had already reached the groundwater 200 meters beneath the Radioactive Waste Management Complex.

A second example where early predictions of limited contaminant transport were later disproved by facts on the ground is the case of tritium at the waste disposal facility in Beatty, Nevada. Despite the fact that it was originally predicted that no tritium would migrate from the disposal area at all, tritium has already been found 48 meters below the site. A third example of this kind of failure was the DOE's prediction that the low rain fall and 90 meter thick unsaturated zone below the waste disposal areas at Hanford in Washington State would prevent any contamination from reaching the groundwater. Unfortunately, some fission products and other radionuclides that have leaked from high-level waste tanks have already reached the water table below Hanford, in some areas, after just 60 years. Finally, a fourth example of the failure of past DOE predictions can be found in its analysis of plutonium migration from the underground nuclear weapons tests conducted at the Nevada Test Site (NTS).

In all of these examples, the conceptual models relied upon by the DOE for decision making failed to accurately predict contaminant transport, and it was only after the discovery of radionuclides spreading into the environment that these models were revised. The failure of these transport models was due in large part to the failure to adequately characterize the systems. As summarized by the National Research Council of the U.S. National Academies of Science

> Simply stated, a transport model is only as good as the conceptualizations of the properties and processes that govern radionuclide transport on which it is based.[64]

In real systems there may be chemical and biological processes that will occur which effect the mobility of contaminants. These processes may themselves be changing over space and time which would further complicate efforts to predict radionuclide transport. There may also be more pathways by which the radionuclides can move than originally expected. For example, plutonium and other transuranics can adsorb onto very small particles known as colloids. These particles are so small that they can move with the ground or surface water thus mobilizing contaminants that would otherwise have been considered to be insoluble and tightly held by the soil or sediments. In other systems flooding or surface erosion may dominate the transport of some radionuclides. Finally, the transport model itself might be adequate, but the information on what parameters to input may not be available from experimental evidence or the information available may not be adequate to properly represent the characteristics of the site.[65]

[64] NAS-NRC 2001 p.92
[65] Smith and Amonette 2006 p. 1 to 2

Concerns regarding the use of models based on insufficient understanding of the behavior of waste are further highlighted by the history of the tank farms at Hanford. Based on laboratory tests using simulated nuclear waste it was estimated that, even using a conservative estimate for the rate of corrosion, the life expectancy of the single shell high-level waste tanks was believed to be on the order of several decades.[66] However, the more complicated chemical and physical processes that took place in the real-world tanks led the first two tanks to fail in the late-1950s after just 13 and 6 years respectively. In fact, within the first 25 years of operation a total of 12 single shell tanks were confirmed to have already leaked high-level nuclear waste into the ground.[67]

Details concerning the issues confronting efforts to predict the behavior of waste in the thick vadose zones under the 200 Areas must be addressed by a separate report since they are beyond the scope of the present work. For now, however, we note that the simplified transport model used in programs like RESRAD (called the constant K_d model) is far too simple to be considered a good model in general, although it may have some applications in modeling local transport in the much thinner vadose zones under parts of the 100 and 300 Areas. A more general discussion of the difficulties and uncertainties that arise in applying the constant K_d model from the different techniques used to measure K_d, the variations in the chemical environment of the soil due both to natural variability and to the unequal disruption of wastes, the importance of non-adsorption/desorption transport processes such as colloid mediated transport, and the physical and chemical changes caused over time by the impacts of biota can be found in the previous IEER analysis of radium and plutonium mobility.[68]

Given the critical importance of the present radionuclide inventory within the river corridor to the accuracy of any potential model for future contaminant distributions, we will turn in the next section to a consideration of the quality of data used in the RCBRA.

Section 3.2 – Data Quality Concerns

In examining the quality of the data supporting the RCBRA we chose to focus primarily on the data for uranium for four principle reasons.[69] First, the main uranium isotopes present at Hanford (U-234, U-235, and U-238) are all extremely long-lived and will therefore still be present regardless of how long the present institutional controls are exerted by the DOE. In fact, the growth of uranium daughter products like radium-226 will only make these contaminants more dangerous over very long time scales. Second, uranium was found by the RCBRA to be an important contributor to the risks from the residual contamination under consideration. For example, in the Rural Residential scenario, uranium isotopes were found to contribute more than 70 percent of the maximum dose at 3 of the 8 waste sites where the maximum dose above background was found to be greater than the 15 millirem per year dose limit (316-5, 316-2, and 316-1). In addition, uranium isotopes made up 82 percent of the average dose at both of the

[66] Gephart 2003 p. 5.10
[67] Gephart 2003 p. 5.39
[68] Smith and Amonette 2006 p. 7 to 8, 10, 12 to 14, and 24
[69] Unless otherwise specified all data presented in this section was taken from the spreadsheets accompanying Appendix F and Appendix G of the RCBRA.

waste sites where the average dose exceeded the 15 millirem per year (316-5 and 316-2).[70] Third, the activity of the three isotopes of uranium are known to follow specific ratios for natural uranium which is what is assumed to be in the contaminated soil and water. Thus, checking the isotopic ratios for uranium measurements provides a valuable and somewhat unique check on the physical reasonableness of the data. Finally, the fourth reason to focus on uranium is that its mass is also measured by chemical means in several of the same sample areas and by converting the radioactive measurements into mass allows a further check on the reasonableness and accuracy of the data. The issues identified in this way should not be viewed as confined to the uranium measurements only, and should be viewed instead as potentially indicative of more general concerns regarding the quality of the data unless there is other evidence to the contrary.

To begin with, the Quality Assurance Project Plan (QAPP) for the RCBRA data was used in an effort to eliminate any suspect points from newly acquired data. As summarized by the RCBRA,

> Analytical results for soil, sediment, water, and biota collected for the RCBRA investigation were evaluated against the quality criteria specified in the QAPP (DOE/RL-2005-42, Section 2). As a measure of data quality, analytical results identified as nondetects in the RCBRA data set (i.e., results for soil, sediment, water, and biota qualified by the laboratory, reviewer, or validator as "U," and subsequently assigned detect status of "FALSE" as described in Section 4.3.2) were compared to the laboratory required detection limits prescribed in the QAPP. Nondetect results reported at values higher than the prescribed detection limit were identified for additional consideration. The adequacy of these results was then determined by comparing the reported nondetect value to the applicable media-based lookup value (such as an ecological benchmark or cleanup value). Nondetect results reported at values less than the media-based lookup values were determined to be useable nondetect results. Those results where the nondetect result exceeded the media-based lookup value were acknowledged as uncertainties in the risk analysis.[71]

Unlike the data collected specifically for this risk assessment, however, measurements taken from previous studies such as the 100-B/C Pilot Project were not screened for quality against the QAPP and instead, "[t]hese data were presumed to meet the minimum quality criteria for analytical performance and reporting as specified in their project-specific planning documentation."[72] As a result of these quality assurance procedures, the RCBRA assumes that the

> Uncertainty related to chemical concentrations in soil and biota samples, including sample collection and laboratory sample preparation and analysis, is generally not a significant contributor to overall uncertainty in risk assessment results. A major reason for this is that QC samples are used to ensure that analytical results are within acceptable levels of precision and accuracy. However, the use of environmental data from a variety of sampling programs over time in

[70] DOE 2007 p. 5-65 to 5-66
[71] DOE 2007 p. 4-14
[72] DOE 2007 p. 4-14

> this assessment introduces a potentially higher degree of uncertainty in the consistency of analytical results than is usual due to differences in sample acquisition methods, sample preparation techniques, and analytical methods over time.[73]

While there are many data points that are reasonable in the data set used by the RCBRA, we have several concerns regarding the quality of enough of the uranium data to raise questions about the overall success of the quality assurance procedures used in the RCBRA. The first problem we identified with the uranium data relates to the isotopic ratios for U-234, U-235, and U-238 reported in various media. For natural uranium, the percentage of the total uranium *radioactivity* attributable to each of the three isotopes should be approximately

U-234 = 48.9 percent
U-235 = 2.2 percent
U-238 = 48.9 percent

For natural uranium, the percentage of the total uranium *mass* attributable to each of the three isotopes should be approximately

U-234 = 0.0054 percent
U-235 = 0.711 percent
U-238 = 99.284 percent

Although the majority of total natural uranium *mass* is in the form of U-238, U-238 and U-234 activities are equal because uranium-234 is a decay product of uranium-238.[74] If measurements show uranium-238 and uranium-234 activities to be about equal, this implies natural uranium. Uranium used in Hanford reactors had an isotopic signature similar to that of natural uranium. In such cases, uranium-235 must necessarily be in the vicinity of 2.2 percent of the radioactivity of the sample, with some allowance for measurement errors (but not above 5 percent). In the present instance, in some U-234 and U-238 measurements for soil, the isotopic ratios measured are in good agreement with these percentages (see Figures 1 and 2). Hence, the U-234 and U-238 data comparison seems to point to natural uranium.

[73] DOE 2007 p. 5-93

[74] In the case of actual soil samples, small differences between uranium-238 activity and uranium-234 activity may be found due to the different recoil energies when alpha particles are emitted. Such differences are not of consequence in the context of the present discussion.

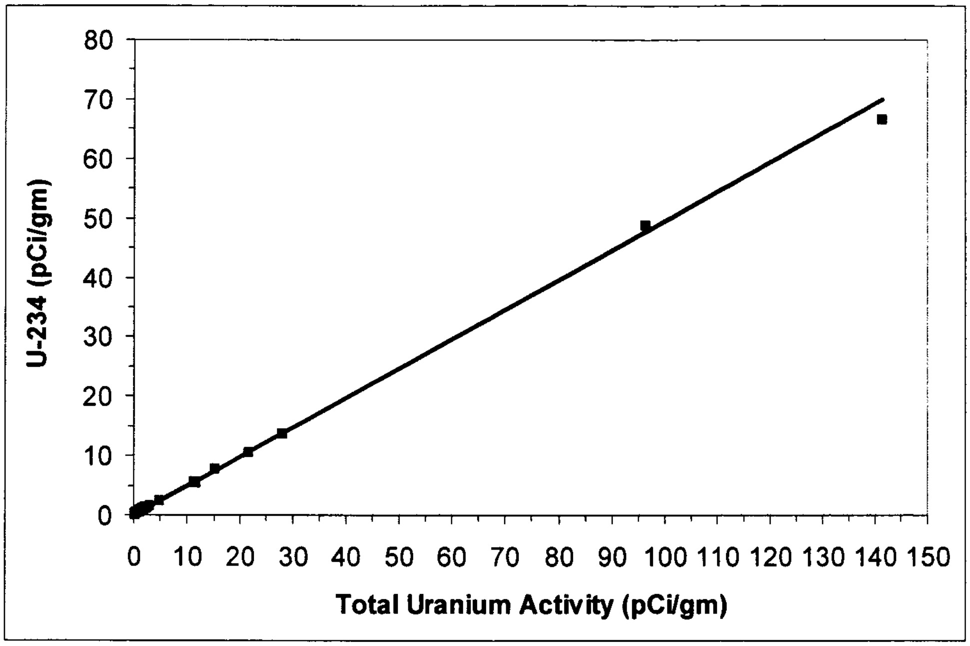

Figure 1: Graph of measured U-234 activity in soil samples taken in the 100 and 300 Areas versus the total uranium activity measured for these samples. The solid line represents the expected trend for natural uranium.

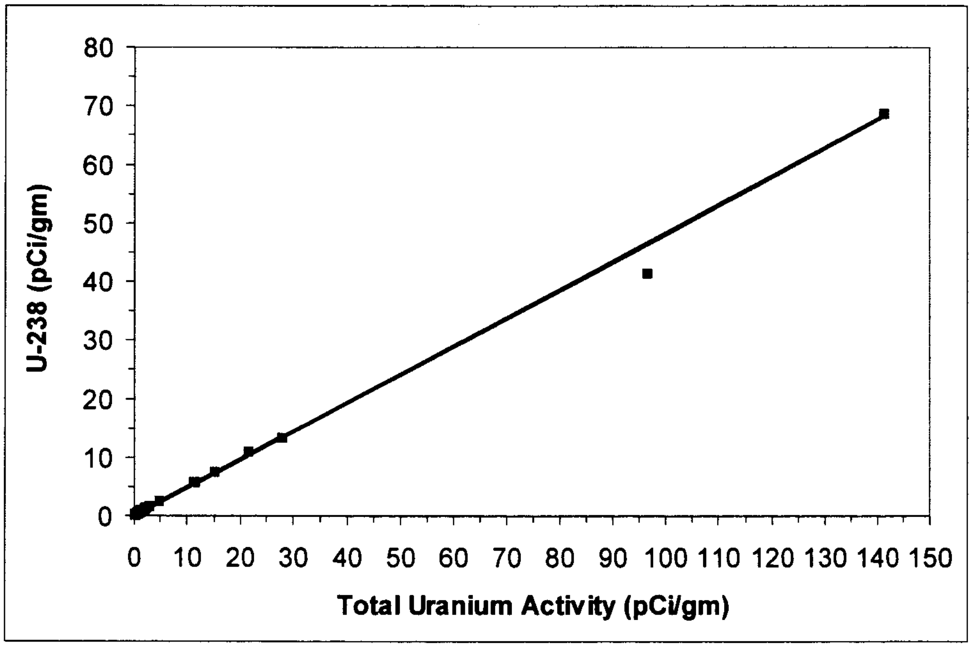

Figure 2: Graph of measured U-238 activity in soil samples versus the total uranium activity measured for the samples. The solid line represents the expected trend for natural uranium.

However, the trend for U-235 activity in soil shows a greater degree of variability, with several data points lying further from the predicted line for natural uranium than would be expected (see Figure 3). This is particularly significant given that there is evidence from the RCBRA that the isotopic ratios for uranium in this data set was investigated as part of the risk assessment and yet the problems identified throughout this section were not mentioned. Specifically, in discussing the chemical toxicity of natural versus depleted uranium the RCBRA notes that "the isotopic uranium data evaluated for this report do not indicate the presence of depleted uranium."[75] The failure to identify these data concerns would raise concerns in either case, but the failure to identify them despite analyzing the isotopic ratios for uranium would significantly heighten these concerns and raise questions as to the competence or attentiveness of those who interpreted the data.

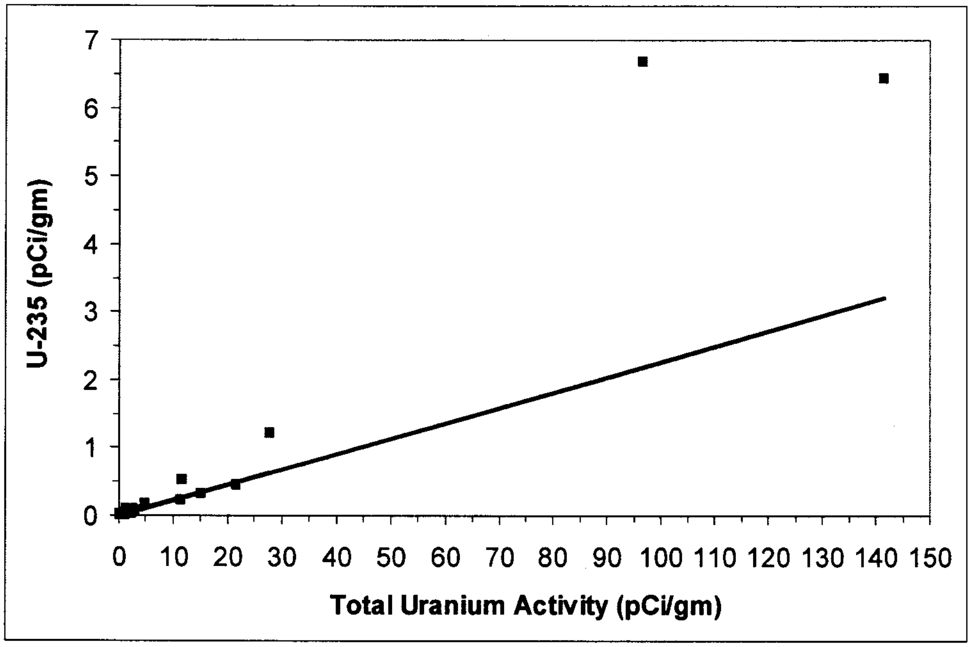

Figure 3: Graph of measured U-235 activity in soil samples versus the total uranium activity measured for the samples. The solid line represents the expected trend for natural uranium.

Similar trends for surface water and groundwater observed with the U-234 and U-238 data are generally reasonable (one notable exception was a data point for groundwater with a U-234 ratio far too low at just 3.3 percent and a U-238 ratio correspondingly far too high at 96.4 percent) while the U-235 measurements for these media showed far greater variability (see Figures 4 and 5). In the case of both the surface water and groundwater, the concerns are compounded by the reporting of zero for the U-235 activity for several data points as discussed further later in this section.

[75] DOE 2007 p. 5-94

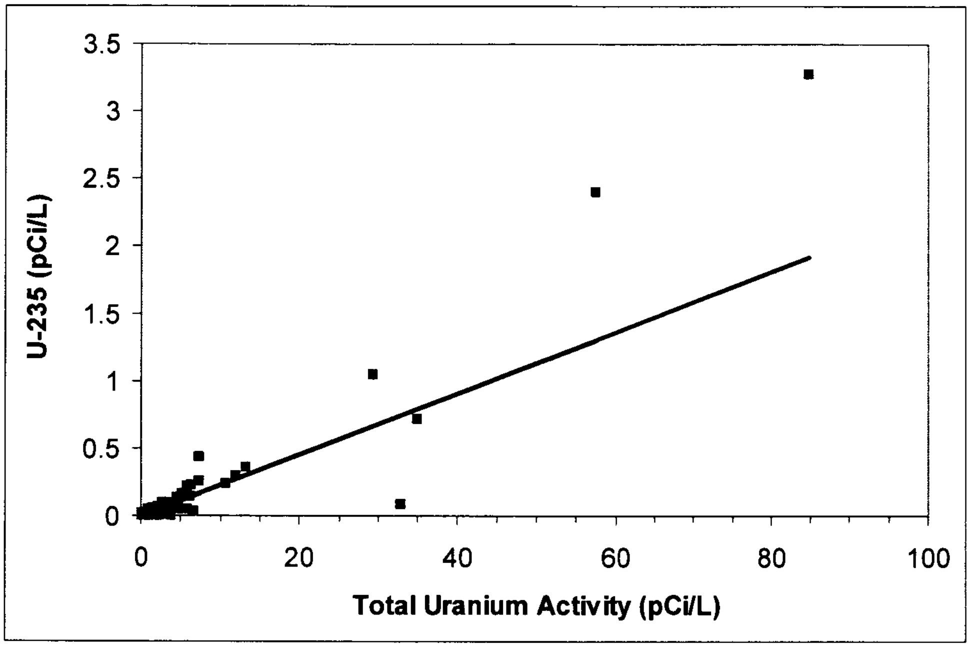

Figure 4: Graph of measured U-235 activity in groundwater samples versus the total uranium activity measured for the samples. The solid line represents the expected trend for natural uranium.

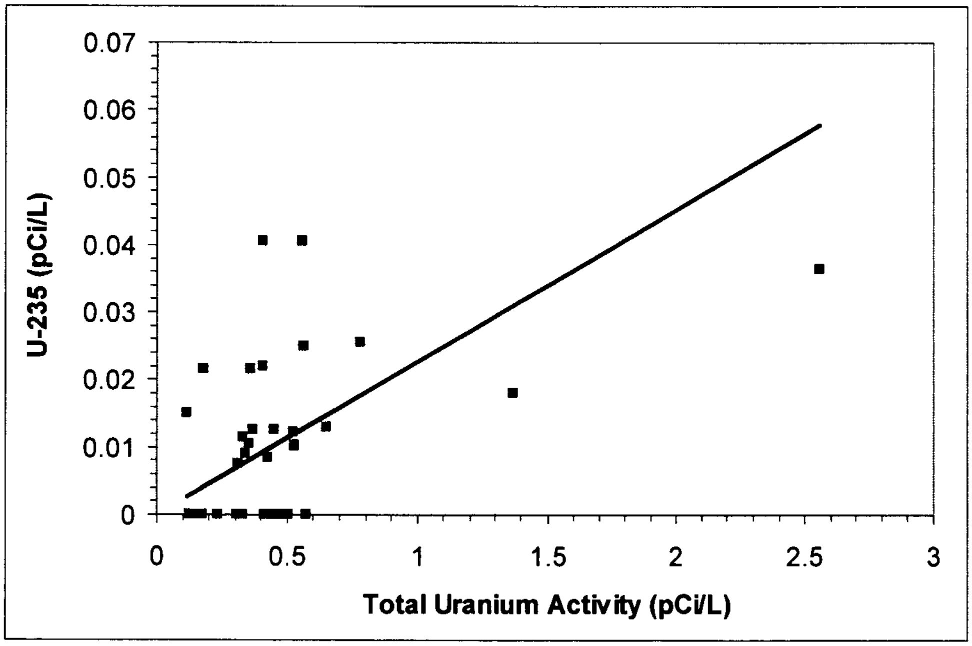

Figure 5: Graph of measured U-235 activity in surface water samples versus the total uranium activity measured for the samples. The solid line represents the expected trend for natural uranium.

The data sets associated with uranium concentrations in plants and fish, which are of particular concern for the protection of Tribal human health, appear to have additional concerns regarding their isotopic ratios. Specifically, in both cases the U-238 activity appears to be systematically underestimated compared to what would be expected in natural uranium measured at Hanford, while the U-235 activity appears to be systematically overestimated (see Figures 6 through 9).

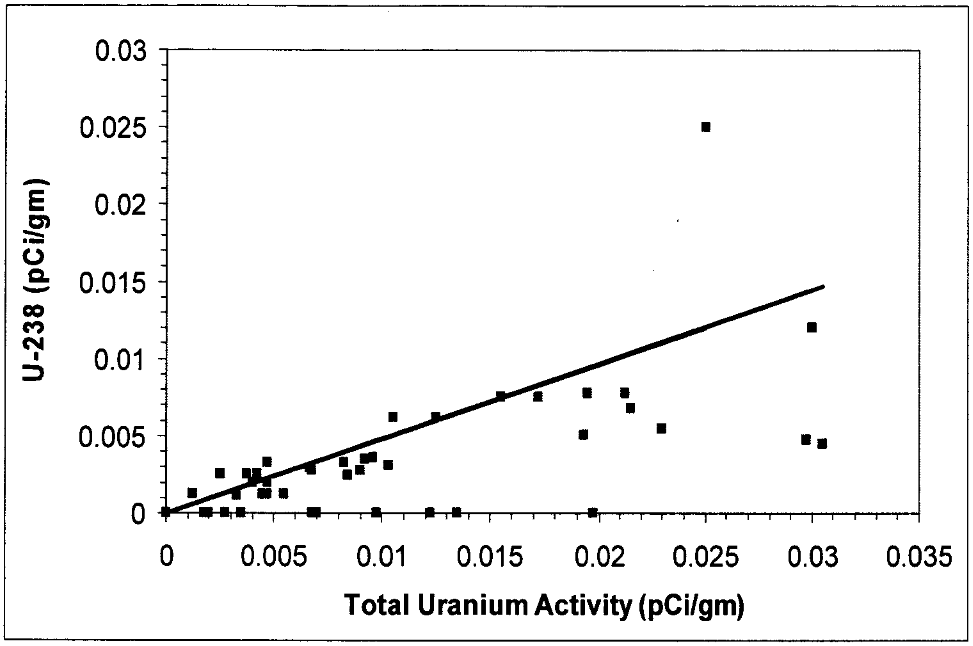

Figure 6: Graph of measured U-238 activity in terrestrial plants versus the total uranium activity measured for the plants. The solid line represents the expected trend for natural uranium.

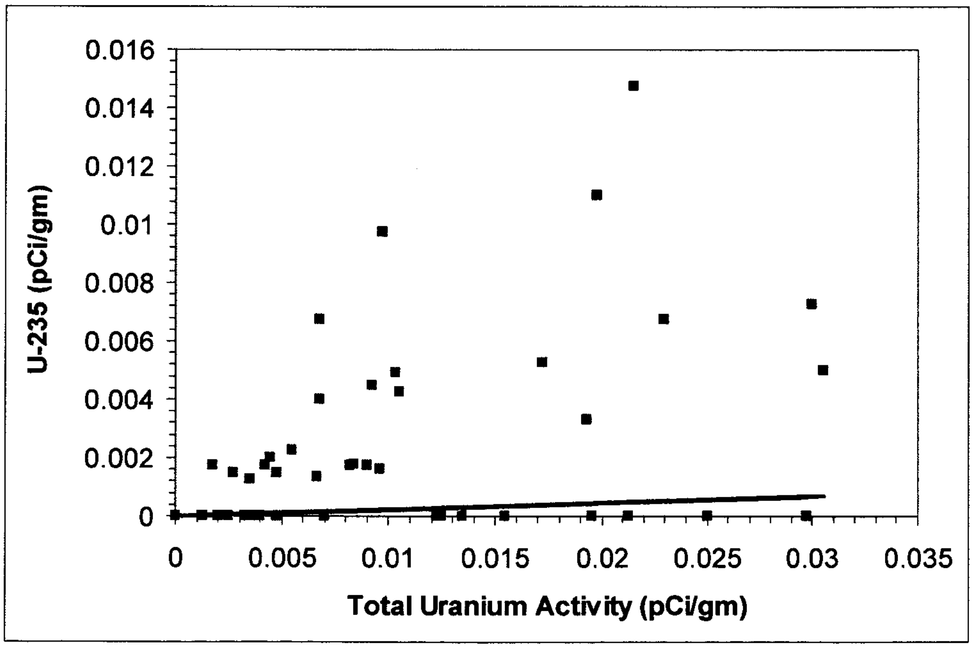

Figure 7: Graph of measured U-235 activity in terrestrial plants versus the total uranium activity measured for the plants. The solid line represents the expected trend for natural uranium.

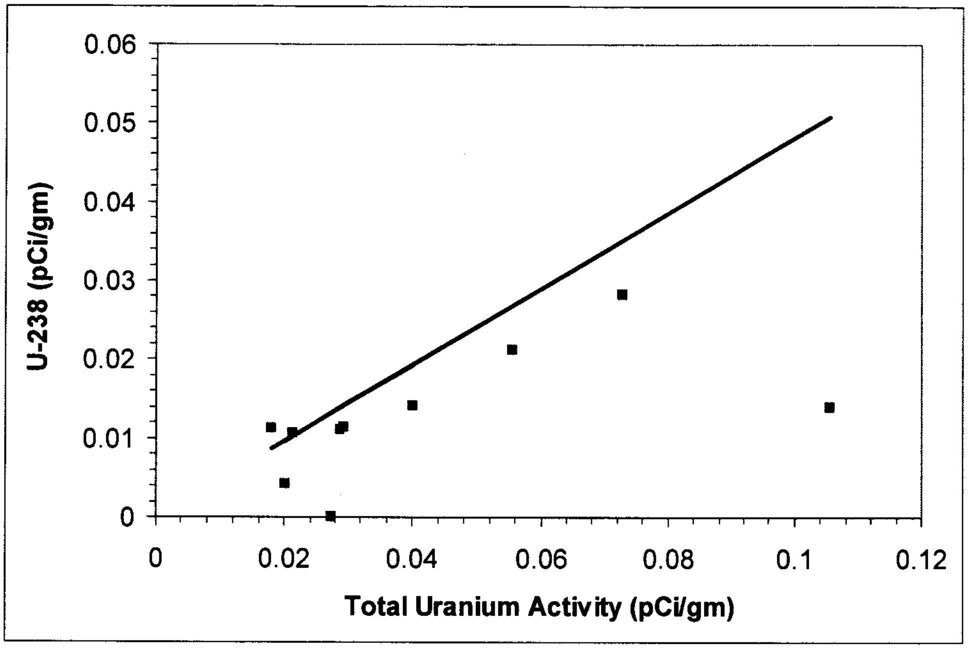

Figure 8: Graph of measured U-238 activity in sculpin (fish) versus the total uranium activity measured for the fish. The solid line represents the expected trend for natural uranium.

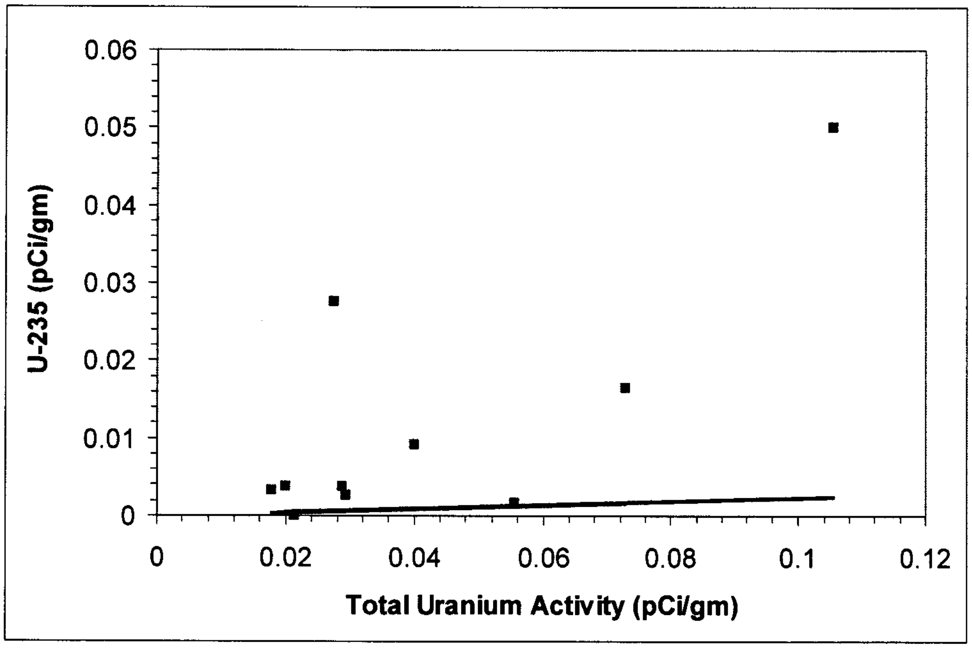

Figure 9: Graph of measured U-235 activity in sculpin (fish) versus the total uranium activity measured for the fish. The solid line represents the expected trend for natural uranium.

While it is true that the data sets with the largest deviations from the expected isotopic ratios are those with the lowest levels of activity, this is not necessarily a simple case of a small signal to noise ratio causing the variability. For example, in the soil data presented in Figure 3 we see the largest deviance for the U-235 ratios at the highest activities not the lowest. In the case of the surface water (Figure 5) we see a number of data points recorded as strictly zero when there is non-zero activity measure for other nearby samples. In the case of the plant and fish samples, the apparently systematic under estimation of U-238 activity and the systematic overestimation of U-235 activity points to the possibility of non-random errors in the data analysis procedures. Finally, it should be noted that in some cases relative U-235 values are so high as to be impossible, since U-235 remains less than 5 percent of the total radioactivity of a uranium sample at any enrichment. For higher enrichments, U-234, not U-235 dominated the radioactivity. For depleted uranium, U-238 dominates. Yet, as can be seen in Figures 7 and 9, a number of concentrations of U-235 are far greater than 5 percent of the total uranium radioactivity. These ratios are not possible.

A note about detection limits appears to be in order here. When a sample had a non-detect, this was indicated as N/A. All other data, where positive numbers are reported, indicate positive detections. But some of these positive detections indicate exceedingly low minimum detection limits. For instance, in Figure 7 there are data points as low as 0.001 pCi/gram of total uranium. This is on the order of one thousand times less than typical values of natural uranium in soil (which are one to a few picocuries per gram). Such low detections of uranium would be expensive and require long counting times. While we have not reviewed laboratory procedures in this report, such a review appears to be warranted in view of the exceedingly low detection limits implied by many samples combined with clearly incorrect isotopic ratios.

A second area of concern we identified regard the quality of the RCBRA data relates to comparisons between the radiological and chemical measurements of total uranium. As noted by the RCBRA,

> In many environmental samples, data obtained for isotopic uranium (in units of activity per mass or activity per volume) could be converted to total uranium data (in units of mass uranium per mass of sample, or mass uranium per volume). In this way, the effects of uranium metal as a kidney toxicant could be assessed in addition to evaluation of radiation dose and cancer risk when only isotopic uranium data are available.[76]

For the 50 samples where information regarding both the isotopic activities of U-234, U-235, and U-238 and the chemical measurement of total uranium are available, this conversion allows for an additional check on the quality of the data. For reasonable data, the mass of uranium calculated from the radiological measurements should be equal to that measured by chemical techniques. As can be seen in Figure 10, however, while there are several points that agree with the expected relationship, there are also a number of points in the RCBRA data set that fall below the expected value.

[76] DOE 2007 p. 5-94

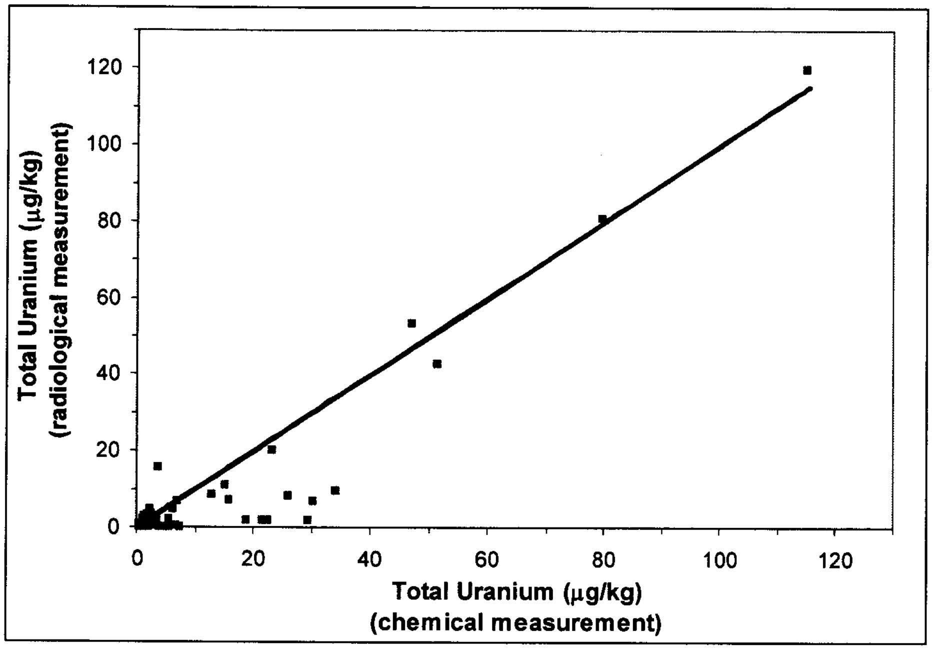

Figure 10: Graph of total inorganic uranium in micrograms per kilogram in all media as measured by radiological and chemical means. The solid line represents the expected where both measures report the same value for the total mass of uranium present in the sample.

In fact, nearly three out of every four measurements have a value for the total amount of uranium deduced from the radiological measurements that is lower than that from the chemical measurements. These are the points below the line in Figure 10. Three of these data points actually recorded zeros for the radioactivity of the sample while reporting non-zero amounts of total uranium mass. This is physically impossible since any amount of uranium has some radioactivity. Excluding these three zeros, the largest difference was a factor of more than 1,100 times between the chemical and radiological measurement techniques. All told, more than one third of the data had differences between the two measurements of more than a factor of five, while the average difference had the chemical measurement greater than the radiological measurement by a factor of 52 times. Even using the geometric rather than arithmetic mean to reduce the significance of the very high ratios observed results in the chemical measurements overestimating the amount of uranium compared to the radiological measurements by a factor of more than 3.6 times. As with previous examples for U-235 and U-238 activity in plants and fish, the apparent systematic underestimation of the total uranium by the radiological measurements compared to chemical measurements raises serious questions about the data analysis procedures involved with the RCBRA.

Our third concern relating to the quality of the RCBRA data revolves around measurements for the amount of uranium taken up into the terrestrial plants from the soil. As noted in the ecological assessment of the RCBRA,

> Upland Terrestrial Plants. No lines of evidence suggest that COPCs are adversely affecting terrestrial plants in upland soils. The general lack of plant contaminant uptake indicates minimal COPC exposure. Some COPCs are detected in plants,

> but tissue concentrations do not differ between upland remediated waste sites and reference sites and generally do not correlate with abiotic media concentrations.[77]

In other words, measurements for the amount of contaminants in the plants sampled did not appear to correlate with the measurements of the amount of that contaminant in the soil. Unless the amounts of these contaminants is so low that all of the results are non-detections, however, this lack of a correlation should be cause for concern rather than a cause to conclude that no uptake of contamination is occurring.

In order to examine this lack of correlation more closely we examined 53 data points where information on both the soil and plant concentrations was available for a particular site. From that data we calculated the fractional uptake of each isotope of uranium. The default plant uptake value used by the RCBRA is 0.017, meaning the concentration of uranium in the plant should be just 0.017 times the concentration in the soil.[78] From the data shown in Figures 11 and 12 it is clear that (1) the implied plant uptake values are not a single constant value, (2) the highest implied uptake ratios are far above the level of 0.017 and are, in fact, higher than 2.1 for U-235 implying a significant bioconcentration of this uranium isotope in the plant, and (3) the implied uptake ratio for each isotope of uranium does not appear to be correlated with that of the other isotopes. This last observation is particularly important since the uptake of uranium by the plant is a chemical process and, thus, it should absorb all isotopes with approximately the same ratio. The fact that the uptake of different isotopes appear virtually uncorrelated with each other raises serious questions about the accuracy of either the soil measurements, the plant measurements, or both.

[77] DOE 2007 p. ES-16

[78] DOE 2007 p. 5-18 to 5-19 and Wang et al. 1993 p. 23 to 24. (Note: This assumes a wet to dry weight conversion of 0.15 as was used in the RCBRA.)

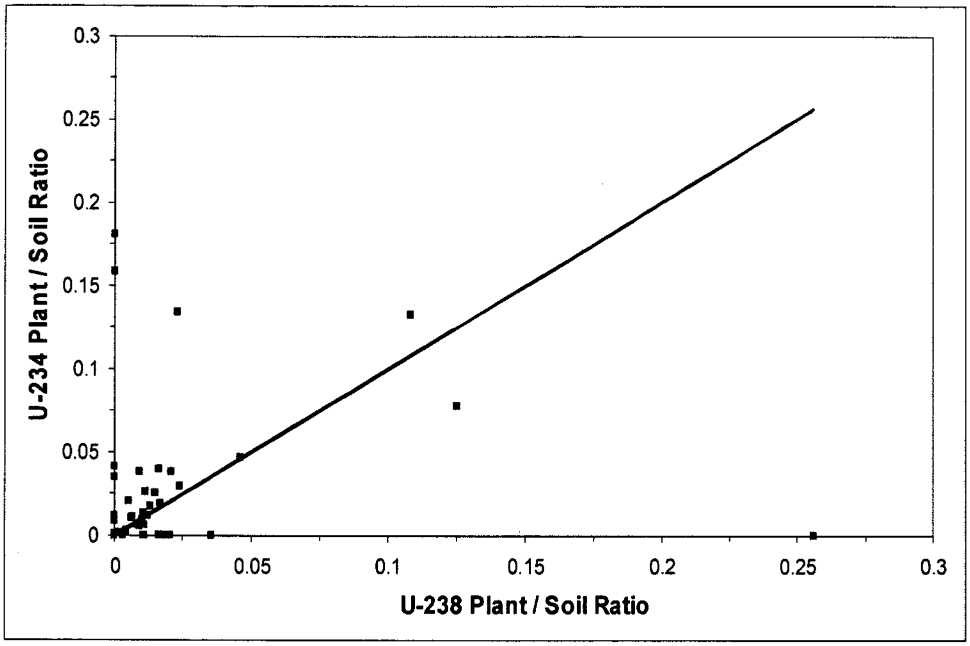

Figure 11: Graph of the plant uptake factor for U-234 compared to that of U-238. The solid line represents the expected trend where both isotopes report the same value for the uptake.

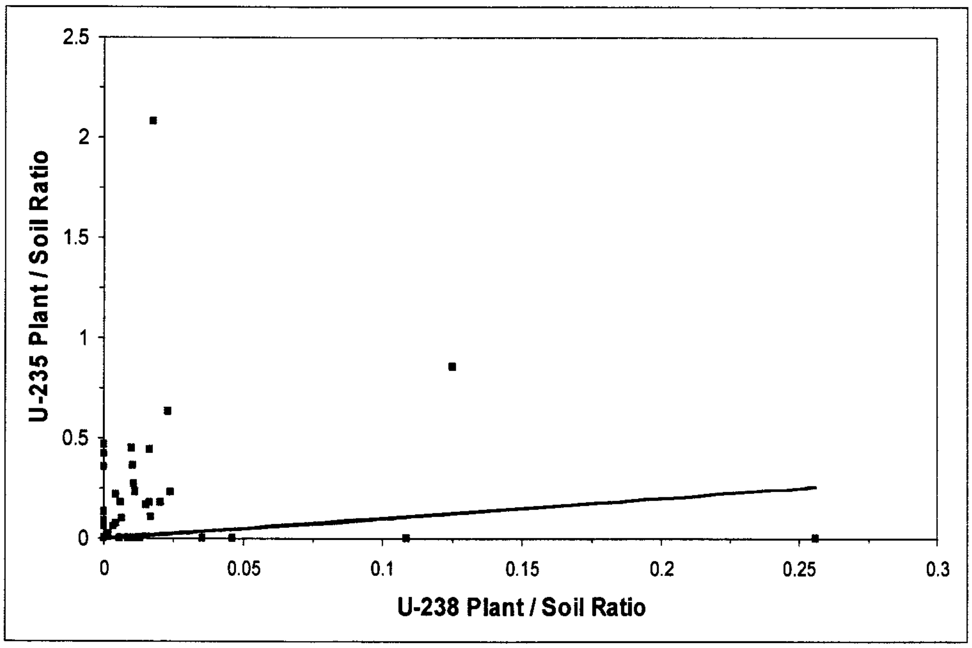

Figure 12: Graph of the plant uptake factor for U-235 compared to that of U-238. The solid line represents the expected trend where both isotopes report the same value for the uptake.

From the RCBRA we know that the "detection limits are typically elevated in [plant] tissues because of matrix interferences".[79] If the lack of correlation between the plant data and the soil

[79] DOE 2007 p. 6-127 to 6-128

data in these figures is due solely to high detection limits then, as noted, we would expect that most of the plant data should have been recorded as non-detections. Thus, the plant data is a likely place to begin with in terms of seeking an explanation for these discrepancies. Resolving this question will be an important goal of future sampling programs. In addition, answering this question concerning data quality in soil to plant transfer will also provide valuable information on the uptake factors appropriate for Hanford plants, which will directly aid in the refinement of the Tribal Rights scenario (also known as the Tribal Exposure Scenario) discussed in section 4.1.3 and the ecological risk assessment discussed in section 6.2.2.

Finally, a fourth concern regarding the data quality in the RCBRA relates to the overall methodology used to collect the soil, water, and other samples. For many of the sampling locations, only a small number of samples were taken and, in many cases, only a single measurement was made. For example, the RCBRA notes that

> There has only been a single sample event for most combinations of analyte and monitoring well. Among the 64 wells, only wells 199-H4-48, A4650, A4681, and C4670 have two samples across all analytes. Well A4587 has two additional samples for just hexavalent chromium. Because of the extremely limited number of samples, it is impossible to evaluate patterns or variability in seasonal or long-term trends in groundwater constituent concentrations in these wells. ***Consequently, there is a high degree of uncertainty in the time-averaged constituent concentrations used in the risk assessment.***[80]

The significance of this limited sampling is highlighted by more recent measurements that have revealed higher concentrations of some contaminants in groundwater than previous detected. As summarized by the RCBRA,

> The groundwater data employed in this assessment, collected at 64 monitoring wells, represent only a small subset of the available groundwater data collected over time in the 100 Area and 300 Area. For example, recent groundwater sampling in the vicinity of the former Sodium Dichromate Transfer Facility (100-D-12) has revealed chromium concentrations significantly higher than any captured in the groundwater data used in this assessment. Similarly, trichloroethene was recently detected in an area east of the 316-3 South Process Ponds near borehole 399-3-18 at concentrations higher than has normally been measured in monitoring wells in the 300 Area, but there are no detected concentrations of this analyte in the data set used in this risk assessment. Therefore, these groundwater risk results should be interpreted as semiquantitative estimates for the purpose of establishing the approximate magnitude of potential groundwater-related risks relative to the risks presented in Section 5.7 [Human Health Risk Assessment Results].[81]

Thus, a program of longer-term and more widely distributed sampling of groundwater under the 100 and 300 Areas is clearly required even to characterize the existing levels of contamination,

[80] DOE 2007 p. 5-113 (emphasis added)
[81] DOE 2007 p. 5-105

not to mention characterizing the levels to be expected in the future as a result of ongoing contaminant transport.

The lack of significant numbers of repeat measurements is not, however, simply a concern for the ground water data. Overall, more than one-third of all data reporting uranium isotopic activity, had only a single measurement for a given sample or location. In addition, nearly 10 percent of the data had at least one of the isotopes reported with zero activity. This is significant because a "zero" should have been considered a non-detect and either eliminated as a valid data point or, at the very least, recorded with the value of the minimum detection limit. Table 1 summarizes the sampling data for all media broken down by isotope as well as for total uranium measured by chemical techniques.

Table 1: Summary of data for uranium concentrations in all media (soil, biota, and water) highlighting additional data quality concerns relating to limited sampling and high variability where multiple measurements were made at a given location.

	All data points		All sample locations with multiple measurements	
	Percent sample locations with only a single measurement	Percent with a mean reported as zero	Percent locations showing a standard deviation over 5 times the mean	Maximum std. deviation/ average standard deviation
U-234	37.2%	5.13%	4.46%	77.6/1.66
U-235	28.2%	17.9%	14.3%	116.6/4.89
U-238	37.9%	5.13%	6.50%	105.5/1.82
Total U[(a)]	42.0%	0.00%	3.45%	15.8/1.33

(a) Total inorganic uranium measured by chemical means.

The significance of the large number of samples with only a single measurement is highlighted by the second half of Table 1. For those samples where more than one measurement was made, the variability of the results was found to be quite significant at many locations. For example, in more than one out of every seven locations where multiple measurements of U-235 activity were made, the upper confidence limit for the radionuclide's concentration was more than five times higher than the mean value. The highest standard deviation was 116.6; this is very large. It makes the 95 percentile value almost 200 times the mean value.[82] The average value of the standard deviation of the U-235 measurements was 4.9 times the mean, still very high. Such variability in repeat measurements from the same locations raises questions about the adequacy of the sampling program and, in particular, whether the contaminant levels being assumed in the RCBRA are adequately representative of what is actually in the environment.

The need for improved sampling is further supported by the fact that the calculations in the RCBRA for its CTUIR scenario found larger doses for the "Reference Areas" for both fish consumption and for terrestrial pathways at all but one operational unit. Specifically, the fish

[82] Assuming a normal distribution. Lognormal distribution results would indicate a greater ratio.

dose in the "Reference Area" (200 millirem per year) was twice as high as that in either the 100 or 300 Areas. Similarly, the average of the ratios of "Reference Area" dose for terrestrial pathways (180 millirem per year) to the dose predicted for the local and broad area pathways in the RCBRA's Tribal scenario was about 4.3.[83] In both cases, this was due, in large part, to high americium-241 levels measured in soil and fish.[84] As such, we recommend that additional sampling of all media be undertaken with multiple measurements at each location in order to gain a better understanding of what contamination remains in the environment in the 100 and 300 Areas. We note that americium-241 is a man-made radionuclide of Hanford origin. To use "reference" area doses that are dominated by man-made radionuclides of Hanford origin gives an incorrect picture of relative contamination in the 100 and 300 Areas, since it makes it appear in many cases that the latter areas have been remediated to below some reference or background level. This is like comparing two populations with excess cancers but designating one of them as a reference, unexposed group. This approach is incorrect. A reference area that can be definitively shown to be uncontaminated with Hanford radionuclides should be chosen.

In summary, we note that the quality assurance program for the data used to support the RCBRA claims that

> All analytical data used in the human health and ecological risk assessments were subjected to a process for ascertaining their usability to support such assessments. All data were required to have, at a minimum, the following attributes in order to be considered "usable."
>
> 1. An analyte name or CAS identification number
> 2. A numerical result without a rejected ("R") qualifier in any field
> 3. Associated units for the results
> 4. A media type
> 5. Definitive locational information.
>
> Even in cases where all five attributes were present, analytical data were at times labeled "not usable" for 1 or more of 15 reasons for which usability codes have been assigned in the database (see Section 4.0). Some of these reasons include inappropriate analytical method, nonstandard units that cannot be converted, physically infeasible results, and mixed media type such as paint chips or concrete.[85]

Despite this effort, however, we have demonstrated the existence of a number of data points that represent "physically infeasible results" such as those having isotopic ratios for U-234, U-235, and U-238 incompatible with what is possible for natural uranium (or depleted or enriched uranium either, in some cases) or those having vastly different estimates for the amount of uranium present when measured by radioactivity or by chemical techniques. The presence of

[83] It should be noted that a rather large number of results are between 40 and 45 millirem per year.
[84] DOE 2007 p. 5-70 to 5-72, 5-89, 5-291, and 5-291. Unless otherwise specified, the averages discussed here are the average values of the ratios of doses obtained in different areas for different scenarios. This is not the same as the ratio of the average values. For instance, the ratio of the average terrestrial pathway doses was about 4 (compared to 4.3 for the average of the ratios).
[85] DOE 2007 p. 5-95

such "physically infeasible results" in the data set used to support this risk assessment raises a number of questions about the validity of the quality assurance program.

The uncertainty analysis conducted in the RCBRA claims that the potential bias from its assumptions regarding the quality of the data is neutral, that is, it is claimed that some uncertainties underestimate risks, others are neutral, and yet others lead to overestimates of risk.[86] However, we have seen that the isotopic measurements of U-235 and U-238 activity in plants and fish as well as the measurement of total uranium by chemical versus radiological means all appear to have systematic biases that act to shift the data preferentially in one direction. These potential biases, along with the physically unreasonable data points and other concerns regarding the field observations discussed in Section 6.2.3, combine to call into question the integrity of the data used to quantify the contamination of the river corridor. These questions must be addressed by a new sample collection and data analysis effort before the final RCBRA is published. This effort should not only focus on known waste sites that have already been remediated, but should seek to quantify the level of contamination through the 100 and 300 Areas in light of the fact that, as noted by Roy Gephart, "[f]ew records were kept documenting solid waste burial activities before 1960" and that until 1967 the records that were kept generally "did not identify all burial locations, dates of waste shipments, or the chemical nature of material dumped."[87]

[86] DOE 2007 p. 5-92 and p. 5-93
[87] Gephart 2003 p. 5.21

Section 4: Issues Regarding the Tribal Exposure Scenarios

Adapting the quantitative risk assessment methodology typically used to guide cleanup efforts to the protection of Native Peoples presents a number of important challenges that must be confronted. First, there are a variety of relatively straightforward issues such as determining how much water a person would consume and how much air a person will breathe when practicing a traditional, subsistence lifestyle (see Section 4.1). Second, there are a number of unique exposure pathways in a Tribal Rights scenario (also known as the Tribal Exposure Scenario) that are not typically included in methodologies designed for what we will call "Reference Man farmers", but are, nevertheless, amenable to traditional quantitative risk assessment methodologies.[88] These Tribal pathways include such things as exposure to waterborne contaminants in sweathouses and the incidental ingestion of plant and animal matter during the manufacture of traditional goods like baskets or digging sticks (see Section 4.2). Third, and finally, there are a number of special considerations that must be taken into account for a Tribal Rights scenario that are not easily translated into quantitative risk analysis such as the protection of the spiritual and cultural value of particular plants, animals, and ecosystems (see Section 4.3). While protection of these values cannot generally be reduced to a simple number or dose limit, they must be treated with the same care and attention as any other element of the risk assessment and, despite their qualitative nature, should be given no less weight in determining cleanup standards. The specific Tribal Rights scenario considered here is that applying to the Yakama Nation.

Before turning to a consideration of the exposure pathways common to both a Tribal Rights scenario and a Reference Man farmer scenario, we will briefly review the nature of the exposure scenarios included by the DOE in the draft risk assessment. Table 2 reproduced from the RCBRA summarizes the various exposure pathways it considered. The only Tribal scenario available at the time was that for the Confederated Tribes of the Umatilla Indian Reservation (CTUIR) and, as such, will be the focus of much of the analysis in this section. We will, however, also make extensive use of the *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment* prepared by Ridolfi Inc. (2007) for the Yakama Nation Environmental Restoration and Waste Management Program and published after the draft RCBRA was completed. The "Rural Residential" scenario is the common subsistence farmer scenario used in a variety of quantitative risk assessments and what we will refer to as the Reference Man farmer.

[88] We will use the phrase Reference Man farmer to describe the exposure scenario commonly used in risk assessments like those implemented by RESRAD including conventional crops grown for food and a reliance on chicken, turkeys, and cows for meat and cows for milk. We call this the Reference Man farmer given the ICRP's definition of Reference Man as being "Western European or North American in habitat and custom." [ICRP 23 p. 4]

Table 2: Spatial Scales for Evaluating Soil-Related Exposure Pathways [89]

Spatial Scale	Exposure Scenario and Receptors: Potentially Complete Pathways				
	Rural Residential (adult; child)	**CTUIR (adult; child)**	**Resident Monument Worker (adult)**	**Industrial / Commercial (adult)**	**Recreational (adult; child)**
Local Area Related to an individual waste site.	Inadvertent Soil Ingestion; Dust Inhalation; Dermal Absorption[1]; External Irradiation; Garden Produce Ingestion; Poultry and Egg Ingestion; Beef and Milk from Penned Cattle	Inadvertent Soil Ingestion; Dust Inhalation; Dermal Absorption[1]; External Irradiation; Garden Produce Ingestion[2]; Beef Ingestion from Penned Cattle[3]	Inadvertent Soil Ingestion; Dust Inhalation; Dermal Absorption[1]; External Irradiation (*for the residential component of this scenario*)	Inadvertent Soil Ingestion; Dust Inhalation; Dermal Absorption[1]; External Irradiation	*No local-area exposure pathways.*
Broad Area Related to an entire operational area.	Beef and Milk from Free-Range Cattle; Dust Inhalation[4]	Use of Native Plants; Ingestion of Meat from Wild Game; Dust Inhalation[4]	Inadvertent Soil Ingestion; Dust Inhalation; Dermal Absorption[1]; External Irradiation (*for the occupational component of this scenario*)	*No broad-area exposure pathways.*	Inadvertent Soil Ingestion; Dust Inhalation; Dermal Absorption[1]; External Irradiation

1 Not evaluated for radionuclides (EPA 1989; Section 10.5.5).
2 Evaluated as a localized surrogate for use of gathered native plants.
3 Evaluated as a localized surrogate for ingestion of meat from wild game.
4 Inhalation exposure from the broad area source term is added to "local area" exposure for the combined Local and Broad Areas calculations.
CTUIR = Confederated Tribes of the Umatilla Indian Reservation

While it is very positive that the DOE has agreed to explicitly include a Tribal scenario in its risk assessments, it noted several times in the RCBRA that it does not endorse the underlying assumptions that Native Peoples would one day be allowed to use the Hanford site so that they

[89] DOE 2007 Table 5-2 (p. 5-166)

can resume practicing a more traditional, subsistence lifestyle.[90] For example, in the executive summary, the RCBRA states that

> Local and regional Tribes having ancestral ties to the Hanford Reach of the Columbia River and surrounding lands have been requested by DOE to provide an exposure scenario(s) reflecting their traditional activities. At this time, only the CTUIR have submitted an exposure scenario report to DOE. As noted previously in this summary, the use of these hypothetical scenarios in this risk assessment does not imply any endorsement of either the scenarios or the underlying assumptions by DOE or other stakeholders with respect to future land use.[91]

While this lack of an endorsement does not necessarily affect the quality of the risk assessment, the RCBRA should acknowledge that the Yakama Nation has rights under the 1855 Treaty with respect to the Hanford Site. Specifically, the treaty states that

> The exclusive right of taking fish in all the streams, where running through or bordering said reservation, is further secured to said confederated tribes and bands of Indians, as also the right of taking fish at all usual and accustomed places, in common with the citizens of the Territory, and of erecting temporary buildings for curing them; together with the privilege of hunting, gathering roots and berries, and pasturing their horses and cattle upon open and unclaimed land.[92]

Thus, the philosophy underlying the cleanup of Hanford should be guided explicitly by the goal of allowing Native Peoples to safely live the lifestyle to which they are entitled. This way of thinking will be particularly important when considering how to incorporate non-quantitative elements into the risk assessment such as the spiritual or cultural value of a site.

The need to ensure that the Tribal Rights scenario used in the revised RCBRA is as complete and realistic as possible is highlighted by the history of interim cleanup standards at Hanford. Specifically, a number of interim cleanup goals in the 100 and 300 Areas have been set in the past by using either the Rural Residential or the Industrial / Commercial scenarios.[93] However, the DOE now acknowledges that even the Tribal scenario used in the RCBRA "may be more restrictive than the rural-residential exposure scenario."[94] In addition, in characterizing its various exposure scenarios, the RCBRA concludes that, while the intensity of the Rural Residential scenario is "High", that of the CTUIR is "Very High".[95]

This difference in the intensity of site use is reflected in the higher radiation doses and chemical risks associated with the CTUIR scenario as compared to the Rural Residential scenario. For example, even excluding the consumption of fish and drinking water, both of which would be higher for Native Peoples, the RCBRA found that the doses received in the CTUIR scenario was

[90] See, for example, [DOE 2007 p. ES-2, ES-5, 2-8, and 2-23]
[91] DOE 2007 p. ES-5
[92] Treaty with the Yakama 1855
[93] DOE 2007 p. 2-10 and 2-23
[94] DOE 2007 p. 2-8 to 2-9
[95] DOE 2007 p. 5-98

always the same or higher than that in the Rural Residential scenario. Specifically, comparing the "local area only" results to those from the Rural Residential scenario, the RCBRA predicts that the doses under the CTUIR scenario are up to 30 times higher than those for the Rural Residential scenario with an average increase of nearly 2.7 times. While it is complicated by the high background doses from americium-241 in the data used by the DOE, it is instructive to compare the "local and broad areas" tribal doses to the Rural Residential scenario. The average of the ratios of the tribal to rural farmer dose is roughly 14.[96] A similar complication with americium-241 doses arises for the fish pathway, but the fish doses predicted in the RCBRA for the CTUIR scenario are between 2.6 and 13 times bigger than those projected for either the Rural Residential or the Recreational - Avid Angler scenarios.[97] Finally, we note that, even ignoring the impact of future migration of contaminants to the groundwater, nearly 70 percent of the wells examined in the RCBRA had incremental doses over background in excess of 4 millirems per year with a maximum dose of 825 millirem per year. As noted above, the migrating contaminants from the Central Plateau will aggravate the problem unless there is a thorough clean up that meets MTCA and CERCLA risk levels and ARARs in the Central Plateau after clean up is complete.

A properly constructed Tribal Rights scenario is very likely to require the most stringent cleanup levels. That should be the starting point. Specifically, some sites that were not considered hazardous enough to warrant cleanup under a scenario with a less intensive use of the site, will likely require a second round of remediation in the future if this is not done. Something akin to this has already occurred at Hanford in terms of the interim cleanup standards in the 300 Area. For example, the RCBRA identified several previously remediated waste sites where the radiation doses or risks from chemical contaminants were "higher than at most other sites and elevated relative to threshold criteria" and that these areas "include sites from the 300 Area, which were remediated to cleanup standards related to industrial land use" among others.[98] Now these sites would require further remediation due to the more stringent land use assumptions that DOE is making in RCBRA.

With this as motivation, we will now turn to our consideration of ways in which the Tribal Rights scenario has been implemented in the RCBRA and how it should be improved in the final risk assessment. We will begin by examining the exposure pathways common to a Tribal Rights and Rural Residential scenario and identify a number of subtleties and complications that arise due to differences between the diet and daily activities involved with an indigenous subsistence lifestyle and those of a Reference Man farming lifestyle. We will then turn to the quantitative pathways which are unique to the Tribal Rights scenario and find that, despite the inclusion of the important pathway of the sweathouse in the RCBRA, there remain other unique pathways completely absent from the risk assessment. Finally, we will end this section with a brief discussion of the qualitative impacts that should be integrated into future risk assessments and introduce the notion of "holistic" risk assessment.

[96] DOE 2007 p. 5-71 to 5-72, 5-244, and 5-257 to 5-258. The ratio of the average value of the tribal dose to the rural famer dose is about 5.3, still very large. Both ratios show the importance of considering a tribal-specific scenario..
[97] DOE 2007 p. 5-89
[98] DOE 2007 p. 5-117

Section 4.1 – Quantitative Pathways Common to Conventional Scenarios

In large part the kinds of activities carried out in a traditional, subsistence lifestyle are similar in kind to those carried out in a Reference Man farming scenario. For example, in both cases there is the potential for consumption of locally grown plants, animals, and fish, the drinking of water and milk from onsite sources, the incidental and intentional ingestion of soil, the inhalation of contaminated dust, and external exposures to radionuclides in the soil and in water. Thus, integrating these pathways into the Tribal Rights scenario is a question of details and not of the basic structure of the risk assessment methodology. However, as with most things, the devil truly is in the details. In this section we will discuss a number of issues and open questions that we have identified with the way in which the RCBRA incorporated the various common exposure pathways into its tribal scenario. Many of these questions revolve around unknowns and data gaps in our current understanding of how radionuclides move through the environment and how they are transferred to plants and animals. For example, in describing the CTUIR scenario, the RCBRA states that

> Exposure routes and receptors have been defined by the CTUIR for a traditional subsistence lifestyle scenario. A complete lifetime is reflected in this scenario, from infancy through old age. Some of the exposure routes are identical to those described for the Rural-Residential, Avid Wild Game Hunter, and Avid Angler scenarios, although the specific exposure media and parameter values may differ in the CTUIR scenario. There are a number of potentially unique exposure media in this scenario including surface water, and wild plants used as medicines, smoking materials, smudges, dyes, and for various crafts. The types of animals hunted and fished, and the tissues eaten or otherwise used, may differ from what is assumed in the Recreational Use scenarios. Similarly, the types of garden produce and livestock, and their uses, are not necessarily analogous to those that are applicable in the Rural-Residential scenario.[99]

In determining how Native People's use of different plants and animals (or animal parts) affect their risk profile, the RCBRA would also be making a major contribution to our understanding of how to protect ecosystems from radioactive waste (see Section 6) given the importance of understanding radionuclide transfer through the food web to Native Peoples and to the plants and other animals present in the environment.

As described in the *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment*, the Yakama lifestyle includes the following day to day activities

- Fishing, including the preparation, consumption, and use of fish for food, medicine, and materials;

- Hunting, including the preparation, consumption, and use of meat, organs, and other parts of the animal for food, medicine, and materials;

[99] DOE 2007 p. 2-29

- Gathering, including preparation, consumption, and use of roots, shoots, stems/stalks, leaves, and berries for food, medicine, and materials;

- Consumption and use of water (surface water and groundwater);

- Other daily activities, such as time spent outdoors (for work and recreation, potentially exposed to dust), and natural materials production (handling and using natural resources to make shelter, clothing, tools, and accessories); and

- Cultural activities, including sweating and participating in various celebrations, ceremonies, and memorials.[100]

We will consider each of these activities in turn, starting with the consumption of fish from the Columbia River, and explore the data gaps and other issues concerning their treatment in the RCBRA.

Section 4.1.1 – Fishing

The consumption of fish is an important part of the Yakama's traditional customs and practices and, thus, great care needs to be taken to ensure the safety of the fish in the Columbia River. As such, there are two major concerns raised by the treatment of the fish pathway in the RCBRA. The first relates to the use of studies of a single fish species, the bottom feeding sculpin, to represent the uptake of radionuclides by all of the various fish species consumed by the Yakama and other Native Peoples. As summarized in the *Yakama Nation Exposure Scenario*,

> The primary fish of importance is salmon, including spring and fall Chinook, coho, sockeye, and chum salmon, steelhead and cutthroat trout. Other anadromous as well as resident fish species of key importance to the Yakama diet include bass, bull trout, smelt, lamprey (eel), suckers, whitefish, and sturgeon. These and other fish species are harvested from the Columbia River and have been identified specifically at the Hanford Reach. The Yakama fish year round, depending upon the fish reproductive cycles.[101]

Some of these animals, such as the lamprey, have unique biologies and some, such as the sturgeon, have very long lives and, as such, the RCBRA acknowledges that there is uncertainty as to "[w]hether harvested sculpin represent contaminant concentrations in all fish at aquatic investigation areas."[102] In particular, the RCBRA notes that

> The sculpin fish tissue data are representative of a fish species with a restricted home range of approximately one-tenth of a kilometer in diameter. Data for Hanford-related contaminants in this species are used to protectively represent

[100] Ridolfi 2007 p. 17
[101] Ridolfi 2007 p. 18
[102] DOE 2007 p. 6-139

> tissue concentrations in other species that may be fished for subsistence or recreational purposes and which have a much broader home range or, in the case of salmon, are anadromous. This could potentially be a source of significant uncertainty.[103]

The authors go on to play down the importance of this uncertainty by noting that

> However, calculated fish tissue cancer risks and hazards are primarily associated with analytes (PAHs and PCBs) that are not key Hanford Site contaminants and are known to be widely distributed in the Columbia River.[104]

Given the contaminants in the groundwater at Hanford that are known to be migrating toward the Columbia River (see Section 3.1), it is critically important that the risk assessment accurately characterize the transfer of radionuclides and other contaminants to the kinds of fish consumed by the Yakama and other Native Peoples. This improved sampling should also seek to fill in data gaps regarding present day contamination such as the fact that the "[i]n the 100-B/C Area, fish tissue data are limited to strontium-90, technetium-99, Aroclor-1254, and Aroclor-1260" and that in the 100-N Area "fish tissue data are limited to only strontium-90 and technetium-99."[105]

Given the need to project doses into the future, it will be important to be able to model the uptake of radionuclides by the fish in addition to making use of present day measurements of contaminants already in the fish. Specifically, it is critical to consider future migration of contaminants from the Central Plateau in order to ensure that drinking water MCLs will not be exceeded in the future, among other things. Further, in seeking to make such projections, it will be important to take into account that, unlike many western fishers, Native Peoples eat more than just the fillet. For example, Harris and Harper point out that parts of the fish such as the "heads, fins, tails, skeletons, and eggs" are used by Native Peoples in the Hanford region for soup.[106] Likewise, the *Yakama Nation Exposure Scenario* notes that "[f]ish consumption includes whole body (i.e., all fish parts) as well as fillet only".[107] This is important because the uptake of radionuclides into the various parts of the fish consumed by the Yakama may be substantially different from the uptake assumed by conventional risk assessment models for some contaminants. For example, Table 3 summarizes data from the Advisory Committee on Radiological Protection of the Canadian Nuclear Safety Commission regarding the bioconcentration factors for freshwater fish in six different tissues.

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[103] DOE 2007 p. 5-97
[104] DOE 2007 p. 5-97
[105] DOE 2007 p. 5-88
[106] Harris and Harper 1997 p. 792
[107] Ridolfi 2007 p. 18

Table 3: Bioconcentration Factors for Natural Radionuclides in Freshwater Fish[108]

	Bone	Flesh	Liver	Kidney	Gonad	Gut
Uranium	20 – 800	0.1 – 25	<0.04 – 0.5	0.1 – 0.5	0.01 – 0.35	0.05 – 0.5
Radium	35 – 1800	1 – 60	1 – 45	3 – 30	5 – 115	7 – 45
Lead	100 – 2500	4 – 100	3 – 420	6 – 780	10 – 150	11 – 206
Thorium	15 – 160	4 – 32	4 – 36	5 – 46	13 – 50	23 – 50

Taking uranium as an example given its potential importance as a contaminant of concern at Hanford, we note that even the geometric mean value for the bioconcentration factor in bone is more than 12 times higher than the default value assumed by programs like RESRAD while the upper end of the bioconcentration factors for the bone is 80 times higher than that used in RESRAD.[109] Significantly, DOE itself has recognized in a different context the need to consider the uptake of radionuclides by the whole-body when the whole animal is being consumed. Specifically, in its 2002 *Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota*, the DOE noted that

> If biological samples are intended to be used to estimate both human and nonhuman exposures, then both muscle and carcass should be analyzed for at least some of the samples, as is practicable. The use of muscle tissue alone may underestimate the B_{iv} [bioaccumulation factor] for nonuniformly distributed elements. This is of particular concern when estimating food-chain transfers for biota; wildlife generally consume the entire organism, not just the muscle tissue. Hence, whole-body concentrations are generally the appropriate measurements for estimating food chain transfers to biota.[110]

The same logic should apply in the case of people who consume the whole fish as well. As a result, the sampling of additional fish species and contaminants recommended above should also seek to determine the concentration of those contaminants in all tissues of the fish to aid in making more accurate projections of future doses.

Section 4.1.2 – Hunting and Wild Game

As with the case of fishing discussed in the previous section, Native Peoples traditionally consume a variety of different kinds of meat making their exposure profile more complex than that of a Reference Man farmer. In particular, the Yakama hunt a wide variety of mammals and birds which are not necessarily well represented by the cow, chicken, and turkey common on rural farms. As described by the *Yakama Nation Exposure Scenario*

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[108] ACRP 2002 p. 20
[109] Wang et al. 1993 p. 34 to 35 and ACRP 2002 p. 20
[110] DOE 2002 p. M2-51

> The Yakama hunt year round, and harvest many species of wild mammals and birds, primarily deer and elk, but also rabbit, goat, sheep, beaver, pheasant, wild turkey, duck, and (in previous times of food scarcity) chipmunk and squirrel, and (historically) bear.[111]

In fact, the only large mammal specifically identified as not hunted by the Yakama is the coyote "because this animal is considered a sacred brother to the people."[112]

In the RCBRA, the CTUIR scenario recognizes the potential for differences between the consumption of wild game and those from the consumption of cows and chickens. For simplicity, the CTUIR scenario assumes a 50/50 split between the consumption of game animals like waterfowl and deer and the consumption of domesticated poultry and cattle.[113] However, there are number of simplifications that the RCBRA makes which limit the realism of its treatment of the wild game pathway.

Starting with game animals, we note that the RCBRA assumes that the contamination of meat from game animals like deer and elk will be exactly the same as that estimated for a free-range cow.[114] This assumption is questionable, however, for two principal reasons. First, the RCBRA contains no discussion of the exposure profiles of cows versus wild game animals in support of its assumption that will result in similar uptake of radionuclides from the environment. Potential differences in the exposure of cows and wild game could result from a number of factors such as

- the types of plant life these animals consume -- different plants in their diets could concentrate radionuclides differently
- the amounts of water they consume, which could be different given their daily activity profiles
- the amount of soil the animals ingest incidentally could be different given their different foraging habits.

This last possibility is particularly important given the fact that soil ingestion by free range cows was found to be such an important driver of human risk in the Rural Residential scenario.[115] Before assuming that the contamination of wild game is the same as that of cattle a detailed assessment should be made of these and other relevant differences in both biology and behavior.

Finally, care should be taken to recognize the uncertainties inherent even in the estimates of radionuclide transfer factors for cows. For example, in the document supporting the default transfer factors used in both the RESRAD program and the RCBRA, Wang et al. cautions that "[i]t is reported that this transfer factor is perhaps the least well documented in the literature

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[111] Ridolfi 2007 p. 19
[112] Ridolfi 2007 p. 19
[113] DOE 2007 p. 5-41 and 5-42
[114] DOE 2007 p. 5-24
[115] DOE 2007 p. 5-63

because of the obvious practical difficulty – the need to sacrifice the meat-producing animals to collect the required experimental data."[116] Wang et al. went on to note that

> Some of the difficulties in deriving the beef/feed transfer factor include the following:
>
> • *The need for equilibrium* – With a few exceptions, the time required for a radionuclide to reach equilibrium in many animal products (e.g., beef) is so long that few experiments can be conducted sufficiently long to approach equilibrium conditions. Hence, a transfer factor derived from comparatively short experiments will underestimate the equilibrium transfer factor.
>
> • *Effect of chemical and physical forms of diet and composition* – The availability of a radionuclide for gut uptake differs markedly, depending on the chemical and physical forms of the radionuclide and on the constituents of the diet. Higher radionuclide concentrations are often found in tissues other than muscle, particularly liver (e.g., for Pu, Am, Co, Ag, Ru) and bone (e.g., Pu, Am). Radionuclide transfer models often underestimate soil adhesion on vegetation ingested by animals. The extent of soil ingestion will be influenced by the species of animal, season, soil type, stocking rates, and pasture management. Consequently, values for soil ingestion will be highly site specific.
>
> • *Influence of age* – The intake of radionuclides by an animal is dependent on the animal's species, mass, age, and growth rate, as well as on the digestibility of the feed. Young animals often have enhanced gut uptake and, hence, higher transfer coefficients than adults. Few available transfer coefficient data take these factors into account.[117]

In addition to concerns regarding the exposure profile of different species noted above, this summary of complications in estimated uptake by animals highlights a further concern important to the Tribal Rights scenarios. Namely, as in the case of fish discussed in the previous section, Native Peoples consume more than just the muscle tissue of the animals they hunt. For example, the Yakama traditionally consume the heart and liver of large game animals like deer and elk, use the intestines and tendon of animals for sausage casing, and, historically, ate both beaver tails and bear claws.[118] As a further example, Harris and Harper note that Native Peoples in the Hanford region use the bone marrow of elk and deer to render shortening for cooking.[119] Since many radionuclides are known to concentrate in particular organs or parts of the body, such as those highlighted in the above quote that concentrate in the bone or liver, it will be important for

[116] Wang et al. 1993 p. 11
[117] Wang et al. 1993 p. 11 to 12
[118] Ridolfi 2007 p. 19 and 20
[119] Harris and Harper 1997 p. 792

the revised RCBRA to determine what the potential transfer factors are for contaminants entering all parts of the animal that are consumed. As with the case of bioaccumulation factors for fish, determining these transfer factors for wild game animals will also be of great value to the ecological risk assessment discussed in Section 6.2.2.

For the part of the Native People's meat consumption that does come from domestic cattle one additional concern regarding the methodology employed in the RCBRA relates to the fact that all of the feed for penned cattle, as opposed to free-range cattle, was "assumed to consist of grasses such as alfalfa grown by the farmer."[120] While this is a reasonable assumption to make in principle, it is important to recall that the RCBRA does not take into account the influence of using contaminated groundwater for irrigation or for the cattle's drinking water needs in its quantitative risk assessments.[121] Ignoring this pathway underestimates both the contamination of the animal's feed as well as the animal's direct uptake from the water. This is of particular concern for future dose projections given the movement of contaminants through the groundwater discussed in section 3.1 and should be included in all future risk assessments in their treatment of domestic cattle.

Turning to the consumption of game birds, a number of similar concerns are raised by the use of domestic poultry as a model for the contamination of wild birds. The first concern with this treatment of game birds is the assumption in the RCBRA that

> In this risk assessment, the poultry transfer factors will be applied to uptake of metals and radionuclides in soil rather than feed. It is assumed that chicken feed is store-bought, rather than produced from grain grown on-site, and that exposure to soil contaminants for free-ranged chickens is a result solely of their foraging habits.[122]

This is questionable assumption even for chickens, but is clearly inappropriate for wild game birds since they will never consume clean, "store-bought" feed. In order to be more realistic, neither domestic or wild birds should be assumed to get their food from offsite sources.

As with the case of game animals discussed above, an assessment of potential exposure profiles of wild birds should be conducted in order to determine what their level of contamination may be. In addition, an assessment of contaminant transfer factors should be carried out to determine how similar their biology is to that of domestic poultry when it comes to the uptake of contamination from the environment. Particular attention should be given in this assessment to the unique exposure profile and characteristics of waterfowl given their role in the diet of Native Peoples. The exposure of waterfowl to both terrestrial and aquatic contaminants could potentially make them very different from a domestic chicken. For example, it was found in past studies of adult ducks at Hanford that radioactive phosphorus was present at concentrations an average of 100 times higher than that found in the water and that fission products concentrated by more than 150,000 times in the tissues of ducklings raised in the waters of an onsite disposal

[120] DOE 2007 p. 5-24
[121] DOE 2007 p. 5-24
[122] DOE 2007 p. 5-22

trench.[123] As a result, greater care should be taken to ensure that the exposure of the Yakama and other Native Peoples via waterfowl and wild game is accurately accounted for in the final RCBRA.

Section 4.1.3 – Gathering

As with the consumption of meat and fish, the Native consumption of plants differs in significant ways from those considered in Reference Man farming scenarios. First, the variety of plant species utilized by Native Peoples is larger and more diverse than most subsistence farming scenarios considers. For example, the *Yakama Nation Exposure Scenario* notes that

> Plant roots, shoots, stems/stalks, leaves, and berries of more than 70 different plant species are harvested seasonally according to plant lifecycles and availability. Plants commonly used as food include Indian celery, biscuitroot, bitterroot, Indian carrot, yellow bell, huckleberries and choke cherries.[124]

There are two points of particular interest in considering the diversity of plants consumed by Native Peoples. The first is the higher percentage of root crops in the diets of Native Peoples compared to other kinds of diets which could affect the amount of contaminants consumed due to plant uptake.[125] The second point of interest is the presence of unique plant types not commonly found in rural farming diets. For example, the Yakama consume "Indian celery" which grows not just on dry land, but also in streams and other small bodies of water.[126] Another example is the Yakama consumption of lichens which, as a class, are known to concentrate heavy metals more than some other types of plant life.[127] In addition to the consumption of unique types of plants, Native Peoples also consume parts of the plant that are not often consumed in Reference Man farming scenarios. For example, as noted by Harris and Harper, the shoots of the cattail are eaten in the spring while later in the growing season the cattail's pollen is used for making breads.[128] The exposure profiles and uptake factors for these plants may be different from those of conventional food crops and, thus, care should be taken to specifically account for these potential differences.

Adding further to the complexity of treating Native Peoples' exposure to contaminants through gathering is their ingestion of plants for purposes not included in Reference Man farming scenarios. For example, as noted by Harris and Harper, "the native diet also includes teas, medicines, spices, sweetening, smoke (from smoked fish, vegetables, and game), and smudges (smoke from aromatic plants), which will increase the overall plant-derived ingestion rate."[129] Examples of such pathways for the Yakama include the use of medicinal plants such as "boiled

[123] Gephart 2003 p. 3.5
[124] Ridolfi 2007 p. 21
[125] Harris and Harper 1997 p. 792
[126] Ridolfi 2007 p. 21 and D-18
[127] Ridolfi 2007 p. 21, Purvis and Halls 1996, Bačkor and Loppi 2009, and Haas, Bailey, and Purvis 1998
[128] Harris and Harper 1997 p. 794
[129] Harris and Harper 1997 p. 792

rose bush" and the consumption of Chinook salmon that has been smoked for preservation.[130] As with the case of unique foodstuffs such as Indian celery and lichens, the consumption of plants not commonly included in a farmer's diet would have to be taken into account with care in order to determine what plant uptake factors were appropriate. It is interesting to note that the RCBRA explicitly mentions many of the consumption pathways for plants as quoted above, but no further mention or treatment of them is included in the risk assessment.

When they were needed in the analysis, plant uptake factors used in the RCBRA were primarily taken from those in the RESRAD program.[131] However, the RESRAD values do not represent the transfer factors for any particular type of plant and are instead "composite values" derived for a variety of different plants "such as leafy vegetables, root vegetables, fruits, grain, and forage plants."[132] This aggregating of different plant types is significant because, as noted by Wang et al.

> The vegetable/soil transfer factor of a radionuclide varies in a complex manner with soil properties and the geochemical properties of the radionuclide in soil. After entering the transpiration stream, radionuclides may not be uniformly distributed within a plant, but instead tend to concentrate in certain organs. Many studies have shown that the vegetable/soil transfer factor also varies with crop type and variety, stage of growth, and plant part, as well as with subsoil characteristics and agriculture practices. Comprehensive data on transfer factors in different crops grown on various soils are available in the literature for relatively few radionuclides. Data for radionuclides for which little or no experimental information exists have been customarily estimated on the basis of the assumption that chemically similar elements act similarly in the soil-plant environment.[133]

The potential significance of these differences can be seen in Table 4 which shows the plant uptake factors for three contaminants of potential concern at Hanford (chromium, strontium, and uranium) and how they vary across different types of plants.

[130] Ridolfi 2007 p. 18 and 21
[131] DOE 2007 p. 5-18 to 5-19
[132] Wang et al. 1993 p. 6
[133] Wang et al. 1993 p. 2 and 6

Table 4: Comparison of transfer factors across various classes of plants as reported in the documentation supporting the default values used in RESRAD and the RCBRA. The composite value is reported in wet weight and therefore cannot be compared simply to the other three columns.[134]

	Composite (in wet weight)	Roots, fruits, and grain for humans (dry weight)	Leafy vegetables (dry weight)	Forage plants and feed for animals (dry weight)
Chromium	0.00025	0.015	0.0075	0.10
Strontium	0.30	0.37	1.6	2.0
Uranium	0.0025	0.0064	0.0085	0.10

From Table 4 we can see that there is a substantial variability in the uptake factors across different types of plants. For example roots, fruits, and grains take up twice as much chromium as leafy vegetables while for strontium roots, fruits, and grains take up less than one-fourth as much as do leafy vegetables. Forage plants for animals tend to have higher concentrations than typical food plants for humans, but can vary greatly by contaminant as to how much higher. For example, strontium in forage plants is only 25 percent higher than in leafy vegetables while it is more than 11 times higher for chromium and uranium. An additional uncertainty in this respect arises because of the use in the RCBRA of a single conversion factor for wet to dry weight for plants. Specifically, the RCBRA uses a conversion factor of 0.15 while the value used in developing the uptake factors in the RESRAD model was 0.428.[135] When specific types of plants are considered there is an even greater difference with estimated wet to dry weight conversion factors ranging from 0.039 for cucumbers to 0.943 for peanuts.[136]

As such, it is critical that the RCBRA accurately measure the uptake factors for a representative sampling of the variety of plants consumed by the Yakama and other Native Peoples. While it is positive that the RCBRA uses direct measurements of contaminant concentrations in native plants when available, there remain several areas of concern. First, as noted above, the RCBRA makes use of the composite plant transfer factors from RESRAD in making predictions about things like the contamination of domestic cattle rather than the more appropriate (and typically higher) values for forage plants. Second, the measurements of radionuclide concentrations in native plants appear to potentially have quality assurance problems as discussed in Section 3.2.[137] And third, there is uncertainty in the use of the RCBRA's measurements with respect to how representative they are of the plants and site as a whole. As noted in the RCBRA, measurements of contaminant concentrations in upland plants are subject to uncertainty due to questions of "[w]hether above-ground vegetative material represents contaminant concentrations in all plant matrices/compartments (e.g., roots, seeds)" and "[w]hether contaminant concentration

[134] Wang et al. 1993 p. 23 to 26
[135] DOE 2007 p. 5-18 to 5-19 and 5-20 to 5-21 and Wang et al. 1993 p. 2
[136] Wang et al. 1993 Table 2 (p. 5)
[137] DOE 2007 p. 5-20

in the two dominant species represents all plant species in investigation areas."[138] Along with the previously noted concerns regarding data quality, these two questions need to be resolved by the final version of the RCBRA. In doing so, particular attention should be paid to plants used by Native Peoples with potentially unique properties such as Indian celery, lichens, and plants used for medicines as well as to plants and parts of plants not commonly consumed in Reference Man farming scenarios such as pollen.

Section 4.1.4 – Soil Ingestion

For long-lived radionuclides that are less mobile in the environment, the soil ingestion pathway can be of particular significance and, thus, care should be taken to ensure that it accurately reflects all avenues of exposure. For the CTUIR scenario, a value of 400 milligram per day is used for the rate of incidental soil ingestion for both children and adults.[139] While this is equal to the EPA's recommended value for the upper percentile of soil ingestion in children, a later review of studies published in the journal *Health Physics*, recommended using a 95th percentile value for soil ingestion for a suburban lifestyle which was more than four times higher than the 400 milligram per day value recommended by the EPA.[140] This is of particular importance for Tribal scenarios given the variety of activities involved in traditional subsistence lifestyles that would tend to result in higher levels of soil ingestion than those resulting from suburban activities.[141] For example, the gathering of wild plants, and in particular the gathering of roots using traditional handheld digging tools, would bring individuals into repeated and prolonged contact with the soil and could thus be expected to increase the overall rate of exposure.[142] In addition the baking of acorns and roots like camas "for several hours in a hot coal-heated and hot rock-heated pit, layered with willow leaves and covered with earth" provides additional pathways for interaction with the soil. As a final example, the Yakama's weekly Washat religious services "usually involve dancing on a dirt floor" and thus creates a further potential exposure to soil that is not captured by the kinds of activities underlying current recommendations for soil ingestion.[143]

As such, while the *Yakama Nation Exposure Scenario* recommends a smaller value for soil ingestion in adults than is used in the RCBRA, and recommends the same 400 milligram per day ingestion level for children, it is our recommendation that a larger value be used in the final version of this risk assessment. This recommendation is based on the level of uncertainty inherent in the existing estimates for soil ingestion and the unique cultural activities that would tend to bring the Yakama into greater contact with the soil than the populations so far studied. To estimate what might be a more appropriate value, we note that the *Health Physics* review recommended a 95th percentile value of more than 1,700 milligrams per day for children; the EPA *Child-Specific Exposure Factors Handbook* has 95th percentile values that are in the 1,400

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[138] DOE 2007 p. 6-127 to 6-128
[139] DOE 2007 p. 5-29 to 5-30 and 5-182
[140] Simon 1998 p. 661-663
[141] Ridolfi 2007 p. 29 to 30
[142] Ridolfi 2007 p. 17 and 22
[143] Ridolfi 2007 p. 21 and 29 to 30

mg/day to over 1,900 mg/day.[144] Ideally soil ingestion rates should be based on a protective level that includes pica children. However, until a Yakama-specific value can be determined a more conservative estimate for the reasonable maximum exposure level for incidental soil ingestion for children would be at least 1,000 milligrams per day, excluding pica children. For comparison, this is roughly equal to the mass in one-quarter of a single Domino brand sugar packet, and thus is not a particularly large amount of material.[145]

In addition to the kinds of incidental soil ingestion described above, it is also important to take into account intentional soil ingestion in children, a behavior known as geophagia or soil pica. Given the significance of this behavior to the soil pathway it is worth quoting a discussion of soil pica from an earlier IEER report at length.

> Geophagia, the intentional ingestion of large quantities of soil, has been documented for centuries and is commonly viewed as a particular manifestation of a behavior known as pica which is the intentional ingestion of all non-food stuffs such as paint, string, and soil. It has been found to occur across "geographic, ethnic and cultural boundaries" and has "been noted not to be a rare event." In its 1985 Superfund Guidance, the EPA acknowledged that short term soil ingestion well above the typical 95th percentile are possible and recommended that risk assessments consider potential exposures of 5 grams per day. In studies of lead poisoning in children, the intentional ingestion of soil and paint chips is commonly viewed as playing a significant role. In its 1997 *Exposure Factors Handbook*, the EPA concluded that "it can be assumed that the incidence rate of deliberate soil ingestion behavior in the general population is low." However, the EPA went on to note that "the prevalence of pica behavior is not known" and that due to the short time period over which children have so far been studied, "[i]t is plausible that many children may exhibit some pica behavior if studied for longer periods of time." As summarized by Calabrese et al.
>
>> Realistic estimates of soil pica are problematic. Estimating the frequency, magnitude, variability, and duration of soil pica has not been the object of extensive research. In the course of three soil ingestion studies, we have observed unambiguous soil pica in two children.... These data suggest that soil pica may vary considerably both between and within individuals and are consistent with observations that generalized pica behavior is common in normal children, but may be more prevalent and of longer duration in mentally retarded children.
>>
>> ...The findings also support the hypothesis that there is considerable interindividual variation with respect to soil pica frequency and magnitude. Thus, for the majority of children, soil pica may occur only on a few days of the year, but much more

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[144] Simon 1998 p. 661-663 and EPA 2008 p. 5-31

[145] For a typical value of soil density (RESRAD default of 1.5 g/cc) , this amounts correspond to a volume of approximately 0.63 cm^3 (0.038 in^3), which is just an eight of a standard teaspoon measure.

> frequently for others. If soil pica is seen as an expected, although highly variable, activity in a normal population of young children, rather than an unusual activity in a small subset of the population, its implications for risk assessment become more significant.

> Estimates for the amount of soil that a pica child might intentionally ingest carry even greater uncertainties than estimates of routine ingestion. Accurately estimating the amount of soil ingestion requires "extensive knowledge of the living conditions and cultural attitudes of the population of interest." Generally, however, the assumptions that have been made are that a child experiencing pica will consume between 5 and 10 grams per day. This has been the assumption adopted by risk assessments and recommendations of the Environmental Protection Agency, the Centers for Disease Control, and the Agency for Toxic Substances and Disease Registry. In 1997, the EPA officially recommended the use of 10 grams per day as the ingestion rate for a pica child. However, smaller estimates (one to five grams per day) and larger estimates (26 to 85 grams per day) have been considered by other sources.[146]

Based on this evidence, we recommend that risk assessments such as the RCBRA that are considering reasonable maximum exposures for children should include a contribution from acute exposures consisting of at least 30 to 40 grams of soil per year in addition to the routine, incidental exposures discussed above. The need to consider this additional soil exposure as acute rather than chronic is important at sites like Hanford where the distribution of contaminants is not uniform and thus the possibility of large quantities of soil being consumed from a single hot spot, even one of relatively small size, should be included for the soil pathway. Both the higher level of routine exposure to soil (1 gram per day) and the acute exposure due to pica (30 to 40 grams per year) should be included for children in the revised RCBRA treatment of the Tribal Rights scenario while only the higher level of incidental soil ingestion needs to be included for adults.

Section 4.1.5 – Other Pathways

In the case of estimating the time spent outdoors on site each day, the RCBRA states that

> For the Rural-Residential and CTUIR scenarios, the RME child value of 3 hours is approximately the 75th percentile of time spent at home in the yard for a child age 1 to 6 years. The adult RME residential value of 3 hours is also approximately the 75th percentile of time spent at home in the yard for adult age categories.[147]

Interestingly, the RCBRA uses larger values for the time spent outdoors for both the adult and child in the Recreational - Casual Use scenario (six hours per day) and the Recreational - Avid Hunting and Recreational - Avid Fishing scenarios (8 hours per day).[148] This is, in part, a

[146] Smith 2005 p. 12 to 13
[147] DOE 2007 p. 5-38
[148] DOE 2007 p. 5-183

reflection of the DOE's choice to focus the risk assessment on individual waste disposal units rather than on the site as a whole. As such, the time spent outdoors on site was interpreted as outdoors "at home in the yard." Specifically, the RCBRA notes that

> The CTUIR scenario implemented in this assessment employs primarily Local Area soil exposure (around the residence), with Broad Area soil exposure assessed for biotically mediated soil exposure pathways. Harris and Harper describe the application of the subsistence lifestyle scenario across five separate age groups with numerous activity categories and associated pathways. This assessment incorporates the exposure pathways and contact rate parameter values described in Harris and Harper. However, rather than attempting to apportion time across numerous potential activities and locations for different individuals, this assessment considers an individual who spends essentially all of their time in and around their residence. This assumption results in a maximally exposed individual with respect to residual contamination associated with an individual waste site.[149]

Given our recommendation to assess the site as a whole and to project doses into the future to account for the transport of contamination through the environment, a more realistic estimate for the maximum time spent outdoors for a subsistence lifestyle would be a value of at least seven hours per day as noted in the *Yakama Nation Exposure Scenario*.[150]

A final note with respect to the estimate of time spent on site relates to the gamma shielding factor used to take into account the reduction in external exposures from radionuclides in the soil as a result of shielding by the floor and walls of the home. As noted in the RCBRA

> The gamma shielding factor accounts for attenuation of external irradiation in the indoor environment from the shielding effects of the residence. The value of the gamma shielding factor may be expected to vary as a function of building construction methods, the geometry of the source term, and the nuclide-specific energy of the gamma emission. A value of 0.4 for the gamma shielding factor is employed based on EPA recommendation for developing soil screening guidelines.[151]

This value reduces the exposure twice as much as the gamma shielding factor used as a default value in RESRAD and thus appears to be too high for use in calculations meant to represent a reasonable maximum exposure.[152] The shielding factor of 0.7 used by RESRAD should be adopted as a more protective estimate in the revised RCBRA. This would provide a better basis for estimating intakes in homes that are not as tight as the typical U.S. home.

Finally, it is important to point out that there are a number of exposure pathways that are common in some forms of risk assessment, but which have not been included at all in the treatment of radionuclides in the RCBRA. For example, no pathway for dermal absorption of

[149] DOE 2007 p. 5-4
[150] Ridolfi 2007 p. 24
[151] DOE 2007 p. 5-38
[152] Yu et al. 1993 p. 135

radionuclides or consumption of contaminated surface water as a result of swimming is included as part of the present risk assessment. Of greater importance, however, is the lack of any consideration of the exposures to the embryo/fetus or to the breast fed infant as a result of maternal exposures. Significantly, the need to consider the breast milk pathway was mentioned explicitly in the *Yakama Nation Exposure Scenario*.[153] As noted in Section 2.2, the International Commission on Radiological Protection has published dose conversion factors that would allow such dose estimates for these populations to be conducted. As such, it is important that the revised RCBRA include both an assessment of the doses to the embryo/fetus as well as an assessment of the doses to young children from the breast milk pathway in order to ensure that the most vulnerable members of the Tribal societies are adequately protected.

Section 4.2 – Quantitative Pathways Specific to the Tribal Rights Scenario

In addition to the exposure pathways considered above, which are similar to those commonly used in Reference Man farmer scenarios, a Tribal Rights scenario must include additional pathways that are unique to the lifestyle and customs of Native Peoples. Among these, two pathways are of particular interest given their repetition as part of daily activities and their cultural importance. The first of these is the inhalation of waterborne contaminants in sweathouses making use of either surface water or groundwater sources. The second pathway is the non-dietary ingestion or inhalation of plant and animal matter as a result of using these natural materials to make clothing, jewelry, tools, or structures. We will consider each of these exposure routes in turn and find that, with care, each is amenable to the kinds of methodologies commonly employed for other pathways in quantitative risk assessments.

Turning first to the sweathouses pathway, the *Yakama Nation Exposure Scenario* notes that

> Use of a sweathouse for physical and spiritual cleansing is an important activity of the Yakama, practiced historically using mobile structures and continuing today with more permanent structures, which are generally used on a daily basis.[154]

Thus, it is very positive that the RCBRA includes a generally reasonable model for this important pathway based on the work of Harris and Harper.[155] However, we have identified three points of potential concern regarding the treatment of sweathouses in the RCBRA. First, the temperature within the sweathouse was assumed to be 150 °F and the exposure time is assumed to be one hour per day.[156] The temperature is important because it effects things like the vapor pressure within the structure and thus the amount of water in the air while the amount of time spent in the sweathouse is important given that it determines the total amount of exposure. While the assumed temperature and exposure time may be reasonable, it is important to ensure that it is representative of the practices of all Native Peoples in the Hanford area, and

[153] Ridolfi 2007 p. 35
[154] Ridolfi 2007 p. 25 to 26
[155] DOE 2007 p. 5-17 to 5-18 and 5-36 to 5-37
[156] DOE 2007 p. 5-18 and 5-31

not just of those within a particular tribe or nation.[157] This is particularly important given the higher estimate for the length of use of sweathouses in the *Yakama Nation Exposure Scenario*.[158]

The second concern with respect to the sweathouse treatment in the RCBRA relates to the fact that, as discussed in Section 3.1, the groundwater concentrations used in this risk assessment are only those for the present day and, as noted in the RCBRA itself these "groundwater contaminant concentrations are dynamic and have not necessarily peaked for all combinations of contaminants and locations in the 100 Area and 300 Area."[159] Given that the sweathouse pathway already contributes "more than 70% of total groundwater risks" for roughly half of the sampling wells in this study, the need to ensure that Native Peoples will be adequately protected from this pathway at the time of peak dose is of particular concern.[160] This concern is intensified in the context of DOE's own estimates for future contamination of river corridor water due to migration of radionuclides (and hazardous chemicals) from the Central Plateau.[161]

Related to this concern is the question of whether or not the level of drinking water consumption for children one to ten years old used in the RCBRA should be increased to account for the additional water children would consume if they were to take part in the use of the sweathouse. Therefore, it is important for the final RCBRA to ensure that either (1) no children under the age of 10 make use of the sweathouse or (2) that the additional water the children would consume as a result of using the sweathouse is taken into account in their exposure factors.[162] Therefore, it is important to ensure for the final RCBRA that the additional water the children would consume as a result of using the sweathouse is taken into account in its exposure factors.

Finally, the third concern relates to the RCBRA's assumption that contaminants are "introduced into the sweat lodge predominately through the water poured over heated rocks that is used to create steam."[163] However, as noted in the *Yakama Nation Exposure Scenario*,

> Respondents noted the use of willow branches to construct the sweathouse frame, which not only provides the structure, but also releases its medicinal component during the steaming process. Fir boughs and blankets and other materials complete the construction.[164]

Given the ability of beneficial chemicals to leach from the natural materials used to construct the sweathouses, it is also possible that some volatile contaminants could be released if they had been taken up into the plants. This may also be an important mechanism for baked roots and acorns given the fact that they are cooked underground in lined earthen pits layered with willow

[157] The differences in temperature are likely to be a relatively minor point since any change would be relatively small in terms of the thermodynamic temperature scale (i.e., in the Kelvin temperature scale), but for completeness this factor should still be made appropriate for the most exposed group.
[158] Ridolfi 2007 p. 26
[159] DOE 2007 p. 5-4
[160] DOE 2007 p. 5-106
[161] DOE 2009 Vol. 2, section U.1.3.
[162] DOE 2007 p. 5-39
[163] DOE 2007 p. 5-17
[164] Ridolfi 2007 p. 26

leaves and soil.[165] While this contaminant leaching pathway may not be of significant importance for non-volatile radionuclides like uranium, it should still be included in the analysis, particularly when combined chemical and radiological hazards are considered, as discussed in Section 5.

This last issue raises a more general point regarding the second set of unique exposure pathways within Tribal scenarios, namely the potential for the inhalation or ingestion of contaminants that were taken up into natural material other than soil and those used for dietary purposes. As summarized by Harris and Harper,

> In addition, there are certain exposures that are potentially underestimated for a broad cross-section of tribal members. For example, animal parts have many nonfood uses that could contribute to personal exposure: teeth and bones are used for decoration and whistles, skin is made into clothing, fish belly fat is rendered and used as a base for body paint, and so on. As with game, plants are used for more than just nutrition. Daily cleaning, preparation and ingestion of stored plants, and crafting of plant materials into household goods occurs throughout the year. The cattail provides an example: in the spring the shoots are eaten, the roots are consumed, and the fibrous stalks are split, woven or twisted into baskets, mats or cookhole layers. Later in the year the pollen is used for breads. Each of these activities involves selecting and gathering the plants from marshy areas, sorting, cleaning, stripping, peeling, splitting, chewing, and using various parts of the plant. Basket weavers typically hold plant materials in their mouths during separation of the inner and outer bark. In addition to the plant itself, the person contacts sediment and water, and generally there will be cuts on the hands from the sharp edges that could facilitate dermal absorption during gathering, preparation, and weaving.[166]

In the case of the Yakama, there are a number of these types of activities that could bring people into contact with a variety of plant and animal materials not commonly consumed as part of dietary exposures. A partial list of these activities include:

- the use of deer or elk brains to cure hides for manufacture into "clothing (moccasins, leggings, chaps, and dresses), shelter (tipis) and accessories (drums)",[167]
- the use of deer or elk antlers, hooves, and teeth as well as materials like rocks and minerals for jewelry and decoration,[168]
- the use of deer or elk antlers and bones for the construction of digging sticks used in the gathering of roots,[169]
- the use of Indian hemp, cedar, corn husks, and bear grass for the manufacture of bags and carrying baskets[170]

[165] Ridolfi 2007 p. 21
[166] Harris and Harper 1997 p. 794
[167] Ridolfi 2007 p. 20. The authors also note that hides from animals like the weasel and otter, which are less commonly hunted, have also been used.
[168] Ridolfi 2007 p. 20 and 25
[169] Ridolfi 2007 p. 22 and 25

- the use of berries and saprophytic shelf fungus for dyes and paints,[171]
- the use of plants such as bulrush, willow, and fir boughs for the manufacture of materials used in structures such as longhouses and sweathouses,[172]
- the use of Indian hemp to make string for hats and fishing nets,[173]
- the use of oak tree and other plant roots for bowls and cooking pottery,[174]
- and the use of willow in the underground cooking of roots and acorns as well as in the construction of tools.[175]

These non-dietary exposures to plants, animals, and other natural materials are nowhere considered in the present RCBRA. This omission should be corrected in the final risk assessment report. In taking these exposure pathways into account care should be taken to model both inadvertent or incidental ingestion and inhalation during both the manufacture of the products as well as from their use. For example, the weaving of water-tight cedar baskets often involves the strips being pulled taut by the weaver's teeth providing opportunities for both ingesting and inhaling pieces of the plant.[176] In addition, the decay of natural materials used for clothing, shelter, jewelry, or paints as well as the use of bowls and cookware made from roots can provide ongoing opportunities for ingestion or inhalation. A final caution in taking these unique exposures into account is the fact that, as discussed in Sections 4.1.1 through 4.1.3, the plant and animal transfer factors are based on a limited set of data and they are generally focused only on the commonly eaten portions of plants and animals. Thus, as part of the effort to include these exposure pathways into the risk assessment framework, additional environmental sampling will be needed in order to more accurately quantify contaminant transfer factors for the non-dietary plants and animals as well as those that are consumed directly. This will need to accompany efforts to gather additional information concerning the cultural practices involved in the manufacture of products made from natural materials as well as their rate of decay in order to quantify the proper exposure factors associated with the wide variety of potential pathways outlined above.

Section 4.3 – Qualitative Considerations within the Tribal Rights Scenario

In section 5 we will address concerns surrounding the lack of consideration in the RCBRA of the potential impacts from combined exposures of people to both radiological and chemical toxins. Similarly, in Section 6 we will discuss concerns relating to the lack of consideration in the RCBRA of the impacts on ecosystems from multiple stressors including, not only radiological and chemical contamination, but also from such things as soil quality changes and the impacts of fishing and dams. Related to these concerns are those involving less quantifiable considerations

[170] Ridolfi 2007 p. 22 and 24
[171] Ridolfi 2007 p. 24 to 25
[172] Ridolfi 2007 p. 21, 24, and 26
[173] Ridolfi 2007 p. 24
[174] Ridolfi 2007 p. 24 to 25
[175] Ridolfi 2007 p. 21
[176] Ridolfi 2007 p. 24

that are, nevertheless, of equal importance to Native Peoples as those that are amenable to quantitative assessment. This more integrated approach, which embraces a broad definition of impacts, can be referred to as "holistic risk assessment". For example, as summarized by Arquette et al.

> Holistic risk assessment has been discussed as a way to integrate human health and ecologic risk and make better decisions that are more protective of people and the earth as an whole. It is clear that to develop more holistic environmental health models, there is a need to identify and measure sociocultural impacts and integrate them with human health and ecologic effects. To incorporate these many different effects, a holistic model would need to examine and include aspects from many fields of study, integrating qualitative research findings with the sciences of toxicology, epidemiology, and ecology. Such an integrated model would need to be based on a very broad and flexible understanding of health, risk, and restoration, while acknowledging that these definitions are culturally based and community specific. This expanded definition of health would be more inclusive than just the absence of disease or injury. It would encompass alternative definitions of health such as that developed by the World Health Organization in the 1940s to include concepts of wellness that integrate physical, mental, social, and ecologic well-being.[177]

The lack of such a broadly inclusive approach to the impacts of contaminants on the Yakama and other Native Peoples at Hanford is a serious weakness in the RCBRA. As noted in the *Yakama Nation Exposure Scenario*,

> The risk assessment process in general also does not consider impacts and risks to the social, cultural, and spiritual practices of the Yakama people, which are considered an important link to personal health. These uncertainties, biases, and omissions noted during from [sic] this study should be taken into account in future studies.[178]

Given the complexity of the challenges facing quantitative risk assessments as highlighted throughout the rest of this work, it is tempting to remain focused on just those parts of the analysis that can be reduced to numbers and compared against generally accepted regulatory limits such as the 15 millirem per year dose limit for all pathways. However, just because some of the impacts considered in a holistic risk assessment are not quantifiable, doesn't mean they are somehow less important to the process. Instead, this means that a new way of evaluating and talking about risk is required when assessing the impacts to Native Peoples. For example, in discussing some of the differences between traditional risk assessment and those that would be more appropriate to Native Peoples, Burger et al. note that

> We suggest that while most economists and other Western scientists value the goods and services that ecosystems provide, subsistence and tribal peoples often have a broader, more holistic view of the interrelationship of natural and cultural

[177] Arquette et al. 2002 p. 262
[178] Ridolfi 2007 p. 14

> resources (see Fig. 1 [not included]). A healthy ecosystem is one that supports its natural plants and animals, as well as sustaining the biophysical, cultural, and spiritual health of native peoples. There are two distinctions that bear comment: (1) people who view natural resources holistically (i.e., subsistence and tribal peoples) often combine many of the traditional goods and services together rather than considering them separately (for example, people go fishing and hunting, visit burial or sacred grounds, and camp while doing so), and (2) many resources considered to be cultural by Western scientists have a natural resource base as an integral part (for example, a sacred ground includes not only any manmade or altered structures, but the physical environment and natural resources surrounding it...).[179]

In the case of the Yakama, this connection between natural resources and cultural resources is explicit, as noted in the *Yakama Nation Exposure Scenario*,

> The Yakama Nation's traditional homeland is an area where ancient cultures have survived for thousands of years. During a long and dynamic tenure, the Yakama Native Americans developed an intimate understanding of the complex relationships between the land and associated natural resources. Resources used by the Yakama are broadly classified as roots, fibers, berries, fish, birds and other animals, minerals, and places of spiritual guidance and strength.[180]

For example, both the spring Chinook salmon and Indian celery are considered to be "first foods" which are celebrated during annual feasts "to recognize the availability and abundance of food at the start of each growing season."[181] In addition, the only large mammal not traditionally hunted on the Hanford site is the coyote because it is considered "a sacred brother to the people" by the Yakama.[182] Finally, there are areas of the Hanford site, including some islands located in the Columbia River, which are considered particularly unique and sacred by the Yakama.[183] As summarized by the *Yakama Nation Exposure Scenario*

> Important geographical locations for the Yakama include Signal Peak on the western heights of Toppenish Ridge and Satus Peak. Historically, when tribesmen gathered together for a full week each July in Toppenish, the tribesmen held council, danced, and played stick and bone games. Traditional customs and beliefs, strictly upheld by the Yakama, have been passed on through oral tradition through the generations for thousands of years. Rattlesnake Ridge, which is currently part of the Hanford Site, is a very sacred site for the Yakama, providing a wealth of plants to gather for food and medicine, and historically a vision site for children to find their "gift."[184]

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[179] Burger et al. 2008 p. 1265
[180] Ridolfi 2007 p. 2
[181] Ridolfi 2007 p. 16 and 21
[182] Ridolfi 2007 p. 19
[183] Ridolfi 2007 p. 35 to 36
[184] Ridolfi 2007 p. 28

In light of such considerations, we recommend that an explicit effort be made in the final RCBRA to take into account not only an improved quantitative assessment of the impacts on Native Peoples from the site as a whole (including all sources of contamination), but also to address the protection of such resources as "places of spiritual guidance and strength" in a holistic fashion. This is consistent with the recommendations of the *Yakama Nation Exposure Scenario* that "[t]he risk assessment should consider qualitative information provided in this exposure scenario, which explains the extent to which the Yakama depend upon the use of the soil and water, plants, fish and other animals."[185] As a guiding principle for this qualitative risk assessment, the revised RCBRA should seek to demonstrate compliance with the following, simply stated goal from the *Yakama Nation Exposure Scenario*

> A safe and healthy subsistence lifestyle should remain an option for the Yakama in their ancestral lands.[186]

[185] Ridolfi 2007 p. 37
[186] Ridolfi 2007 p. 34

Section 5: Issues Regarding Combined Chemical and Radiological Impacts

Currently, most risks from environmental contaminants are evaluated under the assumption that each contaminant can be treated in isolation and that risks from multiple combined exposures can be evaluated simply by adding up the risks associated with each individual contaminant. This assumption, however, ignores the possibility of interactions between the various contaminants that may act to increase the risk relative to the assumption of additivity (typically called a synergistic interaction) or to decrease the risk (typically called an antagonistic interaction).[187] These possibilities are, in fact, explicitly noted by the RCBRA. For example,

> In most risk assessments, the carcinogenic risks from all carcinogenic chemicals are treated as additive and summed to produce an overall estimate of carcinogenic risk from the site. Interactions that alter the toxicity may also occur among chemicals in a mixture. That is, the potential exists for synergistic effects or antagonistic effects. Synergistic effects occur when the combined effects are greater than the toxicity of each component of a mixture individually, while antagonistic effects occur when the combined effects are less than the toxicity of each component of a mixture individually. Failure to consider potential synergistic or antagonistic effects on toxicity may result in either an underestimation or an overestimation (similar to the assumption of additivity) of the risk, respectively.[188]

In addition to simply increasing or decreasing the impacts of each agent, it is important to note that some combined exposures can give rise to wholly new effects that would not be seen when either agent is present in isolation. However, despite the potential importance of interactions among contaminants, the RCBRA adopts simple additivity as the basis for its quantitative assessments of risks as noted, for example, when it states

> Because the uncertainties related to exposure to chemical mixtures affect whether the risk is over- or underestimated, it is important to determine the conditions under which additivity versus synergism may occur. For example, *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* suggests that additivity be assumed as a "default" approach when mixture components are at low doses and when toxicity occurs via the same mechanism. In this assessment, values of HI [hazard index] will initially be calculated across all chemicals and exposure routes. If an HI is potentially significant, the issue of similarity of the toxicological mechanisms of action across the major contributors to the HI will be explored in the uncertainty analysis.[189]

There is no other substantive mention of synergism in the remainder of the RCBRA (although it is mentioned occasionally), nor is there any acknowledgement that there may be interactions

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[187] For an introduction to synergistic interactions in the case of radiation and chemical exposures see [Makhijani, Smith, and Thorne 2006 p. 52 to 58]

[188] DOE 2007 p. 5-54

[189] DOE 2007 p. 5-55 to 5-56

between chemicals and radiation as well as between chemical mixtures.[190] In fact, the RCBRA implies that even the simple addition of risks from chemical mixtures is perhaps overly cautious at times and that "[i]n general, as the number of exposure pathways and individual chemicals in a risk calculation increases, the degree of protective bias will also increase due to the summation across pathways and analytes."[191]

The focus of the RCBRA on simple addition of risks from individual chemicals, not to mention the lack of consideration of potential interactions between chemicals and radiation, is not consistent with the notion of holistic risk assessment outlined in Section 4.3. In fact, the *Yakama Nation Exposure Scenario* explicitly includes an expectation that the Tribal Rights scenario "be used to evaluate risk in a comprehensive manner for the entire Hanford Site, incorporating all sources, radiological and chemical contaminants, exposure pathways, and natural resource uses."[192] More specifically, the authors note that

> Based upon an increased emphasis on the evaluation of chemical mixtures, aggregate exposures, and cumulative risk assessments, it is recommended that DOE use the results of the exposure assessment described in this report to quantify aggregate exposures. These aggregate exposures should combine the exposure of an individual to a specific contaminant by various exposure routes (e.g., summing exposure to an agent via ingestion of water and food, dermal contact, etc.). It should also quantify cumulative risk, which combines the aggregate exposures of multiple chemical or physical agents (i.e., daily activity patterns combined to evaluate an entire lifetime); and determine cleanup based on a holistic paradigm that evaluates the risk assessment combined with an evaluation of community health and environmental restoration, which are intrinsically linked.[193]

In this light, it is interesting to note that the EPA guidance document[194] cited in the above quote from the RCBRA was published in 2000 while more recent work from the EPA has begun to lay the groundwork for eliminating this simple one-chemical-at-a-time, additive style of risk assessment. For example, in 2003 the U.S. Environmental Protection Agency issued its *Framework for Cumulative Risk Assessment* which continues the EPA's movement towards more formally integrating the impacts of combined chemical exposures into its overall risk assessment framework.[195] This framework was meant to be "the first step in a long-term effort to develop cumulative risk assessment guidelines."[196] In summarizing its work, the EPA laid out the scope of what it was hoping to accomplish by noting that

> In this report, "cumulative risk" means "the combined risks from aggregate exposures to multiple agents or stressors." Several key points can be derived from

[190] See, for example, [DOE 2007 p. 5-55 and 5-73]
[191] DOE 2007 p. ES 13
[192] Ridolfi 2007 p. ii
[193] Ridolfi 2007 p. 37 to 38
[194] EPA 2000
[195] EPA 2003
[196] EPA 2003 p. xvii. For a summary of the EPA risk assessment framework and its development see, for example, [Callahan and Sexton 2007].

> this definition of cumulative risk. First, cumulative risk involves multiple agents or stressors, which means that assessments involving a single chemical or stressor are not "cumulative risk assessments" under this definition. Second, there is no limitation that the "agents or stressors" be only chemicals; they may be, but they may also be biological or physical agents or an activity that, directly or indirectly, alters or causes the loss of a necessity such as habitat. Third, this definition requires that the risks from multiple agents or stressors be combined. This does not necessarily mean that the risks should be "added," but rather that some analysis should be conducted to determine how the risks from the various agents or stressors interact. It also means that an assessment that covers a number of chemicals or other stressors but that merely lists each chemical with a corresponding risk without consideration of the other chemicals present is not an assessment of cumulative risk under this definition.[197]

Thus, the RCBRA clearly cannot be classified as a cumulative risk assessment under the EPA framework. While a complete review of the potential synergistic interactions between chemicals or between chemicals and radiation is far beyond the scope of the present work, we will briefly explore in the following sections the potential significance of their omission from the RCBRA and make recommendations as to the effects that should be studied further in order to determine whether they pose risks that are greater than those implied by simple addition.[198]

Section 5.1 – General Considerations Concerning Combined Exposures

To begin with, we note that the most recent recommendations of the International Commission of Radiological Protection concluded that

> Although the potential importance of synergistic effects between radiation and other agents is recognised by the Commission, at the present time there is no firm evidence for such interactions at low doses that would justify a modification of existing radiation risk estimates.[199]

However, in the review of interactions between radiation and chemicals which was cited by the ICRP in making its recommendations, the United Nations Scientific Committee on the Effects of Atomic Radiation specifically cautioned that

> The lack of pertinent data on combined effects does not imply per se that interactions between radiation and other agents do not occur. Indeed, substances with tumour promoter and/or inhibitor activities are found in the daily diet, and cancer risk therefore depends on lifestyle, particularly eating habits. Not only can these agents modify the natural or spontaneous cancer incidence, but they may

[197] EPA 2003 p. xvii

[198] Much of the following discussion in this section of potential synergistic interactions between radiation and chemicals follows the outline of a major study on this issue written by the present author and by Dr. Arjun Makhijani which will be published by IEER later this year.

[199] ICRP 103 p. 57

> also modify the carcinogenic potential of radiation. Such modifications would influence the outcome particularly when radiation risks are projected relative to the spontaneous cancer incidence.[200]

Since the publication of the United Nations review, a great deal has been learned concerning the potential for synergistic interactions between radiation and certain types of chemicals like hormonally active agents (see Section 5.2) and heavy metals (see Section 5.3). In addition, we note that there is already one well known case of synergism between chemicals and radiation that is of sufficient strength that it has been identified even in human epidemiological studies, namely that between cigarette smoke and radon, or more specifically radon daughters.[201]

Before turning to the specific areas of concern at Hanford, it is important to note that an additional complication involving the study of combined exposures is that the interactions between different agents do not necessarily have to occur only between simultaneous exposures. In some cases, exposures that are separated in time (sometimes by as much as years or decades) may still interact with each other. Complicating matters even further is the fact that, in some cases, there may be interactions if exposure to one agent precedes the other, but no interaction if the order is reversed. In addition, there may be cases where the interaction would only occur if the exposures happened during certain critical windows of time such as during fetal development or early childhood, and not if the exposures occurred at any other time. Finally, it is found that interactions between different agents will sometimes result from highly specific biological processes and, as a result, different animals (i.e., rats versus mice) or different biological endpoints (i.e., chromosomal damage versus breast cancer incidence) may disagree with respect to the strength or even the nature of the interaction. In other words, scientists doing different experiments using the same combination of agents may reach fundamentally different conclusions over whether the interactions are synergistic and, even if they do agree on the nature of the interaction, they may disagree on how strong the interactions are.

Given the types of chemicals present on the Hanford site, the two most likely areas where synergistic interactions may occur, and thus where the greatest care must be taken, involve (1) hormonally active agents and (2) heavy metals. We will consider each of these in turn below with a special emphasis on the risks posed by uranium which is unique in this context in that it is both a toxic heavy metal and a radioactive contaminant. Before turning to these two classes of chemicals, however, we note that synergistic interactions may potentially be mediated through impacts on the immune system as well.

There are multiple lines of evidence that indicate the possibility that suppression of the immune system can, in some cases, detrimentally affect the ability of an organism to repair or otherwise mitigate the kinds of complex DNA damage caused by carcinogenic agents like radiation. Evidence from studies of people taking immunosuppressant drugs as a result of organ transplants, people living with HIV/AIDS, as well as people living with a host of other immunodeficiency syndromes, in addition to studies with immune-compromised mice, all point to the potential for synergistic interactions to be mediated through the immune system. This may be a concern at Hanford given the presence of radionuclides like strontium-90 in the

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[200] UNSCEAR 2000 p. 215-216

[201] The case of radiation and smoking was previously reviewed in [Makhijani, Smith, and Thorne 2006 p. 57-58]

environment, which are known to concentrate in the bone and thus potentially damage the components of the immune system associated with the bone marrow.[202] An additional system level pathway for interactions, namely the potential for one chemical to alter the the way the body metabolizes another chemical or to alter the biokinetics of radionuclides in the body will not be considered in this section, but care should be taken to determine whether or not these kinds of interactions may occur for contaminants found at Hanford.

Section 5.2 – Hormonally Active Agents

Hormonally active agents, also known as endocrine disruptors, are a broad class of chemicals related by their ability to impact the hormone system of humans and other animals. Some of the more well known endocrine disrupting chemicals include DES (diethylstilbestrol), dioxin (including 2,3,7,8-Tetrachlorodibenzo-para-dioxin), PCBs (polychlorinated biphenyls, including Aroclor-1254), DDT (dichlorodiphenyltrichloroethane), and BPA (bisphenol A). As a class, these compounds cause a wide variety of effects on the hormonal system and, therefore, great care should be taken in dealing with exposures to multiple hormonally active agents. For example, as summarized by Steingraber

> Endocrine-disrupting chemicals are substances that disregulate some aspect of the endocrine system. They can exert their effects in a number of ways: by mimicking hormones, blocking their uptake by receptors, altering the rate of their synthesis or secretion, interfering with their metabolism or elimination from the body or altering the number of hormone receptor sites and thereby making the body more or less sensitive to its own hormonal signals. Not only can endocrine disruptors sabotage any one hormonal signal through a multitude of tactics, the HPG [hypothalamus-pituitary-gonadal] and HPA [hypothalamus-pituitary-adrenal] axes are designed to respond to a multitude of hormonal signals. Furthermore, any one hormone can send a variety of messages to these axes depending on the timing of its receipt and its concentration in the bloodstream. For example, estradiol from the ovaries sometimes serves as a negative feedback to the hypothalamus, causing it to slow down the tempo of its GnRH [gonadotropin-releasing hormone] pulse generator. At other times, estradiol serves to accelerate hypothalamic maturation, which quickens the pulse generator's tempo. New evidence also suggests that the prepubertal breast responds non-monotonically to estradiol. That is, at low doses, estradiol can induce development of breast tissue, while high doses inhibit it. To add to the intricacy, intermittent exposures may have different effects than continuous exposures.[203]

One class of endocrine disruptors that will be of particular importance are those that mimic estrogen in the body. These hormonally active chemicals are known collectively as xenoestrogens or exogenous estrogens to distinguish them from natural (i.e., endogenous) estrogen present in the body and from naturally occurring estrogen mimicking chemicals found in plants known as phytoestrogens.

[202] For discussions concerning the presence of strontium-90 in the river corridor see [DOE 2007 p. 4-23 and 4-24].
[203] Steingraber 2007 p. 11, 15, and 52

The mechanisms by which hormonally active agents may alter the risks associated with radiation exposure fall into three broad categories:

1. Sensitizing certain tissues like the breast or prostate to subsequent carcinogenic exposures as a result of changes to the cellar structure of organs caused by early developmental exposure to estrogen mimickers.

2. Potentially initiating tumor development through direct or indirect effects on the DNA that is then promoted by subsequent exposure to radiation.

3. Promoting the growth of cells damaged by radiation into fully developed tumors via the stimulatory effects of hormones like estrogen.

Ongoing research at IEER has found that there is good reason to believe that the risks of developing breast, thyroid, prostate, and other hormonally sensitive cancers from sequential or simultaneous exposure to endocrine disruptors and carcinogenic compounds will, at least in some cases, likely be higher than would be expected from consideration of the exposures in isolation. In other words, the potential for synergistic interactions between radiation and endocrine disrupting chemicals appears to be a real concern in at least some cases.

Concern regarding endocrine disrupting chemicals is increased by the fact that mixtures of different hormone mimickers may interact with each other and cause effects even when each individual chemical is present at levels low enough that they might cause little or no-observable effects on their own. Thus, the presence of multiple known endocrine disrupting compounds within the Hanford site is of particular note. For example, contaminants of potential concern identified in the RCBRA and the 100-B/C Pilot Project for soil, water, and sediment includes the following nine known or suspected endocrine disrupting chemicals

- Aroclor-1254 (PCB mixture)
- Aroclor-1260 (PCB mixture)
- Dichlorodiphenyldichloroethane (DDD)
- Dichlorodiphenyldichloroethylene (DDE)
- Dichlorodiphenyltrichloroethane (DDT)
- Dieldrin
- Endrin aldehyde,
- Methoxychlor
- Phenol.[204]

Among these, the PCBs such as Aroclor-1254 may be of particular concern. For example, in calculating the hazards from fish consumption, the RCBRA notes that the high risks found are due, in part, to the "widespread levels of these and other organic compounds [carcinogenic polyaromatic hydrocarbons and PCBs] being present in fish of the Columbia River Basin."[205]

[204] DOE 2007 p. 4-131, 4-133, and 4-134
[205] DOE 2007 p. 5-58 and 5-62

Thus the possibility of synergistic interactions occurring for the types and levels of hormonally active agents and radionuclides present at Hanford needs to be carefully explored by future research. In the interim, precautionary measures appear to be warranted as part of overall efforts to conduct more holistic risk assessments.

Section 5.3 – Heavy Metals

To begin with, it is important to note that we choose to use the term "heavy metal" in a broad sense in this section to include a wide range of harmful metals and semi-metals irrespective of their atomic mass. In other words, we are focusing on the toxicity of these elements rather than their atomic mass. As with the case of hormonally active chemicals discussed above, care should be taken when examining the impacts of different heavy metals to take into account the specific nature of the metals or metal compounds being investigated since they may result in very different effects from one another. That being said, in our forthcoming review, we identify three principle mechanisms by which exposure to heavy metals may alter the risks associated with radioactivity. These three are:

1. The ability of some heavy metals to disrupt certain parts of the DNA repair mechanisms in cells thereby increasing the likelihood that damage done by radiation or other genotoxic agents will go unrepaired or be misrepaired.

2. The ability of some heavy metals to disrupt the endocrine system (including some that can mimic estrogen in the body) resulting in potentially synergistic interactions with radiation as described in the previous section.

3. The ability of some heavy metals to damage the immune system (especially while it is still developing) resulting in potentially synergistic interactions with radiation due to suppression of the immune system's role in correcting or eliminating cells with damaged DNA as described in the introduction to this section.

IEER's research indicates there are good reasons to believe that the risks associated with combined exposures to heavy metals and radiation may, in some case, be higher than what would be expected from simple considerations of these exposures in isolation.

As with the case of hormonally active agents, the presence of multiple heavy metals within the Hanford site is of particular significance to our recommendation that the potential for synergistic interactions with radiation be explored. For example, contaminants of potential concern identified in the RCBRA and the 100-B/C Pilot Project for soil, water, and sediment include the following 14 heavy metals

- Aluminum
- Arsenic
- Beryllium
- Cadmium
- Chromium (including hexavalent chromium)

- Copper
- Lead
- Mercury
- Nickel
- Selenium
- Tin
- Uranium
- Zinc.[206]

In addition to contamination originating on the Hanford site, several of these heavy metals, including "arsenic, cadmium, chromium, lead, silver, mercury, and zinc are leached from natural rock and mine tailings in northern Washington, Idaho, and Canada" and transported onto the site via the Columbia River.[207]

In a number of cases heavy metals have been found to be at sufficient concentrations in the river corridor that they contribute significantly to the risk at certain sites. For example, three of the terrestrial waste sites in the 300 Area and two sites in the 100 Area which were found to have consistently high risk profiles in the RCBRA included doses from arsenic as a major contributor.[208] In addition, there is a known plume of hexavalent chromium within the 100-D, 100-H, and 100-K Areas with elevated levels detected in nine of the 64 wells sampled for the RCBRA including some detections near the 100-B/C Area as well.[209] In addition, the DOE's estimates for non-tank sources of chromium in the Central Plateau show that concentrations of chromium in the river corridor are expected to be well in excess of EPA's MCL for hundreds of years.[210]

With respect to the impact from these existing levels of heavy metals, it was found that high cancer risks from chromium were associated with the sweathouse pathway for some wells and that arsenic ingestion via uptake into plants "was the primary contributor to background cancer risks" in both the Rural Residential and CTUIR scenarios. In addition, "exposure to arsenic, PCBs, and pesticides via wild plants were of most significance to background cancer risks" for a variation of the CTUIR scenario.[211] We note here that we do not agree with DOE's use of the term "background" to include some areas of Hanford that are contaminated with man-made radionuclides, such as americium-241.

Interestingly, in discussing these findings the RCBRA again implies that even the simple addition of risk may be an overestimate and makes no mention of the potential for synergistic interactions between heavy metals and radiation. Specifically, the RCBRA notes that

> There are numerous instances where chemicals and radionuclides both contribute significantly to a cancer risk result. The most common occurrence is the summing

[206] DOE 2007 p. 4-131 to 4-134
[207] Gephart 2003 p. 5.44
[208] DOE 2007 p. 5-60
[209] DOE 2007 p. 2-10, 4-22 to 4-23, and 5-62
[210] DOE 2009 Vol. 2, Figure U-7 (p. U-8)
[211] DOE 2007 p. 5-61

> of cancer risks via arsenic and radionuclides in soil in the Rural-Residential and CTUIR exposure scenarios. Also, excluding the organic chemical results, arsenic and potassium-40 are both important drivers of cancer risk via fish ingestion. In the case of fish ingestion, as discussed below, the principal uncertainty is likely to be related to the chemical form (and associated toxicity) of arsenic in fish tissue. Although the different bases of the chemical and radionuclide slope factors makes the general summation of these cancer risks suspect, arsenic is (like ionizing radiation) a known human carcinogen and has a slope factor based on human epidemiological data. Therefore, uncertainty introduced by the addition of chemical (arsenic) and radionuclide cancer risks is not as large in this assessment as might more generally be the case.[212]

In light of the potential importance of combined exposures, the possibility of synergistic interactions occurring between heavy metals and radionuclides present on the Hanford site needs to be carefully explored by future research. As with hormonally active agents, precautionary measures appear warranted in the interim as part of an effort to conduct more holistic risk assessments.

Section 5.4 – Special Concerns Relating to Uranium

Before concluding our brief overview of combined exposures, it is important to include a brief word on the potentially unique role of uranium in this context given that it is both a toxic heavy metal and a radioactive element. The recent science concerning the health effects of uranium have been reviewed before by IEER and, as such, we will not reproduce those findings here.[213] In short, it is important to note that the drinking water limit for uranium cited in Section 2.1 was derived primarily on the basis of its chemical toxicity and not its radioactivity.[214] Despite this fact, the RCBRA states that

> In many environmental samples, data obtained for isotopic uranium (in units of activity per mass or activity per volume) could be converted to total uranium data (in units of mass uranium per mass of sample, or mass uranium per volume). In this way, the effects of uranium metal as a kidney toxicant could be assessed in addition to evaluation of radiation dose and cancer risk when only isotopic uranium data are available. This conversion is most important when evaluating depleted uranium, because uranium activity relative to mass is reduced relative to natural uranium. However, the isotopic uranium data evaluated for this report do not indicate the presence of depleted uranium.[215]

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[212] DOE 2007 p. 5-104
[213] See, for example, [Makhijani 2003 p. 3], [Makhijani and Smith 2004 p. 4 to 19], and [Makhijani, Smith, and Thorne 2006 p. 65 to 75]
[214] Makhijani, Smith, and Thorne 2006 p. 65
[215] DOE 2007 p. 5-94

The RCBRA does acknowledge that ignoring the chemical toxicity of uranium underestimates the risk this contaminant poses in the environment at Hanford.[216] This recognition, notwithstanding, the claim that the chemical toxicity of uranium is somehow less important for natural uranium than depleted uranium reveals a lack of appreciation for the potential importance the role its heavy metal properties may play in determining the risks of uranium exposure. This potential has long been recognized in reviews dating back 10 years from organizations such as the National Institute of Medicine's Committee on Health Effects Associated with Exposures During the Gulf War, the World Health Organization, the National Research Council, and the Royal Society, which have all endorsed the need for additional research on the chemical toxicity of uranium including its potentially negative impacts on the brain, on the reproductive tract, on the blood forming system, and on the skeleton.[217]

In the present context, we are most interested in the potential self-synergistic interactions uranium may display due to its dual nature as both a radiological and chemical toxin. Interestingly, this single contaminant incorporates many of the concerns we have recommended considering in the broader context of combined exposures. Specifically, the mechanisms by which uranium's chemical properties as a heavy metal may interact with its radiological properties as an alpha emitter include

1. Interactions between the ability of uranium to cause oxidative stress as a heavy metal and for its ability to cause genotoxic damage as a radionuclide, in this case there may be local mechanisms of interaction based on the production of highly damaging water radicals or there may be interactions arising from the potential of these effects to allow uranium to act as both a tumor initiating agent and as a tumor promoting agent.
2. Interactions mediated through impacts of uranium on the hormonal system; of particular interest in this respect is the recent evidence that uranium, like a number of heavy metals, can mimic the behavior of estrogen and thus influence its carcinogenicity through impacts on the endocrine system.
3. Interactions mediated through impacts of uranium on the immune system, since uranium may impact the immune system directly via its heavy metal properties as well as through its ability to concentrate in the bone and thus cause radiation damage to the parts of the immune system associated with the bone marrow.

As in the above discussions of hormonally active agents and of heavy metals, our recommendation to investigate the potential for uranium to display synergistic interactions is supported by the widespread nature and importance of this contaminant in the river corridor at Hanford. As already noted, uranium was among the 14 heavy metals identified as a contaminant of potential concern by the RCBRA and the 100-B/C Pilot Project.[218]

The potential concern with uranium is highlighted by the fact that it contributed more than 70 percent of the maximum dose at three of the eight waste sites with the highest terrestrial doses

[216] DOE 2007 p. 5-92
[217] Fulco, Liverman, and Sox 2000 p. 327, WHO 2001 pp. 148 to 149, Royal Society Part II 2002 pp. 66 to 68, and NAS-NRC 2003 pp. 67 to 68
[218] DOE 2007 p. 4-131 to 4-134

while it contributed 82 percent of the average dose at both of the waste sites showing the highest average doses.[219] Contributing to this is the fact that uranium is present in a groundwater plume affecting the 300 Area.[220] In fact, the data included in Appendix G to the RCBRA show that at least six of the 16 wells reporting inorganic uranium, already have uranium concentrations in groundwater that exceed the 30 micrograms per liter drinking water limit with a maximum concentrations of 115 micrograms per liter. In addition, another five wells showed uranium concentrations between 20 and 30 micrograms per liter. All told, the wells exceeding 20 micrograms per liter of uranium represent nearly 70 percent of the wells for which inorganic uranium concentrations were reported. There is also some evidence that uranium may already be taken up by aquatic biota living in contaminated portions of the Columbia River.[221] In addition, uranium migration from the Central Plateau into the River Corridor groundwater is expected to continue and to remain in excess of the present drinking water limit of 30 micrograms per liter for about 2,000 years.[222]

In light of the potential importance of exposures to uranium and the possibility of synergistic interactions occurring between its own heavy metal and radiological properties, special care should be taken with this contaminant and future work needs to examine what role, if any, these concerns may play in the risks it poses at Hanford. In the interim, precautionary measures similar to those for heavy metal exposures in general appear to be warranted to support the goals of holistic risk assessment.

Section 5.5 – Synergy Uncertainty Factors and Future Research Needs

Given the complexity of the waste streams at Hanford as well as the fact that contaminants are being transported on site from locations upriver, it would not be possible, even in principle to experimentally investigate all of the possible combinations and permutations of exposures to determine whether synergistic or other kinds of interactions will occur. However, in light of the biologically plausible mechanisms of interaction between radiation and hormonally active agents or heavy metals, our recommendation for the interim period is to adopt what the National Research Council of the U.S. National Academies of Science called synergistic "uncertainty factors" in its 1989 *Drinking Water and Health, Volume 9: Selected Issues in Risk Assessment.*[223]

At its most basic, a synergistic uncertainty factor is a scaling factor that reduces the allowed exposure level of certain agents in order to provide a safety margin should they, in fact, interact in a synergistic manner. In summarizing its conclusions regarding the need to include these kinds of uncertainty factors in the setting of regulatory limits, the National Research Council committee noted that

> The issue of toxic interactions - synergistic or antagonistic - is central in the development of a risk assessment strategy for chemical mixtures in drinking

[219] DOE 2007 p. 5-60 and 5-65 to 5-66
[220] DOE 2007 p. 2-10 to 2-11
[221] DOE 2007 p. 5-11
[222] DOE 2009, Vol. 2, Figure U-9, p. U-9.
[223] NAS-NRC 1989 p. 98-100 and 128-129

> water. Even though the concentrations of contaminants in most sources of drinking water for the general public are likely to be very low, there is insufficient evidence about the toxicity of chemical mixtures after long-term, low-dose exposure to support a definite conclusion that toxic interactions are absent under these conditions. For instance, a combination of chemicals, even at low concentrations, could conceivably act to modify the immune system, thereby compromising natural defense systems. Evidence supporting the existence or absence of such a response process is clearly not available for all relevant mixtures. The argument for the consideration of greater than response additive effects is strengthened by the possibility that water sources in heavily polluted areas (e.g., hazardous-waste sites or point-source accidental spills) contain much higher concentrations of contaminants. Thus, at least a small fraction of the population is sometimes exposed to relatively high (parts per million) concentrations of mixtures of chemicals in drinking water.[224]

It is interesting to note here, that this quote highlights one of the potential mechanisms of synergism discussed in this section, namely interactions mediated through damage to the immune system.

In determining what uncertainty factor, if any, to apply, the National Research Council recommended that

> The UF [uncertainty factor] could vary from 1 to 100, depending on the amount of information available and the concentrations of the contaminants. If a great deal of toxicologic information is available on the individual contaminants, if toxic interactions are not likely (on the basis of the knowledge available), or if the concentrations of the contaminants are "low," the UF might be set at 1 (thus assuming simple additivity). If less is known about the toxicity of individual components and the concentrations of the contaminants are higher, the UF might be set at 10. The greater the uncertainty involved (because of the lack of information) and the higher the concentrations of the contaminants, the higher the UF would be set.[225]

The National Research Council committee went on to note that "[s]ocietal and policy concerns about the existence of such [synergistic] interactions could lead to the introduction of further uncertainty factors in risk assessment."[226] While the authors were focused on the toxic effects of chemical mixtures in this report, a similar scheme could be adopted for protection against potential synergistic interactions between radiation and chemicals as well.

In adopting such a scheme, the first question that must be answered is which mixtures or combined exposure scenarios should be included and what is the appropriate size for the associated uncertainty factor when they are. As noted by the National Research Council, the "[u]ncertainty factors should not be uniform, but should increase with increasing exposure and

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[224] NAS-NRC 1989 p. 128
[225] NAS-NRC 1989 p. 129
[226] NAS-NRC 1989 p. 131

decrease with increasing knowledge about the agents in mixtures."[227] In seeking to prioritize which exposure scenarios potentially pose the greatest risk and, thus, should have the largest uncertainty factors, we will adopt the criteria proposed by Sexton and Hattis. In their methodology, a "high-priority mixture" would be one that had the following four attributes:

- Scope of exposure: A large number of organisms, communities, or populations are exposed to the mixture and/or a significant number of susceptible organisms, communities or populations are exposed to the mixture.

- Nature of exposure: The magnitude, duration, frequency, and/or timing of exposure to the mixture raises concerns about possible adverse effects.

- Severity of effects: The known or suspected adverse outcomes of exposure to the mixture are of a nature or consequence that suggests risks are likely to be unacceptable.

- Likelihood of interactions: Adverse effects from exposure to the mixture are not likely to be characterized adequately based on knowledge of known effects of individual mixture components acting separately.[228]

While this set of criteria was developed for chemical mixtures, their logic applies equally well to combined exposures to radiation and chemicals. Therefore, applying these criteria to the contaminants prevalent on the Hanford site, we find that the combination of radiation with either endocrine disruptors or heavy metals would appear to qualify as high-priority mixtures. This is because of the widespread nature of the contamination, the variety of hormonally active chemicals and heavy metals found in the environment at Hanford, the importance of these contaminants to the present day risk even in the absence of interactions, and the likelihood that there may be synergisms between these classes of contaminants and radioactivity given the multiple plausible mechanisms through which interactions could be mediated.

Ongoing research at IEER indicates that the existing experimental evidence on interactions between radiation and either endocrine disruptors or heavy metals indicates that a reasonable value for an interim uncertainty factor may be in the range of 2 to 20. In order to simplify the application of this proposal, we recommend that until more definitive research is completed, a synergy uncertainty factor of five (the geometric mean of 2 and 20 rounded up to the nearest integer) should be adopted for risk assessments at Hanford whenever radionuclides are present in combination with potentially significant levels of hormonally active chemicals or heavy metals.

While the use of this interim uncertainty factor would provide an important level of safety in dealing with the immediate concerns of potential synergistic interactions in the context of holistic risk assessment, future research should seek to build more robust and integrated means of dealing theoretically with combined exposures and should seek to examine the kinds of waste mixtures prevalent on the Hanford site and those likely to be carried onto the site via the Columbia River. Many of the needs for future research in this area have been outlined

[227] NAS-NRC 1989 p. 131
[228] Sexton and Hattis 2007 p. 826

previously by the United Nations Scientific Committee on the Effects of Atomic Radiation and supported by the BEIR VII committee of the National Research Council.[229] Carrying out such a research plan would help to more clearly and consistently define how synergistic uncertainty factors should be set, when they should be used, and how large they should be when included in the setting of cleanup goals. As a final note, we recommend that these investigations focus on the most vulnerable segment of the population. Given the differences between the physiological response of women and men to some chemicals and the evidence of dramatically different impacts with age at exposure in some cases, the need to focus on the most vulnerable populations should be carefully considered in the design of all experiments.[230] In particular, the possibility that exposures during one critical window (such as during in utero development) or that exposure of one particular sex may be the determining factor in setting the size of the synergistic uncertainty factor for the entire population should be carefully explored in all cases.

[229] UNSCEAR 2000 p. 217 and NAS-NRC 2006 p. 330

[230] The need for regulations to protect the most vulnerable members of society regardless of age or sex was the central theme of our previous work, *Science for the Vulnerable* [Makhijani, Smith, and Thorne 2006]

Section 6: Issues Regarding the Ecological Risk Assessment

Our final discussion considers the treatment of ecological risk in the RCBRA. While this section will focus specifically on issues raised by this risk assessment, many of the concerns and recommendations have broader implications for how ecosystem impact assessments should be conducted in general for sites with long-lived and highly complex wastes. Unlike the methodology for the protection of humans which has undergone extensive refinement for over the last century, the question of how to ensure protection of ecosystems has been widely considered for a substantially shorter length of the time and, as such, is far less developed. In summarizing the historical development of radiation protection and the need to develop a methodology for ecological risk assessment, the International Commission on Radiological Protection noted in 2003 that

> From a historical point of view, the anthropocentric focus of radiological protection has been prioritized because of the need to protect humans in different circumstances (medical and occupational exposures, and exposures to the public). In doing so, parts of the environment (the human habitat) probably have been afforded a fairly good level of protection through the application of the ICRP system for protection. Nevertheless, there are clearly circumstances where the Commission's current view is insufficient to protect the environment, or even incorrect. Examples are environments where humans are absent (e.g. aquatic environments), situations where humans have been removed for their own safety (e.g. in the case of intervention), and circumstances where the distribution of the radionuclides in the environment is such that the exposure to humans would be minimal, but other members of the flora or fauna could be considerably exposed. ***Another problem is that the implicit level of protection (i.e. not endangering whole species) is inconsistent with sustainable development and many current environmental protection policies, acts, and regulations.***[231]

As a result of this relatively recent recognition of the need to protect the environment as a whole and not just humans, there is no single strategy for conducting such analyses. For example, in documentation supporting the DOE's approach to ecological risk assessment published in 2002 it notes that

> Nationally and internationally, no standardized methods have been adopted for evaluating doses and demonstrating protection of plants and animals from the effects of ionizing radiation.[232]

In response to this lack of consensus, the ICRP has begun a major international effort to develop a methodology for conducting ecosystem impact assessments. As part of this effort, the ICRP published *A Framework for Assessing the Impact of Ionising Radiation on Non-human Species* in 2003 which advocates a "reference fauna and flora" approach in analogy with the "Reference Man" approach still common in radiological protection of humans (see Section 2).[233] While the

[231] ICRP 91 p. 239 (emphasis added)
[232] DOE 2002 p. M1-3
[233] ICRP 91 p. 250

ICRP notes that "[t]he choice of primary reference organisms for flora and fauna will depend on the future development of environmental protection from radiation"[234] it goes on to conclude that

> Ideally, one might like to select those organisms that were known to be particularly sensitive to radiation, or were known to be vital components of particular ecological communities or expected to receive higher exposures because of their habitat (e.g. sediment-dwelling organisms when radionuclides will accumulate in sediment). But one also has to be pragmatic, and therefore consider the amount of radiobiological information that is already available on them, including data on radiation effects. They would also have to be amenable to future research in order to obtain the necessary missing data. One would also have to consider the extent to which they have some form of public or political resonance, so that both decision makers and the general public at large are likely to know what these organisms actually are, in common language - such as a duck or a crab.[235]

The use of a reference biota approach for ecological protection, however, poses a great many complications and concerns even beyond those that arise with the use of Reference Man for regulations aimed at protecting humans.[236] For example, unlike the case of the radiological protection of humans where cancer is generally accepted as the end point of greatest concern in most cases, there is no current international consensus concerning the kinds of biological effects that are important to consider in the protection of the environment more broadly. In addition, the large number of species present in most ecosystems, the many direct and indirect interactions between those species, the wide range of radio-sensitivities between and even within species, and the interactions (including potentially synergistic interactions) between radiation induced effects and other environmental stressors such as chemical pollutants, fishing/hunting by humans, or ecosystem fragmentation add greatly to the problem of determining suitably protective models for individual reference plants and animals.

In the remainder of this section, we will consider many of these complications and concerns as they relate to the methodology used by the RCBRA, which is a hybrid of dose exposure modeling for reference animals with environmental sampling, laboratory experiments, field observations, and literature review. While this approach has many important strengths it also has some equally significant weaknesses. As in our discussion of the Tribal Rights scenario in Section 4, it is important that a suitably protective risk assessment be conducted at the start to avoid the possibility of having to return to further remediate previously cleaned-up areas in the future. This concern is already hinted at in the RCBRA when it notes that

> Because radionuclide cleanup levels for the protection of human health were considered generally more conservative than ecological cleanup levels, it was concluded at that time that the interim action RODs [records of decision] would also protect ecological receptors. For example, removal of soil and debris

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[234] ICRP 91 p. 250

[235] ICRP 91 p. 251

[236] For a discussion of the problems with the use of Reference Man in the protection of humans see, for example, [Makhijani, Smith, and Thorne 2006] and [Makhijani 2009].

> exceeding the human health-based goals and replacement (i.e., backfilling) with clean material was expected to meet the objective of ecological receptor protection.[237]

However, just as noted in the ICRP quote with which we began this section, the RCBRA goes on to state that "this assumption will be reevaluated in the RI [remedial investigation], as some interim action RODs may not be protective of particular ecological receptors."[238] Thus, the goal of ensuring adequate environmental protection from the start is an important goal of our recommendations for the revised RCBRA.

In this section, we will begin by assessing the hybrid ecological risk assessment framework put forth in the RCBRA and focus in particular on its use of the "no observable effects level" in evaluating the impacts of existing contamination on plants and animals and its lack of considering future contaminant transport and its impacts on the ecosystems of the river corridor. We will then turn to a discussion of issues surrounding both data gaps and data quality in the current ecological surveys. Finally, we will turn to the need for the revised RCBRA to include a more holistic approach to ecosystem risk assessment incorporating interactions between multiple stressors projected over long timescales and over large areas. Given the enormous complexity of the tasks involved with ecological risk assessment we will conclude this section by proposing a research plan for how to begin moving towards a methodology capable of ensuring adequate protection of the environment consistent with the qualitative goals outlined in Section 4.3 of ensuring protection of the natural and cultural resources of the Yakama and other Native Peoples.

Section 6.1 – Assessment Methodology for Ecosystem Impacts

The ecological risk assessment in the RCBRA does not follow the model of dose exposure modeling proposed by the DOE in its 2002 *DOE Standard: A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota.* Instead, as noted above, the RCBRA opted for a hybrid approach focusing on multiple lines of evidence with assessment endpoints "developed from the ecological management goals, the conceptual exposure model, and trophic relationships among ecological receptors."[239] In particular, the risk assessment was based on the following

- **Measures of Effect**:
  - Literature toxicity information
  - Literature tissue effect levels
  - Laboratory toxicity tests
  - In situ riverbed survival
  - Biological condition
    - Gross field measurements
    - Histopathology measurements

[237] DOE 2007 p. 2-8
[238] DOE 2007 p. 2-8
[239] DOE 2007 p. ES-14

- **Measures of Ecosystem/Receptor Characteristics:**
  - Field measures
    - Abundance
    - Diversity
    - Community structure
    - Reproduction observed in field
      - Gender ratios
  - Abiotic data (pH, soil texture, etc.).[240]

In adapting these measures to the task at hand, a collection of specific questions were posed within the RCBRA whose answers were meant to provide the basis for making decisions regarding any potential impacts on the ecosystem. For the "upland zone" these questions were

- Do contaminant concentrations in shallow zone soils decrease plant survival or growth?
- Do contaminant concentrations in shallow zone soils affect soil macroinvertebrate survival, growth, abundance, or diversity?
- Do contaminant concentrations in shallow zone soils and food decrease middle trophic-level (herbivorous, insectivorous, or omnivorous) species (lizard, bird, and mammal) survival, growth, reproduction, relative abundance, juvenile recruitment, or affect balanced gender ratios?
- Do contaminant concentrations in shallow zone soils and food decrease carnivorous bird or mammal survival, growth, or reproduction?"[241]

The same questions were used for the "riparian zone" with the addition of

- Do contaminant concentrations in food decrease aerial insectivore survival, growth, reproduction, or relative abundance?[242]

Finally, the assessment questions for the "near-shore aquatic zone" were

- Do contaminant concentrations in sediments and pore water decrease plant survival or growth?
- Do contaminant concentrations in sediments and pore water affect benthic macroinvertebrate survival, reproduction or growth, diversity, and/or relative abundance?
- Do contaminant concentrations in sediments, pore water, and food decrease amphibian survival, growth, reproduction, or relative abundance?
- Do contaminant concentrations in sediments, pore water, and food decrease carnivorous fish, bird, or mammal survival, growth, or reproduction?

[240] DOE 2007 p. ES-15
[241] DOE 2007 p. 6-3 to 6-4
[242] DOE 2007 p. 6-9

- Do contaminant concentrations in sediments, pore water, and tissue increase histopathological indicators of effect for clams or fish?[243]

As noted, the methodology utilized by the RCBRA has many important strengths which should be preserved and expanded upon for the final risk assessment. For example, its focus on multiple lines of evidence and endpoints including (1) comparisons of contaminant concentrations to values determined to be harmful, (2) laboratory assessments of biotic health, and (3) field measurements for population effects help to ensure that direct impacts on the individual species studied will have a higher probability of being identified than if fewer metrics were used. Second, its assumption that all of the contaminants in abiotic media like soil are bioavailable is properly protective for ecosystem modeling given the lack of information on chemical form for the contaminants and the long-term changes that may result in contaminants due to natural processes.[244] And third, the use of toxicity bioassays in addition to literature reviews to evaluate potential impacts is important given "their ability to provide site-specific information and ecologically relevant effects data" as well as their ability to "offer site-specific information on adverse effects of contaminant mixtures and on contaminant bioavailability for Hanford Site aquatic media."[245]

However, there are also several weaknesses in RCBRA's approach; they should be remedied in the final risk assessment. First, and likely most important, is the fact that it seeks only to evaluate the impacts of existing contamination on the ecosystems of the 100 and 300 Areas without any treatment or consideration of the long-term impacts from contaminant transport and land use changes. We have already noted that the DOE itself projects that contaminant transport from the Central Plateau will severely pollute the River Corridor for thousands of years, unless there is a through cleanup plan.[246] The second weakness relates to the use of no observable effect concentrations from published studies which may underestimate the actual effects on wildlife due to the statistical design of the studies and their typical exclusion of multiple contaminant mixtures or other co-stressors. We will consider both of these concerns below. Additional weaknesses involving the quality of data and its sampling methodology as well as its lack of integrated ecosystem modeling will be explored in the following sections.

Section 6.1.1 – Lack of Consideration of Future Site Changes

As discussed in the context of human risk assessment in Section 3.1, the choice of the RCBRA to evaluate the waste sites individually and to ignore potential contributions from the long-term migration of contaminants both within the river corridor as well as those migrating from the more heavily contaminated 200 Areas is a serious deficiency of this analysis. For example, the RCBRA concluded that "[r]isks to upper trophic-level mammals [in the riparian zone] are negligible on the basis of modeled dietary exposure."[247] Included in this model, however, are

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[243] DOE 2007 p. 6-9
[244] DOE 2007 p. 6-19
[245] DOE 2007 p. 6-23
[246] See Yakama Nation 2010, Attachment 3 for a discussion of the importance of meeting ARARs and the CERCLA and MTCA risk limits in the 200 Areas.
[247] DOE 2007 p. 6-39

badgers which are assumed to be "exposed to multiple media through ecological exposure models, obtaining soil in their diet, ingesting small mammals, and drinking water from the river."[248] Thus, the lack of consideration of doses due to future contamination of the river that may result due to contaminant transport from the 200 Areas may significantly underestimate the risks to animals like the badger. The RCBRA conclusion that risks to these animals are "negligible" is premature at best and cannot be made without considering contaminant transport from other areas into the river corridor.

As another example, the exclusion of natural processes like erosion from consideration also affects the long-term validity of conclusions in the RCBRA because erosion and the mixing of soil due to burrowing animals may lead to larger exposures in the future by bringing more contaminated soils from the subsurface within reach of the typical depths of roots or animal burrows. A related concern is the lack of adequate consideration in the RCBRA for how the plant and animal life present on the site will be expected to change over time as the waste sites are allowed to return to a more natural configuration after remediation and how this may affect the exposure profile of and impacts on future plants and animals. For example, some of the more contaminated waste sites are currently kept free of vegetation significantly reducing their potential impact on local plants and animals, but once remediation is complete efforts to restore these areas will be undertaken.[249] Specifically, as noted in the RCBRA

> Some areas that are no longer used for waste disposal, construction activities, or site operations have begun to revegetate naturally to communities dominated by gray rabbitbrush with an understory of Sandberg's bluegrass, bulbous bluegrass (Poa bulbosa), and cheatgrass.... To promote the reintroduction and colonization of native species, remediated upland CERCLA waste sites are revegetated with the goal of reestablishing sagebrush/Sandberg's bluegrass communities with a mixture of other native grasses and forbs adapted to rocky soils. Following restoration, the vegetation type, density, and species diversity may not be the same as before initial disturbance, due to the change in soil structure. As the restored plant communities mature, however, improvements in shrub coverage will provide important habitat for native wildlife species. In addition, plants that rely on fine-textured soils may occur on restored or naturally recovering sites, but typically are not expected to be abundant.[250]

In addition to plant life and small mammals, the restoration of the waste sites to a more natural habitat will also affect the populations of other animals. This may include an increase in the population of large mammals such as the elk as well as of birds such as the horned lark, western meadowlark, savannah sparrow, and loggerhead shrike.[251] Thus, the ecosystems that will be affected in the future may be substantially different from those common in the operational areas of Hanford today and care must be taken to ensure that no undue harm will occur in these future ecosystem as a result of the contamination left behind based on considerations of only today's plant and animal life. As such, for the protection of humans, as well as that of the rest of the

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248 DOE 2007 p. 6-39
249 DOE 2007 p. 2-4 and 2-16
250 DOE 2007 p. 2-14
251 DOE 2007 p. 2-15

environment, it is essential that a final risk assessment include the modeling of future contaminant transport and other changes to the site in order to ensure that the remediation efforts are adequately protective of the ecosystem over the long timescales for which the contaminants at Hanford will remain hazardous.

Section 6.1.2 – Use of the "No-Effect" Level

With respect to the use of studies concerning the impacts on plants and animals from chemical and radiological toxins, the RCBRA notes that

> It is important to recognize that while this is a baseline risk assessment, it relies primarily on no-effect level benchmarks and TRVs [toxicity reference values], which are normally used in an ecological screening-level assessment. Exceedance of no-effect levels does not necessarily indicate a risk. Use of no-effect levels is another contribution to the conservatism inherent in this risk assessment.[252]

However, it is possible that the reverse is true as well, which is to say, that staying below the reported no-observable effect concentration (NOEC) does not necessarily indicate the complete absence of risk. There are two principal reasons for this statement. The first relates to the fact that many studies of contaminants published in the literature "usually evaluated just one contaminant at a time" as highlighted by the RCBRA itself in discussing its choice to include toxicity bioassays as a high-weighted line of evidence.[253] Thus, as noted in Section 5 in the case of human exposures, the interactions between multiple contaminants can make combined exposure hazardous even if it would have been considered safe had the exposures occurred in isolation. In the case of environmental impact assessments this possibility is made even more important by the fact that contaminants can interact, not only with other contaminants, but also with other kinds of stressors such as overfishing or droughts.

Even in cases where only a single contaminant is present, however, the methodology typically used to determine no-observable effect concentrations can still underestimate the actual risk due to the statistical design of these studies. To explain why this can occur, we note that there are two basic kinds of errors that may be made in these kinds of toxicity studies against which experimenters must protect. The first, called "type I" error, occurs when a substance is determined to have been dangerous at a given concentration when it was, in fact, not dangerous at that level of exposure. But this type of test is not a strong statement about safety. It just states that when we conclude something is unsafe that we are reasonably sure it is unsafe. It does mean we are reasonably sure about the safety of something. In order to make a statement about safety, a different test is needed; that test is for a "type II" error. This test is just the opposite of a type I test. A type II test makes a statement about safety. When the substance (or collection of substances passes a type II test, then we are reasonably sure it is safe. We minimize the risk of saying something is harmless at a given concentration when it is, in fact, dangerous.

[252] DOE 2007 p. 6-21
[253] DOE 2007 p. 6-23

Experiments used to determine the toxicity of a given compound are typically designed to control for errors of type I and not type II.[254] For example, as summarized by Chapman et al.

> Tests tend to be low powered because power, or one minus the type II error, as a concept is not taken seriously and is not clearly understood. In contrast α, or type I error, the probability of obtaining a significant result when there is no difference between treatments, is taken very seriously. It is usually set to be very small, typically 5% or less, and gives protection from drawing the wrong conclusion when no difference between the treatments exists. Thus, current tests tend to protect against drawing the wrong conclusion when a treatment has no effect but give little protection against drawing the wrong conclusion when the treatment does have an effect. In an ideal world a test would give protection in both situations, but in practice this results in high numbers of replicates that are too costly.[255]

As can be seen in this quotation, a Type I test is often suitable in pharmacological situations, where a reasonable level of certainty is sought that a new course of treatment is actually more effective than another. But for ecosystem harm, we want to be reasonably sure that something is actually safe – that it will not cause harm. For this a type II test is more effective.

In addition, Chapman et al. "found that the choice of software and computing options and control options also influences the NOEC determination" and that "[d]ifferent statistical methods (Dunnett's, Duncan's, Tukey's, etc.) will not give the same results."[256] This is important, because it means that, as noted by Crane and Newman, "[t]he ability to detect a statistically significant effect depends not only on biological response but also on the experimental design."[257]

As an example of how this uncertainty can manifest itself in particular circumstances, two laboratories sought independently to determine the NOEC for the effluent from a paper mill in relation to impacts on oyster and mussel larvae. In the case of the oyster larvae, the two labs disagreed about the NOEC by a factor 2.2 while the labs disagreed concerning the no-observable effect concentration for the mussels by a factor of nine.[258] In another example, a review of studies concerning the NOEC for chronic exposure of fish to chemicals like 3,4-dichloroaniline, linear alkylbenzene sulfonate, and sodium pentachlorophenol found concentrations that varied by at least a factor of 3.4, with even larger variations seen among some individual tests.[259] As a result, Crane and Newman concluded that "[t]his analysis suggests that the NOEC is neither a consistent summary statistic nor an indicator of safe concentrations of toxic chemicals."[260]

Thus, the no-observable effect concentration which is measured by these tests can, in some cases, be a poor approximation of the actual concentration at which no effects occur (i.e., of the

[254] Canham, Cole, and Lauenroth 2003 p. 354
[255] Chapman, Caldwell, and Chapman 1996 p. 79
[256] Chapman, Caldwell, and Chapman 1996 p. 78
[257] Crane and Newman 2000 p. 516
[258] Chapman, Caldwell, and Chapman 1996 p. 77
[259] Crane and Newman 2000 p. 516 and 518-519
[260] Crane and Newman 2000 p. 516

true no-effect level, if one exists).[261] As summarized by Crane and Newman, "it would be unreasonable to suggest that such NOECs provide reliable information on safe concentrations of toxic chemicals" and that

> In most cases, a risk assessor using NOEC values will have no way of knowing whether these values are indicative of low, medium, or high effects on the endpoint of interest, but the NOEC is rarely if ever an indicator of no effect.[262]

While the use of no-effect levels from literature and toxicity bioassays is not the sole line of evidence upon which the present ecosystem assessment is based, a discussion of the uncertainties inherent in the use of these values should be included in the revised RCBRA and care should be taken to not give the impression, as was done with the quote at the start of this section, that staying below the reported no-effect levels is necessarily safe.

We recommend that type II error be determined (in addition to type I errors) when determining the risk of ecosystem harm.

Section 6.2 – Data Gaps and Data Quality Issues

Turning to a consideration of the data used in the ecological risk assessment for the RCBRA we will find that there are three outstanding concerns relating to its quality. The first issue involves data gaps that were identified by the RCBRA concerning the four ecological receptors that may already have been impacted by the current levels of contamination on site. These data gaps also include two cases where plant or animal species were not included in the RCBRA's consideration of ecosystem impacts due to failed collection efforts. In all cases, these data gaps are planned to be filled by the Inter-Areas shoreline assessment and will therefore only briefly be considered here. The second issue relates to areas where additional data collection is needed to address uncertainties in the existing data such as measurements of the concentration of contaminants in plants and animals including measurements of the transfer factors for particular organs or organelles. In addition, we will find that longer term field observations are required to ensure accurate characterization of the site as a whole.

Section 6.2.1 – Data Gaps that are Planned to be Filled

In the RCBRA, there were four areas where potential impacts from the existing levels of contamination could not be ruled out within the framework adopted by the DOE. The first of these involved potential impacts on sediment-dwelling, aquatic macroinvertebrates such as clams. For example, it was found that "[c]lam survival was significantly reduced in the chromium plume shoreline locations" but that this "reduced clam survival was significantly correlated with the confounding factor of sediment grain size."[263] In particular, it was found that

[261] Chapman, Caldwell, and Chapman 1996 p. 78
[262] Crane and Newman 2000 p. 519
[263] DOE 2007 p. ES-21

"clam survival residuals (the variability in survival not explained by particle size) showed no statistically significant relationships with pore water chromium or other COPCs" but that "significant negative correlations were observed between clam survival and primarily pesticide COPC concentrations in clam tissue."[264] Similar impacts on survival in the chromium plume were found for the amphipod Hyalella azteca, although, like clams, the reduction in survival of this receptor was also correlated with sediment particle size.[265]

In addition to these observations, it was found from examinations of individual clams that the "loss of digestive tubular epithelial cells" and the "[n]umber of reproductive system follicle cysts -- a fibrous reaction around and within reproductive follicles" were statistically different between the operational and reference sites.[266] Interestingly, one of these two histopathological measurements was elevated for clams collected from the operational areas while the other was elevated for clams collected at the reference location. As a final point, it was noted in the RCBRA that

> The concentrations of all COPCs in benthic macroinvertebrate tissue are less than tissue effect levels (tissue concentrations that have been reported to be associated with adverse effects on aquatic organisms) with the exception of selenium. Selenium concentrations at upstream and downstream locations are greater than tissue effect concentrations published in the literature.[267]

However, as noted above, the use of no-observable effect levels from the literature may not always be a meaningful measure of protection. The open questions relating to the potential impacts on near-shore macroinvertebrates will need to be addressed in the "Inter-Areas shoreline assessment".[268] In particular, care should be taken to clarify the role of particle size and other confounding factors.

The second open question regarding potential impacts from present day contamination relates to aquatic plants. As summarized by the RCBRA

> Uncertainties exist with regard to possible impacts on near-shore plants from sediment COPCs; these uncertainties can be addressed with the expanded sediment bioassay data being compiled for the Inter-Areas shoreline assessment. For sediment, phytotoxicity bioassay (with pakchoi) results suggest that growth was reduced in sediments collected in the strontium plume associated with the 100-N Area. However, there are no relationships between the bioassay results and strontium levels in any of the sediment sampling locations. In addition there are very few macrophytes along most of the operational areas, most likely due to the strong and variable river flows.[269]

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[264] DOE 2007 p. 6-42
[265] DOE 2007 p. 6-43
[266] DOE 2007 p. 6-42 and 6-118
[267] DOE 2007 p. ES-21
[268] DOE 2007 p. ES-22 and 6-46
[269] DOE 2007 p. 6-40

Particular care should be given to exploring these potential impacts on aquatic plants given the fact that, as noted by the DOE in 2002, "[t]here are no DOE or internationally-recommended dose limits established for aquatic plants, primarily due to lack of data on radiation effects to these organisms."[270] In addition, as discussed in Section 4.1.3, Indian celery is a culturally significant plant used both as a food and medicine by the Yakama that grows in springs and streams.[271] Thus, care should be taken to determine whether impacts on near-shore plants may be of relevance to Indian celery as well.

The third identified data gap relating to potentially affected species involves insectivorous bats. Specifically, insect consuming bats had projected hazard indices associated with metals such as antimony and selenium that were significantly higher in the operational areas than those calculated for the reference area.[272] Thus, as noted in the RCBRA "[a] broader scale assessment of bats including the Inter-Areas shoreline assessment is warranted" and that this additional sampling is "important to better understand the sources of the COPCs contributing to risk to bats."[273]

Finally, the fourth data gap concerning potentially affected components of the Hanford ecosystem is related to the lack of information on contaminant concentrations in amphibians and reptiles. In the case of amphibians, this lack of information was the result of failed efforts to collect sufficient numbers of amphibians in the field.[274] This lack of information concerning exposures of amphibians and reptiles, however, was downplayed by the RCBRA. For example, it notes that

> While reptiles are an important component of arid environments like the Hanford Site, the general dearth of toxicity information for lizards and snakes limits the utility of exposure modeling to this group. Amphibians can be found at locations within the Hanford Site, but they too are limited with regard to information on toxicity based on food ingestion pathways. Consequently, reptiles and amphibians were not evaluated in the ecological exposure modeling component of this risk assessment. It is noted that amphibians are broadly protected by some abiotic media benchmarks for direct exposure (e.g., water quality protection levels). This project is directly assessing effects on amphibians from COPCs in pore water using the FETAX bioassay...[275]

Significantly, the FETAX bioassay, a four-day exposure test that evaluates survival, growth, and deformities, did find statistically significant differences between the operational and reference areas in terms of the survival and growth of amphibians. However, the RCBRA goes on to classify these differences as "slight and likely not ecologically relevant" before concluding that "available data do not suggest that COPC concentrations are adversely affecting amphibian survival and growth."[276] As with the other data gaps discussed in this section, the Inter-Areas

[270] DOE 2002 p. M2-23
[271] Ridolfi 2007 p. 21
[272] DOE 2007 p. 6-50 and 6-51
[273] DOE 2007 p. 6-50
[274] DOE 2007 p. ES-22, 6-18, 6-21, 6-46, and 6-137
[275] DOE 2007 p. 6-18
[276] DOE 2007 p. 6-47

shoreline assessment is planning to examine this possible impact further, and care should be taken to determine whether these potential impacts are ecologically relevant or not. The RCBRA conclusion is not warranted at present. Rather, longer term observations are needed in view of the finding of statistically significant differences; a Type II statistical error tests should be also applied: how sure are we that the exposures are not causing harm?

Section 6.2.2 – Additional Data Gaps

In addition to the data gaps identified in the previous section which are planned for further investigation as part of the Inter-Areas shoreline assessment, there are a number of other uncertainties and data quality issues that should also be addressed prior to the conclusion of the revised RCBRA. For example, the RCBRA includes a discussion of additional data gaps resulting from incomplete sampling or concerns regarding laboratory procedures. A partial list of the data that was planned for collection but which was not obtained includes

- Sandberg's bluegrass bioassay results were not useable due to data quality problems with the analytical laboratory.

  ....

- Relative abundance of terrestrial invertebrates was not obtained because sample collection methods included some hand-collected invertebrates in addition to some invertebrates collected using systematic sampling methods (pitfall traps).
- Nest success for kingbirds was not measured due to frequent nest predation by ravens and crows.[277]

In addition to this kind of information, which was never collected or properly analyzed, additional data gaps arise from uncertainties affecting the data that was collected for the RCBRA. In particular, several of the sources of uncertainty that remain concerning plant and fish exposures should be more thoroughly quantified by additional environmental sampling as discussed below. Many of the proposals in our recommended research plan outlined at the end of this section would be of particular use in addressing these questions.

Starting with measurements of the contamination levels of fish and the impacts that might result from that exposure, the uncertainties identified within the RCBRA include

- Tissue effects concentrations based on potentially dissimilar species from those occurring on site
- Analytical measurement uncertainties (e.g., detection limits are typically elevated in tissues because of matrix interferences). Considering the mobility of fish, uncertainty in whether tissues of electroshocked animals represent contaminant concentrations for organisms within investigation areas (versus upstream/downstream organisms).

[277] DOE 2007 p. 4-2

- Whether harvested sculpin represent contaminant concentrations in all fish at aquatic investigation areas.[278]

As noted in Section 4.1.1, one of the most important data gaps in this respect is the question of whether or not the contaminant concentrations in the bottom feeding sculpin are adequately representative of the many different fish common in the Columbia River. Recall that, just considering the species consumed by the Yakama, there are the "salmon, including spring and fall Chinook, coho, sockeye, and chum salmon, steelhead and cutthroat trout" as well as the "bass, bull trout, smelt, lamprey (eel), suckers, whitefish, and sturgeon."[279] As noted by the U.K. Environment Agency in its *Impact Assessment of Ionising Radiation on Wildlife*,

> The greatest uncertainty lies in the values of concentration factor used to calculate internal contamination by radionuclides, and hence internal doses. Concentration factors vary considerably between species and also with environmental conditions, such as water chemistry and soil type. The true values for concentration factor could easily differ from the recommended defaults by an order of magnitude or more in either direction.[280]

Thus, unlike humans where only two varieties (man and woman) generally have to be considered, the wide variety of species in an ecosystem like Hanford's adds a further layer of complication to the assessment of exposures since it is not always obvious which of the species will receive the highest dose or which, if any, is the most at risk once differences in radiosensitivity are included. Of course, male-female differences must also be considered for assessing the reproductive health of species (including some plants) that reproduce sexually.

In addition to the interspecies differences, there is the additional complication of uncertainties in how various radionuclides or chemical contaminants will distribute themselves within the bodies of individual fish as well.[281] This uncertainty was already noted in Section 4.1.1 in the context of the Yakama's consumption of more than just the fillet or steak from the fish they gather. Specifically, we noted above that the DOE recommended in its 2002 *Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota* that

> If biological samples are intended to be used to estimate both human and nonhuman exposures, then both muscle and carcass should be analyzed for at least some of the samples, as is practicable. The use of muscle tissue alone may underestimate the B_{iv} [bioaccumulation factor] for nonuniformly distributed elements. This is of particular concern when estimating food-chain transfers for biota; wildlife generally consume the entire organism, not just the muscle tissue. Hence, whole-body concentrations are generally the appropriate measurements for estimating food chain transfers to biota.[282]

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[278] DOE 2007 p. 6-139
[279] Ridolfi 2007 p. 18
[280] UK Environment Agency 2001 p. 108
[281] Blaylock, Frank, and O'Neal 1993 p. 3
[282] DOE 2002 p. M2-51

From Table 3 in Section 4.1.1, we found that the Advisory Committee on Radiological Protection of the Canadian Nuclear Safety Commission reported bioconcentration factors for uranium in the bones of freshwater fish whose geometric mean was more than 12 times higher than the default value assumed by programs like RESRAD and that the upper limit was 80 times higher than that used in RESRAD.[283] As a result, the collection of site specific data on the uptake of contaminants by the individual organs of environmentally and culturally important fish species at Hanford will aid not only the human risk assessment, but also the ecological risk assessment.

Turning next to the uncertainties acknowledged with the data on contaminant levels in plants, the RCBRA notes that "[t]issue concentrations in plants are subject to:"

- Analytical measurement uncertainties (e.g., detection limits are typically elevated in tissues because of matrix interferences).
- Whether above-ground vegetative material represents contaminant concentrations in all plant matrices/compartments (e.g., roots, seeds).
- Whether contaminant concentration in the two dominant species represents all plant species in investigation areas.[284]

As discussed in Section 3.2, the lack of physically meaningful correlations between the concentrations of various uranium isotopes in soil and those in plants from the same locations represents an important outstanding question concerning the overall quality of the RCBRA's data. This lack of correlation should be resolved by future data collection efforts and should not be written of, as was done in the RCBRA, as evidence that the "lack of plant contaminant uptake indicates minimal COPC exposure."[285] A similar lack of correlation in the RCBRA's data between the measured contaminant levels in invertebrates and the soil in which they live should similarly be resolved by additional sample collection.[286]

In addition, the uncertainty regarding the applicability of the measurements of above ground material from two species to all species and all plant components is one that should also be addressed by additional sampling. As noted in Section 4.1.3, this additional data collection is important for the human risk assessment given the use by Native Peoples of plants with unusual growth environments or biologies like Indian celery, lichens, and shelf fungus. The known variability between plant types (such as leafy vegetables versus root vegetables) provides additional motivation for ensuring that the most exposed and most radiosensitive plant types are being considered in the risk assessment.

Finally, the collection of additional data to address this uncertainty is also important in the context of ecological risk assessments given the fact that, as noted by the International Commission on Radiological Protection, the most "radiosensitive parts of plants are usually the meristem tissues, which are located in the roots and shoot tips" and that "[t]his superficial

[283] Wang et al. 1993 p. 34 to 35 and ACRP 2002 p. 20
[284] DOE 2007 p. 6-127 to 6-128
[285] DOE 2007 p. ES-16 and 6-26
[286] DOE 2007 p. 6-27 and 6-28

location of the meristem makes it particularly vulnerable to radiation exposure from the deposition of radionuclides."[287] In fact, the International Atomic Energy Agency (IAEA) noted in its *Effects of Ionizing Radiation on Plants and Animals at Levels Implied by Current Radiation Protection Standards* that "it appears that in the natural environment the most sensitive plants display acute radiation sensitivities which are similar in magnitude to those found for mammals."[288] Given the importance of plants in tribal life, this is a particularly important issue to keep in mind in designing research on ecosystem effects of contaminants and in creating a remediation plan.

Section 6.2.3 – Issues Concerning the Field Sampling Methodology in RCBRA

In addition to the specific data gaps and uncertainties identified above, we have identified a more general concern regarding the sampling methodology used for the field observations supporting the RCBRA. In particular, it does not appear that the observations were of sufficient length in terms of time or of sufficient scale and resolution to adequately ensure that all potential impacts on the ecosystems would be properly identified. While there is an extensive set of data regarding Hanford and its environs, the primary data used to support the RCBRA was collected over a period of less than one year. For example, the survey of plant cover used as one line of evidence in the ecological risk assessment was conducted over just a single two month period between March and May while the collection of small mammals occurred over less than a five month period from February to June.[289] Significantly, the collection of small mammals at any particular location did not even cover the full five months since the trapping of mice at a given site lasted only long enough for the collection of six mice.[290]

One implication of the short-term nature of the field work supporting the RCBRA has been noted above, namely the lack of successful collection of amphibians for analysis. As noted in the RCBRA, this lack of successes was likely caused by flooding of amphibian breeding sites.[291] Thus, a short-term condition of the river negatively impacted the ability of the RCBRA to evaluate even present day impacts on the ecosystem. While this data gap is planned to be filled by the Inter-Areas shoreline assessment, it nevertheless illustrates one of our concerns regarding the choice to sample over such a limited timescale.

In addition to easily identifiable data gaps such as the lack of amphibians to collect, the complex dynamic interactions between the many elements of an ecosystem make it important that any kind of field observations be conducted over a suitably long timescale to ensure that subtle changes which may build up over time are not missed. For example, as noted by the International Atomic Energy Agency

[287] ICRP 91 p. 233 to 234
[288] IAEA 1992 p. 15
[289] DOE 2007 p. 6-14
[290] DOE 2007 p. 6-15
[291] DOE 2007 p. 4-2

> It might take considerable periods of time to express damage at the population and community levels, under chronic low level irradiation, and consequently most studies have probably been terminated prematurely.[292]

This caution was echoed by the Advisory Committee on Radiological Protection of the Canadian Nuclear Safety Commission. For example, comparing ecological risk assessments to those for humans, the Advisory Committee noted that

> Ecosystem monitoring requires much more effort and therefore should be repeated over intervals of a number of years. Biological communities respond to a host of ongoing changing environmental conditions and changes within communities require several years to be measurable.[293]

Ecosystem effects of contaminants may also be affected by natural environmental stresses. Hence, the patterns of harm that may emerge during a prolonged drought or low river flows of in the aftermath of severe storms may be different than those revealed in sample collection lasting for a year. This reinforces the above argument for studies over an adequately long period. (See further discussion on stresses below.)

Adding to the complexity of efforts to understand the impacts on the ecosystems at Hanford is the fact that substantial changes to some areas are expected in the future. As noted in Section 6.1.1, restoration of the waste sites in the 100 and 300 Areas may result in increases in the population of large mammals such as elk as well as of birds such as the horned lark, western meadowlark, savannah sparrow, and loggerhead shrike.[294]

The need to address these kinds of long-term changes support our recommendation that the modeling of future ecosystem impacts from contaminants and other environmental stressors be conducted as part of the revised risk assessment (see Section 6.4). In order to support this kind of modeling, substantial amounts of additional field data will be needed and that data will have to be collected over a long period. For example, Yodzis noted that

> For those doing practical work with environmental impacts, it is of crucial importance to understand that short-term observations of environmental impacts that can be viewed as press perturbations are close to useless for estimating probable long-term impacts. Moreover, as the present study makes clear, predicting those long- term effects not only requires data on the strengths of many interactions in the system, it requires very accurate data on many interaction strengths. This is a daunting prospect indeed.[295]

Finally, we note that natural populations and communities exhibit considerable spatial as well as temporal variability even in the absence of imposed stresses. Thus, field studies need to be conducted over an area that is large enough to be ecologically significant for the kinds of plants

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[292] IAEA 1992 p. 4
[293] ACRP 2002 p. 61
[294] DOE 2007 p. 2-15
[295] Yodzis 1988 p. 515

and animals of interest at the Hanford site. This question of scale is important because, as noted by the IAEA, "ecological responses to stress over small areas may differ from responses to the same intensity of stress applied to larger areas."[296] In addition, while the field sampling must occur over a wide range, it must also have sufficient resolution to allow for distinctions to be made between individual habitat patches. Adequate resolution is of importance given the goal of protecting certain individual plants and animals from harm rather than just entire populations given the qualitative aspects of holistic risk assessments discussed in Section 6.3. For example, as noted by the DOE

> In protecting populations, considerable averaging over space and time could be allowed and still ensure adequate protection. In protecting individuals, however, it could be more appropriate to allow little or no averaging over space and time. Thus, in protecting individuals, use of the maximum concentrations of radionuclides in the environment at any location and at any time could be more appropriate.[297]

This question of adequate scale and resolution raises questions over the RCBRA's focus on exploring the potential impacts on a waste site by waste site basis. Additional field observations will be required to determine the appropriate scale and resolution of future ecological risk assessments. We will explore this point further in Section 6.4.

Section 6.3 – The Need for a Holistic Ecological Risk Assessment

As with the need to consider a more integrated approach to human risk assessment including the need to consider potential interactions between multiple contaminants as discussed in Section 5, a more holistic approach to ecological risk assessments is required as well, particularly when efforts to project impacts into the future are to be made. In the case of the RCBRA, it was already noted that it included toxicity bioassay as a prominent line of evidence in part because "while studies reported in the literature usually evaluated just one contaminant at a time, these bioassays offer site-specific information on adverse effects of contaminant mixtures and on contaminant bioavailability for Hanford Site aquatic media."[298] However, unlike human risk assessments, it is not only important to consider the impact from exposure to multiple contaminants, but it is also important to include considerations of other environmental stressors such as human activities (fishing, dam construction, etc.) as well as changes to abiotic conditions such as temperature, pH, and dissolved oxygen.

The need to examine the impacts of radiation and other contaminants within the context of multiple environmental stressors acting on the ecosystems is widely acknowledged in publications from regulatory and advisory bodies at both the national and international level. Examples include

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[296] IAEA 1992 p. 7
[297] DOE 2002 p. M2-80
[298] DOE 2007 p. 6-23

- National Council on Radiation Protection and Measurements

  It must be recognized that increased radiation exposure is but one of the many stresses imposed upon aquatic populations by human activities. However, determination of the mode of interaction of radiation, whether it be antagonistic, additive, or synergistic, with other environmental contaminants or stressors, is extremely difficult to assess under conditions of chronic exposure.[299]

- U.S. Department of Energy

  Participants identified a number of methodological problems limiting the value of laboratory studies.... Third, the range of sensitivities of species and life stages in nature is undoubtedly much greater than the range of sensitivities of species and life stages for which laboratory data are available. Nutritional status is known to affect responses of animals to stress; because of parasitism, disease, or variations in food availability, animals in nature are probably often more vulnerable to added stresses such as ionizing radiation than are well-fed laboratory animals.[300]

  Plants and animals may also be simultaneously exposed to other stressors, such as noise and hazardous chemicals. At present, no consensus exists within the scientific community about what the cumulative impacts are of simultaneous exposure to ionizing radiation and other anthropogenic stressors, or how to measure them. This factor should be considered when estimating and describing the risks associated with doses of ionizing radiation, if only qualitatively. In cases where exposure of biota to ionizing radiation exceeds the biota dose limits, a consideration of cumulative impacts from radiation and other stressors present may be warranted.[301]

- U.K. Environment Agency

  Research into the biological effects of ionising radiation has focused on mammals, often in laboratory experiments. It is difficult to extrapolate these data to assess effects on wildlife in natural systems because of the lack of consideration for other stressors that may be present in the natural environment.[302]

- FASSET [Framework for Assessment of Environmental Impact, European Commission 5th Framework Programme]

  The interactions between some environmental factors and acute ionising radiation have been demonstrated in the 1960s - 70s. Several authors reported modifications of the survival and metabolism of test organisms showing results as a function of the type and doses of radiation, the tested

[299] NCRP 109 p. 61
[300] Barnthouse 1995 p. 11
[301] DOE 2002 p. M1-58
[302] UK Environment Agency 2001 p. 122

environmental factors, the tested species, the duration of the experiment, etc.[303]

- International Commission on Radiological Protection

  Effects of ionising radiation on flora and fauna are always modified by the action of a range of ecological factors. Compensatory, additive, or synergistic effects of radiation and other environmental factors may therefore be expected.[304]

- International Atomic Energy Agency

  Also, very little information is available in the area of interactive effects of natural or man-made radiation with other natural or man-made stresses or agents. This may be a serious omission in the light of the multiple forms of pollution which threaten many contemporary populations and ecosystems. This fact should be remembered in making assessments and in the development of protection standards for the environment.[305]

The inclusion of multiple stressors at the Hanford site is particularly important given the range of chemicals and radionuclides present in the waste, the variety of chemicals carried down the Columbia River from areas upstream, and the physical disruption of the site resulting from the construction, operation, and decommissioning of the DOE facilities. For example, as noted by Gephart, "of the 132 potentially hazardous chemicals studied in a fish contaminant study of the Columbia River Basin, the U.S. Environmental Protection Agency detected 92, some at levels of concern."[306] Thus, nearly seven out of ten potentially harmful chemicals were found when looked for in the river's water. In addition to contaminants such as fertilizer and pesticides, a number of heavy metals such as "arsenic, cadmium, chromium, lead, silver, mercury, and zinc" are also known to leach into the Columbia River from both natural sources and from mine tailings upriver from Hanford.[307] As summarized by Gephart, "[t]his underscores the need to assess the water quality and health of the Columbia River as an integrated whole regardless of contamination source."[308] These multiple contamination sources are, of course, in addition to the stress placed on the aquatic ecosystem from such human activities as fishing and the operation of large damns on the river.

As with the discussion of potential interactions that may occur between multiple contaminants in humans, there is evidence pointing to the potential significance of combined exposures in aquatic ecosystems. For example, the insecticide carbaryl (a neurotoxin) has been found in some instances to become more lethal to aquatic animals when exposure to the chemical pollutant is accompanied by exposure to warmer temperatures, variations in pH, higher levels of UV-B, or to chemical predator cues.[309] As summarized by Relyea

[303] FASSET 2003 p. 180
[304] ICRP 91 p. 237
[305] IAEA 1992 p. 4
[306] Gephart 2003 p. 5.44
[307] Gephart 2003 p. 5.44
[308] Gephart 2003 p. 5.44
[309] Relyea 2003 p. 1515 and 1519

> The current study suggests that the lethal concentrations of carbaryl (and perhaps other pesticides) can be much lower than we currently appreciate because traditional toxicology studies frequently isolate animals from their natural ecology (including predator cues). When we include some of the natural ecology, even low concentrations of a pesticide can be highly lethal to amphibians. In short, ignoring the relevant ecology can cause incorrect estimates of a pesticide's lethality in nature, yet it is the lethality of pesticides under natural conditions that is of utmost interest.[310]

An important component of these kinds of interactions between stressors are those mediated by behavior or developmental changes that can alter critical life functions such as foraging activity or predator avoidance.[311] For example, sub-lethal doses of the insecticide carbaryl can alter tadpole behavior making them more susceptible to predators.[312] Specifically, it was found that

> Carbaryl caused a nearly 90% reduction in tadpole activity at the lowest concentration compared to the controls (Fig. 1 [not included]). Because the time unstressed tadpoles spend swimming is directly correlated with time spent feeding, diminished swimming activity can have several consequences by generating delays in growth and development. First, many tadpoles escape predators by being too large to capture or by emerging from ponds as early as possible. Tadpoles must reach a minimum size before metamorphosis, and fast growth shortens the larval period, thus decreasing exposure to predators. Therefore, reduced growth can lead to indirect mortality by prolonging susceptibility to predators either by tadpoles remaining small enough to capture or by lengthening the larval period. Second, because tadpoles often inhabit ephemeral ponds, a short delay in growth or development may result in a failure to emerge before the pond dries. Finally, amphibian adult fitness (e.g., survival to first reproduction, fecundity) diminishes because of slowed growth and development and is correlated with the length of the larval period and the size at metamorphosis. Chronic exposure to carbaryl may strongly impact critical tadpole life history functions and indirectly influence mortality and adult fitness.[313]

Since the importance of these predator-prey interactions cannot be studied in isolation (i.e., without both predator and prey and the environmental settings in which they interact), they are likely to be missed by some traditional laboratory studies of chemical toxicity.

A further complication to studies of these kinds of interactions arises because of the potential for exposures to occur to either predator or prey individually as well as to both simultaneously. For example, as noted by Bridges

> The presence of environmental contaminants may alter predator-prey interactions among aquatic species by altering activity levels of predators or prey, or by

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[310] Relyea 2003 p. 1520

[311] Bridges 1997 p. 1935 and Bridges 1999 p. 205

[312] Bridges 1997 p. 1935 and Relyea 2003 p. 1519

[313] Bridges 1997 p. 1937

> altering predator avoidance behavior. The outcome of a predatory encounter may be dependent upon whether both species are exposed to a contaminant simultaneously, or whether exposure occurs only in one of the species....This study suggests that when tadpoles and newts are exposed to a sublethal level of a contaminant [the neurotoxin carbaryl] simultaneously, that predation rates do not differ from those observed under natural conditions, but exposure of either predator or prey at different times can disrupt predator-prey dynamics.[314]

Finally, we note that adding consideration of competitors as well as predators adds further to the complexity of these kinds of assessments. Thus, a study found that carbaryl pollution could actually aid tadpole growth in some cases because of such effects as its greater toxicity to competing zooplankton and reduced competition between the tadpoles that survive for food and other resources. [315]

The presence of multiple ecosystem stressors is, of course, not limited to the aquatic environment. For example, the hazard index for mammals in higher trophic levels in both the upland and riparian zones were found to be elevated at Hanford. Specifically they were both found to be in the range of 10, while any number greater than one indicates the potential for harmful effects. However, the RCBRA notes that the hazard indices "are similar between remediated waste and reference sites" and that therefore there is little evidence for elevated risk from the waste sites that have been cleaned up.[316] In the present context, however, the elevated hazard indices for these mammals should be taken as a cause for concern given that it would put them at potentially greater risk from any additional stresses from exposure to contaminants in the future. Thus the higher apparent risks for the whole site would tend to argue for greater caution in determining any potential impacts that might occur over the long timescales for which the Hanford contaminants will remain hazardous.

As a final example, the RCBRA notes that the effects from soil quality changes due to operational and remediation efforts include the following

> Some areas that are no longer used for waste disposal, construction activities, or site operations have begun to revegetate naturally to communities dominated by gray rabbitbrush with an understory of Sandberg's bluegrass, bulbous bluegrass (Poa bulbosa), and cheatgrass.... To promote the reintroduction and colonization of native species, remediated upland CERCLA waste sites are revegetated with the goal of reestablishing sagebrush/Sandberg's bluegrass communities with a mixture of other native grasses and forbs adapted to rocky soils. Following restoration, the vegetation type, density, and species diversity may not be the same as before initial disturbance, due to the change in soil structure. As the restored plant communities mature, however, improvements in shrub coverage will provide important habitat for native wildlife species. In addition, plants that

[314] Bridges 1999 p. 205
[315] Relyea 2003 p. 1515 and 1519 and Hatch and Blaustein 2003 p. 1089 and 1091
[316] DOE 2007 p. 6-32 and 6-39

> rely on fine-textured soils may occur on restored or naturally recovering sites, but typically are not expected to be abundant.[317]

In addition to plants that rely on finer soils, the RCBRA goes on to note that

> The soils in the industrialized 100 Area and 300 Area have been disturbed, and most of the area is covered with sandy gravel and cobble soil. This soil matrix limits the diversity of small mammals to species that live on the surface or in very shallow burrows.... Burrowing species such as the Great Basin pocket mouse and the pocket gopher are limited to areas where fine grained soils are at least 30 cm (12 in.) deep. These species are not found at remediated waste sites backfilled with sandy gravels.[318]

As such, the RCBRA concludes that "[a]reas that were not used as waste sites will have less soil disturbance and may support a more robust and diverse community of soil-dwelling fauna."[319] Thus, the residual contamination in the remediated waste sites may have a greater impact in the future than expected given the additional stresses and loss of diversity caused by the choice to backfill the waste sites with sandy gravel rather than more locally appropriate soil types.

While a complete review of the kinds of interactions that may occur between environmental stressors in complex ecosystems is clearly beyond the scope of the present work, the above examples hint at the complexity of this challenge and the need for careful long-term studies and a holistic approach to evaluating ecosystem health. As a part of that holistic approach, we will conclude this section with a brief discussion of the kinds of ecosystem models that should be explored for use at Hanford while details regarding the research needed to develop and validate these models will be considered in our recommendations in the following section.[320] To begin with, we note that the RCBRA makes the following assumption regarding the ecological receptors of greatest interest

> Regarding COPC characteristics, Hanford Site contaminants are predominantly inorganic chemicals such as heavy metals and radionuclides. Because such COPCs do not typically increase in concentration through trophic transfer, the risks posed to higher trophic-level organisms are generally of less concern than risks to organisms lower in the food web. To the extent that inorganic chemicals do accumulate in biotic tissues, there is a greater propensity for invertebrate uptake compared to plant uptake. Therefore, relative to plant-eating wildlife (or to wildlife that eat a variety of foodstuffs), invertivorous (invertebrate-eating) wildlife should experience relatively greater exposure to radionuclides and metals and are valuable assessment endpoint entities because they are potentially more exposed indicators for evaluating the adverse effects of inorganic COPCs.[321]

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[317] DOE 2007 p. 2-14
[318] DOE 2007 p. 2-15
[319] DOE 2007 p. 2-16
[320] For a detailed discussion of the science behind ecosystem modeling see [Ulanowicz 1997] and [Canham, Cole, and Lauenroth 2003]
[321] DOE 2007 p. 2-36 to 2-37

While higher trophic levels are included in the RCBRA, this focus on lower and middle trophic levels is repeated several times throughout the risk assessment.[322] However, in terms of impacts from chronic exposures, it is not necessarily always the most exposed receptors that are greatest concern. In terms of reproductive effects it may be that those with long lifespans and low reproductive rates are of greater concern.[323] Thus, the identification of the receptors at greatest risk, as well as their role in the ecosystem as a whole, is a major motivation for the use of the kinds of ecosystem modeling we recommend.

Significantly, our endorsement of ecosystem modeling is consistent with the recommendations of the ICRP in its 2003 *Framework for Assessing the Impact of Ionising Radiation on Non-human Species*. For example, the ICRP notes that "[a]n important factor in ecology is the interdependence of populations and communities" and that "[a] change in one ecological factor may have a drastic effect on another."[324] It also concludes that

> Effects on higher levels of biological organisation (e.g. populations and ecosystems) occur only if individual organisms are affected, and effects data are generally obtained for individuals rather than for higher levels of organisation. Caution should be made for situations where the effects on individuals might not be easily recognisable but the effects on a population might be manifested. Depending on the circumstances and need, assessments of radiation effects may have to be made at the level of the individual, population, community, or ecosystem. Such assessments may be difficult to achieve and will depend upon many factors, such as the number of individuals within a population that are affected, the nature of the different types of populations within a community, and so on. In the natural environment, the situation can become very complex because of the interaction between each individual and its surrounding ecosystem. The effects can also be modified by the presence of other environmental stressors or by combined effects related to the presence of other pollutants, and by interactions between different trophic levels.[325]

The complexity of these internal interactions between and among the individuals within an ecosystem and between the individuals and their environment can give rise to a variety of unexpected behaviors. For example, even relatively simple ecosystem models have shown emergent phenomena that give rise to effective interactions between populations even though no direct causal mechanism between those populations was included in the model design.[326] As with other complex systems, these emergent properties are not necessarily always deterministic in nature and can result in the possibility of very different outcomes having very similar probabilities of occurring as a response to stresses placed on the ecosystems.[327] In particular, as summarized by Dulvy et al.

[322] DOE 2007 p. 2-38 and 6-1 to 6-2
[323] IAEA 1992 p. 21 to 22
[324] ICRP 91 p. 235
[325] ICRP 91 p. 235
[326] Canham, Cole, and Lauenroth 2003 p. 69, 71, and 304
[327] Canham, Cole, and Lauenroth 2003 p. 446

> There is increasing evidence of rapid and nonlinear shifts from one ecosystem state to another that could hinder species recovery in some ecosystems. These shifts can result from functional removal of a species, disrupting indirect interactions which have a stabilizing role in communities, causing species replacements, trophic cascades and phase shifts.[328]

As such, it has been noted by Canham et al. that these kinds of uncertainties are "poorly addressed by current sensitivity analysis in ecological modeling, which, by and large, fails to address the consequences of simultaneous variability in the driving parameters and variables, and tests for alternative expressions of ecological processes."[329]

The ICRP recognizes the immense complexity of any effort to realistically model ecosystems and, while choosing to focus on the protection of individuals within the environment for now, it keeps open the option for more comprehensive approaches to be developed in the future. Specifically, it notes that

> Although theoretical models representing energy flow in ecosystems, predator-prey interactions, and population dynamics have been developed for a limited number of simplified ecosystems or economically important species, there is in general a lack of data with which to assess the effects of environmental contaminants, including ionising radiation, on these important ecological functions. Consequently, assessments of the ecological effects of contaminants have usually focused on assessment of effects on individuals of the most exposed and/or most sensitive species or life stage, with the conclusion that if the most sensitive species or life stage is protected, ecosystem integrity will also be protected. Research is being conducted to support a more comprehensive approach to the assessment of radiological effects on ecosystem function.[330]

As a result, a major goal of the research plan we propose in the following section is to enable the development of such an integrated approach at Hanford, and to allow projections of future ecological impacts to be conducted. This is in line with our major recommendation for the RCBRA that it must treat the site holistically and project the movement of contamination into the future to ensure that remediation goals set today are adequately protective of both people and the rest of the environment in the future.

Section 6.4 – Outline of Future Research Needs

In seeking to move forward with the development of a more holistic and comprehensive approach to the protection of the Hanford ecosystem we propose the following outline of a research plan designed to fill in the most pressing data gaps and enable a realistic site model to be developed. This plan follows one originally proposed in December 2004 that was created by

[328] Dulvy, Sadovy, and Reynolds 2003 p. 47
[329] Canham, Cole, and Lauenroth 2003 p. 446
[330] ICRP 91 p. 235 to 237

the present author and Dr. Arjun Makhijani, president of the Institute for Energy and Environmental Research, along with significant contributions from Dr. Michael C. Thorne.

The overall direction and general philosophy guiding this work are the rights of the Yakama and other Native Peoples to the resources and the land on which the Hanford facility was built. All efforts to identify and quantify damage done to the environment must be conducted with the full and active participation of the people of the affected tribes so that their specific cultural conceptions of the natural world and their specific lifestyle (economic as well as cultural and spiritual) can play the leading role in setting priorities and the context for the evaluation of ecological harm. The need for such guidance from the affected communities has been recognized by the International Commission on Radiological Protection in its 2003 *Framework for Assessing the Impact of Ionising Radiation on Non-human Species*. In this report the ICRP conclude that

> However, defining what constitutes an acceptable level of harm goes beyond the realm of science and is best dealt with at the environmental management stage when policy decisions take socio-economic factors into account.[331]

The ICRP went on to note that there is "not always a clear distinction between what one might call 'purely scientific' and 'purely value-based' judgments, because science and societal views are interlinked."[332] Of specific relevance to the current issue, we note that the DOE's 2002 guidance on ecological risk assessment cautioned that special care must be taken when evaluating potential damage to economically important species such as salmon as well as to culturally valuable plants such as those used by Native Americans.[333] Finally, we note that this focus would be consistent with our recommendations in Section 6.3 concerning the need to develop a holistic risk assessment that includes the qualitative values of the Yakama as well as the quantitative aspects of dose and risk.

In summary, the efforts that we propose can be broken down into four broad categories:

I. Evaluation of environmental dosimetry including determination of the relative biological effectiveness of alpha particles, the concentration factors for the organs of critical species and radionuclides, and the uptake of radionuclides from the environment including food web bioaccumulation.

II. Evaluation of radiation effects on individual members of species present in the environment surrounding the Hanford site including mortality, morbidity, reduced reproductive capacity, and chromosomal damage both in isolation and in the presence of other environmental stressors.

III. General ecosystem modeling of the affected environs including determination of effective energy, water, and nutrient flow models of the key ecosystems surrounding the Hanford site taking into account the potential that damage may

[331] ICRP 91 p. 249
[332] ICRP 91 p. 255
[333] DOE 2002 p. M1-21

have already occurred to these areas prior to the creation and validation of these models.

IV. Connecting the above efforts in order to

a. Identify the critical species from an ecosystem point of view. These will include species particular to the Hanford area which are highly radiosensitive and those which play a pivotal or keystone role in local plant and animal communities. These critical species will be added to the list of species that are culturally or economically important to the Yakama and other Native Peoples for further detailed study at the radiation dose and effects levels.

b. Identify what type and severity of effects may be expected to occur at the individual, population, community, and ecosystem levels as a result of the radioactive and other contaminants in the Hanford environment (i.e., develop an integrated assessment model).

While we will further outline each of these areas below, it should be noted that they are often overlapping and implicitly interrelated and that they are to be viewed as evolving in an iterative manner as the overall investigation of potential environmental damage proceeds.

With respect to the development of an improved understanding of environmental dosimetry, we recommend the following four research activities be undertaken.

I. Evaluate the contaminant distribution over the area of interest at the Hanford site including both radioactive and chemical pollutants.

a. Ensure that the coverage is of a sufficient area to be ecologically appropriate, taking into account the mobility of species under investigation, and of a sufficient resolution to allow distinction to be made between individual habitat patches.

b. Retain information on the spatial and temporal variations in contaminant concentrations in addition to the averages to allow estimates to be made of the variability in exposures of individuals as well as the average or reference individual.

c. Evaluate contaminant distribution over time to allow transport models to be validated, and thus providing confidence in projections of future doses and facilitating estimation of uncertainties in those projections.

II. Investigate the appropriate relative biological effectiveness (RBE) factor for alpha particles, tritium, and other radionuclides with particular decay characteristics such as Auger electron emitters for the plant and animal species of interest at Hanford and for the biological effects found to be important in the other steps of this research program.[334]

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[334] The relative biological effectiveness factor (RBE) is a used in radiological protection schemes to take into account the greater amount of biologically relevant damage caused by radiation such as alpha particles compared to the same amount of energy absorbed from gamma radiation. The higher the RBE the more damage the radiation causes per unit of absorbed energy

   a. A sensitivity analysis needs to be performed in cases where experimental evidence is lacking. An example would be the use of 5, 10, and 50 for the RBE of alpha particles as recommended by the FASSET review. The possible range was large; the review concluded that the RBE for alpha particles as "unlikely to be greater than ~200." In view of this a higher RBE, of 100 for instance, should also be evaluated.[335]

III. Develop specific dose models for the most sensitive ecological receptor and the most sensitive developmental stage in light of the biological effects found to be of interest in the other steps in this research plan.

   a. These models should include, among other elements, a consideration of the external alpha and beta contribution for the embryo/fetus or larval stage when they are sufficiently small or otherwise constructed so that these radiations can penetrate to an appreciable degree to the radiosensitive organs and tissues of interest.

   b. For developmental stages and biological effects where specific organs are of concern (brain, gonads, skeleton, etc.), appropriate tissue concentration factors need to be determined including biokinetic models for lifecycle stages that are short. This effort will connect closely to the work on bioaccumulation discussed in the following step.

IV. The bioaccumulation of both radionuclides and chemical pollutants present at Hanford needs to be determined for each critical or culturally/economically significant species.

   a. Site and species specific evaluations need to be made given that the bioconcentration factors and/or biokinetic characteristics are known to be highly dependent on soil and water chemistry as well as on other aspects of the species' habitat.

   b. Both organic pollutants and radionuclides need to be considered given that they can behave very differently in the food web with some (mainly organic chemicals) bioconcentrating at higher trophic levels with others (including many radionuclides) being bioexcluded and diminishing in concentration at higher levels. This distinction can have an important impact on the identification of the critical species and ecological functions. Therefore, species specific bioaccumulation of all relevant radionuclides and chemical pollutants on the Hanford site will be essential to developing reliable estimates of the harm to individuals, populations, and communities as well as to the ecosystem as a whole.

   c. A thorough uncertainty analysis should be conducted to examine the impact of variations in abiotic parameters such as the partition coefficient of radionuclides across different soil types which are known to substantially affect the bioaccumulation of radionuclides. The choice of partition coefficient (K_d) will not only affect the transport of contaminants to the

[335] FASSET 2004 p. 63

aquatic biota, but high partition coefficients for bioexcluded radionuclides like plutonium can significantly heighten damage at lower trophic levels.

In the context of our proposals for research into environmental dosimetry, it is important to point out again that improvements in our understanding of contaminant uptake into plants and animals will, not only support the ecological risk assessment, but also improve the human risk assessment as well given the concerns outlined in Sections 4.1.1 through 4.1.3 regarding the use of many different plants and animals by Native Peoples as well as their consumption of more than just the muscle tissue of animals. It is interesting to note here that the uptake factors used in the RCBRA have not been updated in the last 17 years.[336] This is despite the fact that the last revision of the uptake factors made several major changes over the earlier estimates. For example, the uptake factor for chromium and strontium, two contaminants of concern at Hanford, increased by a factor of 9.1 and 26.7 times respectively during the last review of the science in 1993.[337] Similar changes were also made for the uptake of chromium and uranium in fish with increases of 10 and 5 fold respectively. In other cases no changes were made while some uptake factors actually decreased.[338] Updating these factors with more recent information and information specific to the plants and animals of the Hanford site will significantly strengthen the overall assessment of risk.

Turning next to the research needed to further define the effects of radiation on plants and animals, we note that it will not be possible to fill in all of the missing data for all of the species present on site. For example, in the FASSET review, 80 percent of its wildlife groups had either no data or too little data to derive a dose-response relationship for effects like mortality, morbidity, reproductive success, and DNA damage. In many of the remaining 20 percent, the FASSET review found that only "some data" was available and that this data was often focused mainly on animals in the human food chain or on biota that are simple to experiment on, such as rats and mice.[339] Therefore, the recommended studies included in this section are assumed to be primarily limited to those species identified as either culturally or economically important by the Yakama people or those identified as particularly radiosensitive or critical to the functioning of the ecosystem.[340]

I. In addition to direct effects on reproduction, studies should include all radiation effects that can indirectly degrade reproductive success including early mortality and morbidity (cancer, skeletal defects, opercular defects, immune system effects, etc.) as well as neurological damage and behavioral changes.

   a. The effects considered should take into account the differences between short-lived fast reproducing biota and longer-lived slower reproducing biota, including the differences in their likely exposures to both radionuclides and chemical pollutants during different stages in their lifecycles.

[336] Wang et al. 1993 and ResRad 6.4
[337] Wang et al. 1993 p. 27
[338] Wang et al. 1993 pp 18 and 19 for fish, and pages 23 and 24 for examples of no change or decrease
[339] FASSET 2004 p. 42
[340] We note a further complication in that radiosensitivity varies substantially both between species within a particular wildlife group as well as between genetically distinct strains within a single species. What affect this intra-species variability in sensitivity may have in the wild remains an open question.

b. The effects research should also take note of the potential for synergistic or antagonistic effects between chemicals and between chemicals and radioactive contaminants. In particular, care should be taken to include interactions between contaminants originating both on site as well as those transported onto the site from upriver.

II. DNA damage should also be studied as one of the potentially important effects of radiation on the environment given that it has been shown to correlate with decreased vigor and decreased reproductive success in fish from the contaminated White Oak Lake at the Oak Ridge National Laboratory and that the genetic and epigenetic sequences of organisms are interconnected to their environment as explored in other parts of this research plan.[341]

a. The DNA damage needs to be evaluated in multiple organs and at multiple times of the year and stages of growth since these factors are known to affect the interpretation of correlations between DNA damage and higher level effects on plants and animals.[342]

b. Techniques capable of distinguishing between single strand breaks and double strand breaks in field studies are needed since many heavy metals and organic chemicals are known to be genotoxic or carcinogenic, and thus the type of DNA damage can help indicate whether or not radiation is a significant factor in any observed population effects.[343]

III. Particular care must be taken in performing all field studies used to determine the effects of radiation damage to insure that the reference area is suitably similar to the experimental areas to avoid potentially misclassifying effects. For example, experiments with fish at Oak Ridge National Laboratory originally mistook an effect due to the more eutrophic nature of the contaminated White Oak Lake compared to the control site as indicating a positive effect of radiation on fecundity.[344] This may be important at Hanford given the fact that (as noted in Section 3.2) the RCBRA reports higher doses at reference sites than those at operational areas or waste sites in both the terrestrial and fish pathways of the CTUIR scenario.

a. Observation of an apparent stimulatory effects on some species in a contaminated environment should be taken as a possible indication of a negative effect on a more sensitive predatory or competing species and experiments should be conducted to explore this possibility.[345]

b. In this context it is important to recognize that natural populations and communities exhibit considerable spatial and temporal variability even in the absence of additional imposed stresses. As a result, many of the environmental signals of interest tend to be noisy over time and, thus, careful

[341] IAEA 1992 p. 52 to 54, FASSET 2003 p. 116, and Makhijani 2001
[342] Theodorakis, Blaylock, and Shugart 1997 p. 215
[343] Theodorakis, Blaylock, and Shugart 1997 p. 206 and 212
[344] FASSET 2003 p. 116
[345] IAEA 1992 p. 5 to 6

long-term studies are required to establish the effects of anthropogenic stressors within these systems.

IV. In addition to single pollutant studies, there is a need to include studies of the effects of multiple environmental stressors to determine the possibilities of synergisms or other modifications to the effects of radiation. These studies need to evaluate the chemical pollutants and other stressors in the Hanford environment including abiotic factors such as temperature, pH, and dissolved oxygen.

Turning to the research needed to support the development of an integrated ecosystem model, we recommend that the following steps be followed.

I. Conduct field and literature reviews of the types of ecosystem communities present on the Hanford site and in the surrounding areas.

   a. The field studies need to be conducted over a long enough timescale to properly observe important ecosystem characteristics and variability. Short-term data are of limited value in developing the required ecosystem model. A rigorous attempt should be made to establish a pre-Hanford ecosystem baseline for use in comparisons to the stressed environments.

   b. The field studies need to be conducted over an area that is large enough to be ecologically significant, since effects at small scales may differ from those over larger areas. As with the contaminant survey, the ecosystem studies should also have sufficient resolution to allow distinctions to be made between individual habitat patches.

II. Construct an integrated ecosystem model of the energy, water, and nutrient flow through the environment recognizing that the aquatic and terrestrial ecosystems can be linked in numerous ways and that the ecosystem overall is not closed, but can exchange mass and energy across its borders.

   a. Breaking the total ecosystem down into modules must be done with care to insure a high probability that all important indirect effects have been included and that all important connecting paths through the food web have been fully represented.[346]

   b. The model should allow for the radiation effects identified in other parts of this research plan to be modeled for the affected species in order to examine the impact those effects on individuals may have at the population and ecosystem level in order to determine if a critical species and/or effect can be identified.

III. Map the ecosystem needed by each of the important organisms identified in this and other steps (including the culturally and economically valuable species) with a view to establishing its corresponding major structures in the species genome, establishing the common ecosystem elements needed by these species, and investigating the propagation of ecosystem stresses back into the genome structure both in terms of gene expression and gene mutation. This basic research

[346] See, for example, [Yodzis 2000]

will enable the establishment of a systematic method of investigating the effects of perturbations caused by pollutants in the ecosystem on the ensemble of important species. The long-term aim of this research would be to establish a genetic and epigenetic map of the unstressed and stressed ecosystems. Interim steps would be to consider key genes or markers of gene activity in the species identified as critically important.[347]

Finally, in terms of integrating the above lines of research into a holistic ecological risk assessment for the Hanford site, the following efforts will be required.

I. The search for important species and important effects need to take all indirect and emergent effects into account through the use of the ecosystem model developed and validated in the above steps.

   a. For example, if an invertebrate is found to be critical to ecosystem functioning, the potential for an effect on a more radiosensitive plant that indirectly effects the invertebrate would have to be considered[348] as well as even further removed impacts such as effects on predators leading to altered risk avoidance behavior in herbivores leading to impacts on vegetation leading ultimately to effects on the critical invertebrates. It is emphasized that the rules of engagement between species are only poorly understood, and are known to vary greatly between different communities (even among those that contain the same or similar mix of species). Furthermore, it is not clear that communities necessarily have critical or keystone species (i.e., there may be no one species whose elimination would result in a catastrophic decline in community diversity or productivity), rather functional considerations for interacting communities (e.g., soil decomposers) may be the best way of evaluating changes in characteristics.

II. An uncertainty analysis needs to be performed when using data for studies of one type of animal to evaluate impacts on another type of similar biota (i.e., using data for a salmon to describe a trout). In these instances there is the need to address the possibility of large inter-species differences in sensitivity and unexpected differences in exposure pathways. These kinds of large differences have been previously observed in studies of birds and reptiles where effects were limited or unobserved for one species, but significant for others within the same family.[349]

III. Organisms or functional groups found to be crucial to the ecosystem in this overall analysis cannot be eliminated from consideration for lack of data on radiation effects as was done with reptiles in the RCBRA.[350] This issue is why the entire process must iterate until an integrated assessment model based on the transport, dosimetry, effects, and ecosystems model can ultimately be put together aimed at predicting the impact expected from the contamination of the Hanford environs on the individual, population, community, and ecosystem levels.

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[347] See the discussion in [Makhijani 2001]
[348] IAEA 1992 p. 12, 18-19, and 22 and Barnthouse 1995 p. 6
[349] IAEA 1992 p. 21 and UK Environment Agency 2001 p. 48
[350] DOE 2007 p. 6-18

Given the need for long-term data collection and field observation, as well as the enormous complexity of the effort we are recommending, we acknowledge that it is likely that fully implementing this research plan could take considerable time. However, there will be many interim steps along the way whose findings will provide significant information of use to guiding remediation and cleanup efforts that are protective of both humans and the environment.

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