Appendix G

Groundwater Quality Impacts

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The purpose of this appendix is to describe the analysis used to calculate concentrations of key contaminants that could potentially reach the groundwater from the Low Level Burial Grounds (LLBGs) defined in each of the Hanford Site Solid (Radioactive and Hazardous) Waste Program Environmental Impact Statement (HSW EIS) alternative groups. The analysis also assesses the potential impacts to accessible surface water resources from contaminated groundwater. Calculated concentrations of key contaminants are compared with drinking water standards as a benchmark against which water quality may be assessed. These calculations also provide the basis for estimates of potential human health risk and ecological risk for comparison among the alternative groups. Human health and risk consequences are discussed in Section 5.11 (in Volume I of this EIS).

Wastes considered in this assessment include previously disposed of wastes and wastes to be disposed of in the Hanford solid waste (HSW) disposal facilities (for purposes of analysis, it was assumed that new disposal facilities would be operational by October 2007):

- Previously disposed of low-level waste (LLW) (that is, wastes disposed of before 1996), which includes:
  - LLW disposed of in LLBGs between 1962 and 1970 (referred to as pre-1970 LLW in this section)
  - LLW disposed of in LLBGs after 1970, but before October 1987 (referred to as 1970–1987 LLW in this section)
  - LLW disposed of in LLBGs after October 1987, but before 1995 (referred to as 1988–1995 LLW in this section)
- Category (Cat) 1 LLW, which includes:
  - Cat 1 LLW disposed of in the LLBGs after 1995 including Cat 1 LLW forecasted to be disposed of through 2007 (referred to as Cat 1 LLW [1996–2007] in this section)
  - Cat 1 LLW disposed of after 2007 including Cat 1 LLW forecasted to be disposed of through 2046 (referred to as Cat 1 LLW disposed of after 2007 in this section).

- Cat 3 LLW, which includes:
  - Cat 3 and greater-than-Cat 3 (GTC3) LLW disposed of in the LLBGs after 1995 including Cat 3 LLW forecasted to be disposed of through 2007 (referred to as Cat 3 LLW [1996–2007] in this section)
  - Cat 3 and GTC3 LLW disposed of after 2007 including Cat 3 LLW forecasted to be disposed of through 2046 (referred to as Cat 3 LLW disposed of after 2007 in this section).
- Mixed low-level waste (MLLW), which includes:
  - MLLW disposed of after 1996 including MLLW forecasted to be disposed of through 2007 (referred to as MLLW [1996–2007] in this section)
  - MLLW disposed of after 2007 including MLLW forecasted to be disposed of through 2046 (referred to as MLLW disposed of after 2007 in this section).
- Melters from the tank waste treatment program
- Immobilized low-activity waste (ILAW) from the tank waste treatment program.

Inventories of retrievably stored transuranic (TRU) wastes in trenches and caissons located in the LLBGs were not evaluated for their potential groundwater impacts because the TRU wastes will be retrieved and sent to the Waste Isolation Pilot Plant for disposal. TRU wastes are in containers, and the containers are not expected to be breached before retrieval, hence the contents would not be released to the environment. No substantial releases to the vadose zone or groundwater from retrievably stored TRU wastes in HSW facilities have been detected. Additionally, current procedures on the retrieval of these wastes require inspection of waste container integrity and containment. Any detected compromise of containment and/or integrity of the containers would require characterictization and mitigation of any potential releases below retrievably stored TRU waste facilities as a part of site closure.

The groundwater exposure pathway analyzed considers the long-term release of contaminants from the variety of LLW and MLLW, analyzed groundwater transport through the vadose zone underlying the potential sources, and lateral transport through the unconfined aquifer immediately underlying the vadose zone to the Columbia River. The LLBGs are all located in the 200 Areas and the physical area of potential groundwater impacts is the unconfined aquifer bounded laterally by the Rattlesnake Hills in the west and southwest, by the Columbia River in the north and east, and by the Yakima River to the south (see Volume I, Section 4.5, Figure 4.16).

This groundwater assessment was performed using a combination of screening techniques and numerical modeling. The groundwater modeling results predict contaminant concentrations in the groundwater associated with selected alternatives from assumed site closure at 2046 up to 10,000 years after LLBG closure. Although not specifically required by current regulations for LLW management, this assessment examined potential groundwater quality impacts for up to 10,000 years after the operational

period. Current requirements for performance assessment of LLW disposal facilities, as prescribed in DOE Order 435.1 (DOE 2001), focus on potential impacts during the first 1000 years after disposal.

Contaminants released from disposal facilities and other sources (for example, tank wastes, canyon facilities, the US Ecology, Inc. commercial LLW facilities) are included in an assessment of combined potential impacts in Section 5.14 (in Volume I of this EIS).

G.1 Methodology and Approach

The approach and steps taken to assess potential impacts to the groundwater system are provided in this section. The alternatives considered in this assessment are described in detail in Section 3.3 (in Volume I of this EIS).

The analysis framework of this groundwater quality assessment considers three major elements: source-term release, vadose zone transport, and groundwater transport. In addition, this analysis framework considers the eventual impact of predicted concentration levels in groundwater on the water quality of the Columbia River.

G.1.1 Lines of Analysis

The lines of analysis (LOAs) used in this comparative assessment were located on the Hanford Site along lines approximately 1 km (0.6 mi) downgradient of aggregate Hanford solid waste (HSW) disposal areas within the 200 East and West Areas, ERDF, and near the Columbia River located downgradient from all disposal site areas (see Figure G.1). Additional analyses of potential groundwater quality

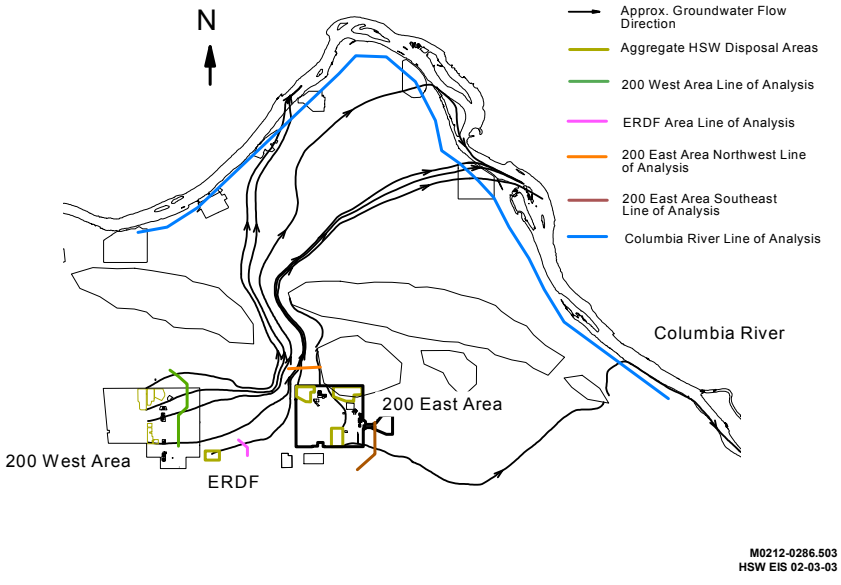

Figure G.1. Lines of Analysis Downgradient of Aggregate Hanford Solid Waste Disposal Areas

impacts for a new combined-use facility (as presented for Alternative Groups D_1, D_2, and D_3), are presented in Section G.5 and provide a perspective on the relative impact at waste management boundaries about 100 m downgradient of the aggregate waste disposal area versus potential impacts at the 1-km LOAs. A similar impact analysis is provided for LLW and MLLW disposed of before 2007 for another perspective.

LOAs were selected based on transport results of unit releases at selected HSW disposal site locations. LOAs approximately 1 km (0.6 mi) downgradient from the overall waste disposal facilities in each area are not meant to represent points of compliance, but rather common locations to facilitate comparison of potential impacts from waste management selections and locations defined for each alternative group.

Predicted constituent concentrations presented for each alternative group from specific water category releases represent maximum concentrations estimated along these LOAs. Because of the variation in the location of the different waste types and category releases for a given alternative group, the estimated maximum concentrations calculated from a specific waste category release may not correspond to the same point on the line analysis for every waste category and alternative group. For the sake of being conservative, however, combined concentration levels presented for each LOA and alternative group reflect the summation of predicted concentration levels regardless of their position on the LOA.

Delineation of potential waste impacts in the 200 East Area required two different LOAs. One LOA, designated as the 200 East Northwest (NW) LOA, is used to evaluate concentrations in groundwater migrating northwest of the 200 East Area. Another LOA, designated as the 200 East Southeast (SE) LOA, is used to evaluate concentrations in groundwater migrating southeast of the 200 East Area.

G.1.2 Overall Analytical Approach

To estimate the concentration of contaminants in the groundwater, it was necessary to link the results of process models of waste release, transport through the vadose zone, and transport through the groundwater system. Two general approaches are available to link these models. One approach involves simulating a contaminant inventory distribution through each of the three process models. The other approach involves simulating a unit release through each of the three process models and superimposing these results with a specific constituent inventory distribution.

The first approach requires that each of the calculations be performed sequentially with each simulation representing a unique inventory distribution and parameter set. This approach is preferred when the number of combinations of inventory distributions and parameter sets is small compared with the total number of simulations required.

The second approach involves development of system output or response and, from that, a unit release that can be simulated for each source area, parameter set, and process model. (In this case, the process models include estimating source release, vadose zone flow and transport, and groundwater flow and transport.) Unit releases in each of the process models can be simulated independently. Then, by making the assumption of linearity, the unit release responses from each individual source area, via each of the process models, can be combined or superimposed using the convolution integral approach

(Lee 1999). The convolution calculational approach is preferred when the number of combinations of inventory distributions and parameter sets is large compared with the number of vadose zone and groundwater flow and transport scenarios that need to be simulated. This second approach was selected for this analysis.

The convolution approach and the implicit assumption of linearity provide a reasonable approach in approximating the long-term release of constituents from solid waste disposal facilities for the following reasons:

- The waste zone environment of solid waste sources in HSW disposal facilities has been characterized as a low-organic, low salt, near neutral geochemical environment (Kincaid et al. 1998) and, as such, processes such as non-linear adsorption and other complex chemical reactions are not expected to have a substantial effect on contaminant release and transport through the vadose zone and groundwater water at the scales of interest (that is, 100 m downgradient of the waste facilities to the Columbia River).

- Wastes disposed of in HSW disposal facilities are largely dry solids and do not have any substantial amount of liquids or complex chemical fluids that could enhance migration of constituents to the underlying water table.

- Waste releases are expected to occur over long periods of time and will likely reach the water table when the effect of past artifical discharges has dissipated and the unconfined aquifer returns to more natural conditions. Using estimates of infiltration through the vadose zone to the underlying groundwater that would reflect long-term average rates of natural recharge would appear reasonable.

The convolution approach used also incorporates the process of solubility control that is assumed to be important in the source release for some constituents. The effect of this process is approximated by applying appropriate solubility controls in the source-term release component of the analysis. This approach can be effectively used without disrupting the superposition process. Solubility-controlled release models were used in the calculation of source-term release of the uranium isotopes in each of the alternatives.

In the convolution integral calculational approach, the concentration in the groundwater at a specific location, *i,* at time, *t,* ($C_{i,t}$) can be estimated using Equations G.1 and G.2:

$$C_{i,t} = \sum_{s=1}^{n} \mathrm{M}_s \sum_{\mathrm{T}=1}^{t} (\mathrm{f}_{s,\mathrm{T}}\, c_{s,i,t-\mathrm{T}+1}) \qquad \text{(G.1)}$$

$$\mathrm{f}_{s,t} = \sum_{\mathrm{T}=1}^{t} (r_{s,\mathrm{T}} f_{s,t-\mathrm{T}+1}) \qquad \text{(G.2)}$$

where $C_{i,t}$ = Concentration at location, *i*, at time, *t*

M_s = Inventory at source, *s*

$c_{s,i,t}$ = Groundwater concentration at *i* based on a unit release from *s* (Coupled Fluid, Energy, and Solute Transport [CFEST] model output)

$r_{s,t}$ = Fractional release of unit inventory in source *s* at time *t* (Release model output)

$f_{s,t}$ = Flux to water table from source, *s,* at time, *t,* based on unit release from *s* (Subsurface Transport Over Multiple Phases [STOMP] model output)

^{n} = number of sources

T = time integration variable.

and where $c_{s,i,t}$ and $f_{s,t}$ are the discrete response functions estimated with the vadose zone and groundwater models based on a unit release. These discrete responses can be quickly combined with Equations G.1 and G.2 (that is, superimposed) in a variety of combinations to estimate system responses to different inventory distributions and parameter sets. (Note that equations G.1 and G.2 are discrete-approximation representations of the classic convolution integral calculational approach used in the calculation of superposition of responses in linear response systems.) The form of equation G.1 was also used to estimate the time-varying flux of a contaminant to the Columbia River by substituting the groundwater concentration based on a unit release from *s* with the calculated flux to the river based on a unit release from *s*. This river flux was combined with average annual river flows in the Columbia River to estimate river concentration levels that provided the basis for potential human health impacts and ecosystem risk from exposure to Columbia River water.

Potential impacts from the subsurface transport pathway were analyzed for the LLBGs. The contaminant inventory for the LLBGs was assumed to be released to the vadose zone according to an appropriate release model. Transport within the vadose zone was estimated with a steady-state, one-dimensional variably saturated vadose zone transport model by assuming a unit release for a range of recharge rates. Travel times for releases of unit mass were defined by arrival of 50 percent of each unit mass. These travel times were used to translate mass releases from the LLBGs into mass releases at the water table in the aquifer. The time-varying mass flux arriving at the water table reflects the entire time history of the mass release from the source area, as well as the calculated travel time in the vadose zone.

Estimates of contaminant release transport from the LLBGs to the groundwater were evaluated. This evaluation was done by first calculating transport of 10-year releases of a unit of dry mass into the unconfined aquifer at the approximate locations of the LLBGs at the water table. These transport calculations were made with a steady-state, three-dimensional saturated groundwater flow and transient transport model. These calculated concentrations, based on a unit release, were then used in the convolution integral calculational method to translate transport of mass releases from the LLW through the vadose zone and the aquifer to specified locations downgradient from the source areas. The concentrations in the groundwater plumes for each radionuclide were translated into doses using methods described in Appendix F.

The sequence of calculations used in the long-term assessment required estimating the potential groundwater quality impacts using a suite of process models that estimated source-term release, vadose zone flow and transport, and groundwater flow and transport. The computational framework for these process models and relationship of software elements, which are schematically illustrated in Figure G.2, are as follows:

1. Microsoft® Excel worksheet
2. Dynamically linked library version of the STOMP code (White and Oostrom 1996, 1997; Nichols et al. 1997)
3. Coupled Fluid Energy and Solute Transport (CFEST) code (Gupta et al. 1987; CFEST, Co. 1997)

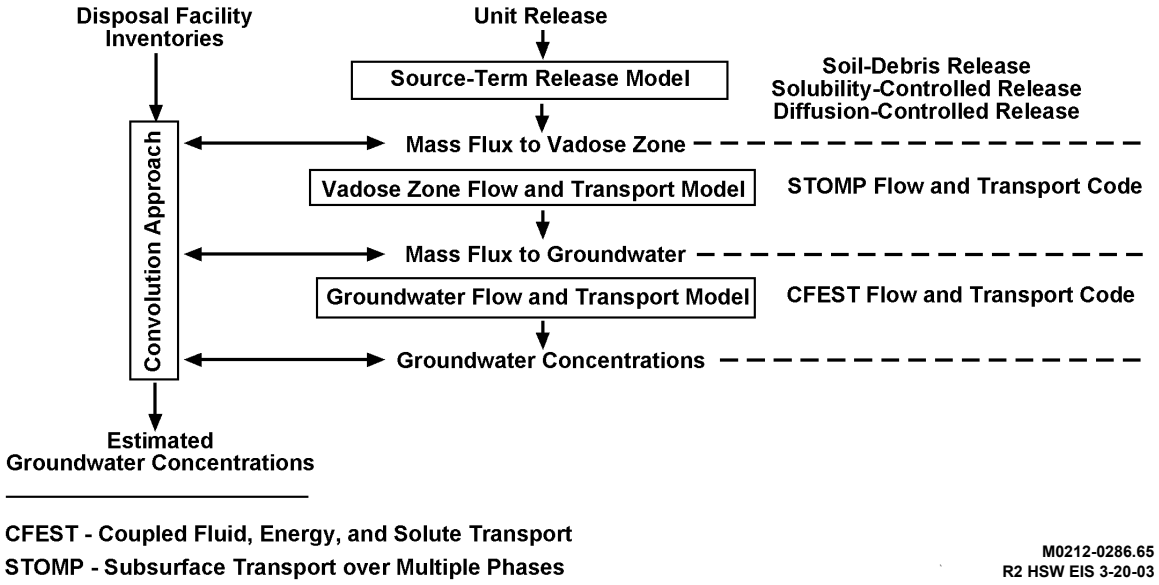

Figure G.2. Schematic Representation of Computational Framework and Codes Used in this HSW EIS

The concentrations in the groundwater plumes for each radionuclide were translated into potential human health impacts, which are summarized in 5.11 and Appendix F.

The methodologies for calculating source-term release, vadose zone transport, and groundwater transport are described in the following sections. Assumptions (for example, geometry, initial conditions, boundary conditions, and parameters) for each calculation are identified and discussed. The implementation of each model for each alternative is described.

G.1.3 Source-Term Release

The source-term is the quantification of when and what constituents (by mass or activity) would be released. This source-term includes the water flux into the vadose zone that results from precipitation

infiltrating the waste and mass or activity solubilized from dissolution of waste in the LLBGs. This section addresses the approach and methods used for source-term release that involve:

- grouping of constituents into categories based on their mobility and screening to determine which constituents should be considered in this analysis
- aggregating potential sources into common source areas
- developing the contaminant inventories for each source area
- selecting appropriate source-term release models to calculate mass flux and fluid flux release as a function of time.

G.1.3.1 Constituent Grouping and Screening

The LLBGs contain over 100 radioactive and non-radioactive constituents that potentially could impact groundwater. Screening of these constituents considered a number of aspects that included 1) their potential for dose or risk, 2) their estimated amount of inventory, and 3) their relative mobility in the subsurface system within a 10,000-year period of analysis.

The assessment was the beneficiary of preceding analyses and field observations including the performance assessments for 200 West and 200 East post-1988 burial grounds (Wood et al. 1995, 1996), the remedial investigation and feasibility study of the ERDF (DOE 1994), the disposal of ILAW originating from the single- and double-shell tanks (Mann et al. 1997) and (Mann et al. 2001), and the Composite Analysis of the 200 Area Plateau (Kincaid et al. 1998). These and other analyses included development of inventory data and application of screening or significance criteria to identify those radionuclides that could be expected to substantially contribute to either the dose or risk calculated in the respective analysis. Clearly, those radionuclides identified as potentially significant in these published analyses are also expected to be key radionuclides in this assessment.

To establish their relative mobility, the constituents were grouped based on their mobility in the vadose zone and underlying unconfined aquifer. Contaminant mobility classes were used rather than the individual mobility of each contaminant because of the uncertainty involved in determining the mobility of individual constituents. The mobility classes were selected based on relatively narrow ranges of mobility.

Some of the constituents, such as iodine and technetium, would move at the rate of water whether in the vadose zone or underlying groundwater. The movement of other constituents in water, such as americium and cesium, would be slowed or retarded by the process of sorption onto soil and rock. A parameter that is commonly used to represent a measure of this sorption is referred to as the distribution coefficient or K_d. This parameter is defined as the ratio of the quantity of the solute adsorbed per gram of solid to the amount of solute remaining in solution (Kaplan et al. 1996). Values of K_d for the constituents range from 0 mL/g (in which the contaminant movement in water is not retarded) to more than 40 mL/g (in which the contaminant moves much slower than water).

The LLW inventory constituents were grouped according to established K_ds for each constituent, or an assumed conservative K_d where a range of K_ds is known for a particular constituent. The constituent groups, based on mobility and examples of common constituents, are described in the following text.

A summary of all constituents and associated groupings (based on K_d values) is provided in Table G.1. The constituent classes used for modeling include:

Mobility Class 1 – Contaminants were modeled as non-sorbing (that is, $K_d = 0$) and would not be retarded in the soil-water system. Contaminant K_d values in this group ranged from 0 to 0.59 mL/g and include all the isotopes of iodine, technetium, selenium, chlorine, and tritium.

Mobility Class 2 – Contaminants were modeled as slightly sorbing (that is, $K_d = 0.6$) and would be slightly retarded in the soil-water system. Contaminant K_d values in this group ranged from 0.6 to 0.99 mL/g and include all the isotopes of uranium and carbon.

Mobility Class 3 – Contaminants were modeled as slightly more sorbing (that is, $K_d = 1$). Contaminant K_d values in this group ranged from 1 to 9.9 mL/g and include all the isotopes of barium.

Mobility Class 4 – Contaminants were modeled as moderately sorbing (that is, $K_d = 10$). Contaminant K_d values in this group ranged from 10 to 39.9 mL/g and include all the isotopes of neptunium, palladium, protactinium, radium, and strontium.

Mobility Class 5 – Contaminants were modeled as strongly sorbing (that is, $K_d = 40$). Contaminant K_d values in this group were 40 mL/g or greater and include all the isotopes of actinium, americium, cobalt, curium, cesium, iron, europium, gallium, niobium, nickel, lead, plutonium, samarium, tin, thorium, and zirconium.

The constituent listing in Table G.1 was further evaluated using estimates of constituent transport times through the thick vadose zone to the unconfined aquifer during the 10,000-year period of analysis. For purposes of this analysis, the infiltration rate selected was 0.5 cm/yr. This rate was assumed, based on recharge estimates for different site surface conditions by Fayer et al. (1999), to reflect a conservative estimate of infiltration for surface conditions that would be expected to persist at the LLBGs during the post-closure period. Estimates by Fayer et al. (1999) indicate that infiltration rates for surface conditions that have a Modified Resource Conservation and Recovery Act (RCRA) Subtitle C Barrier system would be below the assumed 0.5 cm/yr rate used in this screening analysis.

Based on this assumed infiltration rate and estimated levels of sorption and associated retardation for each of the classes above, estimated travel times of all constituents in Mobility Classes 3, 4, and 5 through the thick vadose zone to the unconfined aquifer beneath the LLBGs were calculated to be well beyond the 10,000-year period of analysis. Using the same vadose zone recharge rate of 0.5 cm/yr, average travel times to the water table for constituents within Mobility Classes 3, 4, and 5 are estimated to range from 30,000 to 50,000 years, 250,000 to 400,000 years, and 800,000 to 1 million years, respectively. Thus all constituents in these classes were eliminated from further consideration.

Table G.1. Constituents Categorized by Mobility (K_d) Classes

Constituent	Best K_d Estimate	Range of K_d Estimates	Reference	Half-Life (years)
Mobility Class 1 (K_d = 0.0 mL/g)				
H-3	0	0–0.5	Kincaid et al. (1998)	1.2E+01
Tc-99	0	0–0.6 0–0.1	Kincaid et al. (1998) Cantrell et al. (2002)	2.1E+05
I-129	0.3	0.2–15 0–2	Kincaid et al. (1998) Cantrell et al. (2002)	1.5E+07
Cl-36	0	0–0.6	Kincaid et al. (1998)	3.8E+05
Se-79	0	0–0.78	Kincaid et al. (1998)	6.5E+05
Mobility Class 2 (K_d = 0.6 mL/g)				
C-14	0.5	0.5–1,000	Kincaid et al. (1998)	5.7E+03
U-232 U-233 U-234 U-235 U-236 U-238	0.6	0.1–79.9 0.2–4	Kincaid et al. (1998) Cantrell et al. (2002)	6.9E+01 1.5E+05 2.4E+05 7.0E+08 2.3E+07 4.5E+09
Mobility Class 3 (K_d = 1.0 mL/g)				
Ba-133	1	NA	Wood et al. (1995)	1.0E+01
Mobility Class 4 (K_d = 10.0 mL/g)				
Np-237	15	2.4–21.9	Kincaid et al. (1998)	2.1E+06
Pa-231	15	2.4–21.9	Kincaid et al. (1998)	3.3E+04
Pd-107	10	NA	DOE and Ecology (1996)	6.5E+06
Ra-226	20	5–173	Kincaid et al. (1998)	1.6E+03
Sr-90	20	5–173 10–20	Kincaid et al. (1998) Cantrell et al. (2002)	2.8E+01
Mobility Class 5 (K_d = 40.0 mL/g)				
Ac-227	300	67–1,330	Kincaid et al. (1998)	2.1E.01
Am-241 Am-242m Am-243	300	67–1,330	Kincaid et al. (1998)	4.3E+02 1.5E+02 7.4E+03
Co-60	1200	1,200–12,500	Kincaid et al. (1998)	5.3E+00
Cm-243 Cm-244 Cm-245 Cm-246 Cm-248	300	67–1,330	Kincaid et al. (1998)	2.9E+01 1.8E+01 8.4E+03 4.7E+03 3.4E+05
Cs-135 Cs-137	1500	540–3,180	Kincaid et al. (1998)	2.3E+06 3.0E+01
Eu-152	300	67–1,330	Kincaid et al. (1998)	1.3E+01
Gd-152	100	NA	Wood et al. (1996)	1.1E+14
Nb-94	300	50–2,350	Kincaid et al. (1998)	2.0E+04
Ni-63	300	50–2,350	Kincaid et al. (1998)	1.0E+02

Table G.1. (contd)

Constituent	Best K_d Estimate	Range of K_d Estimates	Reference	Half Life (years)
Mobility Class 5 (K_d = 40.0 mL/g) - continued				
Pb-210	2000	13,000–79,000	Kincaid et al. (1998)	2.2E+01
Pu-238 Pu-229 Pu-240 Pu-242 Pu-244	200	80 – >1,980	Kincaid et al. (1998)	8.7E+01 2.4E+04 6.5E+03 3.7E+05 8.1E+07
Th-229 Th-230 Th-232	1000	40 – >2,000	Kincaid et al. (1998)	7.3E+03 7.7E+04 1.4E+10
Sm-147	100	NA	Wood et al. (1996)	1.1E+11
Sn-126	50	50–2,350	Kincaid et al. (1998)	9.9E+04
Zr-93	1000	40 – >2,000	Kincaid et al. (1998)	1.5E+06
NA = not applicable.				

Of the suite of remaining waste constituents, technetium-99 and iodine-129 in Mobility Class 1 and carbon-14 and the uranium isotopes in Mobility Class 2 were considered to be in sufficient quantity and mobile enough to warrant a detailed analysis of potential groundwater impacts. Although three of the constituents in Mobility Class 1—selenium, chloride, and tritium—are considered very mobile, they were screened out for other factors. Selenium and chloride were not considered in the assessment because the total inventories for both of these constituents were estimated to be less than 1×10^{-2} Ci. Tritium was not evaluated because of its relatively short half-life.

Estimated inventories of hazardous chemical constituents associated with LLW and MLLW disposed of after 1988 being considered under each alternative group would be expected to be found at trace levels. MLLW, which would be expected to contain the majority of hazardous chemical constituents, would undergo predisposal solidification to stabilized waste forms and containment and thermal treatment to remove organic chemical components of the MLLW. This waste treatment would be done to meet current waste acceptance criteria and land disposal restrictions before being disposed of in permitted MLLW facilities. Consequently, potential groundwater quality impacts from these constituents would not be expected to be substantial.

Analysis of MLLW inventories for this assessment did identify two exceptions that included lead and mercury inventories associated with the projected MLLW that were estimated at 336 kg (741 lb) and 2.5 kg (5.5 lb), respectively. Because of its affinity to be sorbed into Hanford sediments, lead falls within Mobility Class 5 (K_d = 40 mL/g) and would not release to groundwater within the 10,000-year period of interest. The inventory estimated for mercury is assumed to be small enough that it would not release to groundwater in substantial concentrations. Even the most conservative estimates of release would yield estimated groundwater concentrations at levels two orders of magnitude below the current drinking water standard for mercury of 0.002 mg/L.

LLW disposed of before October 1987 may contain hazardous chemical constituents, but no specific requirements existed to account for or report the content of hazardous chemical constituents in this category of LLW. As a consequence, analysis of these constituents and estimated impacts based on the limited amount of information on estimated inventories and waste disposal locations would be subject to uncertainty at this time. These facilities are part of the LLW and MLLW facilities in the LLW Management Areas (MA) 1 through 4 that are currently being monitored under RCRA interim status programs. Final closure or remedial investigation of these facilities under RCRA and/or CERCLA guidelines could involve further analysis of the potential impacts of the chemical components of these inventories.

In response to comments received during the public comment periods on the drafts of the HSW EIS, efforts were made to develop an estimate of quantities of potentially hazardous chemicals in previously buried LLW so that potential impacts of such chemicals on groundwater quality could be evaluated. The estimation of these inventories, which used a waste stream analysis estimation method, is summarized in the Technical Information Document (FH 2004). This initial assessment of the estimated hazardous chemical inventory in pre-1988 buried wastes is provided in Section G.6.

G.1.3.2 Source Inventories

The source inventories of key constituents that provided the basis for potential groundwater quality impacts described in this appendix and Section 5.3 are summarized by alternative group in Appendix B. The inventory associated with the specific constituents for each of alternatives was partitioned between the 200 East and West Areas roughly in proportion to estimated disposal areas in the LLBGs that had already received LLW or will receive newly generated LLW. Estimates of LLBG areas for all the alternatives are summarized in Volume 1, Section 5.1, Table 5.1. Distribution of LLBGs for each waste category assumed in the release modeling, described in the section below, in the HSW disposal site areas by alternative are given in Table G.2. The broad categories considered include previously disposed of LLW, newly generated Cat 1 and Cat 3 LLW, and MLLW. The relative percentages of LLBG areas for these three categories provide the basis for the partitioning of LLW volumes and associated constituent inventories. For purposes of this analysis, the GTC3 LLW were considered part of the Cat 3 LLW inventory. Although no specific GTC3 LLW is expected in forecasted wastes, for purposes of this analysis, it was assumed that about 1 m^3 (1.4 yd^3) of GTC3 LLW containing mostly cesium-137 and other non-mobile nuclides would be part of the inventory considered. The inventory of this category is included in the Cat 3 LLW and is not discussed separately.

G.1.3.3 Release Models

Source-release models were selected and used to approximate contaminant releases from the variety of LLW types considered in this analysis. The models considered included a soil-debris release model and a cement release model.

Table G.2. Assumed Distribution of LLBG Areas (ha) of Previously Disposed of LLW, Cat 1 LLW, Cat 3 LLW, MLLW, and Melters in the 200 East and 200 West Areas by Alternative Group

	Previously Disposed of LLW						Category 1 LLW				Category 3 LLW				MLLW				Melters
	1962–1970 LLW		1970–1988 LLW		1988–1995		1996–2007		After 2007		1996–2007		After 2007		1996 to date and future		After 2007		
Disposal Alternative	200 East	200 West	200 East	200 West	200 East	200 West	200 East	200 West	200 East	200 West	200 East	200 West	200 East	200 West or ERDF	200 East	200 West or ERDF	200 East	200 West or ERDF	200 East or ERDF
A (Lower Bound Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7		4.4		39.7		4.4		1.7		1.5	6.0
A (Hanford Only Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7		4.4		39.7		4.4		1.7		1.5	6.0
A (Upper Bound Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7		8.9		39.7		8.9	3.5	1.7		3.0	6.0
B (Lower Bound Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7	0.7	16.7		39.7	0.7	16.7		1.7	5.7		6.0
B (Hanford Only Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7	0.7	16.7		39.7	0.7	16.7		1.7	5.7		6.0
B (Upper Bound Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7	4.0	25.1		39.7	1.1	28.0	3.5	1.7	10.2		6.0
C (Lower Bound and Hanford Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7		4.4		39.7	0.0	4.4		1.7	1.5		6.0
C (Hanford Only Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7		4.4		39.7	0.0	4.4		1.7	1.5		6.0
C (Upper Bound Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7		8.9		39.7	0.0	8.9	3.5	1.7	3.0		6.0
D_1, D_2, and D_3 (Lower Bound Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7	3.0			39.7	3.0			1.7	1.1		6.0
D_1, D_2, and D_3 (Hanford Only Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7	3.0			39.7	3.0			1.7	1.1		6.0
D_1, D_2, and D_3 (Upper Bound Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7	6.2			39.7	6.2		3.5	1.7	3.0		6.0
E_1, E_2, and E_3 (Lower Bound Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7	3.0			39.7	3.0			1.7	1.1		6.0
E_1, E_2, and E_3 (Hanford Only Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7	3.0			39.7	3.0			1.7	1.1		6.0
E_1, E_2, and E_3 (Upper Bound Volume)	7.1	2.2	20.9	16.6	19.6	39.7		39.7	6.2			39.7	6.2		3.5	1.7	3.0		6.0
No Action	7.1	2.2	20.9	16.6	19.6	39.7		39.7		39.7		39.7				1.7			

G.1.3.3.1 Soil-Debris Model

In the soil-debris model, LLW is assumed to be mixed with soils. Waste sources included in this model were assumed to be permeable to percolating water. Thus, all surfaces of the waste were assumed to come into contact with percolating water. If contaminant inventories in the source were high enough, leaching of the contaminant through the bottom of the source was controlled by the solubility of the contaminant in soil water. Otherwise, leaching was controlled by partitioning of the radionuclides between aqueous and sorbed phases. The inventory was assumed to be perfectly mixed throughout the source volume during the entire release period—assuming perfectly mixed conditions reduced the likelihood that solubility would control the release. The mathematical basis of this release model is described in detailed in Appendix D of Kincaid et al. (1998).

The soil-debris model was used to estimate release of all non-grouted contaminants from previously disposed of LLW, Cat 1 LLW, Cat 3 LLW, and MLLW. The key parameter in the use of the soil-debris release model, besides the depth of the waste, is the rate of infiltrating water through the LLBGs. Table G.3 provides a summary of assumed waste depths and infiltration rates used in the soil-debris model for each alternative.

This assessment focuses on the long-term release of contaminants from new LLBGs during the post-closure period. This assumption of minimal leaching and migration prior to site closure is reasonable for the majority of LLW and MLLW being considered. Containment and waste forms used in Cat 1 and Cat 3 LLW would be expected to be sufficient to contain and isolate disposed of LLW during the operational period. MLLW facilities, which involve the collection and management of leachate during and following the operational period, are also expected to control the amount of waste leaching during the period of operations. Thus, an infiltration rate of 0.5 cm/yr was used for the Cat 1 LLW, Cat 3 LLW, and MLLW within the No Action Alternative.

Because less rigorous requirements for waste contaminant and content were in effect prior to 1988, contaminants contained in solid LLW disposed of in LLBGs before 1988 offer the highest potential for leaching and release into the vadose zone prior to site closure. This analysis evaluated the potential impacts of these earlier disposals by evaluating the effect of higher infiltration rates during the period of operations. The leaching of these categories of LLW prior to site closure has the potential to be influenced by relatively high infiltration rates during and shortly after the disposal period when bare soil conditions persist. Infiltration rates into coarse surface sediments maintained free of vegetation, as would be expected during and shortly after the disposal period, is estimated to be in the order of 5 cm/yr, based on data from a non-vegetated, gravel-covered lysimeter study conducted on the Hanford Site (Fayer and Walters 1995; Fayer et al. 1999). Eventually, infiltration through the LLBGs would be expected to be reduced to lower levels as surface cover conditions return to a more natural vegetative state.

For the No Action Alternative, an infiltration rate used in release modeling of the pre-1970 and 1970-1988 LLW was increased to 0.5 cm/yr after the operational period and during the post-closure period. This infiltration rate is a reasonable rate (Fayer and Walters 1995; Fayer et al. 1999) to use in the post-closure period when natural vegetative cover would be expected to persist.

Table G.3. Summary of Waste Depth and Infiltration Rates Used in the Soil-Debris Release Model

	Waste Depth (meters)	Infiltration Used in Waste Release Models (cm/yr)							
		Prior to 2046	2046–2546	2547–2646	2647–2746	2747–2846	2847–2976	2947–2946	3046–12046
Action Alternatives									
Wastes Disposed of Before 1995									
Pre-1970	6	5	0.01	0.05	0.1	0.2	0.3	0.4	0.5
1970–1987	6	5	0.01	0.05	0.1	0.2	0.3	0.4	0.5
1988–1995	6	5	0.01	0.05	0.1	0.2	0.3	0.4	0.5
Wastes Disposed of Between 1996 and 2007	6	NA	0.01	0.05	0.1	0.2	0.3	0.4	0.5
Wastes Disposed of After 2007									
Alt Group A	15.6	NA	0.01	0.05	0.1	0.2	0.3	0.4	0.5
Alt Group B	6	NA	0.01	0.05	0.1	0.2	0.3	0.4	0.5
Alt Group C	15.6	NA	0.01	0.05	0.1	0.2	0.3	0.4	0.5
Alt Group D_1	15.6	NA	0.01	0.05	0.1	0.2	0.3	0.4	0.5
Alt Group D_2	15.6	NA	0.01	0.05	0.1	0.2	0.3	0.4	0.5
Alt Group D_3	15.6	NA	0.01	0.05	0.1	0.2	0.3	0.4	0.5
Alt Group E_1	15.6	NA	0.01	0.05	0.1	0.2	0.3	0.4	0.5
Alt Group E_2	15.6	NA	0.01	0.05	0.1	0.2	0.3	0.4	0.5
Alt Group E_3	15.6	NA	0.01	0.05	0.1	0.2	0.3	0.4	0.5
Melter Trench (All Alternatives Groups)	18.6	NA	0.01	0.05	0.1	0.2	0.3	0.4	0.5
No Action Alternative									
Wastes Disposed of Before 1995									
Pre-1970	6	5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
1970–1987	6	5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
1988–1995	6	5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Wastes Disposed of After 1996	6	NA	0.5	0.5	0.5	0.5	0.5	0.5	0.5

NA = No specific infiltration rate is applicable for release of waste disposed of after 1995 during the period of operation for the alternative groups. Because of assumptions related to waste containment and active management of leachate collection during the operational period, no waste release is assumed to occur until after the start of the post-closure period in year 2046.

For all LLW and MLLW under all action alternatives, it is assumed that LLBGs would have a long-term surface barrier at site closure that would limit infiltration rates through the disposed of wastes. The assumed barrier is a Modified RCRA Subtitle C Barrier system. Recharge from this barrier system is expected to be very low and comparable to long-term recharge estimates for the Hanford Protective Barrier. A recent analysis by Fayer et al. (1999) for ILAW disposal has estimated a long-term infiltration at 0.01 cm/yr through this type of a system with an established natural (that is, shrub-steppe plant community) cover condition.

No guidance is available for specifying barrier performance after its design life. However, an immediate decrease in performance is not expected, and it is likely that this specific barrier will perform as designed far beyond its design life. Without data to understand and predict long-term performance of the specific barrier, a conservative assumption is the performance of the barrier would degrade stepwise after reaching its design life, and until the recharge rate matches the natural recharge rate in the surrounding environment. This approach is based on the assumption that a degraded cover will eventually return to its natural state and behave like the surrounding environment. The period of degradation was assumed to be the same as the design life. At the time of site closure, all waste disposal facilities are assumed to be covered with the Modified RCRA Subtitle C Barrier system. To approximate the effect of the cover on waste release, the following assumed infiltration rates, as illustrated in Figure G.3, were used in the waste release modeling. For 500 years after site closure, an infiltration rate of 0.01 cm/yr was used to approximate the effect of cover emplacement over the wastes and its potential impact on reducing infiltration. After 500 years, the cover is assumed to begin to degrade. Between 500 to 1000 years after site closure, infiltration rates were increased from 0.01 cm/yr to 0.5 cm/yr to approximate a 500-year period of cover degradation and a return infiltration rate reflective of natural vegetated surface soil conditions over the wastes. The final rate of 0.5 cm/yr was used for the remaining 9000-year period of analysis.

Additional analyses were performed to provide perspective on potential impacts using two assumptions: 1) no cover system is installed and 2) a cover system is used and remains intact for 10,000 years (see Section G.4.)

A number of the alternatives considered specify the use of liner systems to control waste release during the period of operations. However, no credit for the effect of these liner systems was considered in this long-term analysis. Although the liner systems as described in Section 3.1 might last (that is, contain leachate for removal) for several hundreds of years if properly managed, this analysis assumed that the emplaced liners would fail during the 100-year active institutional control period and would have little effect on the long-term waste release during the 10,000-year period of analysis.

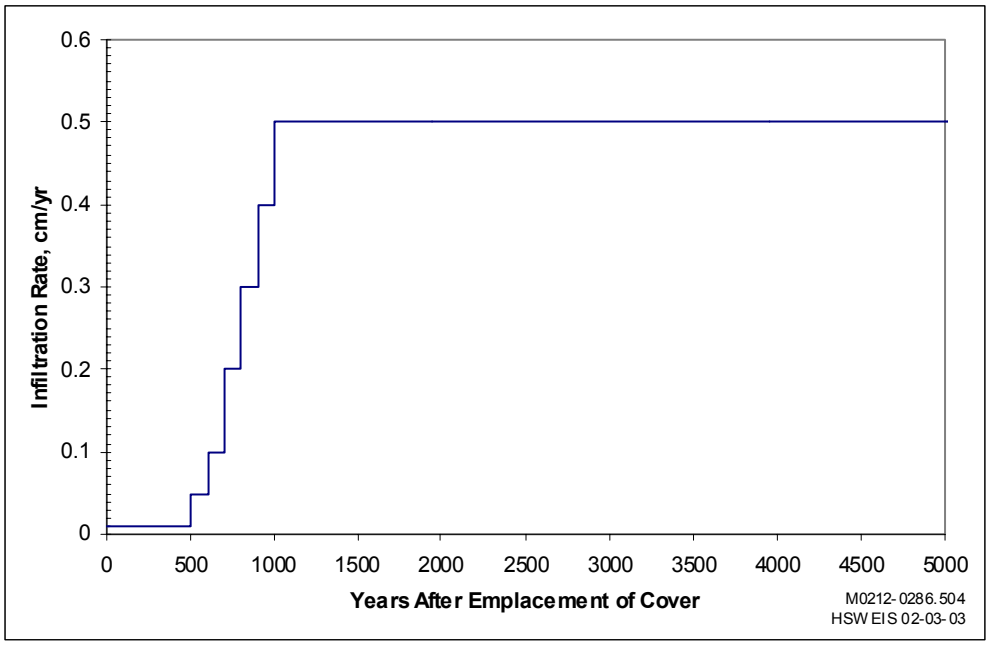

Figure G.3. Changes in Infiltration Rates Assumed in Source-Term Release to Approximate the Modified RCRA Subtitle C Barrier System Degradation

In the case of uranium isotope release calculations, sufficient inventories of uranium in a number of LLW categories were estimated with the soil-debris model using solubility controls. For all LLW categories except Cat 3 LLW, a solubility-controlled concentration of 64 mg/L was used for all uranium isotopes. This estimate was developed and described for Hanford-specific conditions in Wood et al. (1996) for use in the performance assessment of solid waste burial grounds in the 200 East Area. In the Cat 3 LLW, the geochemical environment created by the presence of cement associated with the high-integrity containers (HIC) and the in-trench grouting is expected to reduce the release of uranium at much lower concentration limits. The solubility-controlled concentration used for Cat 3 LLW was 0.23 mg/L, which was based on an estimate (2.34×10^{-4} g/L) developed and described in Wood et al. (1996) for use in the performance assessment of solid waste burial grounds in the 200 East Area.

To account for the expected delay in release of Cat 3 LLW, because it is contained within HICs or grouted in place, the soil-debris release model used a 300-year delay before releases were initiated. This delay is consistent with the estimated 300-year lifetime of LLW containment effectiveness of the HIC or in-trench grouting.

For some categories (Cat 3 LLW and Cat 3 MLLW) in each of the alternatives, LLW containing elevated levels of technetium-99 will be placed in a grout matrix before being placed in the LLBGs. For this type of grouted waste, a release model referred to as the cement-release model was used to approximate the source release. The underlying basis of the cement-release model assumes that (1) the permeability of the grouted waste is much lower than that of the surrounding soil, (2) the permeability of the waste is low enough that advective water flow within the waste form is essentially zero, and (3) the

pore space connectivity in the cementitious waste form is sufficiently high enough to allow contaminant mobility within the waste form by diffusion. The mathematical basis of this release model is also described in detail in Appendix D of Kincaid et al. (1998).

G.1.3.3.2 Cement-Release Model

In the cement-release model, percolating water is assumed to move around the grouted waste, and contaminants are leached only from the outer surface. As this occurs, contaminants inside the waste form are assumed to diffuse toward the outer surface. Therefore, overall contaminant release from the source zone is assumed to be controlled by the effective diffusion coefficient of the contaminant in the waste form.

Specific values of the effective diffusion coefficient in cement-release model type waste forms for each radionuclide were chosen from the values originally reported by Serne et al. (1989). These values had previously been incorporated into a computer database known as the Multimedia-Modeling Environmental Database Editor (MMEDE) (Warren and Strenge 1994). For the source-term calculation effort of this analysis, the MMEDE database was queried to produce an electronic file of tabulated diffusion coefficients for relevant radionuclides (that was subsequently incorporated into the source-term calculation spreadsheet). This study used diffusion coefficient values as reported in Buck et al. (1997). Diffusion coefficients of 1×10^{-11} and 1×10^{-12} cm^2 s^{-1} for technetium-99 and iodine-129, respectively, were used. For some radionuclides (for which no specific values were available), the diffusion coefficient was fixed at a reasonable conservatively high default value (5×10^{-8} cm^2 s^{-1}).

Effect of Organic Chemicals on Long-Term Groundwater Quality Impacts

The effect of chemicals, particularly organic chemicals, on enhancing the mobility of normally sorbed or immobile constituents in transport was raised as an important technical issue for solid waste disposal facilities during public review and comment of the first draft HSW EIS. Detailed evaluations of tabulations of metal-organic complex stability constants for organic compounds (Martell 1971; Martell and Smith 1977; Smith and Martell 1982) suggest that most of the stability constants are weak for organics typically contained in LLW and MLLW. The more typical organic compounds found in LLW and MLLW are non-polar and relatively hydrophobic molecules. Organics that fit into this category (that is, carbon tetrachloride, trichloroethane, and other volatile organics) generally cannot form a complex with metals and radionuclides and enhance their mobility. However, such non-polar and/or hydrophobic organic compounds if disposed in large quantities and in high concentration could potentially affect radionuclide and metal migration by creating a reducing zone in the sediments or groundwater especially if biological activity is occurring. Field evidence suggests that this has not occurred to any significant extent at any waste site at Hanford (see Serne and Wood 1990 and references therein). Thus this type of enhanced transport is not expected to be important in affecting field-scale transport of constituents of concern from HSW EIS disposal sites. A small subset of organic compounds, commonly referred to as complexing/chelating agents, do have the ability to enhance the mobility of some normally sorbed or immobile constituents. Some notable examples of such agents include EDTA, HEDTA, DTPA, oxalic acid, and tributyl phosphate. The ability of these complexing agents to affect the general mobility of normally immobile or sorbed radionuclides and metals is a function of many factors, including:

- the type and amount of organic complexing agent is present
- the stability of the complex and the kinetics of its formation and disassociation back to free molecules
- pH, REDOX, and microbiological conditions
- the amount of free liquids or fluids contained within the wastes.

In one instance onsite, the presence of complexing agents (EDTA and/or ferro-ferric-cyanide) in a liquid waste stream discharged to the ground is suspected of enhancing the transport of a cobalt-60 plume from the northern part of the 200 East Area. However, the combination of complexing agents and liquid discharge at this waste site is unique and cannot be interpreted as being representative of geochemical or vadose zone flow and transport conditions that would be expected at solid waste burial grounds.

At this time, there is no specific evidence that would support enhanced movement of moderately to strongly sorbed radionuclides or metals (for example. cesium, strontium, europium, uranium, or plutonium) due to the presence of organic complexing agents in solid wastes within LLBGs. In fact, no field-scale evidence has been found at other solid LLW sites across North America that would support this hypothesis (Serne et al. 1990; Serne et al. 1995). Estimated inventories of hazardous chemical constituents and particular organic complexing agents associated with LLW and MLLW disposed of after 1988 are thought to be quite small. MLLW, which would be expected to contain the majority of chemical constituents, will undergo predisposal solidification to stabilize waste forms and thermal treatment to remove organic chemical components of the MLLW. This waste treatment would be done to meet current waste acceptance criteria and land disposal restrictions before disposal in permitted MLLW facilities. Consequently, the effect of organic complexing agents and potential groundwater quality impacts from organic chemicals, in general, would not be expected to be substantial for solid wastes.

LLW disposed of prior to October 1987 might contain chemical constituents and organic complexing agents, but because no specific requirements existed to account for or to report their content, it is difficult to assess impacts. As a consequence, analysis of these constituents and estimated impacts based on the limited amount of information on estimated inventories and waste disposal location would be subject to large uncertainty at this time. These facilities are part of the LLW and MLLW facilities in LLW Management Areas 1–4 that currently are being monitored under RCRA interim status programs. Final closure or remedial investigations of these facilities under RCRA and/or CERCLA guidelines could involve further evaluation and eventually require analysis of the impacts of the chemical components of these disposed inventories.

Relation of the HSW-EIS to Current Performance Assessments for LLW and MLLW Disposal

The long-term radiological impacts of solid wastes disposed of in LLBGs in the 200 East and West Areas since October 1987 have been evaluated in two active performance assessments (*Performance Assessment for the Disposal of Low-Level Waste in the 200 West Area Burial Grounds* [Wood et al. 1995] and *Performance Assessment for the Disposal of Low-Level Waste in the 200 East Area Burial Grounds* [Wood et al. 1996]). These performance assessments were approved by DOE; a copy of the disposal authorization statement is attached to this appendix.

The proposed disposal of immobilized low-activity waste (ILAW) derived from the tank Waste Treatment Plant in a disposal facility sited southwest of the PUREX Plant within the 200 East Area also has been evaluated using a performance assessment (Mann et al. 2001). This performance assessment also was approved, as shown in the attached disposal authorization statement. Ongoing maintenance for all three of these performance assessments includes continual evaluation and production of annual reports on new data and information on projected disposal inventories, geochemical, and waste form performance data and information and their relevance to current performance assessment results and conclusions (Wood and Van Vliet 2002; Mann 2002).

Projected waste inventories, selection of disposal methods, or trench designs that might result from this HSW EIS would be addressed under performance assessment compliance requirements as specified in DOE Order 435.1 (DOE 2001). Long-term performance assessment of radiological impacts from disposal facilities is a part of several requirements specified under DOE Order 435.1 for Hanford Site low-level waste disposal facilities to ensure the protection of workers, the public, and the environment.

Analysis of the most current baseline disposal practices that use conventional trenches for both solid wastes and ILAW show that for current waste inventory projections, operational waste acceptance criteria and waste acceptance practices continue to be compliant with performance objectives.

G.1.4 Vadose Zone Modeling

Contaminants released from the various LLBGs were transported downward through the vadose zone to the water table. The primary mechanism for transport in the vadose zone was water flow in response to gravitational and capillary forces. After the LLW disposal operations cease, steady-state hydraulic conditions resulting from different surface covers (including re-vegetation) that affect recharge were represented in the model. Recharge directly from precipitation or snowmelt infiltrates into the vadose zone. The recharge rate varies for the assumed surface cover conditions for each of the LLBGs. The data used in the vadose zone model are described in the remainder of this section.

The vadose zone was modeled as a stratified one-dimensional column. In this analysis, it was not appropriate to represent the vadose zone as multidimensional because of the large number of LLBG sites modeled and the limited characterization of the vadose zone. Multidimensional modeling of the vadose zone has been performed for some waste sources and types (Mann et al. 1997; Mann et al. 2001) but was not practical for this analysis for the large number of sites in question. A one-dimensional approach would also be expected to yield results that would be more conservative than those produced with multi-dimensional approaches which consider lateral spreading of infiltration and contaminant transport.

The remainder of this section describes the stratigraphy, hydraulic properties, recharge, and geochemical conditions used in this analysis.

G.1.4.1 Stratigraphy

Because of the large number of sites to be modeled in this assessment, the technical approach used for the vadose zone stratigraphy was similar to the approach used in the Composite Analysis by Kincaid et al. (1998). The stratigraphy used was an approximation that was consistent with the major geologic formations found in the vadose zone beneath the Central Plateau in the areas of question and was based on work documented in Thorne and Chamness (1992), Thorne et al. (1993), and Thorne et al. (1994). In the composite analysis, the stratigraphies for several areas of the 200 East and 200 West Areas were defined as a set of strata consistent with the nearest available well log from 18 well logs (Kincaid et al. 1998). Each of the well logs included location, ground surface elevation, and the thickness of the various major sediment types.

A summary of the geologic well logs used in the composite analysis appears in Table G.4. At each profile location, seven sediment types, and one rock type (basalt) were identified and used to define the stratigraphy. The acronyms of the sediment types provided in Table G.5 are associated with the following sediment types: 200 West Area Hanford Sand (WHS) sediment, 200 West Area Early Palouse (WEP) sediment, 200 West Area Plio Pleistocene (WPP) sediment, 200 West Area Ringold (WR) sediment, 200 East Area Hanford Sand (EHS) sediment, 200 East Area Ringold (ER) sediment, and 200 East Area Hanford Gravel (LEHG or EHG) sediment. East Hanford Gravel sediment type also appears in the table as LEHG, but the same soil moisture characteristics are applied to both. At most, four different sediment types occurred above the basalt at any location. In the vadose zone model, the basalt rock type was regarded as impermeable and was used to define the default bottom of the vadose zone profile. If the water table fell below the top of the basalt, as in the case for LLBGs located in the northern part of the 200 East Area, the vadose zone was still assumed to be limited to the basalt surface.

Two of the composite well logs developed for the composite analysis were selected for use in this assessment based on their proximity to the LLBGs. The specific well logs used to approximate the vadose zone stratigraphy at the LLBGs, which are noted in the first two rows of the table, are 218-E-12B in the 200 East Area and 218-W-5 in the 200 West Area and the ERDF.

G.1.4.2 Hydraulic Properties

Modeling water flow and radionuclide transport through the vadose zone required a description of the relationship among moisture content, pressure head, and unsaturated hydraulic conductivity. These relationships, called soil moisture characteristics, are highly nonlinear. In this analysis, non-hysteretic relationships were assumed for Hanford Site soils because few measurements to characterize hysteresis have been made for such soils, and it is believed to be of secondary importance. The hydraulic properties

Table G.4. Geologic Well Logs for the Vadose Zone Model

Composite Well Log	Surface Elevation (m)	Northing (m)[a]	Easting (m)[b]	Sediment 1[c]	Thickness (m)	Sediment 2	Thickness (m)	Sediment 3	Thickness (m)	Sediment 4[d]	Thickness (m)
218-W-5[e]	224.9	137024	565658	WHS	19	WEP	4	WPP	7	WR	85
218-E-12B[f]	191.9	137238	574643	EHG	10	EHS	6	LEHG	54	ER	0.01
218-E-10	190.7	137468	572924	EHG	10	EHS	6	LEHG	59	ER	0.01
299-E13-20	226.4	134313	573610	EHG	10	EHS	6	LEHG	80	ER	60
299-E19-1	224.1	135086	572820	EHG	10	EHS	6	LEHG	91	ER	51
299-E24-7	218.2	135561	574407	EHG	10	EHS	6	LEHG	60	ER	56
299-E25-2	205.9	136062	575514	EHG	10	EHS	6	LEHG	60	ER	36
299-E26-8	188.8	136687	575522	EHG	10	EHS	6	LEHG	44	ER	14
299-E28-16	214.3	136562	573135	EHG	10	EHS	6	LEHG	71	ER	12
299-E28-22	213.5	136321	574041	EHG	10	EHS	6	LEHG	83	ER	17
299-W6-1	214.1	137510	567214	WHS	14	WPP	4	WR	121		
299-W11-2	217.8	136671	567407	WHS	34	WEP	4	WPP	7	WR	110
299-W14-7	206.6	135655	567034	WHS	38	WPP	2	WR	118		
299-W14-8A	221.0	135688	568013	WHS	47	WEP	5	WPP	5	WR	106
299-W15-15	212.8	135752	566089	WHS	42	WEP	3	WPP	8	WR	100
299-W18-21	203.8	134979	566098	WHS	36	WEP	5	WPP	3	WR	100
299-W21-1	213.1	134397	568141	WHS	53	WEP	8	WPP	8	WR	100
299-W22-24	211.0	134411	567648	WHS	42	WEP	13	WPP	12	WR	104

(a) Refers to north coordinate in Washington State Plane NAD83 coordinate system.
(b) Refers to east coordinate in Washington State Plane NAD83 coordinate system.
(c) Refers to the upper sediment layer.
(d) Refers to the lowest sediment layer simulated.
(e) Composite well log used in analysis of the 200 West Area LLBGs.
(f) Composite well log used in analysis of the 200 East Area LLBGs.
EHS = 200 East Area Hanford Gravel Sediment.
LEHG = Lower 200 East Area Hanford Gravel Sediment.
ER = 200 East Area Ringold Sediment.
WHS = 200 West Area Hanford Sand Sediment.
WPP = 200 West Area Plio-Pleistocene Sediment.
WEP = 200 West Area Lower Palouse Sediment.
WR = 200 West Area Ringold Sediment.

Table G.5. Sediment Types and Unsaturated Flow Model Parameters Used in the Composite Analysis[(a)]

Sediment Name (Code)	van Genuchten alpha (-)	van Genuchten n (1/cm)	Residual Water Content (cm^3/cm^3)	Saturated Water Content (cm^3/cm^3)	Saturated Hydraulic Conductivity (cm/s)	Bulk Density (g/cm^3)	Gravel %[(b)]
200 East Area Hanford Gravel (EHG)	8.11E-03	1.58	0.0146	0.119	1.76E-03	1.97	41.70
Lower 200 East Area Hanford Gravel (LEHG)	8.11E-03	1.58	0.0146	0.119	1.76E-03	1.97	41.70
200 East Area Hanford Sand (EHS)	1.30E-01	2.10	0.0257	0.337	1.19E-02	1.78	17.30
200 East Area Ringold (ER)	8.19E-03	1.53	0.0262	0.124	3.97E-04	2.04	43.30
200 West Area Hanford Sand (WHS)	1.44E-02	2.20	0.0519	0.382	3.98E-04	1.64	3.60
200 West Area Early Palouse (WEP)	6.27E-03	2.53	0.0300	0.379	9.69E-05	1.68	2.00
200 West Area Plio-Pleistocene (WPP)	1.55E-02	1.78	0.0616	0.337	5.79E-02	1.65	8.40
200 West Area Ringold (WR)	3.14E-02	1.65	0.0236	0.226	5.76E-02	2.04	43.30

(a) Data are from Khaleel and Freeman (1995). A normal distribution was assumed for the parameters "van Genuchten n," "Residual Water Contents," and "Saturated Water Content," and the mean was calculated accordingly. A log-normal distribution was assumed for the parameters "van Genuchten alpha" and "Saturated Hydraulic Conductivity," and the mean was calculated accordingly. If the sample size was less than 10, the parameters "van Genuchten alpha" and "Saturated Hydraulic Conductivity" were determined using the geometric mean.

(b) Only fine particles were assumed to contribute to sorption of contaminants of concern. The impact of larger particles was corrected using gravel percent.

of Hanford Site soils are highly variable, both between the Hanford and Ringold formations and within each of the formations (Khaleel and Freeman 1995). For purposes of this analysis, the values of each of the parameters provided in the table were the values used.

In this analysis, different sediment types were used to define the one-dimensional columns beneath the LLBGs. The hydraulic properties of the sediment types were assumed to be uniform with each sediment layer. Preferential flow paths in the form of wells and clastic dikes were not considered in this analysis because use of one-dimensional models cannot represent their local influence in a three-dimensional environment. The potential influence of preferential flow paths, especially clastic dikes, has been addressed in the performance assessments for the solid waste burial grounds (Wood et al. 1995; Wood et al. 1996) and, more recently, by Ward et al. (1997) for post-1988 LLW. Wood et al. (1995) and Wood et al. (1996) concluded that clastic dikes were insufficiently large and insufficiently continuous to provide a true preferential pathway.

The model of soil hydraulic properties based on the van Genuchten (1980) and Mualem (1976) analytical expressions was used as the basis for the relationships among moisture content, pressure head, and unsaturated hydraulic conductivity. This model has been applied in previous vadose zone studies at the Hanford Site. Parameters for the van Genuchten and Mualem models have been determined by fitting

experimental data for Hanford Site sediments to the classic analytic expressions of these models. These results are described in several Hanford Site documents, but the parameters used in this analysis were compiled by Khaleel and Freeman (1995).

For this analysis, unsaturated flow parameters were established for each of the vadose zone sediment types previously defined. Sediment types and the associated unsaturated flow modeling parameters used in this analysis are those shown in Table G.5. It should be noted that laboratory-measured moisture retention and saturated conductivity data in Table G.5 have been corrected for the gravel fraction (greater than 2 mm) present in the bulk sample.

G.1.4.3 Recharge Rates

This assessment primarily focuses on the long-term transport of contaminants from the LLBGs through the underlying vadose zone to the unconfined aquifer after the end of the operational period in 2046. For wastes disposed of after 1995, which are assumed to have sufficient containment to delay waste release and transport through the vadose zone until after the site closure, the assumption is reasonable. For these waste releases, initial conditions were based on expected conditions after the operational period and assumed a steady-state natural recharge condition with no contaminants in the vadose zone. The assumed long-term recharge that would govern the migration of contaminants through the vadose zone to the underlying water table would be controlled by the expected regional surface conditions surrounding the LLBGs. For conditions dominated by natural vegetation, this is conservatively estimated to be in the order of 0.5 cm/year, as currently estimated, for vegetative surface conditions (Fayer and Walters 1995; Fayer et al. 1999). The net recharge or infiltration rate would vary, representing a range of surface cover conditions from undisturbed surfaces with natural vegetation, to disturbed surfaces maintained free of vegetation, to engineered surface barriers designed for long-term service.

Because waste containment as described above was not systemically used prior to 1995, release of contaminants contained in solid LLW disposed of in LLBGs prior to 1995 were estimated by evaluating the effect of higher infiltration rates through the waste and vadose zone during operations. Results of analyses of earlier disposal facilities used release and vadose zone infiltration rates of 5 cm/yr, a rate reflective of managed bare surface soil conditions over the older disposal areas during the operations phase. This assumption for mobile contaminants (such as technetium-99 and iodine-129) disposed of before 1995 resulted in arrival of these contaminants several hundred years before mobile contaminants disposed of after 1995.

G.1.4.4 Distribution Coefficients

In this analysis, the linear sorption isotherm model was used in transport calculations. This model was selected because it was the only approach for which model parameters (distribution coefficients) were available for the LLBG contaminants. The distribution coefficients (K_d) used for the vadose zone analysis are summarized in Table G.1 (see Section G.1.3.1).

G.1.4.5 Vadose Zone Model Implementation

The vadose zone flow and transport model was implemented with the STOMP code (White and Oostrom 1996; White and Oostrom 1997; Nichols et al. 1997). Implementation of the vadose zone model with a unit release resulted in estimates of the annual contaminant flux to the water table that were used in the convolution integral method for linear superposition described previously.

The STOMP code was developed under the Volatile Organic Compounds (VOCs) Arid Demonstration Project through the DOE Office of Technology Development (White and Oostrom 1997). STOMP is based on the numerical solution of the three-dimensional Richards' equation for fluid flow (Richards 1931) and the advection-dispersion equation for contaminant transport. Although STOMP is capable of three-dimensional simulations, it is also designed to be efficient in performing one- and two-dimensional simulations. The code is based on an integral-volume, finite-difference method and is designed to simulate a wide variety of multidimensional, nonlinear, nonisothermal, and multiphase situations. STOMP was selected for this analysis because of computational efficiency and flexibility, its prior application to the Hanford Site vadose zone (Ward et al. 1997), and its thorough documentation (Nichols et al. 1997), (White and Oostrom 1997), and (White and Oostrom 1996).

Because of the large number of sites to be modeled in this assessment, the technical approach used for the vadose zone stratigraphy was similar to the approach used in the composite analysis by Kincaid et al. (1998). The stratigraphy used was an approximation that was consistent with the major geologic formations found in the vadose zone beneath the Central Plateau in the areas of question and was based on work documented in Thorne and Chamness (1992), Thorne et al. (1993), and Thorne et al. (1994). A summary of the geologic well logs used in the composite analysis appears in Table G.5. To approximate the vadose zone at the LLBGs in the 200 East and West Areas, two of the composite well logs developed for the composite analysis were selected for use in this assessment based on their proximity to the LLBGs. The specific well logs used to approximate the vadose zone stratigraphy at the LLBGs, which are noted in the first two rows of the table, are 218-E-12B in the 200 East Area and 218-W-5 in the 200 West Area and the ERDF.

Water table elevations for future conditions at the LLBGs were calculated with the groundwater flow model. This information was used in the vadose zone transport calculations to define the bottom of the vadose zone. The elevation of the top of the vadose zone at the LLBGs was calculated from land surface elevations and depth to the bottom of the source, which was tabulated for the LLBG areas.

Results of vadose zone transport of a unit release to the water table for the assumed long-term recharge rate of 0.5 cm/year using assumed soil columns and properties in the 200 East and West Areas is presented in Figure G.4. Average travel times for the releases of unit mass of contaminants within Mobility Class 1, as defined by the arrival of 50 percent of each unit mass, is on the order of 500 to 600 years in the 200 East Area and 800 to 900 years in the 200 West Area.

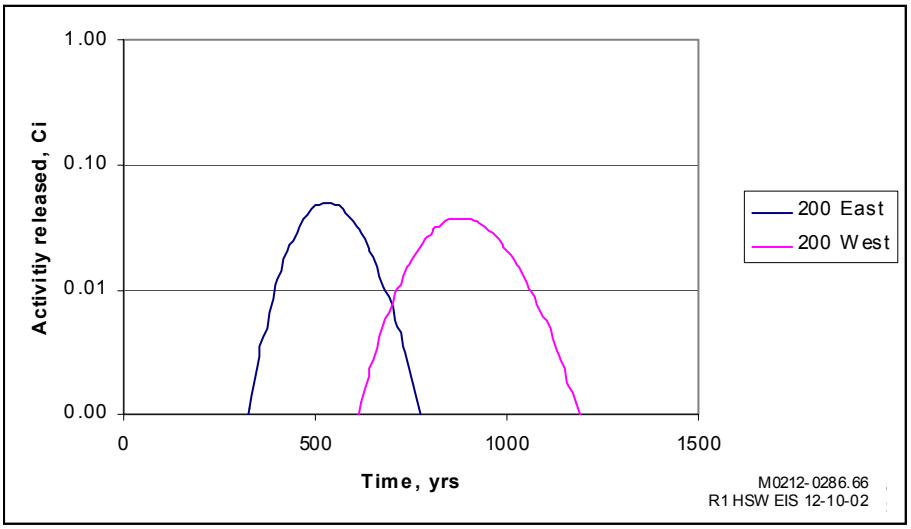

Figure G.4. STOMP Code Results for Releases to the Water Table for a Unit Release from LLBGs for an Assumed Recharge Rate of 0.5 cm/yr

G.1.5 Groundwater Modeling

Contaminant transport through the saturated unconfined aquifer was simulated with the sitewide groundwater flow and transport model, described in Cole et al. (2001a) for the 200 East and the 200 West LLBGs.

A three-dimensional conceptual model was developed for the unconfined aquifer that included stratigraphy, the upper and lower aquifer boundaries, and a table of material units and corresponding flow and transport parameters. The conceptual model was used to guide the setup of the numerical model. A grid spacing of 375 m (1230 ft) was established for the Hanford Site and overlain onto a site map containing physical features and the LLBGs.

G.1.5.1 Conceptual Model

G.1.5.1.1 Hydrogeologic Framework

Hydrogeologic units defined for use in the model were designated by numbers and are briefly described in Table G.6. More detailed descriptions of the sediments were presented in Volume I, Section 4.5 of this HSW EIS, and a graphic comparison of the model units taken from Thorne et al. (1993) against the stratigraphic column defined in Lindsey (1995) is shown in Figure G.5.

Although nine hydrogeologic units were defined, only seven (Units 1, 4, 5, 6, 7, 8, and 9) are found below the water table during post-Hanford conditions (Cole et al. 1997). Odd-numbered Ringold model units (5, 7, and 9) are predominantly coarse-grained sediments. Even-numbered Ringold model units (4, 6, and 8) are predominantly fine-grained sediments with low permeability. The Hanford formation

Table G.6. Major Hydrogeologic Units Used in the Sitewide Three-Dimensional Model

Unit Number	Hydrogeologic Unit	Lithologic Description
1	Hanford Formation	Fluvial gravels and coarse sands
2	Palouse Soils	Fine-grained sediments and eolian silts
3	Plio-Pleistocene Unit	Buried soil horizon containing caliche and basaltic gravels
4	Upper Ringold Formation	Fine-grained fluvial/lacustrine sediments
5	Middle Ringold (Units E and C)	Semi-indurated coarse-grained fluvial sediments
6	Middle Ringold (Lower Ringold Mud)	Fine-grained sediments with some interbedded coarse-grained sediments
7	Middle Ringold (Units B and D)	Coarse-grained sediments
8	Lower Mud Sequence (Lower Ringold and part of Basal Ringold Muds)	Lower blue or green clay or mud sequence
9	Basal Ringold (Unit A)	Fluvial sand and gravel
10	Columbia River Basalt	Basalt

combined with the pre-Missoula gravel deposits were designated as Model Unit 1. Model Units 2 and 3 correspond to the early Palouse soil and Plio-Pleistocene deposits, respectively. These units lie above the current water table. The predominantly mud facies of the upper Ringold unit identified by Lindsey (1995) was designated Model Unit 4. However, a difference in the definition of model units was the lower, predominantly sand, portion of the upper Ringold unit described in Lindsey (1995) was grouped with Model Unit 5 that also includes Ringold gravel/sand Units E and C. This action was taken because the predominantly sand portion of the upper Ringold is expected to have hydraulic properties similar to Units E and C. The lower mud unit identified by Lindsey (1995) was designated Model Units 6 and 8. Where they exist, the gravel and sand Units B and D, found within the lower Ringold, were designated Model Unit 7. Gravels of Ringold Unit A were designated Model Unit 9, and the underlying basalt was designated Model Unit 10. However, the basalt was assigned a very low hydraulic conductivity and was essentially impermeable in the model.

The lateral extent and thickness distribution of each hydrogeologic unit were defined based on information from drillers' well logs, geologists' logs, geophysical logs, and an understanding of the geologic environment. These interpreted areal distributions and thicknesses were then integrated into EarthVision™ (Dynamic Graphics, Inc., Alameda, California), a three-dimensional, visualization software package that was used to construct a database of the three-dimensional hydrogeologic framework.

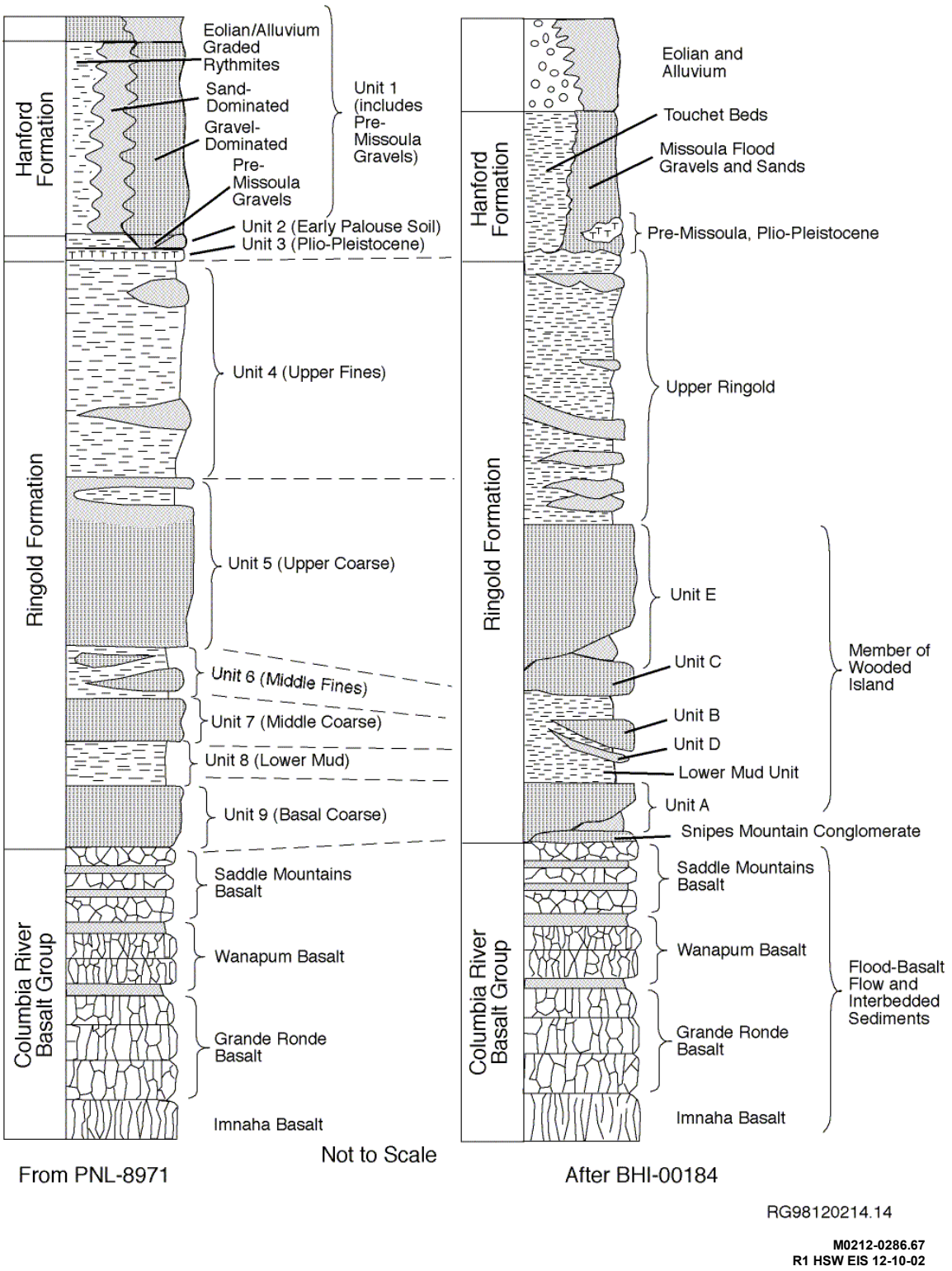

Figure G.5. Comparison of Generalized Hydrogeologic and Geologic Stratigraphy (from Thorne et al. [1993] and after Lindsey [1995])

G.1.5.1.2 Recharge and Flow System Boundary Conditions

The past development of the sitewide model considered both natural and artificial recharge to the aquifer. Natural recharge to the unconfined aquifer system occurs from infiltration of 1) runoff from elevated regions along the western boundary of the Hanford Site; 2) spring discharges originating from the basalt-confined aquifer system, also along the western boundary; and 3) precipitation falling across the site. Some recharge also occurs along the Yakima River in the southern portion of the site. Natural recharge from runoff and irrigation in the Cold Creek and Dry Creek Valleys, upgradient of the site, also provides a source of groundwater inflow. Natural recharge from precipitation on the site is highly variable, both spatially and temporally, and depends on local climate, soil type, and vegetation.

The other source of recharge to the unconfined aquifer has historically come from wastewater disposal. The large volume of artificial recharge from wastewater discharged to disposal facilities on the Hanford Site over the past 60 years has substantially impacted groundwater flow and contaminant transport in the unconfined aquifer system. This volume of artificial recharge decreased significantly in the past 10 years, and the water table has been declining steadily over several years. The unconfined aquifer system eventually will be expected to reach more natural conditions after site closure. Because flow conditions simulated for this assessment focused on conditions that are likely to exist after Hanford Site closure and well into the future, the effect of past and current wastewater discharges on the unconfined aquifer system were not considered in this assessment.

Peripheral boundaries defined for the three-dimensional model are shown in Figure G.6, together with the three-dimensional flow-model grid. The flow system is bounded by the Columbia River on the north and east and by the Yakima River and basalt ridges on the south and west. The Columbia River represents a point of regional discharge for the unconfined aquifer system. The amount of groundwater discharging to the river is a function of local hydraulic gradient between the groundwater elevation adjacent to the river and the river-stage elevation. This hydraulic gradient is highly variable because the river stage is affected by releases from upstream dams.

Because of the regional-scale nature and long-time frame being considered in the current assessment, site-wide flow and transport modeling efforts did not attempt to consider the short-term and local-scale transient effects of the Columbia River system on the unconfined aquifer. However, the long-term effect of the Columbia River as a regional discharge area for the unconfined aquifer system was approximated in the three-dimensional model with a constant-head boundary applied at the uppermost nodes of the model at the approximate locations of the river's left bank and channel midpoint. Nodes representing the thickness of the aquifer below the nodes representing mid-point of the river channel were treated as no-flow boundaries. This boundary condition is used to approximate the location of the groundwater divide that exists beneath the Columbia River where groundwater from the Hanford Site and the other side of the river discharge into the Columbia. The long-term, average river-stage elevations for the Columbia River implemented in the sitewide model were based on results from previous work performed by Walters et al. (1994) for the Columbia River with the CHARIMA river simulation model. The Yakima River was also represented as a specified-head boundary at surface nodes approximating its location. Like the Columbia River, nodes representing the thickness of the aquifer below the Yakima River channel were treated as no-flow boundaries. Short-term fluctuations in the river levels do not influence modeling results.

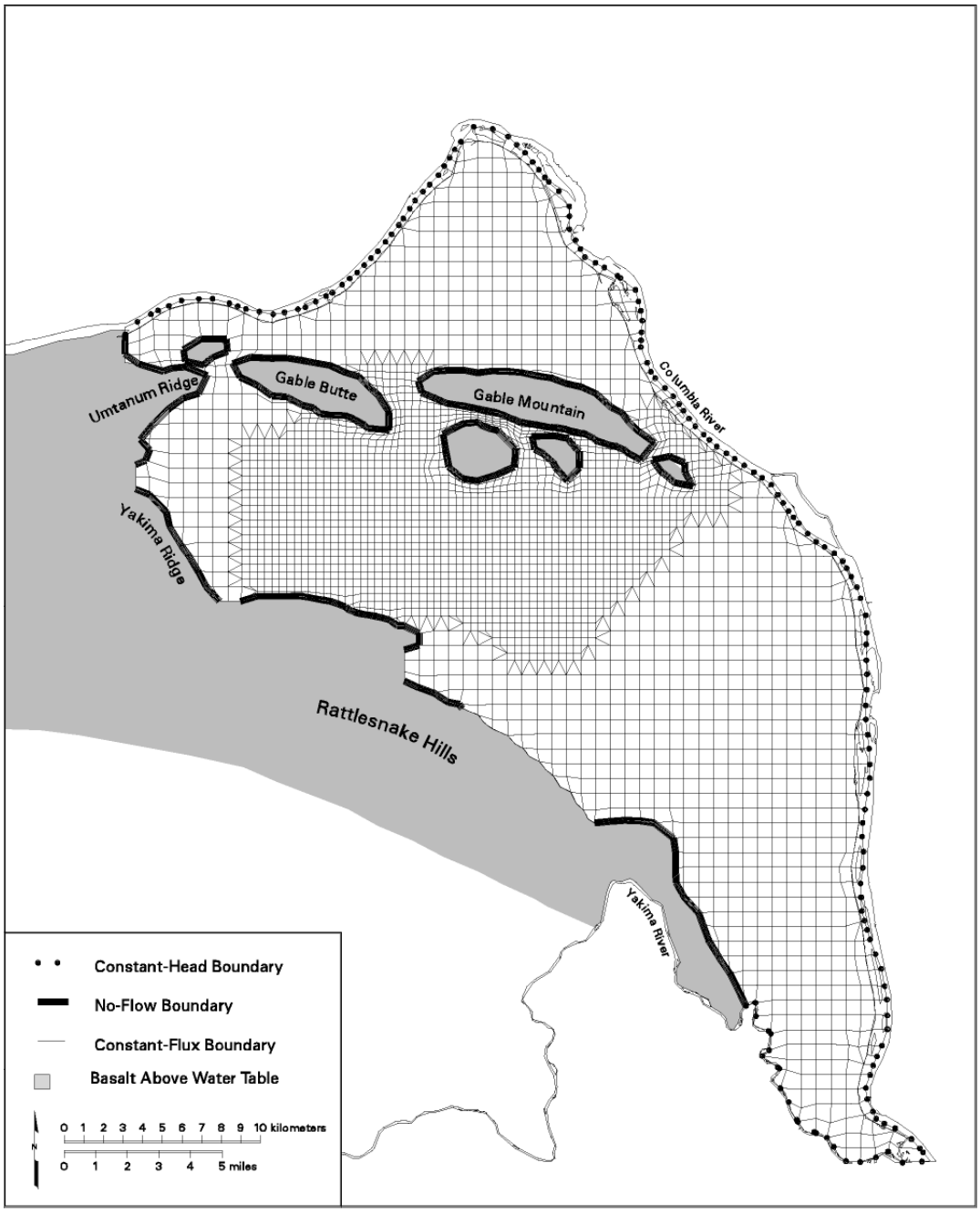

Figure G.6. Peripheral Boundaries Defined for the Three-Dimensional Model (after Cole et al. [1997])

At Cold Creek and Dry Creek Valleys, the unconfined aquifer system extends westward beyond the boundary of the model. To approximate the groundwater flux entering the modeled area from these valleys, both constant-head and constant-flux boundary conditions were defined. A constant-head

boundary condition was specified for Cold Creek Valley for the steady-state model calibration runs. The fluxes resulting from the specified-head boundaries in the calibrated steady-state model were then used in the steady-state flow simulation of flow conditions after Hanford Site closure. The constant-flux boundary was used because it better represents the response of the boundary to a declining water table than does a constant-head boundary. Discharges from Dry Creek Valley in the model area, resulting from infiltration of precipitation and spring discharges, are approximated using the same methods.

The basalt underlying the unconfined aquifer sediments represents a lower boundary to the unconfined aquifer system. The potential for interflow (recharge and discharge) between the basalt-confined aquifer system and the unconfined aquifer system is largely unquantified but is postulated to be small relative to the other flow components estimated for the unconfined aquifer system. Therefore, interflow with underlying basalt units was not included in the current three-dimensional model. The basalt was defined in the model as an essentially impermeable unit underlying the sediments.

G.1.5.1.3 Flow and Transport Properties

To model groundwater flow, the distribution of hydraulic properties, including horizontal and vertical hydraulic conductivity, storativity, and specific yield, was needed for each hydrogeologic unit defined in the model. In addition, to simulate movement of contaminant plumes, transport properties were needed, including contaminant-specific distribution coefficients, bulk density, effective porosity, and longitudinal and transverse dispersivities.

In the original model calibration procedure described in Wurstner et al. (1995), measured values of aquifer transmissivity were used in a two-dimensional model with an inverse model-calibration procedure to determine the transmissivity distribution. Hydraulic head conditions for 1979 were used in the inverse calibration because measured hydraulic heads were relatively stable at that time. Details concerning the updated calibration of the two-dimensional model are provided in Cole et al. (1997). The resulting transmissivity distribution for the unconfined aquifer system is shown in Figure G.7.

Hydraulic conductivities were assigned to the three-dimensional model units so that the total aquifer transmissivity from inverse calibration was preserved at every location. The vertical distribution of hydraulic conductivity at each spatial location was determined, based on the transmissivity value and other information, including facies descriptions and hydraulic property values measured for similar facies. A complete description of the seven-step process used to vertically distribute the transmissivity among the model hydrogeologic units is described in Cole et al. (1997).

The current version of the sitewide model relies on a three-dimensional representation of the aquifer system that was calibrated to Hanford Sitewide groundwater monitoring data collected during Hanford operations from 1943 to the present. The calibration procedure and results for this model are described in

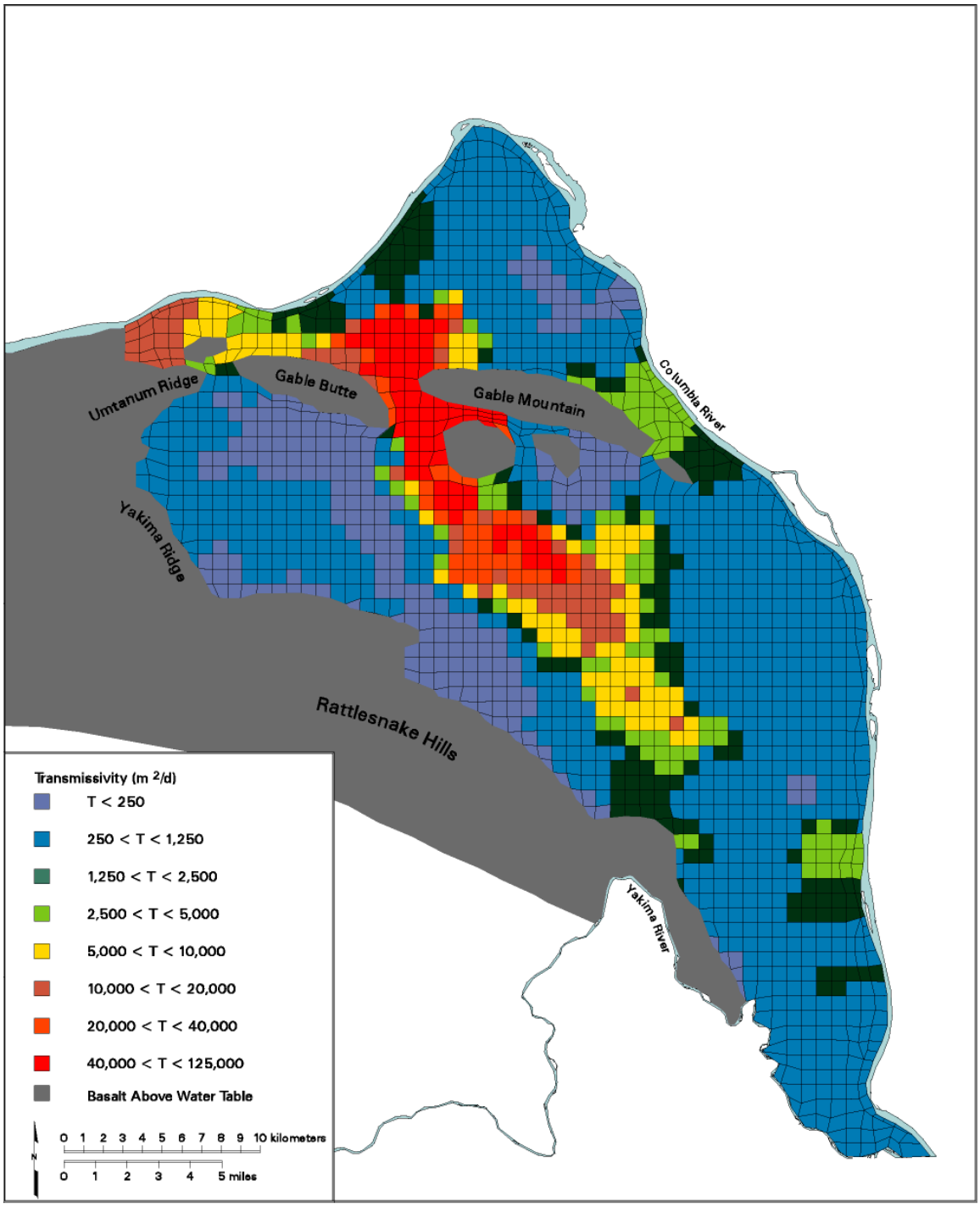

Figure G.7. Transmissivity Distribution for the Unconfined Aquifer System Based on Two-Dimensional Inverse Model Calibration (after Wurstner et al. [1995])

Cole et al. (2001a). This recent work is part of a broader effort to develop and implement a stochastic uncertainty estimation methodology in future assessments and analyses using the sitewide groundwater model (Cole et al. 2001b). Resulting distribution of hydraulic conductivities from this recent calibration effort is provided in Figures G.8 and G.9.

Information on transport properties used in past modeling studies at the Hanford Site is provided in Wurstner et al. (1995). Estimates of model parameters were developed to account for contaminant dispersion and adsorption in all transport simulations. Specific model parameters examined included longitudinal and transverse dispersivity (D_L and D_T) and contaminant retardation factors (R_f). Calculation of effective R_f required estimates of contaminant-specific distribution coefficients, as well as estimates of effective bulk density and porosity of the aquifer materials. The remainder of this section briefly summarizes estimated transport properties.

For this analysis, a longitudinal dispersivity, D_L, of a little less than 100 m (95 m) (310 ft) was selected using this typical approach for estimating longitudinal dispersivity based on the scale of interest. Although transport results produced in this analysis span a range of scales, the key scale of interest is the minimum distance between some of the source areas in the Central Plateau and the location of the buffer zone boundary surrounding this area. For some sources in 200 East Area, the distance of interest is on the order of 1 to 2 km away. Thus, a dispersivity value used in the original analysis was selected to be approximately equal to 10 percent of the minimum travel distance of interest of about 1 km (0.6 mi).

The longitudinal dispersivity was also consistent to be within the range of recommended grid Peclet numbers ($Pe < 4$) for acceptable solutions. The 95-m (310-ft) estimate is about one-quarter of the grid spacing in the finest part of the model grid in the Central Plateau where the smallest grid spacing is about 375 m x 375 m (1230 ft x 1230 ft).

The corresponding transverse dispersivity used in the analysis was selected to be consistent with general available regulatory and technical guidance. EPA guidance (Mills et al. 1985) on the subject suggests a 1 to 3 ratio for D_T to D_L. Freeze and Cherry (1979) report that transverse dispersivities used are normally lower than the longitudinal dispersivity by a factor of 5 to 20 (that is, 0.2 to 0.05). Walton (1985) states that reported ratios of D_T to D_L vary from 1 to 24 but that common values are 0.2 and 0.1. Considering this information, a transverse dispersivity, D_T, used in Composite Analysis simulations was assumed to be about 20 m (65.6 ft), which is approximately 20 percent of the selected longitudinal dispersivity.

The longitudinal dispersivity was also consistent and within the range of recommended grid Peclet numbers ($Pe < 4$) for acceptable solutions. The 95-m (310-ft) estimate is about one-quarter of the grid spacing in the finest part of the model grid in the Central Plateau where the smallest grid spacing is about 375 m x 375 m (1230 ft x 1230 ft).

In addition to the estimated distribution coefficient, calculation of contaminant-specific retardation factors used in the model requires estimates of the effective bulk density and porosity. For purposes of these calculations, a bulk density of 1.9 g/cm^3 was used for all simulations. The effective porosity was

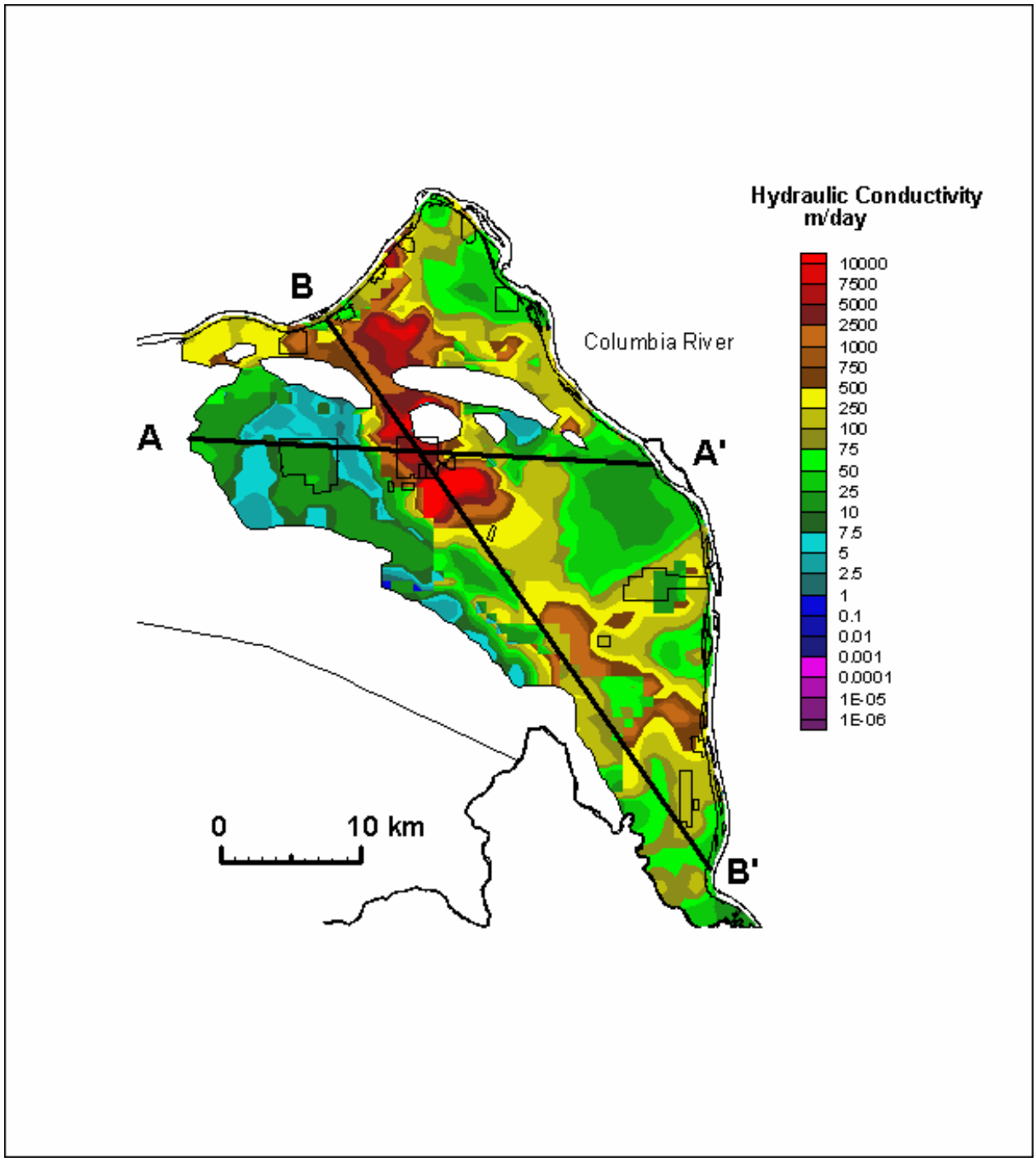

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Figure G.8. Distribution of Estimated Hydraulic Conductivities at Water Table from Best-Fit Inverse Calibration of Sitewide Groundwater Model (after Cole et al. [2001a])

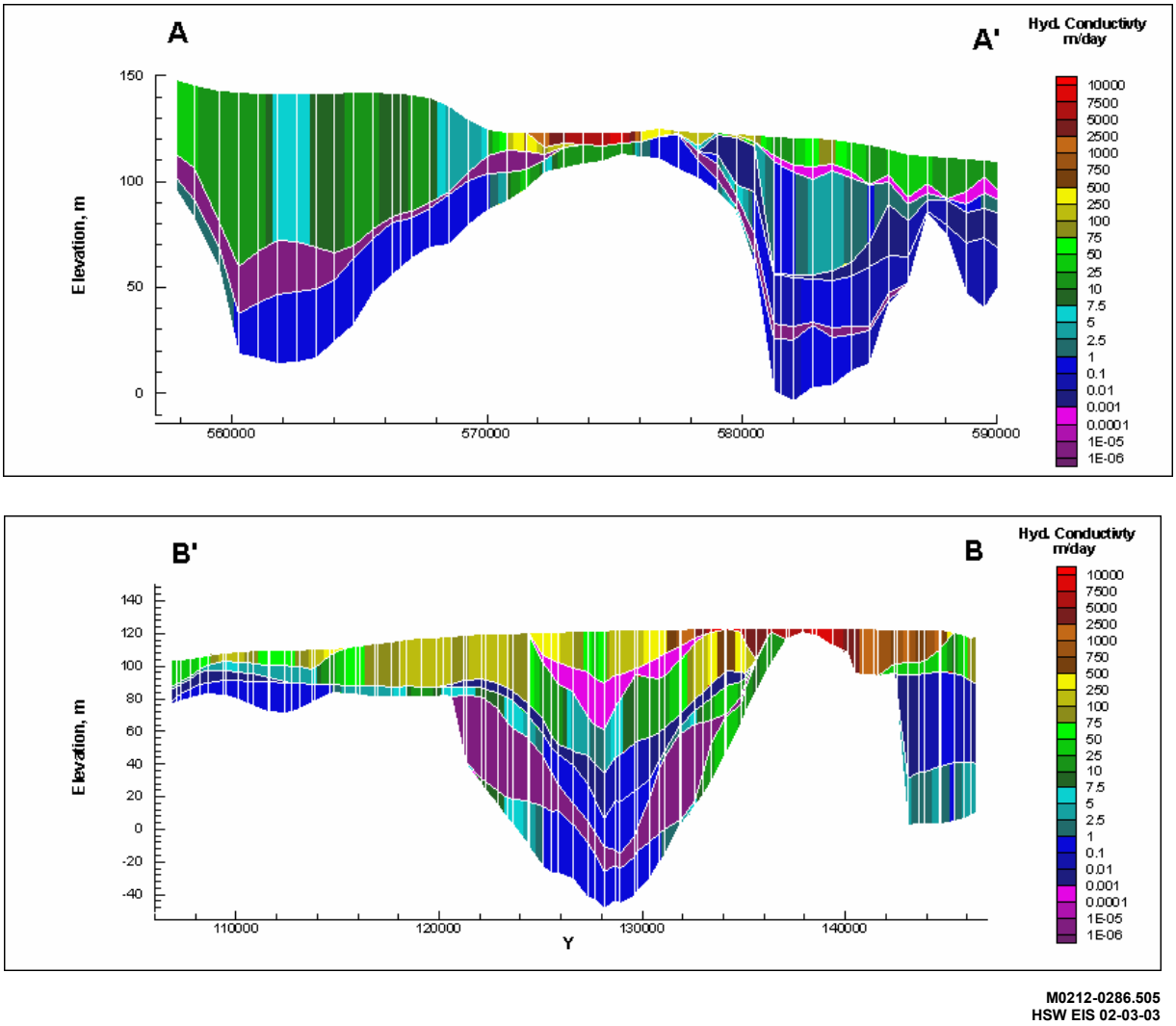

Figure G.9. Distribution of Estimated Hydraulic Conductivities Along Section Lines A-A' and B-B' from Best-Fit Inverse Calibration of Sitewide Groundwater Model (after Cole et al. [2001a])

estimated from specific yields obtained from multiple well aquifer tests. These values range from 0.01 to 0.37. Laboratory measurements of porosity that range from 0.19 to 0.41 were available for samples from a few Hanford Site wells and were also considered. The few tracer tests conducted indicate effective porosities ranging from 0.1 to 0.25. Within the model, a porosity value of 0.1 was used for the Ringold Formation (Model Units 4 through 9) and a porosity value of 0.25 was used for the Hanford formation (Model Unit 1). For the expected lower water table conditions during the post-Hanford period, the Early Palouse and Plio-Pleistocene hydrogeologic units (Model Units 2 and 3) only existed above the projected water table and were not considered in the analysis. Values of distribution coefficient, bulk density, effective porosity, and dispersivity used in this analysis are discussed in more detail in Cole et al. (1997).

G.1.5.2 Simulation of Post-Closure Flow Conditions

Past projections of water table conditions after site closure have estimated the impact of Hanford operations ceasing and the resulting changes in artificial discharges that have been used extensively as a part of site waste management practices. Simulations of transient-flow conditions from 1944 through the year 3050 were conducted by Bryce et al. (2002). The three-dimensional model shows an overall decline in the hydraulic head and hydraulic gradient across the entire water table within the modeled region. Results of these simulations suggest that the water table would reach steady state between 100 to 350 years in different areas over the Hanford Site. These results were generally consistent with findings for the similar conditions in earlier modeling by Cole et al. (1997) and Kincaid et al. (1998).

Given the expected long delay of contaminants reaching the water from the LLBGs, the hydrologic framework of all groundwater transport calculations was based on a postulated post-Hanford, steady-state water table as estimated with the three-dimensional model. These conditions would only reflect estimated boundary condition fluxes (for example, natural recharge and lateral boundary fluxes) and not the effect of past and current wastewater discharges on the unconfined aquifer system.

Flow modeling results also suggest that as water levels drop in the vicinity of central areas in the model where the basalt crops out above the water table, the saturated thickness of the unconfined aquifer will decrease and the aquifer may actually dry out in certain areas. This thinning/drying of the aquifer is predicted to occur in the area just north of the 200 East Area between Gable Butte and the outcrop south of Gable Mountain, and there is the potential of this northern area of the unconfined aquifer becoming hydrologically separated from the area south of Gable Mountain and Gable Butte. Because of the uncertainty in the potential natural recharge and boundary fluxes from upgradient areas, the potential for movement of contaminants either through the gap or to the east toward the Columbia River is also uncertain. To address this uncertainty, two predicted water tables for these post-Hanford steady-state conditions, as illustrated in Figures G.10 and G.11, were considered.

The first scenario, shown in Figure G.10, estimates flow conditions where basalt sub-crops estimated to be above the water table north of the Central Plateau are consistent with those used in the most recent assessments by Bryce et al. (2002). Under this scenario, the overall flow attributes of the water table surface lead to groundwater flow and transport through the gap between Gable Mountain and Gable Butte from most areas in the 200 East and 200 West Areas. This scenario was the flow condition used in all groundwater flow and transport calculations presented in the following sections.

In the second scenario, shown in Figure G.11, flow conditions are reflective of assumed basalt sub-crops just north of the 200 East Area that are more widespread and effectively cut off the flow and transport from both the 200 East and 200 West Areas to the north through the gap between Gable Mountain and Gable Butte. The overall flow attributes of this water table surface leads to a predominant easterly flow direction from nearly all areas within the 200 East and 200 West Areas. The effect of this scenario on calculated results, while not considered in all results presented in Section G.2, is briefly discussed in the following section and in a discussion of results for Alternative Group A in Section G.2.1.

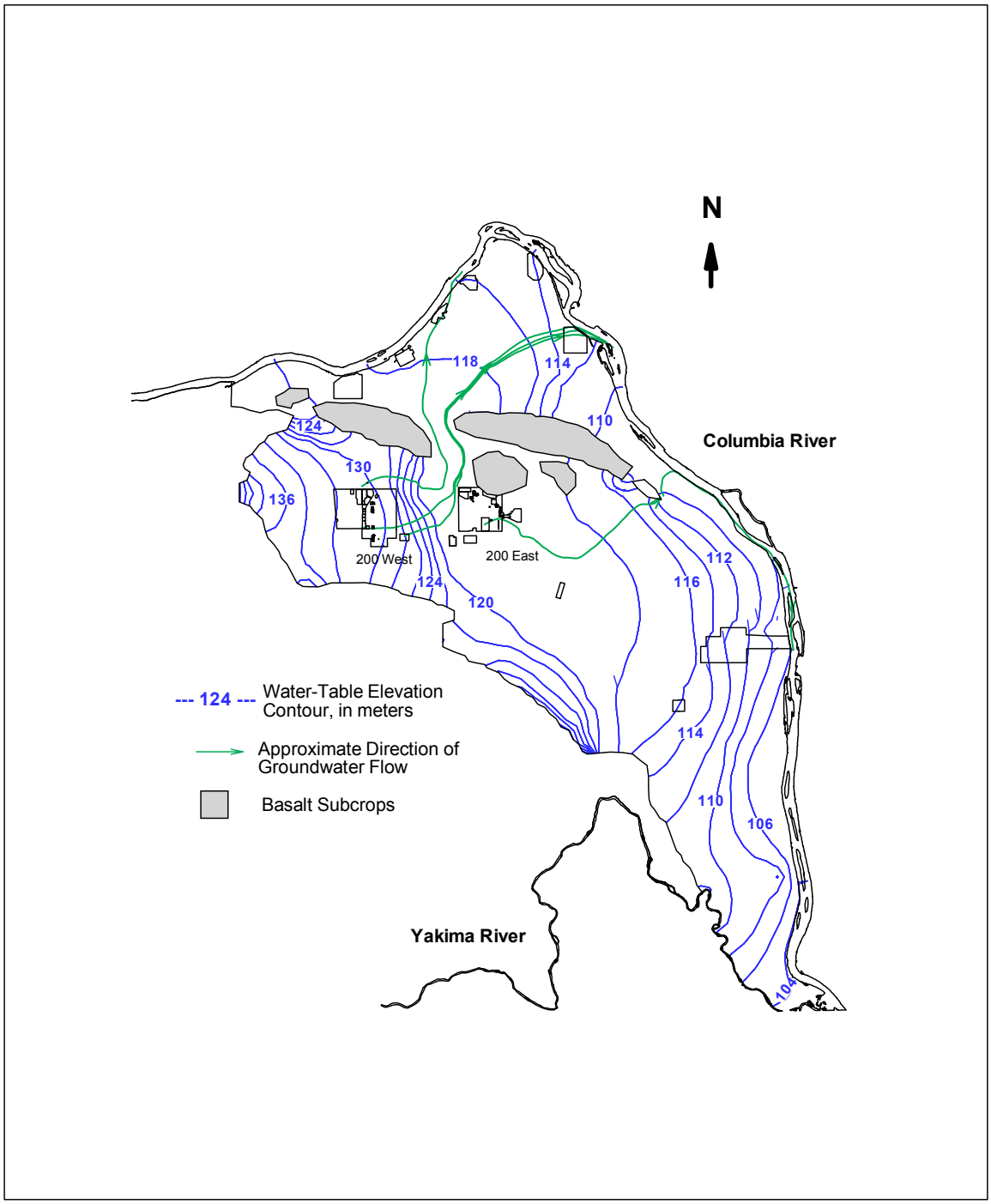

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Figure G.10. Predicted Post-Hanford Water Table Conditions (Predominant Northerly Flow)

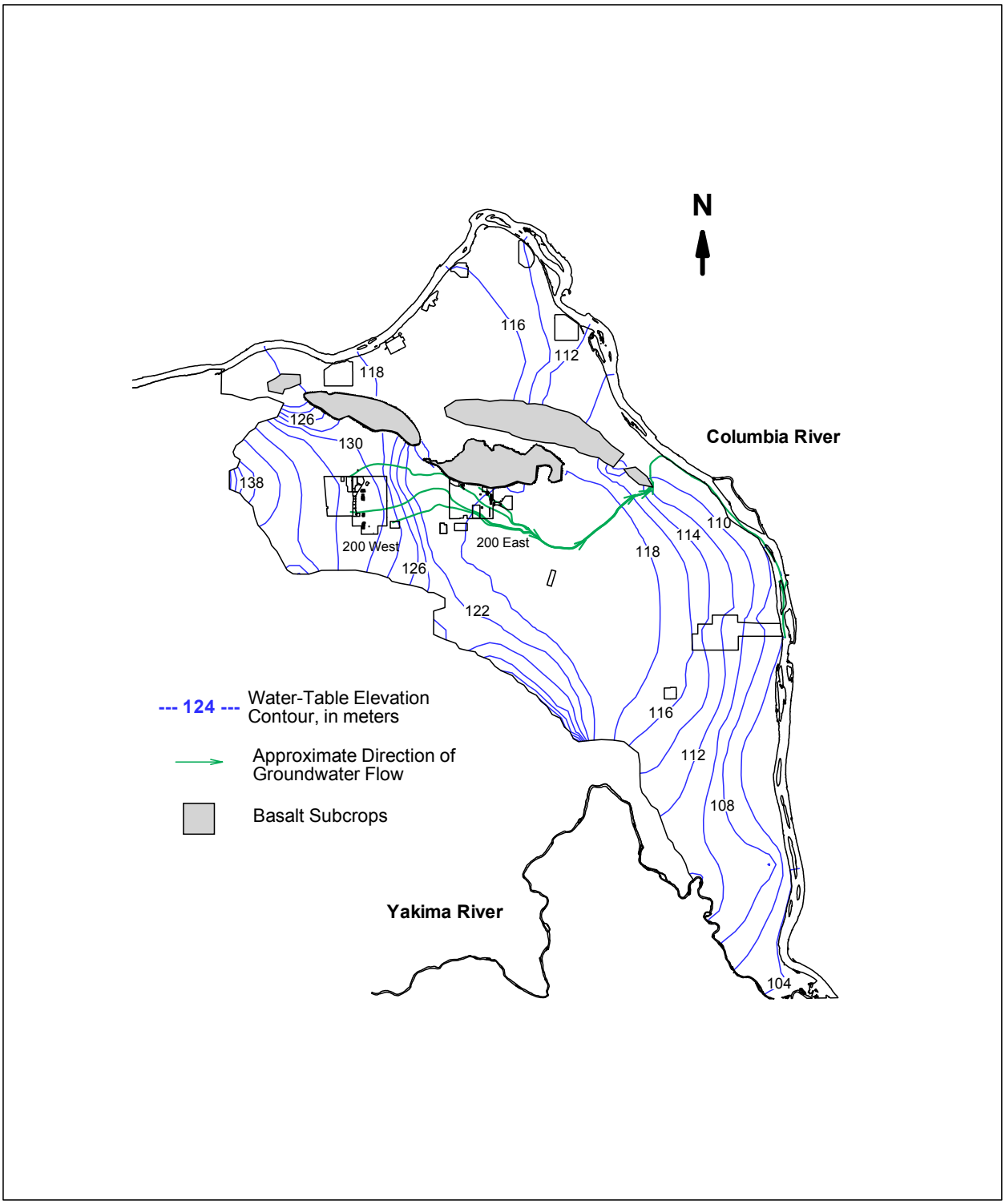

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Figure G.11. Predicted Post-Hanford Water Table Conditions (Predominant Easterly Flow)

G.1.5.3 Simulation of Unit Releases

To allow groundwater transport calculations to be used in the convolution approach for linear superposition (see Section G.1.2), a unit release was simulated with the three-dimensional model and the estimated post-Hanford, steady-state water table condition. These simulation results are used to relate the effect of known release (1 curie over a 10-year period) to predicted concentrations at various points in the aquifer system. Example results of simulated groundwater concentrations in response to a unit release of a long-lived, mobile (non-sorbing) contaminant over a period of 10 years from MLLW disposal sites in the 200 West and 200 East Areas are illustrated in Figures G.12 and G.13, respectively. These simulations were made using the groundwater conceptual model with a predominant northerly flow pattern out of the Central Plateau.

The same calculations were also made using the alternative groundwater conceptual model with easterly flow from the 200 East Area. Results of this model at the same MLLW disposal locations in the 200 West and East Areas are illustrated in Figures G.14 and G.15, respectively.

Results of these unit releases were evaluated to identify the maximum concentrations over time for use in the convolution approach along the LOAs downgradient of the 200 East and West Areas and ERDF HSW disposal areas (see Figure G.6) as appropriate for each alternative group. Because the location of different waste categories within each of the aggregate HSW disposal areas varies as specified for each alternative group, the locations of maximum concentration along the LOAs may not necessarily correspond to the same location for each waste category specified within and across alternative groups. This is particularly true for breakthrough curves developed for LOAs near the Columbia River where the location of maximum concentration varies in time as the simulated plumes migrate north to the Columbia River. The specific calculations presented here were used to evaluate groundwater transport of contaminants in Group 1 (technetium-99 and iodine-129). Similar calculations were made to evaluate groundwater transport of the same Group 1 contaminants and for contaminants in Group 2 (carbon-14 and uranium isotopes) for other waste category locations in the overall convolution approach.

A comparison of unit release breakthrough curves for Group 1 constituents at the 200 East and West Area, ERDF, and Columbia River LOAs for the two alternative groundwater conceptual models are presented in a series of plots in Figures G.16 and G.17 for all waste categories to illustrate differences in results for the two-groundwater conceptual models. Under the first alternative model, potential impacts from LLW disposed of in the 200 East Area LLBGs are evaluated at the 200 East Area NW LOA. Potential impacts from LLW disposed of near the PUREX Plant are evaluated at the 200 East Area SE LOA. Under the second alternative, where groundwater flow is toward the east from the 200 Areas, potential impacts from LLW disposed of in the 200 East Area LLBGs or near the PUREX Plant are evaluated at the 200 East Area SE LOA.

Results of these unit releases were evaluated to identify the maximum concentrations over time for use in the convolution approach along the LOAs downgradient of the 200 East and West Areas and the ERDF HSW disposal areas (see Figure G.1) as appropriate for each alternative group. Because the location of different waste categories within each of the aggregate HSW disposal areas varies as specified for each alternative group, the locations of maximum concentration along the LOAs may not necessarily correspond to the same location for each waste category specified within and across alternative groups.

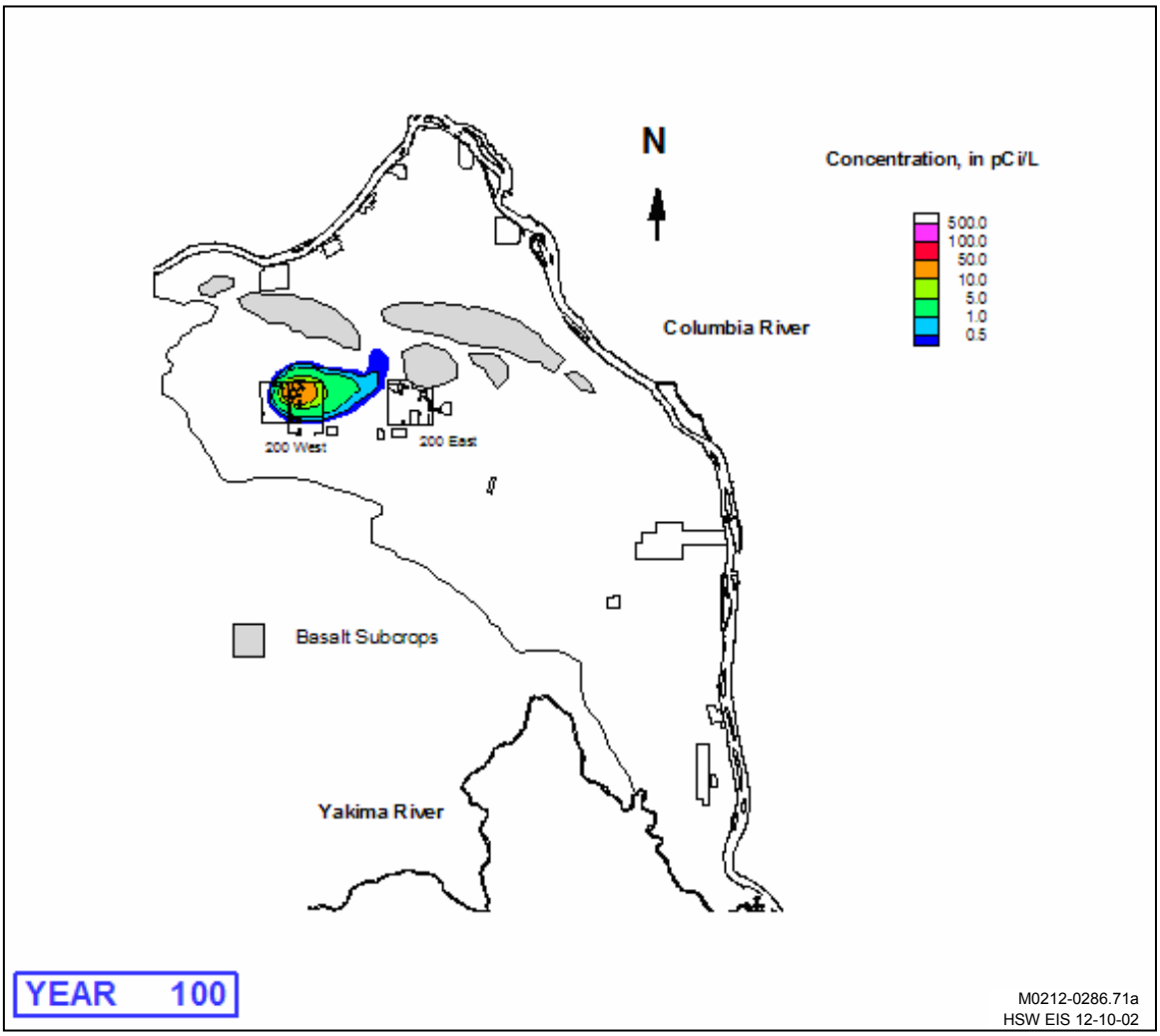
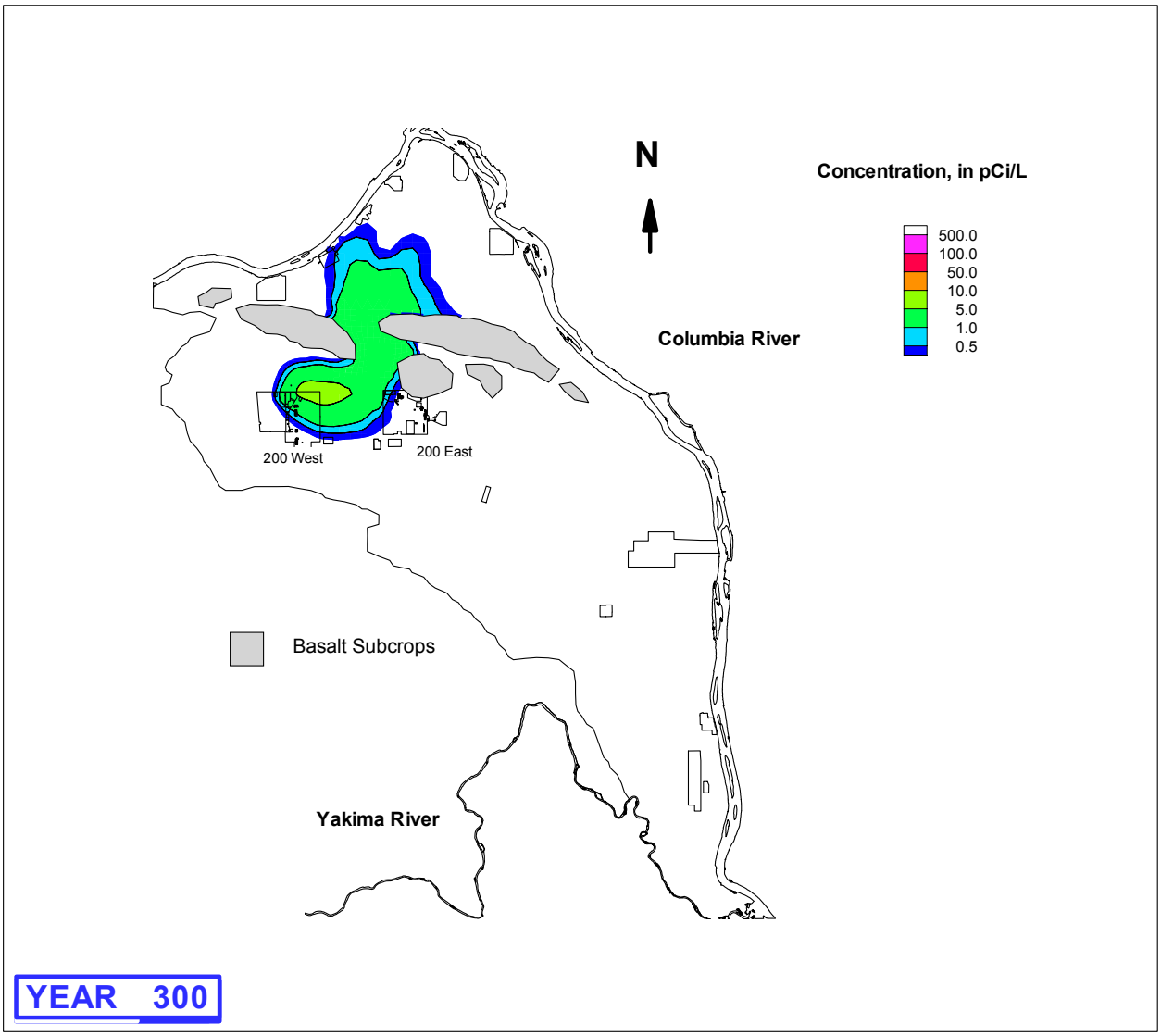


Figure G.12a. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 West Area at 100 Years After Release Using a Groundwater Model with a Predominant Northerly Flow from the Central Plateau

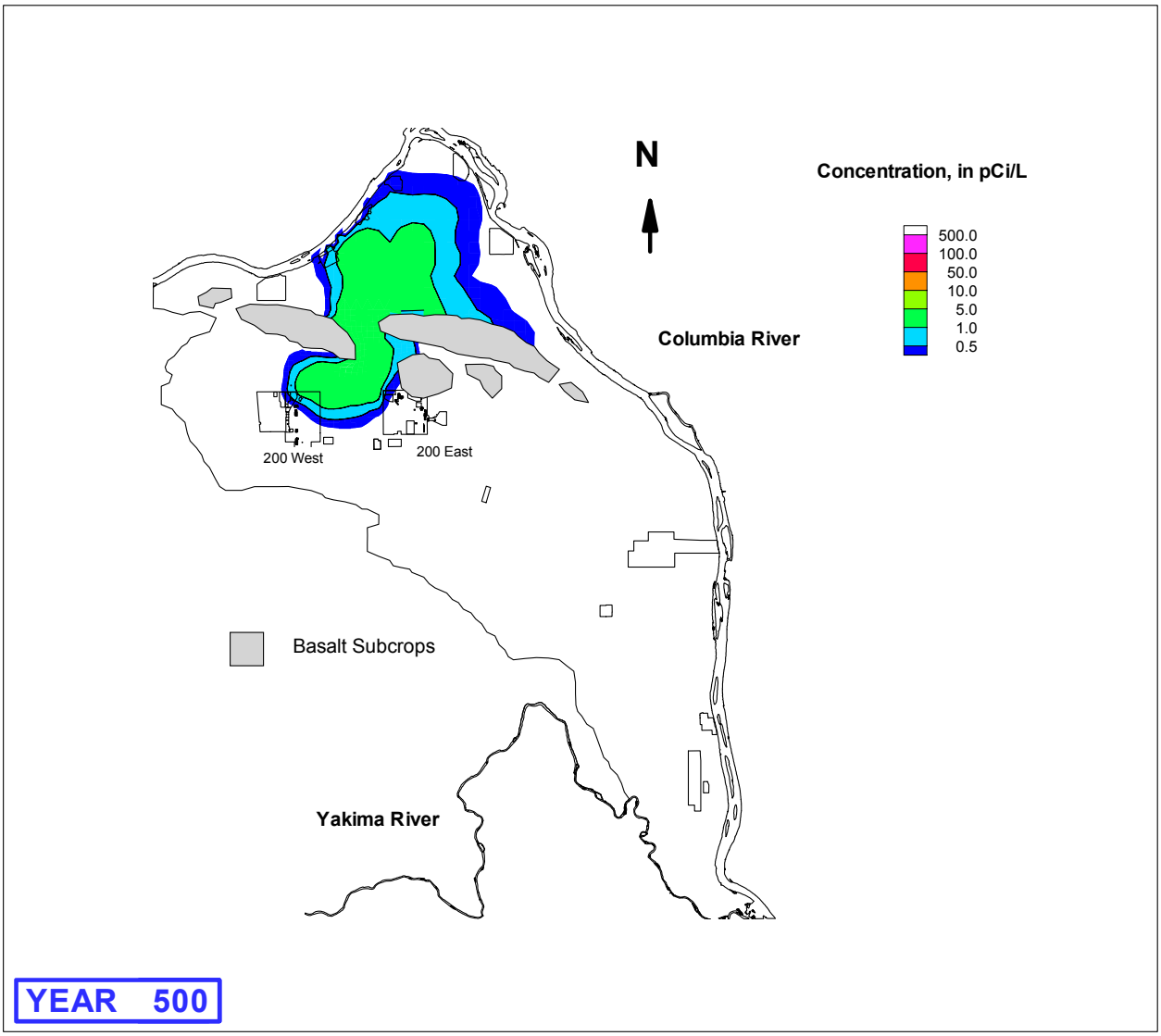
(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.



M0212-0286.71b
HSW EIS 12-10-02

Figure G.12b. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 West Area at 300 Years After Release Using a Groundwater Model with a Predominant Northerly Flow from the Central Plateau

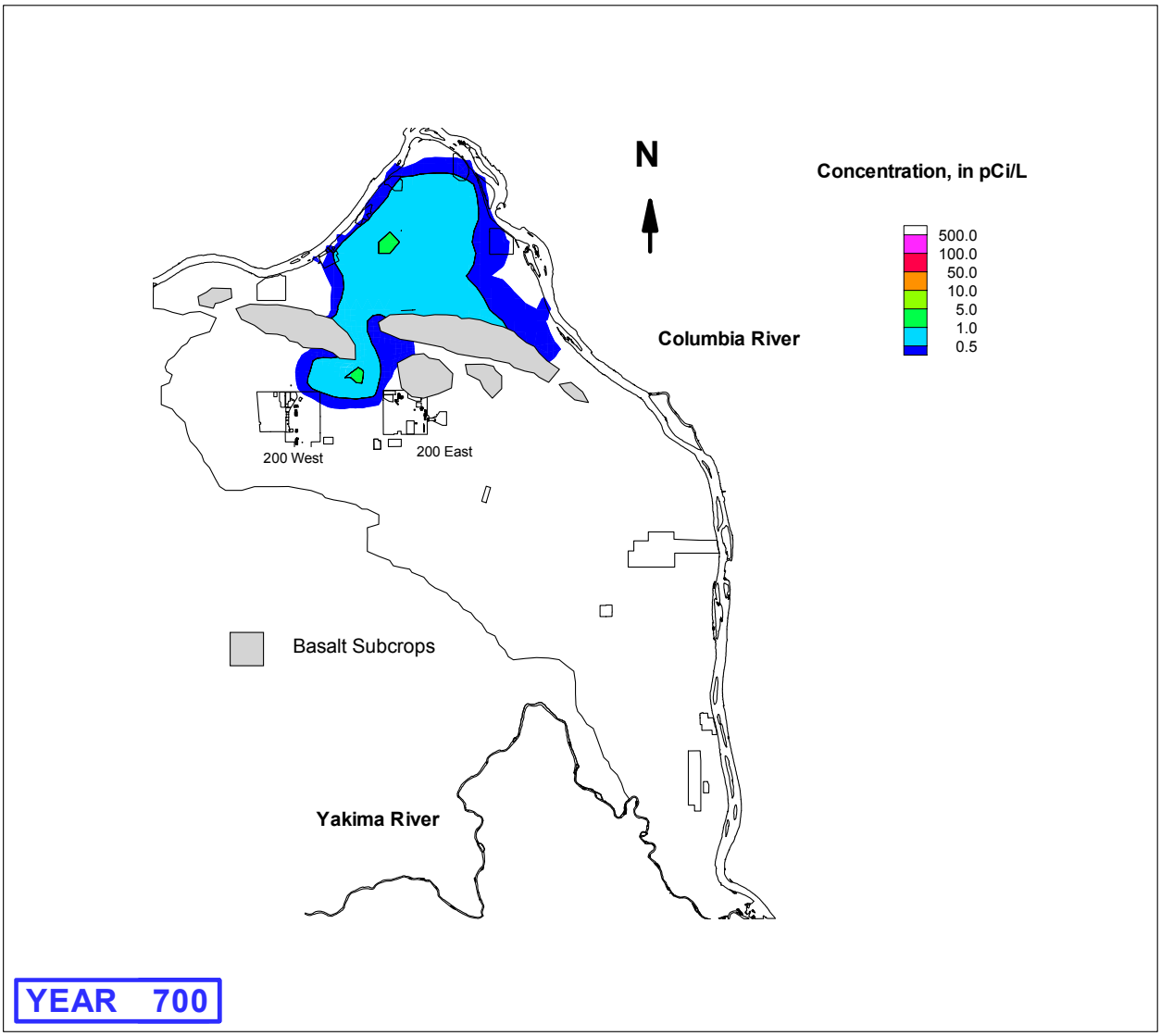
(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system of an unretarded long-lived contaminant. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.



M0212-0286.72
HSW EIS 12-10-02

Figure G.12c. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 West Area at 500 Years After Release Using a Groundwater Model with a Predominant Northerly Flow from the Central Plateau

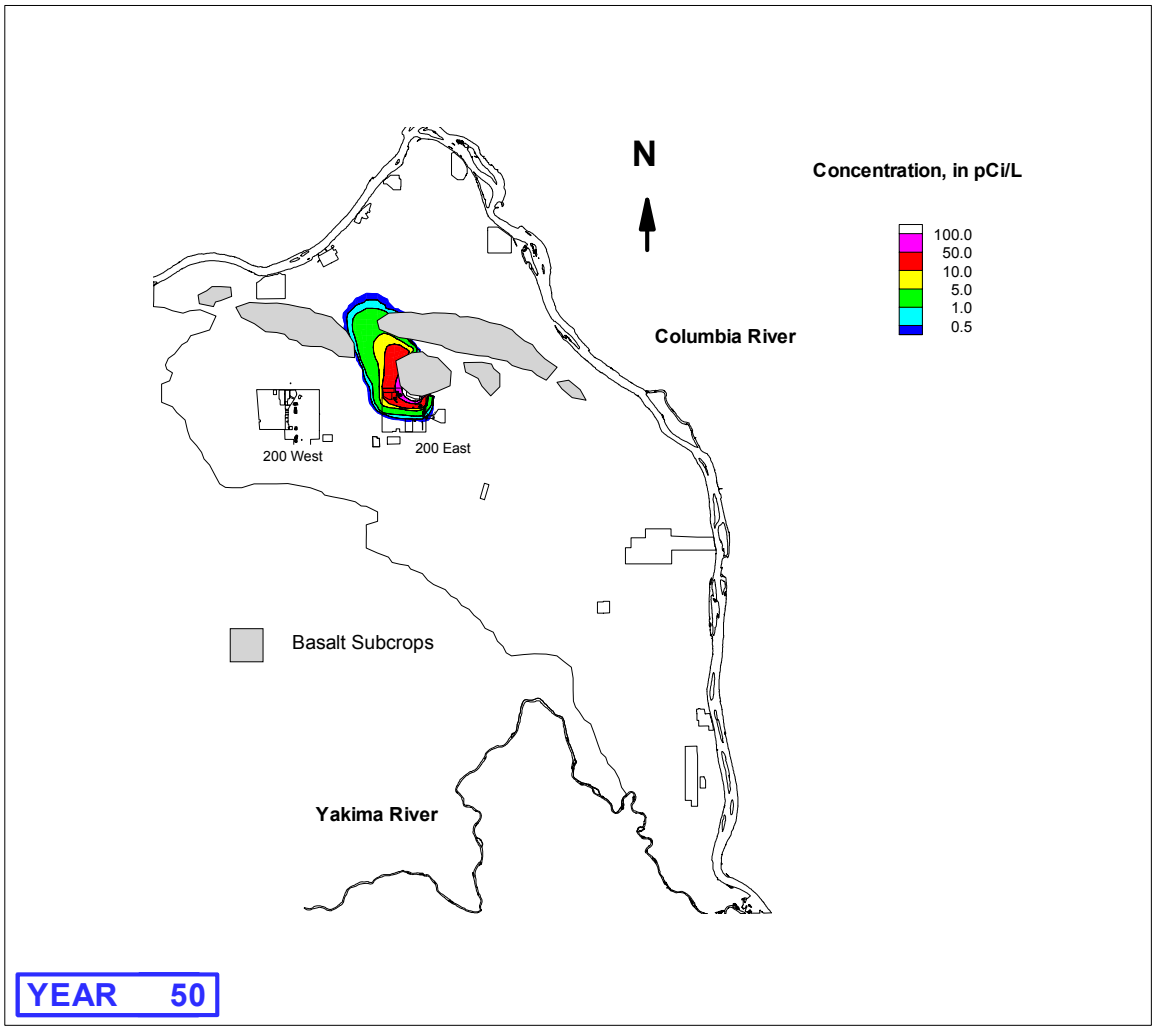
(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.



M0212-0286.73
HSW EIS 12-10-02

Figure G.12d. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 West Area at 700 Years After Release Using a Groundwater Model with a Predominant Northerly Flow from the Central Plateau

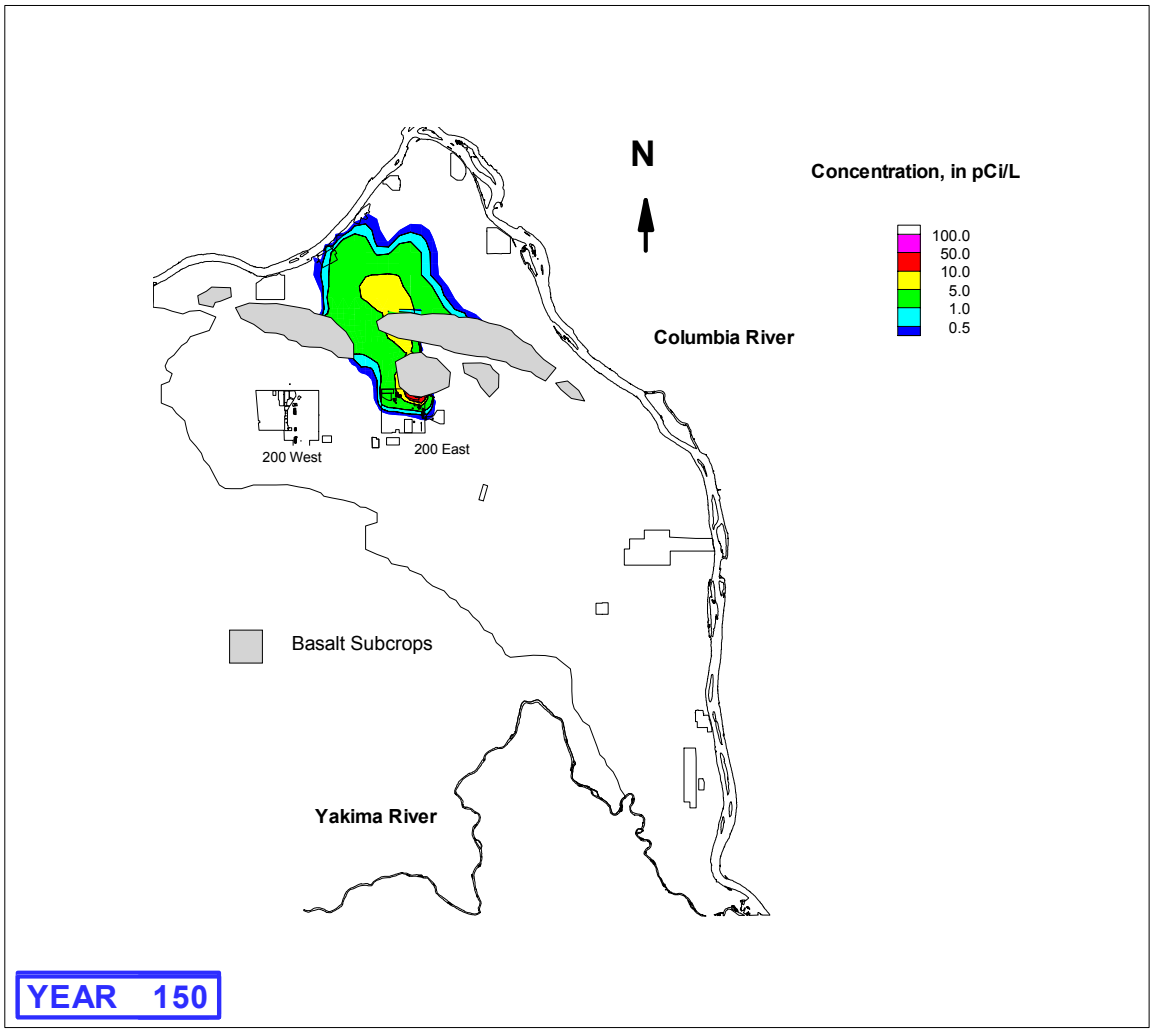
(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.



M0212-0286.74
HSW EIS 12-10-02

Figure G.13a. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 East Area at 50 Years After Release Using a Groundwater Model with a Predominant Northerly Flow from the Central Plateau

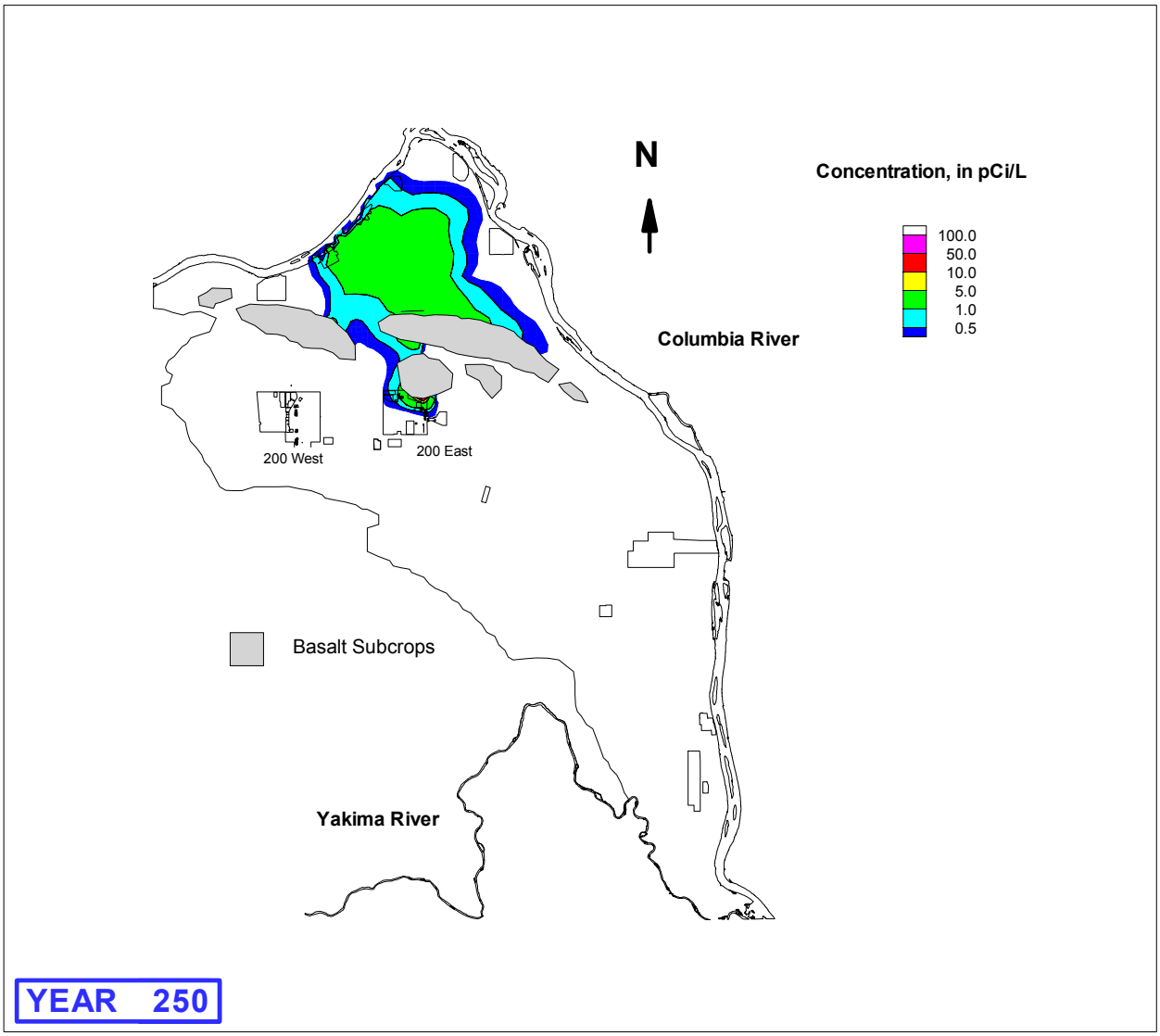
(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.



M0212-0286.75a
HSW EIS 12-10-02

Figure G.13b. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 East Area at 150 Years After Release Using a Groundwater Model with a Predominant Northerly Flow from the Central Plateau

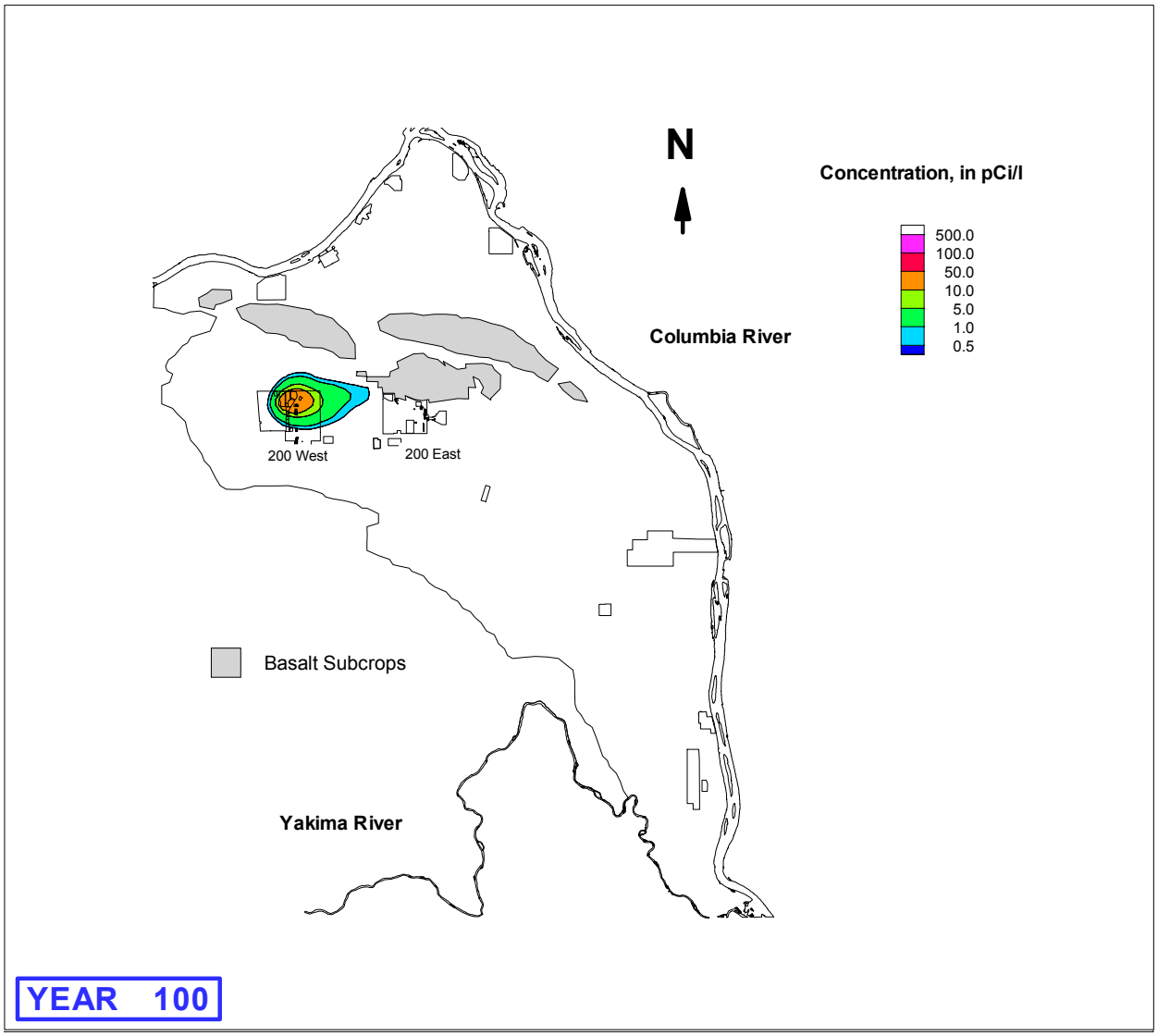
(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.



M0212-0286.75b
HSW EIS 12-10-02

Figure G.13c. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Group 1[(a)] from MLLW in the 200 East Area at 250 Years After Release Using a Groundwater Model with a Predominant Northerly Flow from the Central Plateau

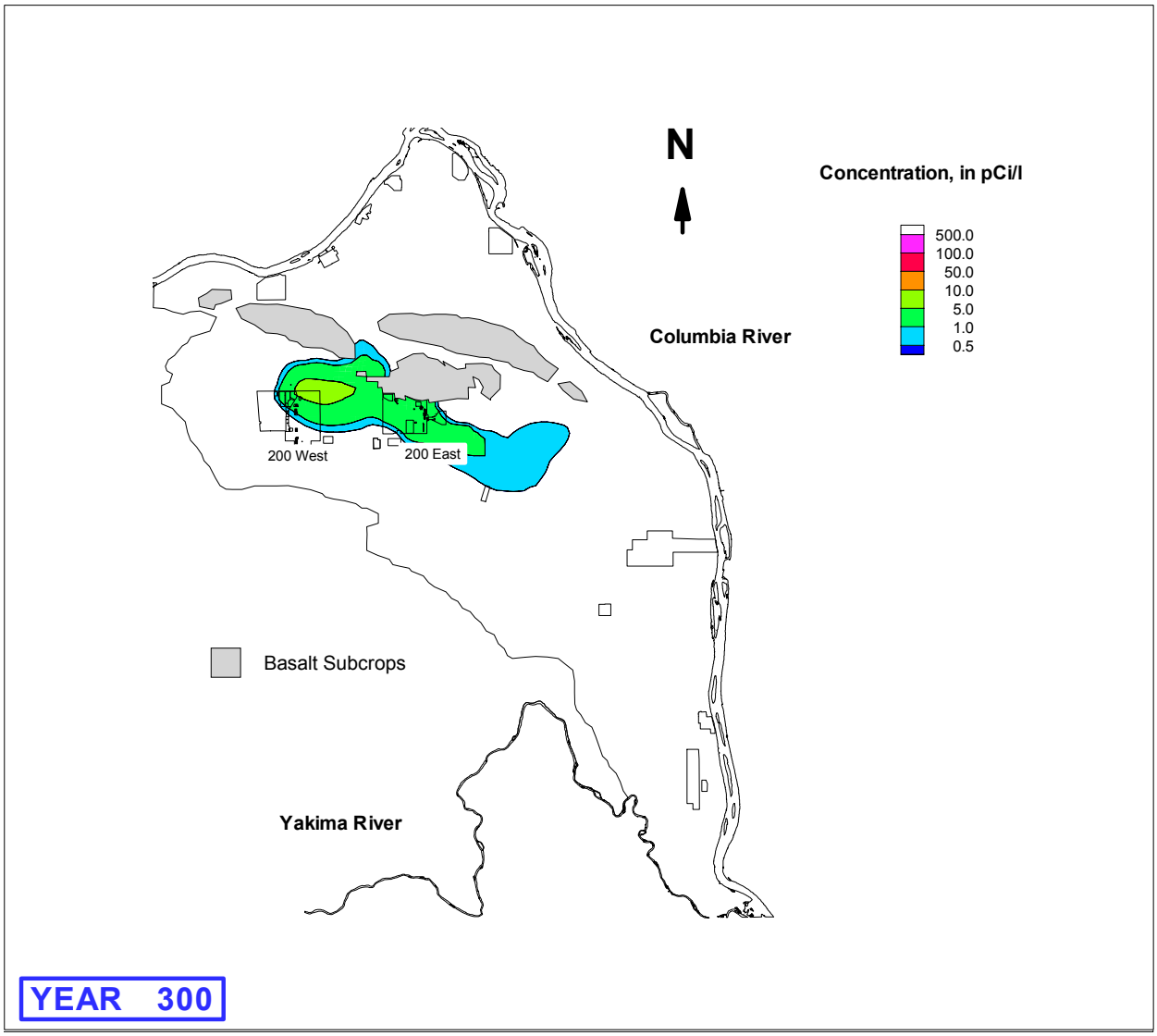
(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.



M0212-0286.75c
HSW EIS 12-10-02

Figure G.14a. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 West Area at 100 Years After Release Using a Groundwater Model with a Predominant Easterly Flow from the Central Plateau

(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.



M0212-0286.75d
HSW EIS 12-10-02

Figure G.14b. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 West Area at 300 Years After Release Using a Groundwater Model with a Predominant Easterly Flow from the Central Plateau

(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.

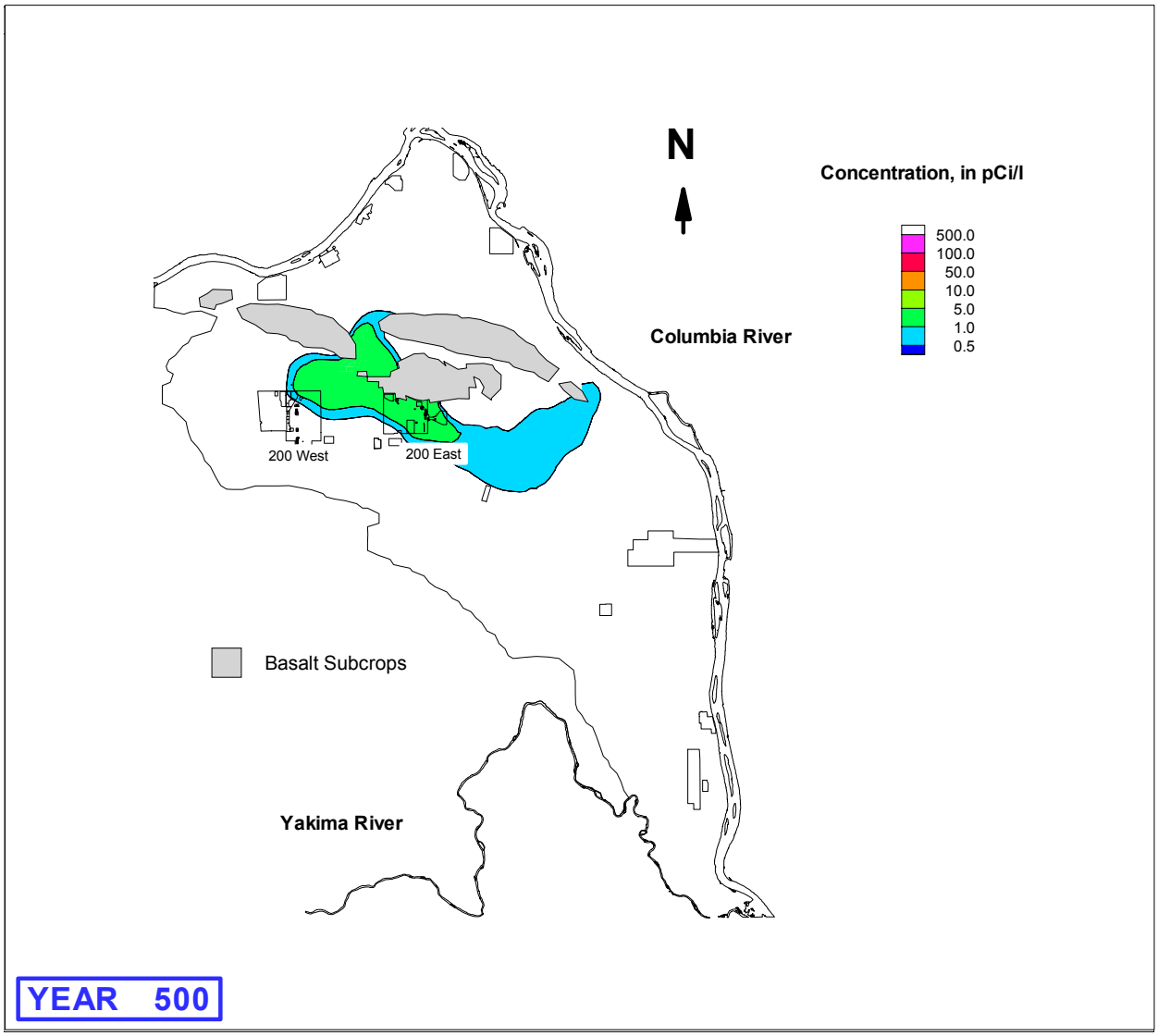
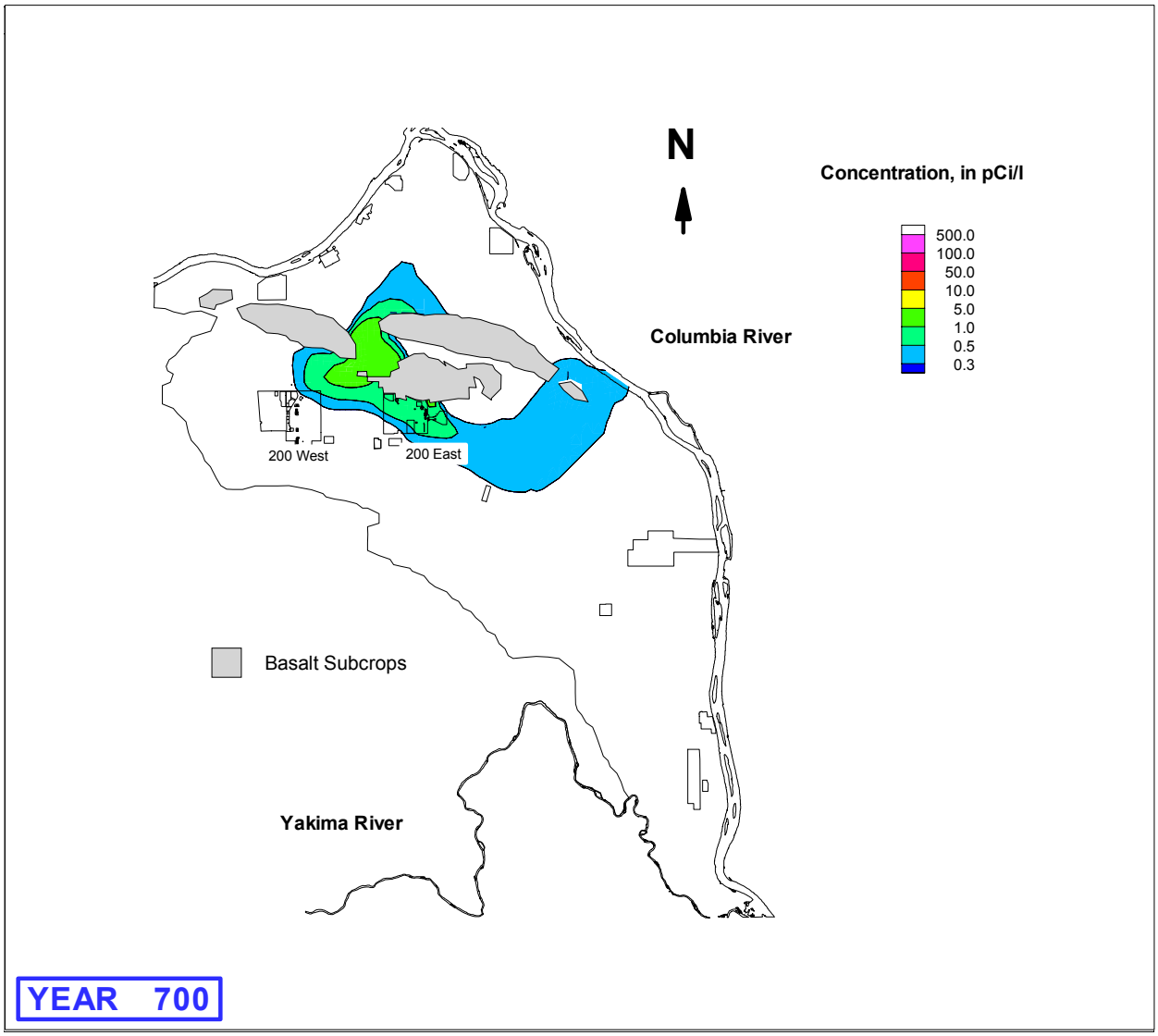


Figure G.14c. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 West Area at 500 Years After Release Using a Groundwater Model with a Predominant Easterly Flow from the Central Plateau

(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.



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HSW EIS 12-10-02

Figure G.14d. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 West Area at 700 Years After Release Using a Groundwater Model with a Predominant Easterly Flow from the Central Plateau

(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.

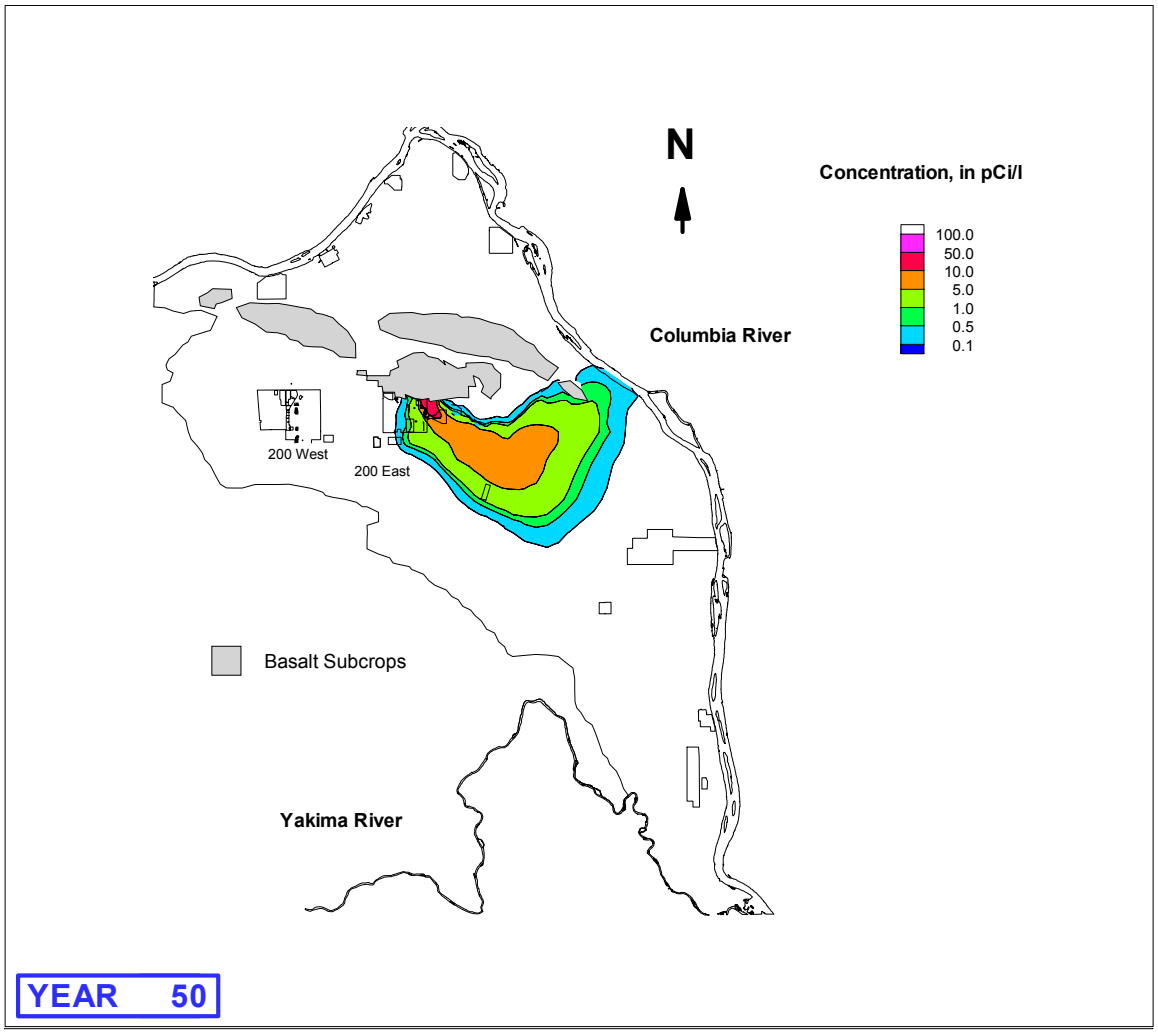


Figure G.15a. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 East Area at 50 Years After Release Using a Groundwater Model with a Predominant Easterly Flow from the Central Plateau

(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.

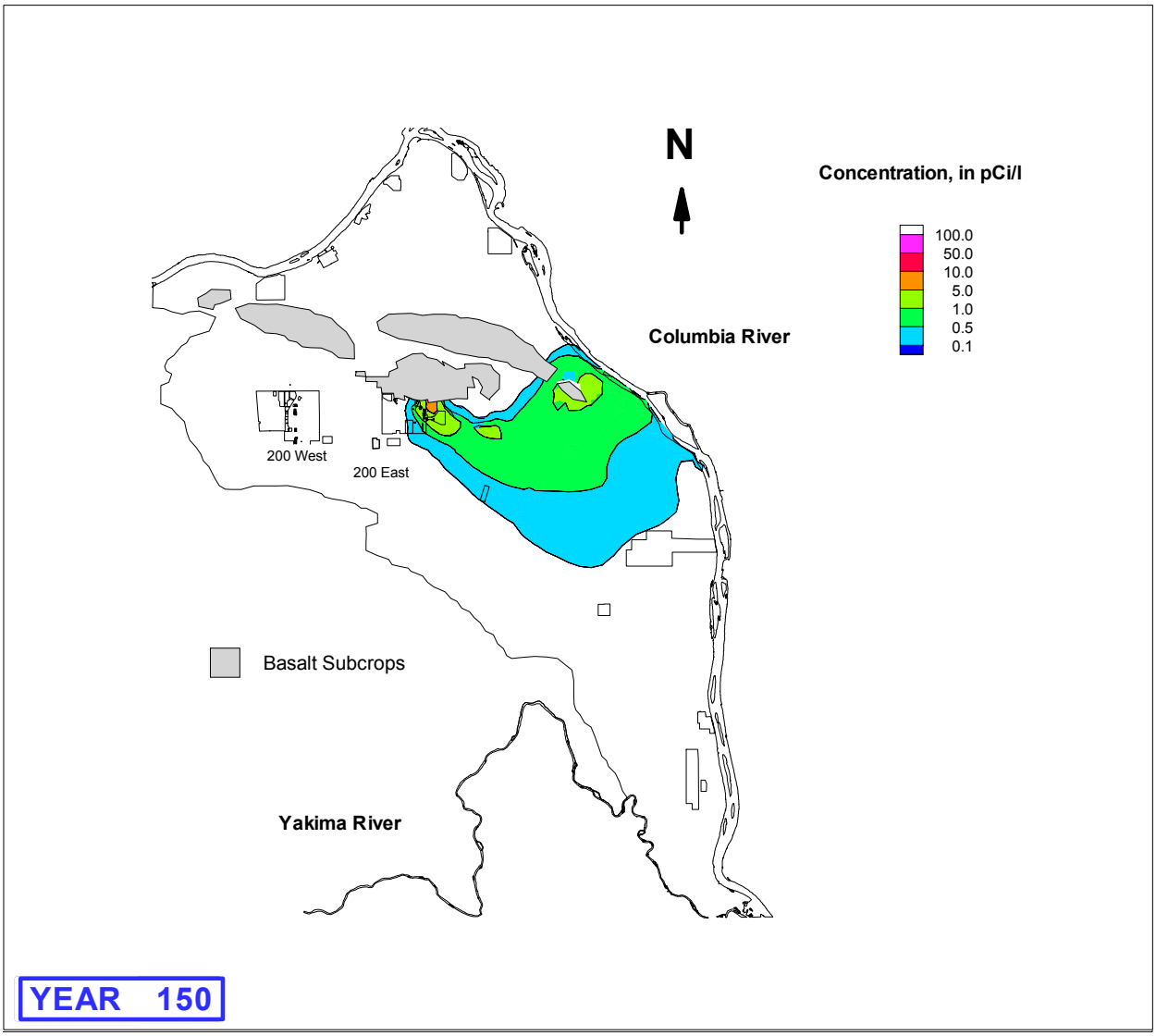
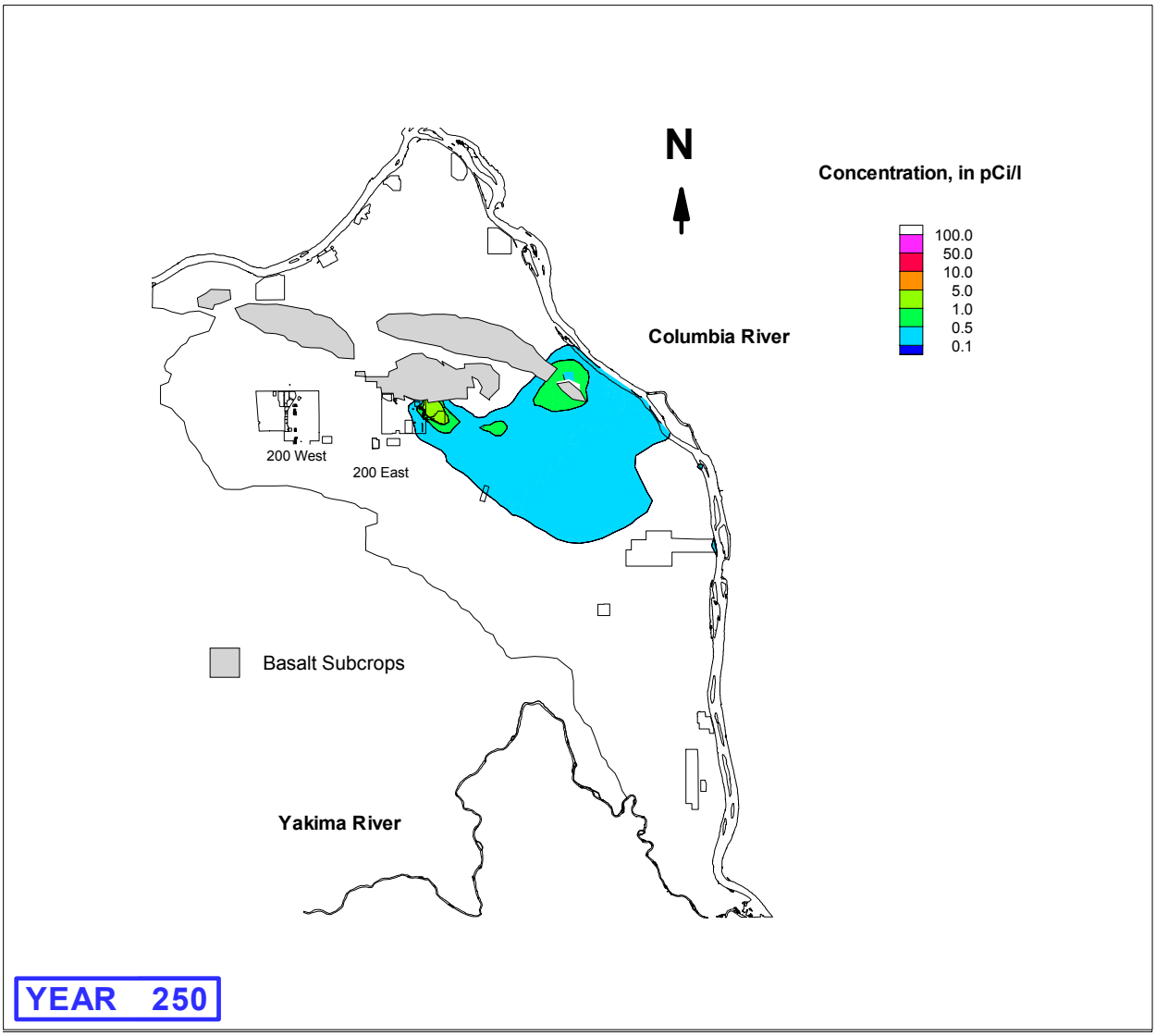


Figure G.15b. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 East Area at 150 Years After Release Using a Groundwater Model with a Predominant Easterly Flow from the Central Plateau

(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.



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HSW EIS 12-10-02

Figure G.15c. Simulated Transport of a 10-Year Unit Release (1 Curie) of a Contaminant Representative of Mobility Class 1[(a)] from MLLW in the 200 East Area at 250 Years After Release Using a Groundwater Model with a Predominant Easterly Flow from the Central Plateau

(a) These simulation results relate the effect of an assumed release (1 curie over a period of 10 years) of a hypothetical, long-lived contaminant in Mobility Class 1 to predicted concentrations at various points in the aquifer system. These results provide the basis for the groundwater transport component of the convolution approach described in Section G.1.2.

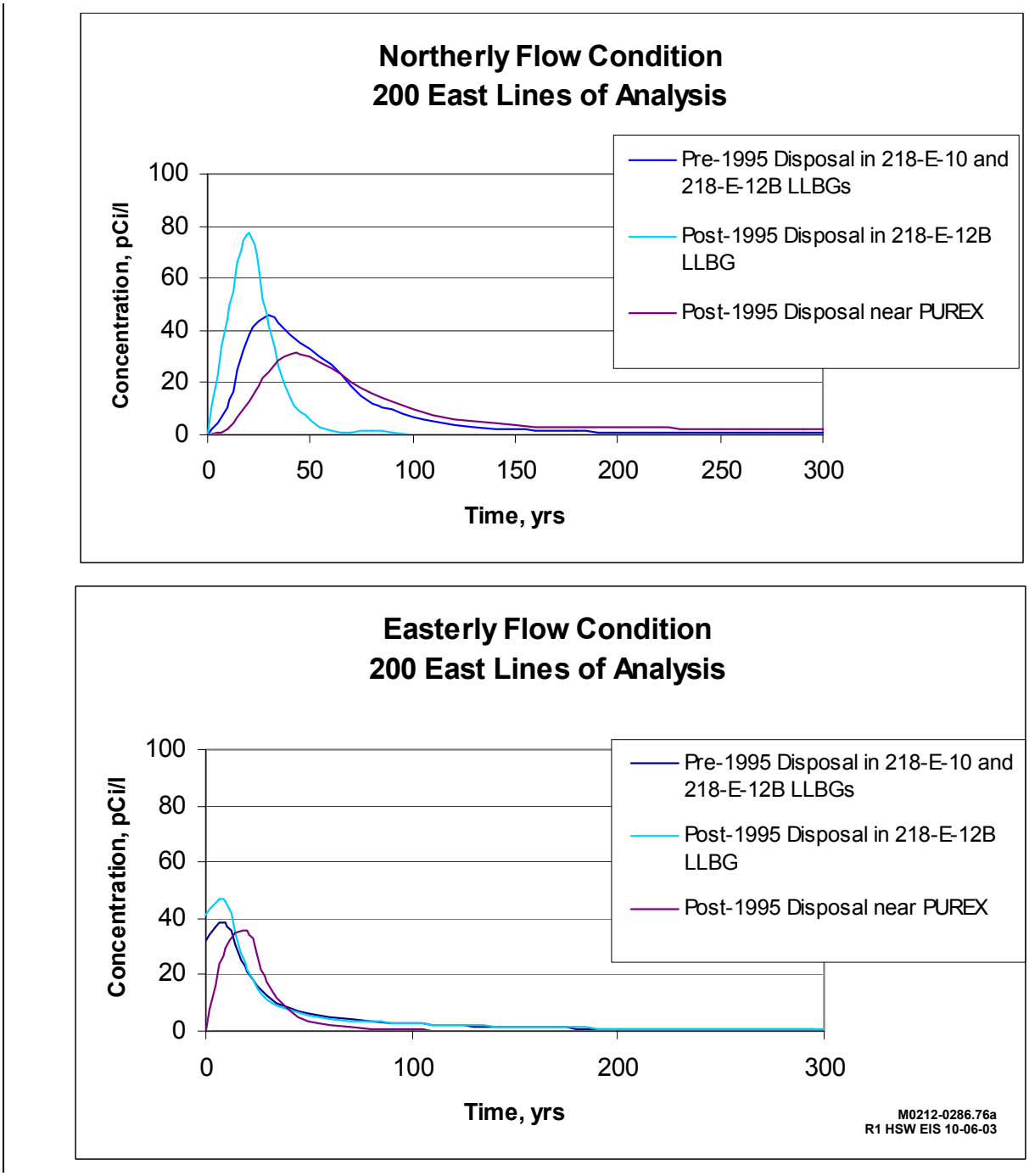


Figure G.16a. Comparison of Predicted Concentrations from Unit Releases from the 200 East Area at 200 East LOAs Using Groundwater Models with a Predominant Northerly and Easterly Flow from the Central Plateau

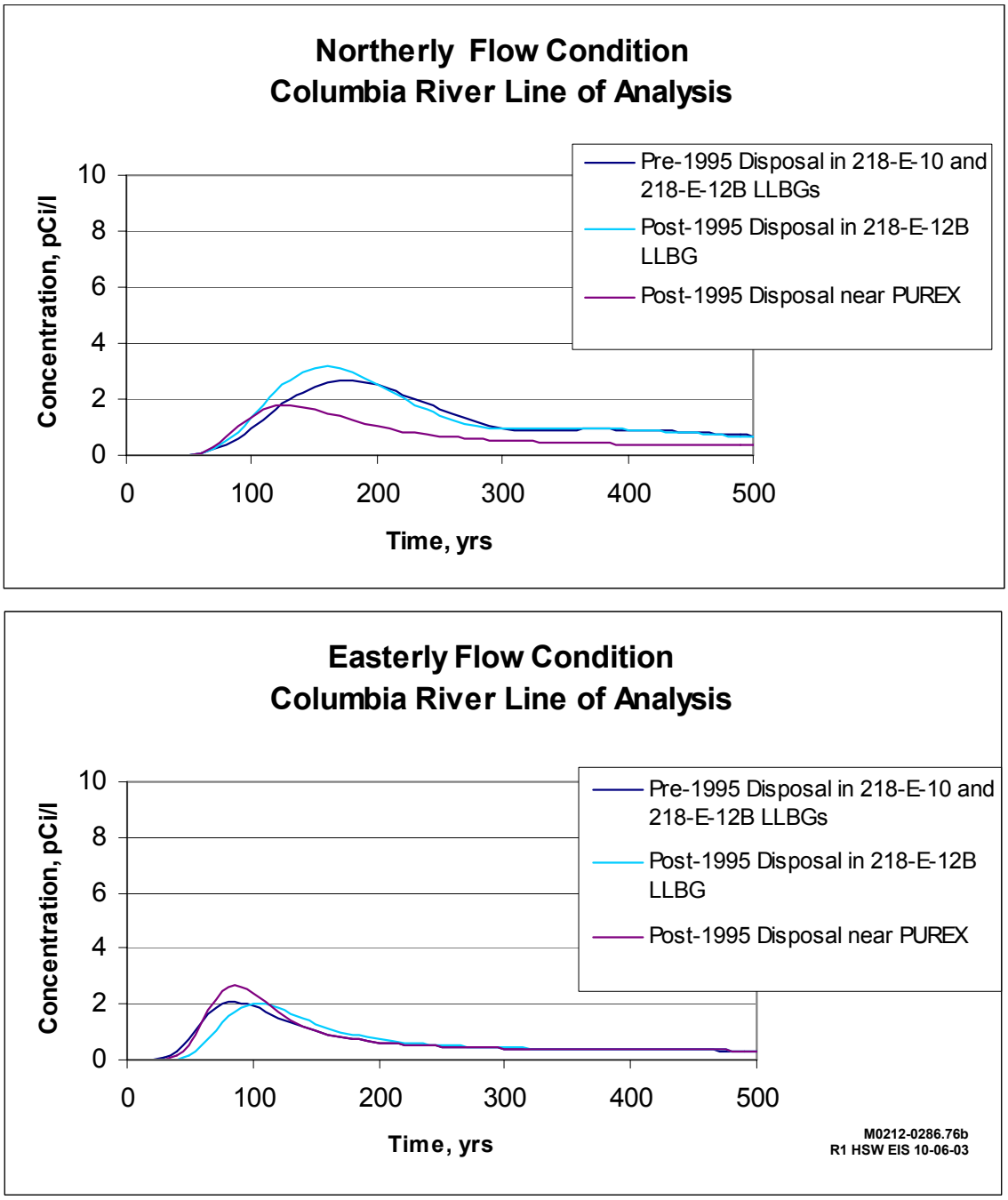
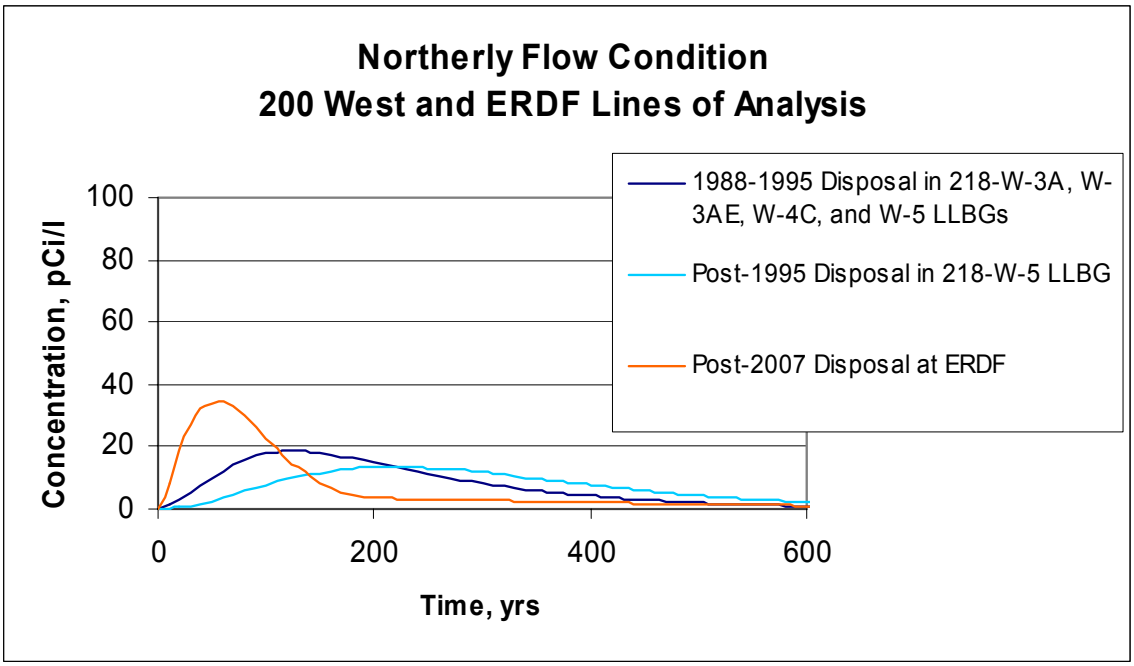


Figure G.16b. Comparison of Predicted Concentrations from Unit Releases from the 200 East Area at Columbia River LOA Using Groundwater Models with a Predominant Northerly and Easterly Flow from the Central Plateau



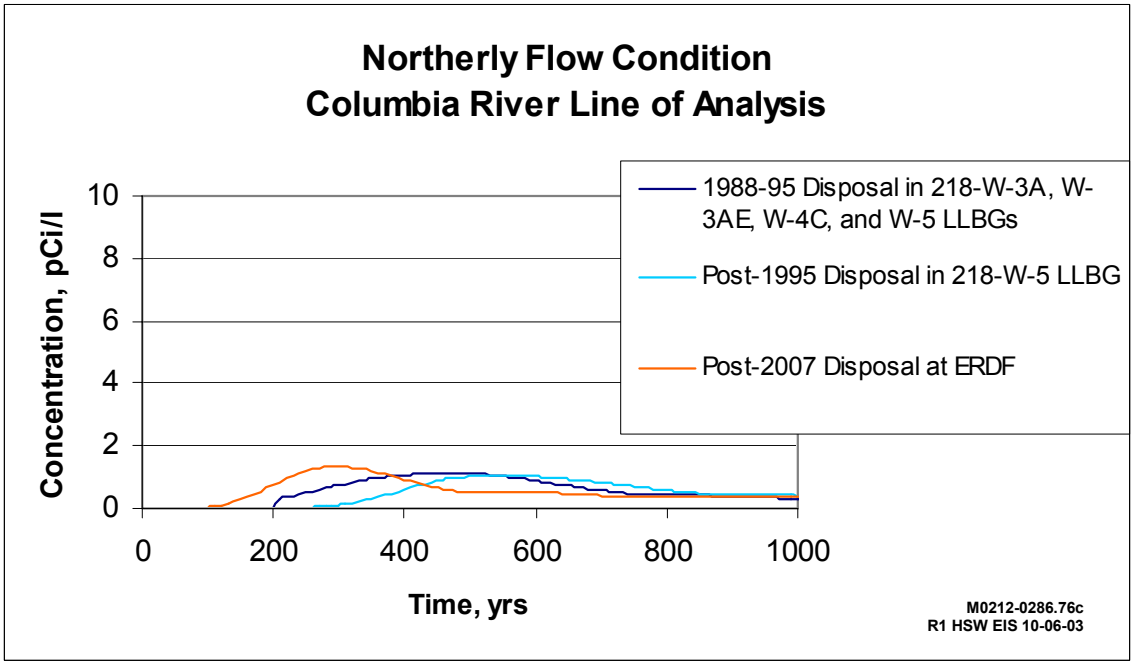


Figure G.17a. Comparison of Predicted Concentrations from Unit Releases from the 200 West Area at the 200 West and ERDF LOAs Using Groundwater Models with a Predominant Northerly and Easterly Flow from the Central Plateau

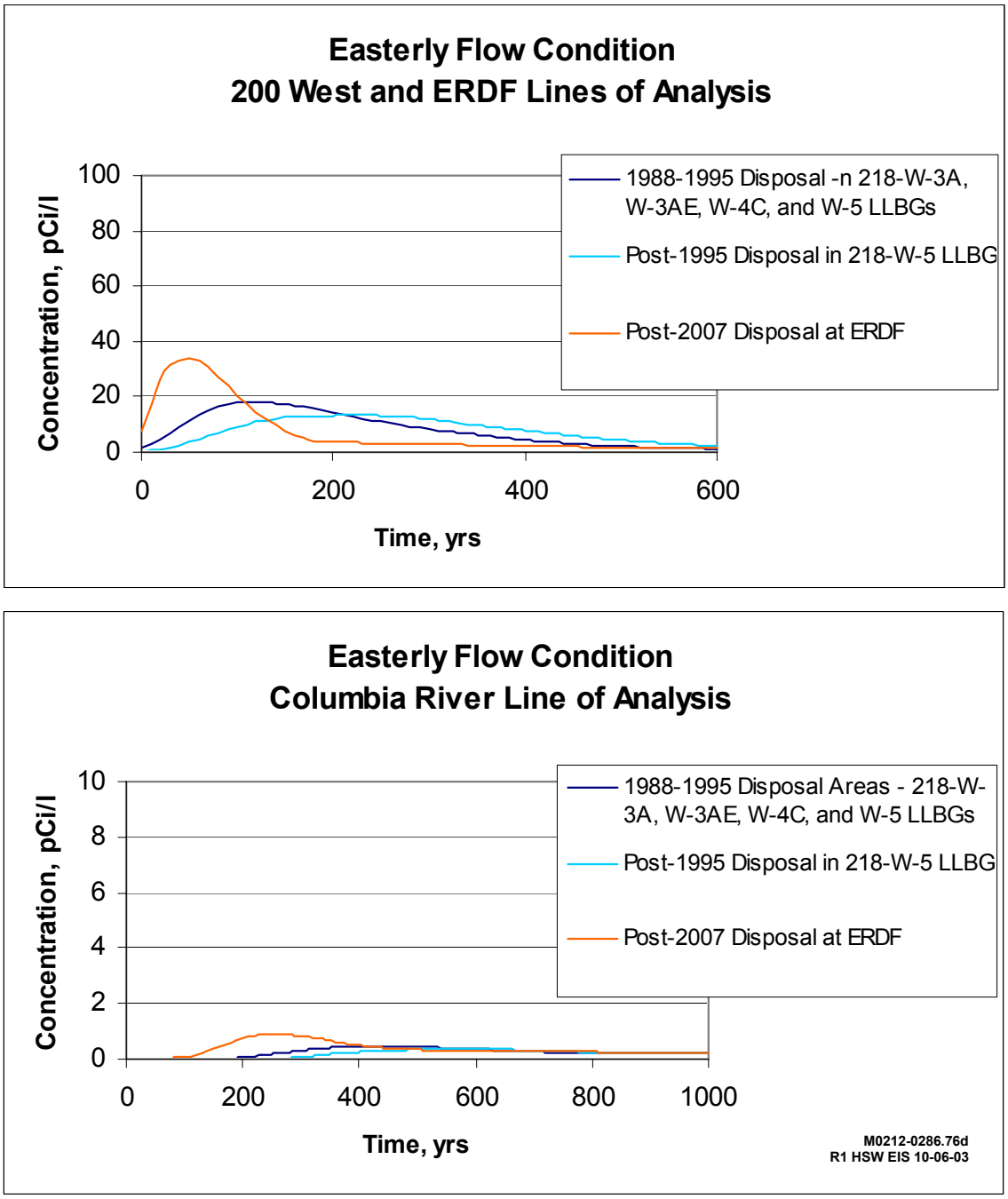


Figure G.17b. Comparison of Predicted Concentrations from Unit Releases from the 200 West Area at the Columbia River LOA Using Groundwater Models with a Predominant Northerly and Easterly Flow from the Central Plateau

This is particularly true for breakthrough curves developed for the LOA near the Columbia River where the location of maximum concentration varies in time as the simulated plumes migrate north to the Columbia River.

G.2 Potential Groundwater Quality Impact Results

Potential impacts on groundwater are provided in the following sections as peak concentrations of contaminants in well water and the time of occurrence. Because of the variation in the location of the different waste types and category releases for a given alternative group, the estimated maximum concentrations calculated from a specific waste category release, provided in Tables G.7 through G.38, may not correspond to the same point on the line of analysis for every waste category and alternative group. Combined concentration levels presented in the following text for each LOA and alternative group reflect the summation of estimated concentration levels regardless of their position on the LOA.

The alternatives, waste types, and disposal conditions are briefly stated to establish the framework for comparing the results. The tables and figures referred to in the following discussion are provided at the end of this section.

G.2.1 Alternative Group A

LLW considered in Alternative Group A includes wastes to be disposed of in several categories:

- Pre-1970 LLW
- 1970–1987 LLW
- 1988–1995 LLW
- 1996–2007 Cat 1 and Cat 3 LLW
- Cat 1 and Cat 3 LLW and MLLW disposed of after 2007 in deeper (18 m) (59 ft) and wider trenches in existing LLBGs 218-E-12B and 218-W-5
- melters disposed of after 2007 in a 21-m (69-ft) deep trench near the PUREX Plant
- ILAW disposed of after 2007 in a disposal facility near the PUREX Plant.

Tabular results of groundwater quality impacts for Alternative Group A are summarized in Tables G.7 through G.9 for wastes disposed of before 2008 and in Tables G.10 through G.12 for wastes disposed of after 2007. Graphical results of groundwater quality impacts are provided in Figures G.18 through G.22 for wastes disposed of before 1996 and in Figures G.23 through G.32 for wastes disposed of after 1996. Results for this alternative group include:

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater at the 1-km (0.6-mi) LOAs downgradient from the waste sites for wastes disposed of before 2008 (Table G.7) and wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.10).

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater near the Columbia River for wastes disposed of before 2008 (Table G.8) and wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.11).

- Predicted peak river fluxes of key radionuclides from an LLBG to the Columbia River for wastes disposed of before 2008 (Table G.9) and wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.12).

G.2.1.1 Wastes Disposed of Before 1996

Constituents released from wastes disposed of before 1996 that have the highest potential impact on groundwater quality are technetium-99 and iodine-129. Estimated combined technetium-99 and potential iodine-129 levels at the 200 East Area NW LOA peaked at about 110 years and at about 220 years at the 200 West Area LOA. Combined concentration levels of technetium-99 were relatively low (less than 20 pCi/L) downgradient from both areas and were a small percentage of the benchmark maximum contaminant level (MCL) for technetium-99 (900 pCi/L). The combined concentration levels of iodine-129 at the 200 East Area NW LOA was about 60 percent (0.6 pCi/L) of the benchmark MCL. This concentration level resulted from releases of the iodine-129 inventory in 1970–1987 LLW. The combined concentration levels of iodine-129 at the 200 West Area LOA was about 50 percent (0.5 pCi/L) of benchmark MCL. This concentration level also resulted from releases of the iodine-129 inventory in 1970–1987 LLW.

Technetium-99 and iodine-129 combined concentrations were well below benchmark MCLs by the time they reached the Columbia River. Overall concentration levels at the Columbia River LOA reached their peaks in about 260 years. Contaminant levels from sources in the 200 West Area reached their peaks near the river LOA between 500 and 600 years.

The combined concentration of carbon-14 and the uranium isotopes were found to peak at about or beyond 10,000 years. Carbon-14 concentrations at all 1-km LOAs were well below the drinking water standard (DWS) of 2000 pCi/L. Combined concentration levels of uranium-238, the dominant uranium isotope, were also well below the benchmark MCL at the 200 East and West Area LOAs at 10,000 years.

Combined contaminant flux for technetium-99 and iodine-129 inventories in previously disposed of LLW reaching the Columbia River within the 10,000-year period of analysis were estimated as follows:

- 95 Ci of technetium-99 (peak loading of 0.1 Ci/yr at around 520–530 years)
- 20 Ci of iodine-129 (peak loading of 0.06 Ci/yr at about 260 years).

This amount of constituent loading does not adversely affect water quality in the Columbia River.

Respective results presented for wastes disposed of before 2008 for Alternative Group A are only presented once in Tables G.7, G.8, and G.9 since these results are the same for all action alternative groups (that is, Alternative Groups A, B, C, D_1, D_2, D_3, E_1, E_2, and E_3). Thus the discussion of results for Alternative Groups B through E will focus on results from LLW and MLLW that would be disposed of after 2007 and not repeat results for wastes disposed of before 2008.

G.2.1.2 Wastes Disposed of After 1995

Potential groundwater quality impacts from wastes disposed of after 1995 also were highest for technetium-99 and iodine-129. Technetium-99 levels at the 200 East Area NW LOA were about 8 percent (75 pCi/L) of the benchmark MCL for the Hanford Only waste volume. The source for these elevated levels is from technetium-99 from MLLW that would be disposed of after 2007. Technetium-99 levels at the 200 West Area LOA were about 33 percent (300 pCi/L) of the benchmark MCL. The source of these impacts is primarily from technetium-99 releases from Cat 3 LLW that would be disposed of after 2007. Predicted technetium-99 levels were very similar for all volumes but were slightly higher for the Upper Bound volume.

Iodine-129 levels at the 200 East Area NW LOA were about 80 percent of the DWS of 1 pCi/L for the Hanford Only volume. The main contributor to these concentration levels is MLLW that would be disposed of after 2007. Iodine-129 levels at the 200 West Area LOA were about 40 percent of the DWS of 1 pCi/L for the Hanford Only volume. The main contributor to these concentration levels is MLLW disposed of between 1996 and 2007 (see Table G.7).

Iodine-129 levels were slightly higher at the 200 East Area NW LOA and slightly lower at the 200 West Area LOA for the Upper Bound volume. This result is reflective of changes in partitioning iodine-129 inventory for the MLLW (1996–2007) waste category between the 200 East and West Areas for the Upper Bound volume (see Table G.7).

Technetium-99 and iodine-129 concentrations were well below benchmark MCLs by the time they reached the Columbia River. Overall concentration levels at the Columbia River LOA from sources in the 200 East Area reached their peaks between 1550 and 1600 years. Contaminant levels from sources in the 200 West Area reached their peaks the Columbia River LOA between 1600 and 2100 years.

Concentration levels of carbon-14 and uranium isotopes at the 1-km (0.6-m) LOAs did not reach their peak values until after the 10,000-year period of analysis and were well below benchmark MCL at 10,000 years.

Combined contaminant flux for technetium-99 and iodine-129 inventories in previously disposed of LLW reaching the Columbia River within the 10,000-year period of analysis were estimated as follows:

- 120 Ci of technetium-99 for the Hanford Only and Upper Bound volumes (peak loading was about 0.04 Ci/yr at about 1750 years)

- 0.2 Ci of iodine-129 for Hanford Only and Upper Bound volumes (peak loading was about 0.0001 Ci/yr at about 1650 years).

This amount of constituent loading does not adversely affect water quality in the Columbia River.

A qualitative analysis of these results using the alternative groundwater conceptual model described in Sections G.1.3.1 and G.1.3.2 would suggest the following:

- Arrival times and estimated concentration levels at the 1-km (0.6-m) well location downgradient for LLW and MLLW disposed of in the 218-E-12B LLBG would be expected to change because these source areas under an easterly flow condition would be closer to an aggregate HSW disposal area boundary and thus be close to the 1-km (0.6-m) well LOA. Changes would be expected to be similar to the earlier rises in concentration levels and slight increases (20 to 30 percent) of concentration levels calculated for unit releases from HSW disposal site areas of the 218-E-12B LLBG. For this alternative group, these types of changes would be expected for nearly all LLW and MLLW categories disposed of in the 218-E-12B LLBG. The most substantial potential impacts would be for key sources that were identified above, including 1) 1970–1987 LLW, 2) MLLW disposed of between 1996 and 2007, and 3) MLLW disposed of after 2007.

- No substantial changes would be expected for estimated concentration levels and impacts estimated from HSW disposal areas in the 218-E-10 LLBG in the 200 East Area and all disposal locations in the 200 West Area and at ERDF.

Respective results presented for wastes disposed of before 2008 for Alternative Group A are only presented once in Tables G.7, G.8, and G.9 since these results are the same for all action alternative groups (that is, Alternative Groups A, B, C, D_1, D_2, D_3, E_1, E_2, and E_3). Thus discussion of results for Alternative Groups B through E will focus on results from LLW and MLLW that would be disposed of after 2007 and not repeat results for LLW and MLLW disposed of between 1996 and 2007 unless the wastes include inventories that are the dominant in a particular HSW disposal area.

G.2.2 Alternative Group B

LLW considered in Alternative Group B includes the same waste considered in Alternative Group A but disposes of Cat 1 and Cat 3 LLW and MLLW in conventional trenches after 2007 in LLBGs 218-E-12B and 218-W-5 and the ILAW disposal facility located just south of the CWC.

Tabular results of groundwater quality impacts for Alternative Group A are summarized in Tables G.7 through G.9 for wastes disposed of before 2008 and in Tables G.13 through G.15 for wastes disposed of after 2007. Graphical results of these impacts are provided in Figures G.18 through G.22 for wastes disposed of before 1996 and in Figures G.33 through G.38 for wastes disposed of after 1996. Results for this alternative group include:

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater at the 1-km (0.6-mi) LOA downgradient from wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.13).

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater near the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.14).

- Predicted peak river fluxes of key radionuclides from an LLBG to the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.15).

G.2.2.1 Wastes Disposed of Before 1996

Results for wastes disposed of before 1996 for Alternative Group A are presented in Tables G.7, G8, and G.9 and apply to Alternative Group B.

G.2.2.2 Wastes Disposed of After 1995

Because of assumptions in the source-term release and vadose zone modeling used for LLW and MLLW disposed of between 1996 and 2007 for Alternative Group B, results for this alternative group were the same for those waste categories calculated for Alternative Group A.

For waste disposed of after 1995, results showed slightly higher concentration values of both technetium-99 and iodine-129 from key wastes at all LOAs. Under this alternative group, groundwater quality was most impacted by releases of technetium-99 and iodine-129 from the disposed of LLW and MLLW. Technetium-99 levels at the 200 East Area NW LOA were about 11 and 13 percent (95 and 116 pCi/L) for the Hanford Only and Upper Bound volumes, respectively. The primary source of these elevated levels is from inventories in MLLW that would be disposed of after 2007. These higher concentration levels are generally consistent with the broader surface area of releases associated with the use of conventional trenches under this alternative group.

Technetium-99 levels at the 200 West Area LOA were estimated to be about 33 percent (300 pCi/L) of the benchmark MCL of 900 pCi/L for both the Hanford Only and Upper Bound waste volumes. These values are slightly less than levels estimated for Alternative Group A. However, this would be expected since the source of these impacts is primarily from the technetium-99 inventories in the Cat 3 LLW that would be disposed of after 2007, and the use of conventional trenches under this alternative group would result in some of the inventory associated with Cat 1 and Cat 3 LLW that would be disposed of after 2007 being emplaced in the 200 East Area.

Iodine-129 levels at the 200 East Area NW LOA were 42 and 47 percent (0.42 and 0.47 pCi/L) of the benchmark MCL of 1 pCi/L for the Hanford Only and Upper Bound waste volumes, respectively. The main contributor to these concentration levels is the release of iodine-129 inventories in ungrouted parts of the MLLW that would be disposed of after 2007. Iodine-129 levels at the 200 West Area LOA were less than 8 percent (0.08 pCi/L) of the benchmark MCL for the Hanford Only waste volume. The main

contributor to these concentration levels is from iodine-129 inventories in the ungrouted part of the MLLW disposed of between 1996 and 2007 (see Table G.7).

Iodine-129 levels were slightly higher at the 200 East Area NW LOA and slightly lower at the 200 West Area LOA for the Upper Bound volume. This impact is reflective of changes in the partitioning of iodine-129 inventory for the MLLW (1996–2007) waste category between the 200 East and West Areas for the Upper Bound volume (see Table G.7).

Concentration levels of carbon-14 and uranium isotopes at the 1-km (0.6-m) well downgradient from source areas of projected LLW and MLLW did not reach their peak values until after the 10,000-year period of analysis. Concentration levels for both constituents were well below benchmark MCLs at 10,000 years.

Concentrations of all constituents were well below benchmark MCLs by the time they reached the Columbia River LOA. Overall concentration levels at the Columbia River LOA from sources in the 200 East Area reached their peaks at about 1400 years. Contaminant levels from sources in 200 West Area sources reached their peaks near the river at about 1500 years.

Combined contaminant flux for technetium-99 and iodine-129 inventories in wastes disposed of after 1995 reaching the Columbia River within the 10,000-year period of analysis were estimated as follows:

- 120 Ci of technetium-99 for the Hanford Only and Upper Bound volumes (peak loading was about 0.04 Ci/yr at about 1690 years)

- 0.2 Ci of iodine-129 for Hanford Only and Upper Bound volumes (peak loading 0.0001 Ci/yr at about 1630 years).

This amount of constituent loading does not adversely affect water quality in the Columbia River.

G.2.3 Alternative Group C

LLW considered in Alternative Group C includes the same wastes considered in Alternative Group A but disposes of Cat 1 and Cat 3 LLW and MLLW in single, lined, expandable trenches after 2007 in LLBGs 218-E-12B and 218-W-5. The melters would be placed in a lined trench, and ILAW would be placed in a single, expandable, lined trench near the PUREX Plant.

Tabular results of groundwater quality impacts for Alternative Group C are summarized in Tables G.7 through G.9 for wastes disposed of before 2008 and in Tables G.16 through G.18 for wastes disposed of after 2007. Graphical results of these impacts are provided in Figures G.18 through G.22 for wastes disposed of before 1996 and in Figures G.39 through G.44 for wastes disposed of after 1996. Results for this alternative group include:

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater at the 1-km (0.6 mi) LOA downgradient from wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.16).

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater near the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.17).

- Predicted peak river fluxes of key radionuclides from an LLBG to the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.18).

G.2.3.1 Wastes Disposed of Before 1996

Results for wastes disposed of before 1996 for Alternative Group A are presented in Tables G.7, G.8, and G.9 and apply to Alternative Group C.

G.2.3.2 Wastes Disposed of After 1995

Because of assumptions in the source-term release and vadose zone modeling used for LLW and MLLW disposed of between 1996 and 2007 for Alternative Group C, results for this alternative group were the same for those waste categories calculated for Alternative Group A (see Tables G.7, G. 8, and G.9). Results for LLW and MLLW that would be disposed of after 2007 for this alternative group were essentially the same as the results presented in Tables G.10 through G.12 for Alternative Group A. These results are consistent since the analysis assumption about waste depth and projected land use for waste that would be disposed of after 2007 are the same for both alternative groups.

G.2.4 Alternative Group D_1

LLW considered in Alternative Group D_1 includes the same wastes considered in Alternative Group A but disposes of Cat 1 and Cat 3 LLW and MLLW in a single, lined, modular combined-use facility after 2007 near the PUREX Plant. The melter trench and the ILAW disposal facility would be placed in the same general area.

Tabular results of groundwater quality impacts for Alternative Group D_1 are summarized in Tables G.7 through G.9 for wastes disposed of before 2008 and in Tables G.19 through G.21 for wastes disposed of after 2007. Graphical results of these impacts are provided in Figures G.18 through G.22 for wastes disposed of before 1996 and in Figures G.45 through G.50 for wastes disposed of after 1996. Results for this alternative group include:

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater at the 1-km (0.6 mi) LOA downgradient from wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.19).

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater near the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.20).

- Predicted peak river fluxes of key radionuclides from an LLBG to the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.21).

G.2.4.1 Wastes Disposed of Before 1996

Results for wastes disposed of before 1996 for Alternative Group A are presented in Tables G.7, G.8, and G.9 apply to Alternative Group D_1.

G.2.4.2 Wastes Disposed of After 1995

Because of assumptions in the source-term release and vadose zone modeling used for LLW and MLLW disposed of between 1996 and 2007 for Alternative Group D_1, results for this alternative group were the same for those waste categories calculated for Alternative Group A (see Tables G.7, G. 8, and G.9).

The highest potential impact for this alternative group reflects the emplacement of all wastes that would be disposed of after 2007 in the vicinity of the PUREX Plant. Potential impacts from LLW and MLLW would be dominated by technetium-99 and iodine-129.

Combined concentration levels for technetium-99 were about 18 to 20 percent (170 to 180 pCi/L) of the benchmark MCL at the 200 East Area SE LOA for the Hanford Only and Upper Bound volumes. The primary source for these elevated levels is from inventories in MLLW that would be disposed of after 2007. Two peaks reflect technetium-99 inventories in both Cat 3 LLW and MLLW that would be disposed of after 2007 near the PUREX Plant.

Combined technetium-99 concentration levels at the 200 Area West LOA were about 5 and 3 percent (42 and 31 pCi/L) of the benchmark MCL for the Hanford Only and Upper Bound volumes. These values are slightly less than levels estimated for Alternative Group A. The source of these impacts is primarily from the technetium-99 inventory in MLLW disposed of between 1996 and 2007 (see Table G.7). Decreased concentrations for the Upper Bound volume reflect the emplacement of some of the MLLW inventory in the 200 East Area.

Combined iodine-129 concentration levels at the 200 East SE LOA were about 28 percent (0.28 pCi/L) of the benchmark MCL for the Hanford Only and Upper Bound waste volumes. The main contributor to these concentration levels is iodine-129 inventories in ungrouted parts of the MLLW that would be disposed of after 2007.

Combined iodine-129 levels at the 200 West Area LOA were about 15 and 8 percent (0.15 and 0.08 pCi/L) of the benchmark MCL for the Hanford Only and Upper Bound waste volumes. The main contributor to these concentration levels is from ungrouted iodine-129 inventories in MLLW disposed of

between 1996 and 2007 (see Table G.7). Combined iodine-129 levels were slightly higher at the 200 East Area SE LOA and slightly lower at the 200 West Area LOA for the Upper Bound waste volume. These results are reflective of changes in partitioning of iodine-129 inventory for the MLLW (1996–2007) waste category between the 200 East and West Areas for the Upper Bound waste volume (see Table G.7).

Combined concentration levels of carbon-14 and uranium isotopes at all LOAs from source areas of projected LLW and MLLW did not reach their peak values until after the 10,000-year period of analysis. Concentration levels for both constituents were well below the benchmark MCLs at 10,000 years.

Technetium-99 and iodine-129 concentrations were well below benchmark MCLs by the time they reached the Columbia River. Overall concentration levels at the Columbia River LOA from sources in the 200 East Area reached their peaks near the river between 1400 and 1500 years. Contaminant levels at the same LOA from sources in the 200 West Area sources reached their peaks between 2100 and 2200 years.

Combined contaminant flux for technetium-99 and iodine-129 inventories in previously disposed of LLW reaching the Columbia River within the 10,000 period of analysis were estimated as follows:

- 100 Ci of technetium-99 for the Hanford Only and Upper Bound volumes (peak loading was about 0.03 Ci /yr at about 14,700 years)

- 0.1 Ci of iodine-129 for Hanford Only and Upper Bound volumes (peak loading was 0.0001 Ci/yr at about 1540 years).

This amount of constituent loading does not adversely affect water quality in the Columbia River.

G.2.5 Alternative Group D_2

LLW considered in Alternative D_2 includes the same wastes considered in Alternative Group A but disposes of Cat 1 and Cat 3 LLW and MLLW in a single, lined, modular combined-use facility after 2007 in LLBG 218-E-12B. The melter trench and the ILAW disposal facility would be placed in the same general area.

Tabular results of groundwater quality impacts for Alternative D_2 are summarized in Tables G.7 through G.9 for wastes disposed of before 2008 and in Tables G.22 through G.24 for wastes disposed of after 2007. Graphical results of these impacts are provided in Figures G.18 through G.22 for wastes disposed of before 1996 and in Figures G.51 through G.56 for wastes disposed of after 1996. Results for this alternative group include:

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater at the 1-km (0.6-mi) LOA downgradient from wastes disposed of after 2007for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.22).

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater near the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.23).

- Predicted peak river fluxes of key radionuclides from an LLBG to the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.24).

G.2.5.1 Wastes Disposed of Before 1996

Potential impact results presented for wastes disposed of before 1996 for Alternative Group A in Tables G.7, G.8, and G.9 also apply to Alternative Group D_2.

G.2.5.2 Wastes Disposed of After 1995

Because of assumptions in the source-term release and vadose zone modeling used for LLW and MLLW disposed of between 1996 and 2007 for Alternative Group D_2, results for this alternative group were the same for those waste categories calculated for Alternative Group A (see Tables G.7, G. 8, and G.9).

The highest potential impacts for this alternative group reflect emplacement of LLW and MLLW that would be disposed of after 2007 in the 218-E-12B LLBG. These potential impacts would be primarily from technetium-99 and iodine-129.

Combined technetium-99 levels at the 200 East Area NW LOA were about 16 and 19 percent (148 and 169 pCi/L) of the benchmark MCL for the Hanford Only and Upper Bound volumes. The primary source for these elevated levels is from inventories in Cat 3 LLW and MLLW that would be disposed of after 2007.

Combined concentration levels of technetium-99 at the 200 West Area LOA were about 5 and 3 percent (42 and 31 pCi/L) of the benchmark MCL for the Hanford Only and Upper Bound volumes, respectively. These values are slightly less than levels estimated for Alternative Group A. The source of these impacts is primarily from the technetium-99 inventory in MLLW disposed of between 1996 and 2007 (see Table G.7). Decreased concentrations for the Upper Bound volume reflect the emplacement of some of the MLLW inventory in the 200 East Area.

The highest combined iodine-129 levels at the 200 East Area NW LOAs were about 28 percent (0.28 pCi/L) of the benchmark MCL for the Hanford Only and Upper Bound waste volumes. The main contributor to these concentration levels is ungrouted iodine-129 inventories in MLLW that would be disposed of after 2007.

The highest combined iodine-129 levels were about 15 and 8 percent (0.15 and 0.08 pCi/L) of the benchmark MCL at the 200 West Area LOA for the Hanford Only and Upper Bound waste volumes. The main contributor to these concentration levels is ungrouted iodine-129 inventories in MLLW disposed of between 1996 and 2007 (see Table G.7).

The highest iodine-129 levels were slightly higher at the 200 East Area NW LOA and slightly lower at the 200 West Area LOA for the Upper Bound volume. This is reflective of changes in the partitioning of the iodine-129 inventory for the MLLW (1996-2007) waste category between the 200 East and West Areas for the Upper Bound volume (see Table G.7).

Concentration levels of carbon-14 and uranium isotopes at the 1-km (0.6-mi) LOA did not reach their peak values until after the 10,000-year period of analysis. Concentration levels for both constituents were well below the benchmark MCLs at 10,000 years.

Technetium-99 and iodine-129 concentrations were well below the benchmark MCLs by the time they reached the Columbia River. Overall concentration levels at the Columbia River LOA from sources in the 200 East Area reached their peaks between 1500 and 1600 years. Contaminant levels from sources in the 200 West Area reached their peaks near the river at about 2000 years.

Combined contaminant flux for technetium-99 and iodine-129 inventories in previously disposed of LLW reaching the Columbia River within the 10,000-year period of analysis were estimated as follows:

- 100 Ci of technetium-99 for the Hanford Only and Upper Bound volumes (peak loading was about 0.03 Ci/yr at about 1520 years)

- 0.11 Ci of iodine-129 for Hanford Only and Upper Bound volumes (peak loading was 0.0001 Ci/yr at about 1640 years).

This amount of constituent loading does not adversely affect water quality in the Columbia River.

G.2.6 Alternative Group D_3

LLW considered in the Alternative D_3 includes the same wastes considered in Alternative Group A but disposes of Cat 1 and Cat 3 LLW and MLLW in a single, lined, modular combined-use facility after 2007 at ERDF. The melter trench and the ILAW disposal facility would also be placed at ERDF.

Tabular results of groundwater quality impacts for Alternative Group D_3 are summarized in Tables G.7 through G.9 for wastes disposed of before 2008 and in Tables G.23 through G.25 for wastes disposed of after 2007. Graphical results of these impacts are provided in Figures G.18 through G.22 for wastes disposed of before 1996 and in Figures G.57 through G.64 for wastes disposed of after 1996. Results for this alternative group include:

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater at the 1 km (0.6 mi) LOA downgradient from wastes disposed of after 1996 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.23).

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater near the Columbia River for wastes disposed of after 1996 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.24).

- Predicted peak river fluxes of key radionuclides from an LLBG to the Columbia River for wastes disposed of after 1996 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.25).

G.2.6.1 Wastes Disposed of Before 1996

Potential impact results presented for wastes disposed of before 1996 for Alternative Group A in Tables G.7, G.8, and G.9 also apply to Alternative Group D_3.

G.2.6.2 Wastes Disposed of After 1995

Because of assumptions in the source-term release and vadose zone modeling used for LLW and MLLW disposed of between 1996 and 2007 for Alternative Group D_3, results for this alternative group were the same for those waste categories calculated for Alternative Group A (see Table G.7, G. 8, and G.9).

The highest potential impacts for this alternative group reflect emplacement of LLW and MLLW that would be disposed of after 2007 at ERDF. Impacts were primarily from technetium-99 and iodine-129.

No LLW and MLLW were disposed of after 1996 in the 200 East Area for the Hanford Only volume under this alternative group. Combined technetium-99 levels at the 200 East Area NW LOA were about 2 percent (15.7 pCi/L) of the benchmark MCL for the Upper Bound volume. The primary source for these elevated levels is from inventories in MLLW disposed of between 1996 and 2007 (see Table G.7).

Combined technetium-99 levels at the 200 West Area LOA were about 5 and 3 percent (42 and 31 pCi/L) of the benchmark MCL for the Hanford Only and Upper Bound volumes. These values are slightly less than levels estimated for Alternative Group A. The source of these impacts is primarily from the technetium-99 inventory in MLLW disposed of between 1996 and 2007 (see Table G.7). Decreased concentrations for the Upper Bound volume reflect the emplacement of some of the MLLW inventory in the 200 East Area.

Combined technetium-99 levels at the ERDF LOA were about 28 percent (250 pCi/L) of the benchmark MCL for the Hanford Only and Upper Bound volumes. The primary source for these elevated levels is from inventories in Cat 3 LLW that would be disposed of after 2007.

No LLW and MLLW were disposed of after 1996 in the 200 East Area for the Hanford Only waste volume under this alternative group. Combined iodine-129 levels at the 200 East Area NW LOA were about 5 percent (0.05 pCi/L) of the benchmark MCL for the Upper Bound waste volume. The main contributor to these concentration levels is from ungrouted iodine-129 inventories in MLLW disposed of between 1996 and 2007.

Combined iodine-129 levels at the 200 West Area LOA were 15 and 8 percent (0.15 and 0.08 pCi/L) of the benchmark MCL for the Hanford Only and Upper Bound waste volumes. The main contributor to these concentration levels is from ungrouted iodine-129 inventories in MLLW disposed of between 1996 and 2007 (see Table G.7).

Combined iodine-129 levels at the 200 West Area LOA were slightly higher at the 200 East Area NW LOA and slightly lower for the Upper Bound volume. This result reflects assumed changes in the partitioning of the iodine-129 inventory for the MLLW (1996–2007) waste category between the 200 East and West Areas for the Upper Bound volume (see Table G.7).

Combined iodine-129 levels at the ERDF LOA were 92 and 94 percent (0.92 and 0.94 pCi/L) of the benchmark MCL for the Hanford Only volume. The main contributor to these concentration levels is from inventories in MLLW that would be disposed of after 2007.

Concentration levels of carbon-14 and uranium isotopes at all LOAs downgradient from source areas of projected LLW and MLLW did not reach their peak values until after the 10,000-year period of analysis. Concentration levels for both constituents were well below benchmark MCLs at 10,000 years.

Combined technetium-99 and iodine-129 concentrations were well below benchmark MCLs by the time they reached the Columbia River. Overall concentration levels from sources in the 200 East Area reached their peaks near the river at about 1400 years. Contaminant levels from sources in the 200 West Area reached their peaks near the river about 2000 years.

Combined contaminant flux for technetium-99 and iodine-129 inventories in previously disposed of LLW reaching the Columbia River within the 10,000-year period of analysis were estimated as follows:

- 120 and 130 Ci of technetium-99 for the Hanford Only and Upper Bound volumes, respectively (peak loading was about 0.04 Ci /yr between 2000 and 2100 years)

- 0.14 Ci of iodine-129 for Hanford Only and Upper Bound volumes (peak loading was 0.0001 Ci/yr at about 2100 years).

This amount of constituent loading does not adversely affect water quality in the Columbia River.

G.2.7 Alternative Group E_1

LLW considered in Alternative Group E_1 includes the same wastes considered in Alternative Group A but disposes of Cat 1 and Cat 3 LLW and MLLW in a single, lined modular trench after 2007 in LLBG 218-E-12B. The melter trench and the ILAW disposal facility would be placed at ERDF.

Tabular results of groundwater quality impacts for Alternative E_1 are summarized in Tables G.7 through G.9 for wastes disposed of before 2008 and in Tables G.28 through G.30 for wastes disposed of after 2007. Graphical results of these impacts are provided in Figures G.18 through G.22 for wastes disposed of before 1996 and in Figures G.65 through G.72 for wastes disposed of after 1996. Results for this alternative group include:

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater at the 1-km (0.6-mi) LOA downgradient from wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.28).

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater near the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.29).

- Predicted peak river fluxes of key radionuclides from an LLBG to the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.30).

G.2.7.1 Wastes Disposed of Before 1996

Potential impact results presented for wastes disposed of before 1996 for Alternative Group A in Tables G.7, G.8, G.9 also apply to Alternative Group E_1.

G.2.7.2 Wastes Disposed of After 1995

Because of assumptions in the source-term release and vadose zone modeling used for LLW and MLLW disposed of between 1996 and 2007 for Alternative Group E_1, results for this alternative group were the same for those waste categories calculated for Alternative Group A (see Tables G.7, G. 8, and G.9).

Potential impacts for this alternative group reflect emplacement of LLW and MLLW that would be disposed of after 2007 in LLBG 218-E-12B and the disposal of melters and ILAW at ERDF. Results for LLW and MLLW that would be disposed of after 2007, excluding the melters, are identical to results for the same wastes in Alternative D_2. The highest potential impacts resulted from releases of technetium-99 and iodine-129.

Combined technetium-99 levels at the 200 East Area NW LOA were about 16 and 19 percent (150 and 170 pCi/L) of the benchmark MCL for the Hanford Only and Upper Bound volumes, respectively. The primary source of these elevated levels is from inventories in Cat 3 LLW and MLLW that would be disposed of after 2007.

Combined technetium-99 levels at the 200 West Area LOA were about 5 and 3 percent (42 and 31 pCi/L) of the benchmark MCL for the Hanford Only and Upper Bound volumes, respectively. These values are slightly less than levels estimated for Alternative Group A. The source of these impacts is primarily from the technetium-99 inventory in MLLW disposed of between 1996 and 2007 (see Table G.7). Decreased concentrations for the Upper Bound volume reflect the emplacement of some of the MLLW inventory in the 200 East Area.

Combined technetium-99 levels at the ERDF LOA were about 0.3 percent (2.7 pCi/L) of the benchmark MCL for both the Hanford Only and Upper Bound volumes. The primary source for these elevated levels is from inventories in the melters that would be disposed of after 2007.

No LLW and MLLW were disposed of after 1996 in the 200 East Area for the Hanford Only waste volume under this alternative group. Combined iodine-129 levels at the 200 East Area NW LOA were about 5 percent (0.04 pCi/L) of the benchmark MCL for the Upper Bound waste volume. The main contributor to these concentration levels is from ungrouted iodine-129 inventories in MLLW disposed of between 1996 and 2007 (see Table G.7).

Combined iodine-129 levels at the 200 West Area LOA were 15 and 8 percent (0.15 and 0.08 pCi/L) of the benchmark MCL for the Hanford Only and Upper Bound waste volumes, respectively. The main contributor to these concentration levels is from ungrouted iodine-129 inventories in MLLW disposed of between 1996 and 2007 (see Table G.7).

Combined iodine-129 levels at the 200 West Area LOA were slightly higher at the 200 East Area NW LOA and slightly lower for the Upper Bound volume, which is reflective of changes in the partitioning of the iodine-129 inventory for the MLLW (1996–2007) waste category between the 200 East and West Areas for the Upper Bound volume (see Table G.7).

Combined iodine-129 levels were 22 percent (0.22 pCi/L) at the ERDF LOA for both the Hanford Only and Upper Bound waste volumes. No iodine-129 inventory was estimated for melters that would be disposed of at ERDF after 2007 for this alternative group.

Concentration levels of carbon-14 and uranium isotopes at the 1-km (0.6-m) well downgradient from source areas of projected LLW and MLLW did not reach their peak values until after the 10,000-year period of analysis. Concentration levels for both constituents were well below the benchmark MCLs.

Technetium-99 and iodine-129 concentrations were well below the benchmark MCLs by the time they reached the Columbia River. Overall concentration levels at the Columbia River LOA from sources in the 200 East Area reached their peaks near the river at about 1400 years. Contaminant levels from sources in the 200 West Area reached their peaks near the river at about 2000 years.

Combined contaminant flux for technetium-99 and iodine-129 inventories in previously disposed of LLW reaching the Columbia River within the 10,000-year period of analysis were estimated as follows:

- 120 and 130 Ci of technetium-99 for the Hanford Only and Upper Bound volumes, respectively (peak loading was about 0.04 Ci/yr between 2000 and 2100 years)

- 0.14 Ci of iodine-129 for Hanford Only and Upper Bound volumes (peak loading was 0.0001 Ci/yr at about 2100 years).

This amount of constituent loading does not adversely affect water quality in the Columbia River.

G.2.8 Alternative Group E_2

LLW considered in Alternative Group E_2 includes the same wastes considered in Alternative Group A but disposes of Cat 1 and Cat 3 LLW and MLLW in a single, lined modular trench after 2007 near the PUREX Plant. The melter trench and the ILAW disposal facility would be placed at ERDF.

Tabular results of groundwater quality impacts for Alternative Group E_2 are summarized in Tables G.7 through G.9 for wastes disposed of before 2008 and in Tables G.31 through G.32 for wastes disposed of after 2007. Graphical results of these impacts are provided in Figures G.18 through G.22 for wastes disposed of before 1996 and in Figures G.73 through G.80 for wastes disposed of after 1996. Results for this alternative group include:

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater at the 1-km (0.6-mi) LOA downgradient from wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.31).

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater near the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.32).

- Predicted peak river fluxes of key radionuclides from an LLBG to the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.33).

G.2.8.1 Wastes Disposed of Before 1996

Various results presented for wastes disposed of before 1996 for Alternative Group A in Tables G.7, G.8, G.9 also apply to Alternative Group E_2.

G.2.8.2 Wastes Disposed of After 1995

Because of assumptions in the source-term release and vadose zone modeling used for LLW and MLLW disposed of between 1996 and 2007 for Alternative Group E_2, results for this alternative group were the same for those waste categories calculated for Alternative Group A (see Tables G.7, G. 8, and G.9).

Potential impacts for this alternative group reflect emplacement of LLW and MLLW disposed of after 2007 near the PUREX Plant and the disposal of melters and ILAW at ERDF. Results for LLW and MLLW that would be disposed of after 2007, excluding the melters, are identical to results for the same wastes in Alternative Group D_1 (see Section G.2.4). Results for the melters were the same as those calculated for Alternative Group E_1 (see Section G.2.7).

G.2.9 Alternative Group E_3

LLW considered in Alternative Group E_3 includes the same wastes considered in Alternative A but disposes of Cat 1 and Cat 3 LLW and MLLW in a single, lined modular trench after 2007 at ERDF. The melter trench and the ILAW disposal facility would be placed near the PUREX Plant.

Tabular results of groundwater quality impacts for Alternative Group E_3 are summarized in Tables G.7 through G.9 for wastes disposed of before 2008 in Tables G.34 through G.36 for wastes disposed of after 2007. Graphical results of these impacts are provided in Figures G.18 through G.22 for wastes disposed of before 1996 and in Figures G.81 through G.88 for wastes disposed of after 1996. Results for this alternative group include:

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater at the 1-km (0.6-mi) LOA downgradient from wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.34).

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater near the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.35).

- Predicted peak river fluxes of key radionuclides from an LLBG to the Columbia River for wastes disposed of after 2007 for Hanford Only, Lower Bound, and Upper Bound volumes (Table G.36).

G.2.9.1 Wastes Disposed of Before 1996

Various results presented for wastes disposed of before 1996 for Alternative Group A in Tables G.7, G.8, G.9 also apply to Alternative Group E_3.

G.2.9.2 Wastes Disposed of After 1995

Because of assumptions in the source-term release and vadose zone modeling used for LLW and MLLW disposed of between 1996 and 2007 for Alternative Group E_2, results for this alternative group were the same for those waste categories calculated for Alternative Group A (see Tables G.7, G. 8, and G.9).

Potential impacts for this alternative group reflect emplacement of LLW and MLLW disposed of after 2007 near the PUREX Plant and the disposal of melter MLLW and ILAW at ERDF. Results for LLW and MLLW that would be disposed of after 2007, excluding the melters, are identical to results for the same wastes in Alternative Group D_3 (see Section G.2.6).

Results for Alternative Group E_3 for combined technetium-99 and iodine-129 concentration levels for Hanford Only and Upper Bound volumes are summarized in Section 5.3, Figures 5.20 and 5.21. Additional information can be found in several tables and figures referenced in Section G.2.9.

Combined technetium-99 levels were slightly less than 3 percent (22 pCi/L) of the benchmark MCL at the 200 East Area SE LOA for the Hanford Only waste volume. The potential impact for the Hanford Only waste volume reflects the melter and ILAW disposals near the PUREX Plant. The highest combined iodine-129 levels at the 200 East Area SE LOA were about 20 percent (0.2 pCi/L) of the benchmark MCL for both the Hanford Only and Upper Bound waste volumes as a result of the ILAW disposal near PUREX.

G.2.10 No Action Alternative

LLW considered in the No Action Alternative includes wastes to be disposed of in several categories:

- LLW disposed of before 1970
- LLW disposed of after 1970 but before 1988
- LLW disposed of between 1988 and 1995
- Cat 1 LLW disposed of in conventional trenches between 1996 and 2007
- Cat 3 LLW and GTC3 LLW disposed of in conventional trenches between 1996 and 2007
- MLLW disposed of in conventional trenches between 1996 and 2007
- Cat 1 and Cat 3 LLW and MLLW disposed of in conventional trenches in LLBGs 218-E-12B and 218-W-5.

Contaminants considered in the LLW categories include estimated inventories associated with Hanford Only and Lower Bound waste volumes. Contaminants considered in the MLLW category include estimated inventories associated with Hanford Only and Lower Bound waste volumes.

Tabular results of groundwater quality impacts for the No Action Alternative for all waste categories are summarized in Tables G.37 through G.39. Graphical results of these impacts for all waste categories are provided in Figures G.89 through G94. Results for the No Action Alternative include:

- Predicted peak concentrations of key radionuclides from an LLBG in groundwater at the 1-km (0.6-mi) LOA downgradient from the waste sites for LLW disposed of before 1996 for the Hanford Only and Lower Bound volumes and LLW and MLLW disposed of after 1995 for the Hanford Only and Lower Bound volumes (Table G.37).
- Predicted peak concentrations of key radionuclides from an LLBG in groundwater near the Columbia River for wastes disposed of before 1996 for the Hanford Only and Lower Bound volumes and after 1995 for the Hanford Only and Lower Bound volumes (Table G.38).

- Predicted peak river fluxes of key radionuclides from an LLBG to the Columbia River for wastes disposed of before 1996 for the Hanford Only and Lower Bound volumes and after 1995 for the Hanford Only and Lower Bound volumes (Table G.39).

G.2.10.1 Wastes Disposed of Before 1996

The highest potential groundwater quality impacts from wastes disposed of before 1996 are related to technetium-99 and iodine-129 releases. Estimated concentrations of technetium-99 and iodine-129 peak at about 110 years at the 200 East Area NW LOA and about 220 years at the 200 West Area LOA. Combined levels of technetium-99 were less than 2 percent (18 pCi/L) at the 200 East Area NW and West LOAs. Combined levels of iodine-129 at the 200 East Area NW LOA were about 50 percent (0.5 pCi/L) of the benchmark MCL.

Combined levels of iodine-129 at the 200 West Area LOA were about 50 percent (0.5 pCi/L) of the benchmark MCL. This concentration level is from releases of the iodine-129 inventory in LLW disposed of between 1970 and 1987.

Carbon-14 and uranium isotopes concentration levels were found to peak at about or beyond 10,000 years. Carbon-14 concentrations were well below the benchmark MCL of 2000 pCi/L at the 200 East and West Area LOAs. Concentration levels of uranium-238, the dominant uranium isotope, were also well below the benchmark MCL of 30 pCi/L at the 200 East and West Area LOAs at 10,000 years. Uranium-238 concentration levels reached their peak of about 3 pCi/L between 14,000 and 16,000 years at the 200 West Area LOA.

Technetium-99 and iodine-129 concentrations were well below benchmark MCLs by the time they reached the Columbia River. Overall concentration levels from sources in the 200 East Area reached their peaks at the Columbia River LOA at about 260 years. Contaminant levels from sources in the 200 West Area reached their peaks at the Columbia River LOA between 500 and 600 years.

Combined contaminant flux for technetium-99 and iodine-129 inventories in previously disposed of LLW reaching the Columbia River within the 10,000-year period of analysis were estimated as follows:

- 1 Ci of technetium-99 (peak loading at 0.001 Ci/yr between 520–530 years)
- 0.5 Ci of iodine-129 (peak loading at 0.001 Ci/yr at around 260 years).

This amount of constituent loading does not adversely affect water quality in the Columbia River.

G.2.10.2 Wastes Disposed of After 1995

The highest potential groundwater quality impacts from LLW and MLLW disposed of after 1995 resulted from releases of technetium-99 and iodine-129. Combined technetium-99 levels at the 200 East Area NW LOA were about 8 percent (77 pCi/L) of the benchmark MCL for the Hanford Only volume. The primary source for these elevated levels is from inventories in MLLW disposed of after 1995.

Combined technetium-99 levels were about 25 percent (225 pCi/L) of the benchmark MCL at the 200 West Area LOA. The source of these impacts was primarily from the technetium-99 inventory in Cat 3 LLW disposed of after 1995.

The highest combined iodine-129 levels were about 6 percent (0.06 pCi/L) of the benchmark MCL at the 200 West Area LOA for the Hanford Only waste volume. The main contributor to these concentration levels is from inventories in MLLW disposed of after 1995.

Concentration levels of carbon-14 and uranium isotopes at the 1-km (0.6-m) LOAs downgradient from source areas of LLW and MLLW disposed of after 1995 did not reach their peak values until after the 10,000-year period of analysis. Concentration levels for both constituents were well below the benchmark MCLs at 10,000 years.

Technetium-99 and iodine-129 concentration levels were well below the benchmark MCLs by the time they reached the Columbia River. Overall concentration levels at the Columbia River LOA from sources in the 200 East Area reached their peaks at the Columbia River LOA at 260 years for ungrouted forms of technetium-99 and iodine-129 and at about 850 years for grouted forms of the inventories. Contaminant levels from sources in the 200 West Area reached their peaks near the river between 1660 and 1820 years.

Combined contaminant flux for technetium-99 and iodine-129 inventories in previously disposed of LLW reaching the Columbia River within the 10,000-year period of analysis were estimated as follows:

- 100 Ci of technetium-99 for the Hanford Only waste volume (peak loading was about 0.03 Ci/yr at about 1820 years)

- 0.07 Ci of iodine-129 for the Hanford Only waste volume (peak loading was 0.0001 Ci/yr at about 1660 years).

This amount of constituent loading does not adversely affect water quality in the Columbia River.