MARTIN MARIETTA

ES/ER-22/V2&D1
ORNL/ER/Sub-87/99053/5/V2

ENVIRONMENTAL
RESTORATION
PROGRAM

RCRA Facility Investigation Report for Waste Area Grouping 6 at Oak Ridge National Laboratory, Oak Ridge, Tennessee

Volume 2
Sections 4 Through 9

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FOR THE UNITED STATES
DEPARTMENT OF ENERGY
UCN-17560 (6 7-91)

ORNL/ER/Sub--87/99053/5/V2

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RCRA Facility Investigation Report for Waste Area Grouping 6 at Oak Ridge National Laboratory, Oak Ridge, Tennessee

Volume 2: Sections 4 Through 9

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Date Issued—September 1991

Prepared by
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Oak Ridge, Tennessee
under subcontract 30B-99053C

Prepared for
U.S. Department of Energy
Office of Environmental Restoration and Waste Management
under budget and reporting code EW 20

MARTIN MARIETTA ENERGY SYSTEMS, INC.
managing the
Oak Ridge National Laboratory   Oak Ridge Y-12 Plant
Oak Ridge K-25 Site   Paducah Gaseous Diffusion Plant
under contract DE-AC05-84OR21400
and the
Portsmouth Gaseous Diffusion Plant
under contract DE-AC05-76OR00001
for the
U.S. DEPARTMENT OF ENERGY

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4. NATURE AND EXTENT OF CONTAMINATION

4.1 INTRODUCTION

Section 4 describes the type, concentration, and spatial distribution of contamination at WAG 6, as developed from RFI data, ORNL RCRA compliance monitoring data, and data from previous investigations. Past waste disposal activities in WAG 6 created a complex contaminant source through which releases occur to the surrounding environment. Environmental media have become contaminated, resulting in contaminants, in many cases, migrating beyond their original disposal areas. The information in this section forms the basis for the assessment of human health and environmental impacts; it also contributes to the definition of the scope of site closure and corrective action alternatives.

Section 4.1 provides an overview of WAG 6 RFI environmental data, including a description of the WAG 6 data base and an evaluation of data quality. Environmental data consist of results from chemical or radiological analysis of samples from various media. Section 4.1 also identifies specific contaminants detected and describes general observations made regarding the data. Inventory information related to source areas is summarized in Sect. 4.2. Section 4.3 presents the nature and extent of on-WAG contamination by environmental media (groundwater, surface water, sediments, and soil). Section 4.4 summarizes potential off-WAG migration of contaminants along the WAG boundary. Appendix 4A is an assessment of contaminant sampling and data analysis quality.

4.1.1 WAG 6 RFI Environmental Data

4.1.1.1 Data base

The WAG 6 analytical data base consists of two elements: the environmental data base, which contains the actual analytical data, and the sample tracking record. The WAG 6 RFI environmental data base is recorded in the Technical Memorandums associated with this RFI report. Table 4.1 lists the Technical Memorandums in which data are presented.

The two categories of analytical data collected are radiological and chemical. Radiological data include gross alpha and gross beta activity, tritium, and specific radionuclide isotopic analyses. Chemical data consisted of analytical results for TCL organic compounds and TAL inorganic analytes; selected samples were analyzed for RCRA Appendix IX analytes. TAL, TCL, and Appendix IX analytes are listed in Sect. 2.

Radiological data for liquid samples are reported in picocuries per liter (pCi/L). For solid samples, data are reported in picocuries per gram (pCi/g). Both units are measurements of the level of radioactivity. Radioactivity is determined as the rate at which a radionuclide decays, or disintegrates. One picocurie is equivalent to 2.22 dpm. A direct equivalence between radioactivity and chemical concentration (i.e., the concentration of the radionuclide on a weight basis) can be calculated only on a radionuclide-specific basis.

Radiological results are generally reported with an accompanying error factor that represents a statistical value equal to two standard deviations of the sample's activity.

Absence of the error term means that the reported radiological result is the Minimum Detectable Activity (MDA) for that analyte in that sample. The actual activity is some value less than the MDA. The MDA is a function of the measurement instrument background and will vary from sample to sample. The best estimate of the radioactivity in a sample is the reported value. The range of what that value may be, with 95% confidence, can be estimated by adding the error factor to the reported value to obtain the upper limit, and subtracting the error factor from the reported value to obtain the lower limit. For example, if the results are reported as 10± 5 pCi/g, the best estimate for the value is 10 pCi/g, and the range of what that value may be, with 95% confidence, is 5 to 15 pCi/g. If the error factor is greater than the reported value, the activity of that radionuclide is statistically indistinguishable from zero and the result is considered a non-detect. In the analyses of radiological samples at ORNL, laboratory background concentrations of the radionuclide being analyzed for are subtracted from the analytical result. This may result in negative concentrations of a radionuclide being reported, indicating that the radiological sample in question may contain an amount of that radionuclide indistinguishable from background concentrations.

Chemical data for liquid samples, such as samples of groundwater or surface water, are presented in units of micrograms of the analyte per liter of liquid (μg/L). Chemical data for solid samples, such as samples of soil or sediment, are presented in units of micrograms of the analyte per kilogram of solid (μg/kg) on a dry weight basis. Both μg/L and μg/kg are approximately equivalent to one part in a billion parts (ppb) by weight. Where appropriate, chemical data for liquids and solids are also reported in milligrams per liter (mg/L) and in milligrams per kilogram (mg/kg), respectively. Both mg/L and mg/kg are approximately equivalent to one part in a million parts (ppm) by weight.

Frequently, a concentration of an organic constituent is reported as, for example, "<5 μg/L." This indicates that the constituent was not detected at or above the detection limit for that constituent (in this case, 5 μg/L).

4.1.1.2 Data quality assessment

An assessment of the quality of WAG 6 RFI environmental data was performed to provide data users with insight regarding the usability of the data. This quality assessment does not address data generated by the ORNL RCRA compliance monitoring program, which is discussed in subsequent sections. Both chemical and radiological RFI data were evaluated by assessing the following data quality indicators: precision, accuracy, representativeness, completeness and comparability (PARCC) (EPA *Guidelines for Data Useability in Risk Assessment* 1990a). A detailed presentation of the findings and conclusions of this assessment appears in Appendix 4A.

All chemical data generated during the WAG 6 RFI were submitted for validation. Any problems associated with the analytical procedure that were felt to affect the integrity of the data were dealt with through the use of reviewer qualifiers. Qualifiers revealed problems associated with the analysis such as blank contamination, violation of holding times, poor analytical accuracy, and low precision. In the validation process, trip blank and method blank contamination were evaluated to ensure that false positive detects would not be reflected in the analytical data. Spike recoveries and relative percent differences (RPD) between duplicate analyses were used to evaluate accuracy and precision. The guidance documents used in the assessment and qualification of chemical data are the *Laboratory Data Validation Functional Guidelines for Evaluating Inorganics Analyses* and the *Laboratory Data Validation Functional Guidelines For Evaluating Organics Analyses* (EPA 1988b, c). Validated data that were rejected or deemed unusable were not included in this report.

Subsequent to the validation and PARCC review of RFI chemical and radiological data, limitations or uncertainties associated with the data were evaluated. This evaluation was performed to determine whether deficiencies identified would have a significant impact on the ability to characterize and quantify the contaminants present on-WAG. Based on this evaluation (Appendix 4A), it was determined that there were sufficient chemical and radiological data of known quality to describe the nature and extent of contamination and to perform the base line risk assessment.

4.1.2 Contaminants Detected

Tables 4.2, 4.3, 4.4, 4.5, 4.6 and 4.7 summarize the concentration range (by medium) of TCL organic compounds, TAL inorganic analytes, Appendix IX constituents, and radiological constituents that were detected at least once in environmental samples, either as part of the WAG 6 RFI or the ORNL RCRA compliance monitoring program. These tables also summarize the total number of times a given compound or analyte was detected and the number of times it was analyzed for. Reference sample results are included in the tables. Rejected data and tentatively identified compounds (TICs) are not included in the summaries. TICs are not included because there are no bases for comparison in either environmental criteria or reference samples. Subsequent sections of Sect. 4 do present results for TICs whenever their nature and extent are important considerations. One example is ethyl ether, a TIC which is included in the groundwater discussion because it is a component of scintillation fluids.

In Tables 4.2 through 4.7, minimum and maximum concentrations for both reference and site samples are listed by medium for comparison to each other and various environmental and health-based criteria. For radionuclides, these health-based, criteria-derived concentrations correspond to a one-in-a-million (i.e., 1×10^{-6}) excess total lifetime cancer risk and are based on pathway-specific unit risk factors provided for drinking water and soil in EPA's *Health Effects Assessment Summary Tables* (EPA 1990b). For chemical and radionuclide contaminants, the following criteria are also included:

- Safe Drinking Water Act (SDWA) Maximum Contaminant Limits (MCLs)
- Proposed SDWA MCLs
- Tennessee Secondary Drinking Water Standards
- Federal Ambient Water Quality Criteria for the protection of freshwater organisms
- Proposed RCRA Subpart S, Appendix A action levels for water and soil

The RCRA action levels are derived from those given in Appendix A of proposed RCRA Subpart S (EPA 1990b). The assumptions, methodology, and health-based criteria (slope factors, reference doses) used to derive the Appendix A values are given in Appendixes D, E, and F, respectively, of proposed Subpart S. These health-based criteria were checked against those published by ORNL in *Toxicity Values from the U. S. EPA Integrated Risk Information System and Health Effects Assessment Summary Table* (ORNL 1991a). If a health-based criterion for a compound had changed, the RCRA Action Level was recalculated using the Subpart S Appendix D and E assumptions and methodology with the new health-based criterion.

From evaluation of Tables 4.2 through 4.7, it is evident that although a large number of contaminants were detected in WAG 6 samples, many were detected only once or at concentrations below criteria and in concentrations reported as estimated below the method detection limit. To identify contaminants for evaluation during the nature and extent analysis, a three-part screening methodology was employed. This methodology is described in the following paragraphs.

First, a statistical comparison was made to determine which contaminants were present at statistically greater concentrations in WAG 6 samples as compared to reference samples. (The reference sampling program is described in Sect. 2.) Any contaminant present at a statistically greater concentration in the on-WAG samples for a given medium was included for consideration in the nature and extent analyses. The statistical comparison was limited to those sampling locations and samples used in the human risk assessment (Sect. 6). For on-WAG, this included 34 of 45 monitoring wells, all of the base flow surface water results, all of the shallow (0 to 6 ft.) soil samples, and sediment samples from the EWB. The monitoring wells not included were the four deep wells (no contaminants detected), one well where no sampling was performed, three shallow wells which were always dry, one well north of the EWB, and two wells east of the branch of WOC along the eastern side of WAG 6 meant to detect cross-stream movement of contaminants. The statistical analysis is described further below.

Second, because the statistical comparison did not include all data collected for WAG 6, an additional comparison was made to ensure that no contaminants that should be evaluated for nature and extent were excluded by the statistical comparison. This comparison included all data collected for WAG 6 and involved a simple comparison of the maximum concentration for each contaminant with the maximum reference value for that contaminant. If the maximum concentration was more than one order of magnitude greater than the maximum reference value for a given contaminant, that contaminant was included for consideration in the nature and extent analysis.

Third, various environmental criteria and health-based, criteria-derived concentrations were compared. (The criteria used were explained previously in this section and are included in Tables 4.2 through 4.7). A contaminant was included in the group to be evaluated during the nature and extent analysis if its maximum concentration was greater than any of the criteria or criteria-derived values. Table 4.8 summarizes the results of the contaminant screening for WAG 6 groundwater, surface water, soils, and sediments.

During the nature and extent analysis, inorganics were further screened by removing from the evaluation those constituents that are essential human nutrients (Table 4.8). The naturally occurring radionuclides were retained in the group because concentrations detected in the environment are often greater than the health-based, criteria-derived concentrations. However, because only thorium-230 is statistically greater than reference, discussion of these constituents will be limited in the nature and extent evaluation.

The chemicals detected in each of the four environmental media were evaluated to determine their distribution (i.e., normal, log-normal).. Contaminant concentrations were generally found not to be normally distributed. The data evaluation also indicated that the occurrences of VOCs, SVOCs, and inorganic compounds and herbicides at detected concentrations on WAG 6 are sporadic.

These results directed the selection of the statistical method used to test for differences between reference and WAG 6 chemical concentration levels. Chemicals found to have statistically significant differences between the site and reference concentrations were included in the list of contaminants for the nature and extent evaluation and the risk assessment.

In cases where data are not normally distributed, alternative estimators and tests are based on rank statistics. "Rank" (or "order" or "nonparametric") statistics use the rank of an observed concentration rather than the reported value. The rank is defined as the number associated with the observation when all measurements have ordered in increasing value. For example, in the following sequence of ordered observations, the ranks range from 1 to 7, where 7 equals the number of observations:

Concentration:	1.5	3.7	8.1	9.7	29.3	37.9	55.6
Rank:	1	2	3	4	5	6	7

Because nonparametric tests rely on ranks rather than absolute values, no assumptions about the distribution of the underlying observations are made. This feature makes nonparametric methods extremely useful where data are not normally distributed.

The Mann-Whitney test, a nonparametric statistical test, was used to compare WAG 6 data against reference data. The Mann-Whitney test relies on the "rank" (or "order") of a set of concentrations for a specific analyte. Specifically, the Mann-Whitney test compares the WAG 6 and reference data and determines if the ranks represented by one of the data sets is significantly higher than the ranks found in the other set. Statistical significance, it is assumed, indicates that the contaminant is present on WAG 6 at concentrations elevated above reference concentrations. The level of significance (alpha) of the test was 0.05. Alpha is normally fixed at 0.05 per the *Risk Assessment Guidance for Superfund* (EPA 1989b). This means that there is a 5% (1 in 20) chance that it will be concluded that concentrations of contaminants are higher than reference when they actually are not (a false positive result). The chemicals not found to be of statistically significant difference were not automatically excluded from the evaluation for nature and extent and the risk assessment, but they, in some cases, were subjected to a more detailed evaluation (see Sects. 4 and 6).

The Mann-Whitney test consists of assigning the ranks across all observations, summing the ranks in the two groups, and testing the sum against a critical value to determine the probability that the rank sums from the two groups could have occurred strictly by chance. For example, in the seven observations shown above, if the first four concentrations (representing ranks 1 through 4) had come from reference well samples and the largest three concentrations (ranks 5 through 7) had come from WAG 6 samples, the Mann-Whitney test would have concluded that reference and WAG 6 sample results were statistically significantly different at a probability level of 0.034. That is, the ranks indicated from the samples would have arisen in the samples from two populations which are not significantly different only 3.4 out of 100 times.

While the Mann-Whitney test is a reliable alternative to the t-test for non-normally distributed data, the test is sensitive to "tied ranks." When ties (multiple observations at the same concentration) occur, ranks are found by assigning the average rank to all of the values. For example, in the following, the ranks range from 1 to 7, but because six of the values occurred at 5, the six lowest values share the average rank (1 + 2 + 3 + 4 + 5 + 6) / 6 = 3.5, and the maximum value carries the rank of 7:

Concentration:	5	5	5	5	5	5	550
Rank:	3.5	3.5	3.5	3.5	3.5	3.5	7

The extent to which ties may bias the results from the test depends on the number of observations from the two sampled populations. The effect of ties increases with increasing difference in the number of samples collected in the two areas. The Mann-Whitney test statistic can be adjusted for the number of ties in the data set. Adjustments for ties have been made where necessary.

Specific sample locations used in the Mann-Whitney test are indicated on Table 4.9. Tables 4.10 through 4.13 present the Mann-Whitney test results from the comparisons of reference and WAG 6 samples for groundwater, surface water, soil, and sediment chemicals, respectively. The chemicals evaluated for nature and extent and included in the risk assessment from this list have an alpha of less than 0.05.

The resulting VOCs, SVOCs, and inorganic analytes and radiological constituents evaluated for nature and extent are presented in Table 4.14. Most of the contaminants listed in Table 4.14 are discussed in the remainder of this section. Those not discussed were found to be of little significance to the nature and extent discussion.

The base line human health evaluation (Sect. 6) and the base line environmental evaluation (Sect. 7) produced their own lists of contaminants for consideration in their respective analyses. These are presented in the respective sections.

4.1.3 General Observations Pertaining to Nature and Extent

During compilation and analysis of site data to develop an understanding of the nature and extent of contamination, certain fundamental observations regarding the data were noted. These observations and general comments regarding data interpretation are described here to provide a framework for understanding data presented in subsequent sections.

4.1.3.1 On-WAG contamination and off-WAG migration

Environmental sampling during the WAG 6 RFI focused on assessing both on-WAG contamination and potential off-WAG contaminant migration at the WAG 6 boundary. The characterization of the nature and extent of on-WAG contamination provides the basis for the assessment of human health and environmental impacts to on-WAG receptors. Additionally, it supports scoping of alternatives for closure and corrective action by delineating primary release areas and areas where contaminant concentrations may exceed action levels.

Characterization of off-WAG migration of contaminants provides the basis for assessment of the contribution of contamination from WAG 6 to adjacent WAG 2 (WOC drainage basin) and allows an assessment of potential human health and environmental impacts to receptors located at the WAG 6 boundary and beyond. In addition, description of the off-WAG migration of contaminants supports regulatory compliance program monitoring.

Data collected during the WAG 6 RFI and previous investigations clearly indicate the release of contaminants from source areas to the environment on-WAG and the subsequent migration of some of these contaminants off-WAG.

4.1.3.2 Vertical distribution of contamination

The data indicate that there is very little radiological contamination of surface soils at WAG 6. The radiation walkover survey conducted for SWSA 6 and the EWB areas indicated only isolated surface soil "hot spots," defined as radiation readings three times above background levels. (The on-WAG 6 "hot spots" were removed as part of site preparation activities for construction of the ICM caps; the EWB hot spots were flagged by survey personnel and the areas were roped off by ORNL ES&H personnel.) The observed absence of radiological contamination of the ground surface at WAG 6 is consistent with site waste disposal methods and practices. That is, wastes were disposed below ground level in trenches and auger holes in a controlled manner and subsequently covered with a soil cover several feet thick.

The primary zone of contamination at WAG 6, therefore, is the zone below the surface soil cover (i.e., below about 3 ft), including the regolith in both the saturated and unsaturated zones and the shallow bedrock—primarily at locations downgradient to waste disposal areas. Numerous groundwater samples and subsurface soil samples indicate radiological and chemical contamination in these zones. There are discernible areal patterns to the contamination, as discussed subsequently. A significant observation regarding subsurface contamination is the decrease in contamination with depth. This trend is most apparent in groundwater, as exemplified in Fig. 4.1, which plots tritium concentrations with depth.

(Tritium is by far the most ubiquitous and mobile contaminant at WAG 6.) This trend is consistent with the understanding of the WAG 6 hydrogeologic system, which is described in Sect. 3 and in which active groundwater flow is local and restricted to the uppermost saturated interval in regolith and bedrock.

4.1.3.3 Variation of groundwater contaminant concentrations with time

As described in Sect. 2, the majority of WAG 6 groundwater monitoring wells have been sampled at least four times. Reviews of analytical results for these wells have identified no clearly defined temporal trends or relationships to physical variations in the hydrogeologic system (such as seasonal and episodic water level fluctuations, increased leaching associated with precipitation, etc.). Rather, contaminant concentrations show an apparently random fluctuation from sampling event to sampling event.

The lack of temporal trends and patterns is consistent with the conclusions drawn by Solomon et al. (1988), who noted that, upon repeated sampling of leachate from selected trenches, there were large and erratic variations in radionuclide concentrations within a given trench. It is likely that the variations noted in trench leachate concentrations are related to the sporadic breakdown of waste disposal containers and varying rates of water infiltration into the trenches.

4.1.3.4 Delineation of contaminant plumes

A contaminant plume is a discrete zone of contamination that typically results from the flow of subsurface water through a source area. Contaminants are transported from the source in the direction of subsurface flow; the resulting zone of subsurface contamination—the contaminant plume—develops a geometry that is controlled by site-specific conditions including, most importantly, the site hydrogeologic regime.

Contaminant data collected during the RFI and presented in this section indicate widespread contamination of shallow groundwater on-WAG, generally at low concentrations. Within this shallow groundwater system, there are discernible areas where particular contaminants predominate and there are areas of relatively elevated contaminant concentrations.

As described in Sect. 3, the hydrogeologic framework of WAG 6 is complex. Groundwater flow is local, with short flow paths from recharge points in higher elevation areas to discharge points along site streams and low areas or to WOL. Groundwater flow is controlled by a combination of prevailing hydraulic gradient and primary or relict fractures in the bedrock and regolith. Groundwater flows along fractures from areas of higher head to areas of lower head. Fracture orientations and density are variable both laterally and vertically at WAG 6. This results in tortuous flow paths (stair-stepping in three dimensions), thereby precluding prediction of groundwater flow directions and rates with any certainty on a local (i.e., point-to-point) basis. Further, the complexity imposed by the fracture network results in highly discrete contaminant transport along fractures that cannot be predicted from well to well. Consequently, detection of contamination in a given monitoring well (regardless of screen length) is extremely fortuitous and probably does not correlate to adjacent wells that may or may not have intercepted the same fracture network.

Therefore, precise delineation of plumes is not feasible; however, areas of concern have been identified. Should collection of contaminated groundwater at WAG 6 be required, it should be possible to use available data to develop a remedial design that accommodates potential deviations from expected conditions.

4.1.3.5 Intermedia transfer of contaminants

An understanding of intermedia transfer of site contaminants (e.g., transfer of contaminants from leachate to groundwater to surface water) is a prerequisite for the effective identification, definition, and evaluation of remedial alternatives. Considerable discussion related to this topic and based upon data gathered during the RFI is presented in the analysis of contaminant fate and transport (Sect. 5). The primary interactions discussed are the leaching of contaminants from buried wastes to groundwater; the discharge of contaminated groundwater to surface water; and the transfer of contaminants between surface water and sediments.

4.2 SOURCE AREAS

Appendix 1A presents information regarding WAG 6 source areas, including information on waste disposal units and locations, waste types and packaging, and available waste inventory data. As a precursor to discussion of site contamination, the following paragraphs recap key information and concepts regarding source areas.

The ORNL Waste Disposal Log documents the radionuclide contents of each disposal trench and auger hole; however, the activities entered into the log generally represent estimated values only. The waste disposal log indicates that, as of December 1990, greater than 85% of the waste disposed of at SWSA 6 was disposed of in auger holes. Consequently, from a radiological perspective, SWSA 6 may be viewed as consisting of several small areas of high-activity waste (primarily auger holes), which are located in a much larger area containing low-activity waste (primarily trenches) (ORNL 1986). Figure 4.2 illustrates the major classifications of radiological waste disposal areas and repeats the labeling of waste disposal areas and drainages employed in this report.

Chemical wastes are known to have been disposed of both in auger holes and trenches. Oils, cleaning solutions, alcohols, paint thinners, kerosene, jet fuel, acids, sodium, and miscellaneous solvents and other chemicals were disposed of in a set of auger holes classified as "solvent auger holes" (ORNL 1986). Some chemical wastes were poured into the auger holes and others were placed in the auger holes in containers ranging from small bottles to 55-gal drums. The exact nature and volume of waste disposed in each solvent auger hole is unknown.

Limited waste disposal records indicate that biological trenches were known to have received waste scintillation fluids, which are composed of various aromatic hydrocarbons. Records indicate that from 1972 through May 1983, approximately 48,000 liters of scintillation fluids were disposed. Thirty-five biological trenches are identified in waste disposal records as having received scintillation fluids. Figure 4.3 illustrates areas where chemical wastes are known to have been disposed.

Trench leachates sampling has been conducted in past environmental sampling to aid in chemical and radiological characterization of source areas. Previous investigations conducted at WAG 6 that involved trench leachate sampling are summarized in Appendix 1B. Key previous investigations that produced radiological data relevant to characterization of source term concentrations are summarized below.

- Tamura et al. (1980) sampled leachate from 26 trenches and analyzed the leachate for strontium-90, cobalt-60, and cesium-137.
- Solomon et al. (1988) sampled leachate from eight trenches and analyzed the samples for tritium, strontium-90, cobalt-60, and cesium-137.
- Davis et al. (1989) collected soil samples from five trenches located in the northeastern portion of WAG 6 (19 Trench area). Soils were analyzed for cobalt-60, cesium-137, and strontium-90.
- Ashwood and Spalding (1990) collected trench leachate samples from 26 intratrench piezometers in the eight ICM capped areas. Samples were analyzed for gross alpha, gross beta, tritium, strontium-90, cesium-137, and cobalt-60.

Past environmental sampling to aid in chemical characterization of source areas has also been limited primarily to sampling of trench leachates or trench bottom soils.

- Solomon et al. (1988) sampled leachate from eight trenches and analyzed the samples for the EPA TCL. Ten VOCs and six SVOCs were identified.
- Ashwood and Spalding (1990) analyzed samples collected from 26 intratrench piezometers for TCL VOCs and SVOCs and TAL metal...
- Davis et al. (1989) analyzed four trench bottom soil samples collected in the 19 Trench area for VOCs and SVOCs.

Trench leachate and trench soil sampling results are discussed where appropriate in the remainder of this section. Trench leachate concentrations should be interpreted with the following caution: trench leachate data collected by Solomon et al. (1988) showed large and erratic variations in radionuclide concentrations with changing hydraulic conditions. Furthermore, the data indicated no correlation between the concentrations detected in trench leachates and the concentrations reported for those trenches in the ORNL waste disposal log. The variability of trench leachate data with time, combined with the large number of trenches at WAG 6, led Solomon et al. (1988) to conclude that trench leachate sampling was not a reliable method of characterizing source term for modeling.

Release rates will likely increase or decrease unpredictably in the future for different contaminants in different areas of SWSA 6. These variations would correspond to the unpredictable rates at which different types of waste packages deteriorate. Based on ORNL experience at SWSA 4, it may be anticipated that, on a site-wide basis, WAG 6 environmental contamination will continue to increase in the near term (assuming no closure or corrective action). In the very long term (i.e., hundreds of years), however, radioactive decay and chemical decomposition will result in lowered concentrations.

4.3 NATURE AND EXTENT OF ON-WAG CONTAMINATION BY MEDIUM

This section presents findings related to the investigation of the nature and extent of radionuclide and chemical contamination in WAG 6 groundwater, surface water, sediments, and soil. At the end of the discussion for each medium, a summary presents the major findings for that medium.

4.3.1 Groundwater

4.3.1.1 Previous investigations

Since 1976, there have been numerous groundwater investigations at WAG 6. Table 4.15 lists WAG 6 groundwater investigators, the wells that were involved, and the dates the investigations occurred. Table 4.16 presents the constituents that were analyzed for during each of the studies. Appendix 1B summarizes the scope of previous groundwater studies and presents data collected during the investigations.

4.3.1.2 Radiological contamination

Both man-made and naturally occurring radionuclides have been detected in groundwater samples from wells at WAG 6. Detected man-made radionuclides include tritium, cobalt-60, strontium-90, cesium-137, and transuranics, which include americium-241, curium-244, plutonium-238, and plutonium-239/240. Naturally occurring radionuclides detected in groundwater samples at WAG 6 include potassium-40 and radionuclides belonging to the uranium series (uranium-238, thorium-234, uranium-234, thorium-230, radium-226) and the thorium series (thorium-232, radium-228, thorium-228, radium-224).

Concentration standards for drinking water for types of radioactivity (alpha or beta radiation) as well as for several specific radionuclides have been established by the Safe Drinking Water Act (52 FR 25690 1987). MCLs have been established for gross alpha (15 pCi/L), tritium (20,000 pCi/L), strontium-90 (8 pCi/L), radium (5 pCi/L, includes sum of radium-226 and radium-228 concentrations), and man-made beta (4 mrem per year). For other man-made radionuclides, a 4-mrem-per-year MCL exists. Naturally occurring radionuclides that do not have MCLs promulgated have been compared to background concentrations to determine if detected concentrations are indicative of contamination. [The MCL for radium is currently under review at EPA. Consideration is being given to raising the MCL to 40 pCi/L for the sum of radium-226 and radium-228 (20 pCi/L each)].

4.3.1.2.1 Nature of radiological contamination

Gross alpha and gross beta. Gross alpha and gross beta activity was detected in all wells sampled. Both naturally occurring and man-made radionuclides, including transuranic radionuclides, contribute to either gross alpha and/or gross beta concentrations. Figures 4.4 through 4.7 show the variability in gross alpha and gross beta concentrations detected in WAG 6 RFI and ORNL RCRA compliance wells. (For ease of display, ORNL RCRA results are presented separately from RFI results. This is the case for other radionuclides, also.)

Gross alpha concentrations analyzed in groundwater samples from ORNL RCRA compliance wells did not exceed 15 pCi/L except in two wells (wells 832 and 857). In these two cases, the error terms associated with these measurements exceeded the measurement and for practical purposes can be considered non-detected measurements.

Although not pervasive, gross alpha concentrations exceeding 15 pCi/L were detected in groundwater samples collected during ORNL RCRA monitoring and the RFI. In some cases, naturally occurring radionuclides accounted for the gross concentrations exceeding 15 pCi/L. These are discussed more fully below (see **Naturally occurring radionuclides**).

Man-made radionuclides. Tritium was detected in a majority of groundwater samples collected at the site (Figs. 4.8 and 4.9) and exceeded the 20,000 pCi/L MCL in numerous wells. Concentrations ranged from non-detectable levels to over 4,000,000 pCi/L. Cesium-137, cobalt-60, and strontium-90 also have been detected in groundwater samples at WAG 6, but none of these radionuclides are widely distributed in groundwater.

The Safe Drinking Water Act MCL for strontium-90 in groundwater is 8 pCi/L. Other an-made, beta-emitting radionuclides are limited in acceptable concentrations equal to a se of 4 mrem per year. To determine if cobalt-60 or cesium-137 concentrations detected [illegible] the site exceeded the 4 mrem per year dose level, a concentration expressed in pCi/L equal [illegible] a dose of 4 mrem per year was calculated, assuming drinking 2 liters of water each day [illegible] one year (EPA 1989b). Concentrations in pCi/L that equal 4 mrem/year exposure for cobalt-60 and cesium-137 were calculated to be 203.3 pCi/L and 109.6 pCi/L, respectively. Maximum detected concentration for cesium-137 did not exceed the 4 mrem per year MCL. The maximum detected concentration for cobalt-60 exceeded the 4 mrem per year MCL in samples from only one well, Well 842, and was exceeded consistently throughout the well's sampling history. The 8 pCi/L MCL for strontium-90, including total radioactive strontium, was exceeded in six wells at the site (846, 848, 850, 858, 1225, and 1237). However, in wells 846, 850, and 858, only the first sampling event exhibited concentrations exceeding 8 pCi/L. In subsequent sampling events, strontium-90 was not detected, suggesting that previous results were anomalous.

The range in concentrations observed for cesium-137, cobalt-60, and total strontium (including strontium-90) in groundwater samples has been observed to vary over time. This variation is displayed in Figs. 4.10 through 4.16). While the range in variability is not as great for total strontium, the apparent variability present in cobalt-60 and cesium-137 concentrations is due primarily to concentrations that also have large error terms—most of which were collected during the January/February 1989 sampling period. In most instances, the associated error terms for cobalt-60 and cesium-137 during that sampling period indicate overlapping concentration ranges with lower concentrations, which suggests the elevated concentrations may not be an accurate indication of conditions actually present in these groundwater samples.

Transuranic radionuclides. An analysis of the nature and extent of transuranic radionuclides in groundwater at WAG 6 is based on analyses conducted in RFI wells and RCRA compliance wells sampled by the RFI team during this investigation only (see Sect. 2). Not all ORNL RCRA compliance wells were sampled for transuranic radionuclides; therefore, their extent at WAG 6 may be underestimated. Transuranics detected include americium-241, curium-244, plutonium-238, and plutonium-239/240.

Naturally occurring radionuclides (gross alpha and radium). Groundwater samples from ORNL RCRA compliance wells were not analyzed for naturally occurring radionuclides except for total radium concentrations, which includes the sum of radium-224 and radium-226. Total radium concentrations do not display large variations by well or among wells (Fig. 4.16). Total radium (radium-224 and radium-226) concentrations reported in ORNL RCRA monitoring data, did not exceed the 5 pCi/L MCL; however, when radium-226 and radium-228 were analyzed for during RFI sampling the 5 pCi/L MCL was exceeded in some wells.

Naturally occurring radionuclides of the uranium and thorium series, which are primarily alpha emitters, were analyzed for to determine if gross alpha concentrations exceeding 15 pCi/L could be attributed to the presence of naturally occurring radionuclides. In some instances, the higher gross alpha concentrations could be attributed to radium-224 which was detected at concentrations approaching that of the gross alpha concentrations. In other instances in which gross alpha concentrations exceeded 15 pCi/L, radionuclides analyzed for were not detected in sufficient concentrations to account for the increased alpha activity. These occurrences are discussed below.

4.3.1.2.2 Extent of radiological contamination

Naturally occurring radionuclides (gross alpha and radium). Monitoring wells in which maximum detected gross alpha concentrations exceed 15 pCi/L are presented on Fig. 4.17. The wells in which naturally occurring radionuclides account for concentrations exceeding 15 pCi/L are also presented but are not considered to be contaminated by site operations.

The wells in which gross alpha concentrations have not been accounted for by naturally occurring radionuclides are wells at the high-activity and low-activity trench areas of the central waste disposal area, and wells adjacent to the biological trench area of the west waste disposal area (Fig. 4.17). Except for two wells (Wells 1228 and 849), man-made transuranic radionuclides have been detected in groundwater samples from these wells. It is not known if additional man-made radionuclides are present or that naturally occurring radionuclides attribute to higher alpha concentrations.

Figure 4.17 shows wells in which the sum of radium-226 and radium-228 exceeded 5 pCi/L. These are Well 1241 located south of the EWB, Well 1225 located west of the high activity trench area of the central waste disposal area, Well 1233 located west of the 49 Trench area, and Well 1229 located east of the biological trenches of the west waste disposal area. Except for radium-228, which was detected at 860 pCi/L in Well 1225, radium-226 concentrations in the other wells exceeded 5 pCi/L. Radium-228 was not detected or was very close to the detection limit (3 pCi/L) in the other wells. Highest radium-226 concentration was 113 pCi/L in Well 1129. Concentration levels in other wells were below 17.3 pCi/L.

Man-made radionuclides—tritium. Tritium has been analyzed for and detected in numerous groundwater samples during ORNL RCRA compliance monitoring, the RFI monitoring, and in numerous independent studies conducted by ORNL researchers such as Poreda, Cerling, and Soloman 1988; Boegly 1984; Doyle and Taylor 1986; Davis et al. 1989; Solomon et al. 1988; Yager and Craig 1989; Ashwood 1989; Vaughan et al. 1982; Dreier and Toran 1989; and Morrissey 1990—to name a few. Tritium is the most widely distributed man-made radionuclide at the site.

The Safe Drinking Water Act maximum concentration level (MCL) for tritium is 20,000 pCi/L. Figure 4.18 presents analytical results on tritium levels. The maximum concentration detected at any location is represented by a dot whose relative size indicates a concentration range (see figure legend). The smallest dot corresponds to non-detected levels or to detected levels below the 20,000 pCi/L MCL.

Tritium has been detected above the MCL in groundwater samples from wells adjacent to nearly all waste disposal areas at WAG 6. Groundwater samples from up-gradient wells on the northwest perimeter of the site and from wells located at the southwest section have not contained concentrations exceeding the MCL. There are also wells located in the interior of WAG 6 in which elevated concentrations of tritium have not been detected.

Highest concentrations of tritium appear to be associated with the high- and low-activity trench areas; tritium levels exceeding the MCL have also been detected in groundwater samples adjacent to biological trenches. Maximum detected levels of tritium in groundwater samples from wells adjacent to biological trench areas have not exceeded 100,000 pCi/L. Tritium concentrations exceeding 100,000 pCi/L have been detected in groundwater samples in wells adjacent to the high-activity trench areas of the central waste disposal area, the 19 Trench area, the northern half of the east disposal area, the low-activity trench area of the central waste disposal area, the low-activity trench area of the west waste disposal area, and the Tumulus area (Fig. 4.18). Tritium concentration levels exceeding 1,000,000 pCi/L have been detected in groundwater samples from wells located at the northern-most trench areas and the low-level trenches of the central waste disposal area.

Tritium has been detected in groundwater samples from the TARA-series wells, in the northeastern part of the WAG at concentrations up to 7,290,000 pCi/L (Davis et al. 1989). This area also has the highest potential for migration of tritium off-WAG. Davis et al. (1989) sampled the TARA-series wells at the 19 Trench area and identified a tritium plume emanating from this area. ORNL RCRA compliance wells in the area confirm the elevated concentrations. Highest detected concentrations occur nearest to the trench area and to the southeast toward Drainage B. Concentrations generally decrease to the north and northwest towards the EWB, although concentrations exceeding 100,000 pCi/L have been detected in surface water samples collected from the EWB.

The northeast auger hole area of the east waste disposal area also contains levels of tritium above the MCL. A tritium plume appears to be migrating easterly-southeasterly towards the unnamed tributary. Wells to the south of the auger hole area generally contain low levels of tritium. Tritium from the auger hole area could be reaching groundwater in Well 1244, located approximately 300 ft east and separated by the unnamed tributary. This well may also be receiving tritium from adjacent WAG 7 located to the east of the well. Well 1244 has been sampled once, and 51,200 pCi/L tritium was detected.

Groundwater samples collected from some wells at the low-activity trench areas in the central part of WAG 6 also contain levels of tritium exceeding 100,000 pCi/L. Two wells (Wells 848 and 849) at the low-level trenches of the central waste disposal area contain levels of tritium exceeding 1,000,000 pCi/L. The high levels observed at the 49 Trench area of the central waste disposal have not been detected in wells located to the west across Drainage FB, except for Well 854, which probably intercepts high levels of tritium from the low-activity trenches of the west waste disposal area. Surface water samples collected adjacent to the 49 Trench area from Drainage FB have had tritium concentrations detected in excess of 400,000 pCi/L. Water collected from the outfall of the French Drain have contained the highest concentration detected on site (14,000,000 pCi/L) strongly suggesting that the 49 Trench area is a major source area of tritium.

Independent studies conducted by Yager and Craig (1989) and Ashwood (1989) at the Tumulus area have identified high levels of tritium associated with Well 1036 (Fig. 4.18). Increasing concentrations of tritium have been detected in groundwater samples from Well 1036, which is located on the eastern side of the Tumulus area. From 11 samples collected between June 1987 and May 1990, tritium concentrations have increased from a low of 486 pCi/L to a high of 226,800 pCi/L. Reportedly, tritium has not been disposed of in this area. All the other wells except for Well 1039, which is located to the west along strike to Well 1036, have had concentrations below 11,000 pCi/L with no discernible trends. Concentration levels have been increasing in Well 1039 also, but to date have remained below the MCL. A possible source of the tritium could be the outfall from the French Drain area located northeast of the Tumulus area. Because none of the Tumulus wells south of Well 1036 have displayed increasing trends in tritium concentration, it appears that along-strike migration may be a primary pathway.

On occasion, tritium has been detected in groundwater samples from wells located along the southeastern and southern boundary of WAG 6 at concentrations exceeding 20,000 pCi/L. The monitoring network in this area presently suggests that this area is not receiving releases of tritium in excess of the MCL.

Man-made radionuclides—cesium-137, cobalt-60, and strontium-90. Cesium-137 has been detected in groundwater samples from four wells at WAG 6. These are wells 846, 856, 836, and 860 (Fig. 4.19). Wells 846 and 856 are reference wells located upgradient from waste disposal sites. Well 836 is located at the southern waste disposal area, and Well 860 is located in the southwest corner of WAG 6 over 700 ft west of the nearest waste disposal area. Except for Well 836, cesium-137 was detected in the first sampling event of these wells, but subsequent sampling events have not detected cesium-137. Cesium-137 was detected in a third sampling event of Well 836, but previous and subsequent sampling events did not detect cesium-137. Figure 4.10 (ORNL RCRA monitoring results) and Fig. 4.11 (RFI results) illustrate the range in variation of cesium-137 analyses. Cesium-137 was not detected in RFI monitoring wells.

Concentrations of cobalt-60 and strontium-90 detected in groundwater samples from a few wells were significantly above the level at which the majority of other sample concentrations were detected (Figs. 4.12 through 4.15). The associated error terms for the higher concentrations indicate a high degree of certainty for the detection.

Occurrences of cobalt-60 and strontium-90 are limited to the north and northeast sections of WAG 6. In particular, the high-activity trench area and the eastern half of the central waste disposal area, the 19 Trench area, and the central part of the east waste disposal area contain wells in which strontium-90 and cobalt-60 have been detected in groundwater samples (Fig. 4.19).

Cobalt-60 distribution in groundwater is presently confined to the vicinity of the 19 Trench area and the central part of the east waste disposal area (Fig. 4.19). Concentrations detected in Well 843, located to the east of the 19 Trench area, have been marginally elevated from what has typically been observed at the site overall (Figure 4.12). Data collected by Davis et al. (1989) from TARA wells located adjacent to the site indicate low levels of cobalt-60, below the magnitude of that detected in Well 843 (Fig. 4.20). This may reflect a temporal difference for the TARA data, which were collected only once in 1987. Groundwater samples from adjacent Well 844, which monitors a zone deeper than Well 843, have not had cobalt-60 detected. Migration is most likely southerly and easterly towards discharge points along Drainage B, and possibly off-WAG through the groundwater system.

Cobalt-60 concentrations are highest along the eastern boundary of WAG 6, adjacent to the east waste disposal area (having been detected in wells 842 and 1243—Fig. 4.19). It is doubtful that cobalt-60 detected in groundwater samples from this area originated from the 19 Trench area, because of the distance from the 19 Trench area to monitoring wells at the east waste disposal area and because Drainage B separates the two areas. Therefore, it appears that the auger hole area of the eastern waste disposal area is a separate source. Cobalt-60 occurrence is presently confined to the upper part of the groundwater system; it has not been detected in deeper Well 841, which is adjacent to Well 842. Cobalt-60 was not detected in wells located south of Well 1243 (Fig. 4.21). Migration of cobalt-60 contaminated groundwater off-WAG 6 to the east is likely.

Strontium-90 has been detected in groundwater samples from three wells (1225, 1237, and 848). Groundwater samples from Well 1225, located on the western boundary of the high-activity trench area of the central waste disposal area (Fig. 4.19), has had the highest concentrations of strontium detected (up to 5400 pCi/L). Strontium-90 was not detected in

Well 847, located to the east of the high-activity trench area. Downgradient wells to the south are possibly too deep to detect any possible plume moving in that direction (Fig. 4.22).

Well 848 is located south of the low-activity trenches of the eastern part of the central waste disposal area. Strontium-90 has been detected in adjacent Drainage DB to the east. Fig. 4.23 illustrates strontium-90 occurrences with depth in this area.

Strontium-90 has also been detected in Well 1237, located upgradient to the west waste disposal area. This well has been sampled only once; wells paired with 1239 have not had strontium-90 detected.

Transuranic radionuclides. Transuranics, consisting of americium-241, curium-244, plutonium-238, and plutonium-239/240, were detected in groundwater samples from five wells located at the high-activity trench area and the 49 Trench area of the central waste disposal area; the west waste disposal area (in both upgradient and downgradient wells at the low-activity trenches the biological trench area), and the auger hole area of the east waste disposal area (Fig. 4.24). Concentrations are generally low in most wells (below 3 pCi/L) but are higher in Well 1225, where americium-241 was detected at 23.9 pCi/L. Americium-241 occurrences in groundwater samples from wells 1227, 1233, 1237, and 1242 are questionable. Previous analytical results from wells 1227 and 1233 indicated americium-241 not present above the detection limit of 1 pCi/L. Wells 1237 and 1242 have been sampled only once. Well 1237 is also located upgradient to any waste disposal area. Plutonium-238 was detected in groundwater samples from wells 1225 and 1229, and curium-244 was detected in Well 1225.

Relatively higher concentrations of transuranic radionuclides were detected in groundwater samples from Well 1225, located on the western perimeter of the high-activity trench area (Fig. 4.24). Strontium-90 and tritium exceeding MCL concentrations have also been detected in groundwater samples from this well. Americium-241, curium-244, plutonium-238, and plutonium-239/240 were detected. Gross alpha concentrations observed in groundwater samples from adjacent Well 847, located on the eastern side of the high-activity trench area, suggest that transuranics, if present, do not contribute significantly to alpha activity in the groundwater (maximum detected gross alpha in Well 847 was 10.8 pCi/L, which is within background concentration ranges).

4.3.1.3 Chemical contamination

4.3.1.3.1 Nature of chemical contamination

This section is subdivided into a discussion of the inorganic and organic constituents analyzed in groundwater. These constituents include metals, VOCs, SVOCs, pesticides, PCBs, organophosphorus pesticides, dioxins, furans, and herbicides. Of the organic compounds detected, the VOCs comprise the largest group, and consequently much of the discussion is devoted to them. Of the inorganic compounds, the only group discussed are the metals. Miscellaneous inorganic parameters, such as pH, conductivity, etc., are not addressed as they are indicators of natural groundwater geochemistry.

Volatile organic compounds. VOCs were detected in a majority of groundwater samples collected from ORNL RCRA compliance and RFI monitoring wells at WAG 6. Some occurrences can be attributed to common laboratory contaminants or field sampling induced contamination and have been eliminated from further consideration. Common laboratory compounds have been identified in disposal trench leachate samples at WAG 6 (Soloman 1989) and may have entered the groundwater system. Therefore, when these compounds have been detected and cannot be identified as induced contamination, their presence in the groundwater is credible.

The VOCs detected in groundwater samples collected from ORNL RCRA compliance and RFI monitoring wells included two TICs that are components of scintillation fluids disposed of at the site. Numerous other TICs have been detected and have undergone data validation procedures; most have been identified as laboratory-induced contamination (Appendix 4A) and are subsequently not discussed.

Table 4.2 lists the VOCs detected in groundwater at the site. The compounds most frequently occurring by well include tetrachloroethene (12 wells), 1,2-dichloroethene (10 wells), trichloroethene (10 wells), chloroform (10 wells), methylene chloride (8 wells), and benzene (6 wells). All other VOCs were detected in five wells or less.

Of the VOCs detected, MCLs exist for eight (Table 4.17). Six compounds were detected that exceed MCLs; these are (the first number in parenthesis following the VOC is the MCL for that compound, the second number is the maximum detected concentration in μg/L): trichloroethene (5/1300), trichloromethane (100/190), benzene (5/110), vinyl chloride (2/88), 1,2-dichloroethane (5/44), and carbon tetrachloride (5/96). Compounds detected for which MCLs are defined but will not become effective until January 30, 1992, include toluene, ethyl benzene, 1,2-dichloroethene, and tetrachloroethene. Concentrations of 1,2-dichloroethene and tetrachloroethene have exceeded the to-be-effective MCLs. Ethyl benzene has been detected at its MCL.

VOC concentrations ranged from a low of 0.4 μg/L 1,1,1-trichloroethane in Well 843 to a high of 6800 μg/L tetrachloroethene in Well 1233. The maximum detected concentrations for all compounds detected in any single well, including results from all sampling events, have been summed to present a "worst case" condition for that well (Table 4.17). Comparisons with other wells are made using this value because all wells were not sampled concurrently, and in some instances data are available from five to six sampling events for a well, and no concentration trends are apparent in the data.

The wells with the greatest summed VOC concentrations (as presented in Fig. 4.25) are: Well 1233 (8326 μg/L) located just west of the central waste disposal area; Well 849 (1661 μg/L) located south of Well 1233 at the central waste disposal area; Well 850 (6778 μg/L) located east of the biological trench area of the east waste disposal area; Well 842 (808.5 μg/L) located east of the auger hole area of the east waste disposal area; and Well 648 (4545 μg/L) located south of the high activity trench area of the central waste disposal area. All other summed VOC concentrations did not exceed 400 μg/L.

Semivolatile organic compounds. SVOCs were detected in groundwater samples from wells analyzed during detection/compliance monitoring and the RFI. Many of these compounds can be attributed to contamination through the sampling and/or analysis procedures—having been detected in various blanks. Several SVOCs were detected that could not be directly attributable to the sampling or analysis procedures but which are common laboratory and environmental contaminants. These compounds are bis(2-ethylhexyl)phthalate, di-n-butyl phthalate, and diethyl phthalate. Phthalates are plasticizers that have been identified as common laboratory contaminants (EPA 1990). The maximum detected concentration level for any of the phthalates was estimated at 7 μg/L. SVOCs present in groundwater at the site include 2-methyl phenol, dimethyl benzene, and naphthalene. Their occurrence is limited to wells 852 and 1228, located at the biological trench area of the west waste disposal area, and Well 852, located southwest of the biological trench area of the central waste disposal area. Maximum detected concentrations ranged from an estimated low of 1 μg/L for naphthalene in Well 852 to a high of 1000 μg/L for naphthalene in Well 850. Maximum summed SVOC concentrations ranged from an estimated low of 7 μg/L in Well 1228 to a high of 1022 μg/L in Well 850.

Other chemical compounds. The herbicides 2,4,5-TP (Silvex), 2,4,5-T, and 2,4-D have been detected in groundwater samples during ORNL RCRA compliance sampling. The introduction of herbicides to groundwater at WAG 6 has been attributed to localized land applications prior to installation of caps on the wells (ORNL 1989).

Except for two occurrences of the PCB arochlor 1254, at levels estimated at 1 μg/L or lower, no other PCBs were detected at the site. Compounds analyzed for but not detected at the site include dioxins, furans, and organophosphorus pesticides.

Metals. Metals occur naturally in the groundwater environment and were detected in every groundwater sample analyzed for metals. To determine if any metals were present in concentrations indicative of releases to the environment or in concentrations harmful to human health and the environment, comparisons were made to reference conditions at the site and to other criteria, such as the Safe Drinking Water Act MCLs. For the purposes of this study, the only metals of concern are metals for which MCLs or RFI guidance health-based criteria exist. After these comparisons were made, it was determined that three metals are present in groundwater at levels of concern: barium, cadmium, and lead. The maximum concentrations detected for these metals are as follows (the first number in parenthesis represents the MCL reported in micrograms per liter, the second is maximum concentration): barium (1000/27,100), cadmium (10/94.5), lead (50/100). Arsenic, mercury, and silver were also detected above their respective MCLs; however, detections greater than MCLs were sporadic and were localized. All other metals were detected at levels that did not significantly exceed background concentrations (generally less than 1 order of magnitude) or did not exceed MCLs.

4.3.1.3.2 Extent of chemical contamination

The discussion on extent of chemical contamination in groundwater at WAG 6 is presented by analyte type. First is a discussion of the organic chemicals, of which the VOCs constitute the major group. The VOC discussion is followed by discussion of SVOCs, herbicides, and metals.

Volatile organic compounds. Groundwater monitoring wells in which VOCs were detected are presented in Fig. 4.25, which shows a graphical representation (rose diagram) of the various VOCs detected and their maximum concentration. Each segment of the rose diagram represents a distinct VOC; the distance radially represents the maximum concentration observed for that constituent for that well. The concentration scale is logarithmic.

Specific VOCs plotted on the rose diagram are: 1,2-dichloroethene, trichloroethene, tetrachloroethene, chloroform, total xylenes, toluene, benzene, and acetone. The primary VOCs were chosen based on frequency of occurrence. An additional segment includes a sum of the remaining constituents detected in that well and is labeled as "others." These other compounds include: 1,1-dichloroethene, 1,1-dichloroethane, 1,2-dichloroethane, carbon tetrachloride, methylene chloride, 1,1,1-trichloroethane, vinyl chloride, ethyl benzene, dichloromethane, 2-butanone, trichloromethane, carbon disulfide, 1,1,2,2-tetrachloromethane, 1,1,2-trichloroethane, and ethyl chloride, including ethyl ether, diethyl ether, and 1,4-dioxane, which are TICs. The VOCs have been arranged to discern distribution patterns. Generally, chlorinated alkenes occupy the right side of the rose; aromatics occupy the left. The order of arrangement of VOCs is consistent from one rose to the other.

Releases of VOCs from some of the areas have VOC suites distinctly different from those of other areas. Scintillation fluids (commonly various aromatic hydrocarbons) have been disposed in biological trenches, and typical components of these fluids have been detected in groundwater in the vicinity of the biological trenches (located in the southern sections of the west waste disposal area and the central waste disposal area). On the other hand, the low- and high-activity trench areas and the auger hole areas contain a different suite of VOCs, dominated by the chlorinated alkenes trichloroethene, tetrachloroethene, and 1,2-dichloroethene. Some aromatic compounds are present at auger hole areas.

The discussion on extent of VOC contamination is presented by proximity to drainage features, generally from an upgradient to downgradient position at the site. The discussion starts with the northernmost waste disposal unit of the central waste disposal area (the high-activity trench area) and proceeds southward along the FB drainage to the south waste disposal area. The discussion continues with the northeastern half of the WAG (the 19 Trench area) southward across Drainages B and C to the east waste disposal area (Fig. 4.25).

- **Drainage FB—north high activity trench area.** The high-activity trench area is the northernmost waste disposal area adjacent to drainage FB. Here, VOCs have been detected in monitoring wells located to the west (Well 1225) and to the south (Well 648) (Fig. 4.25). Ten different VOCs were detected in Well 648, which is downgradient from a field of solvent auger holes. Concentrations ranged from 6 μg/L xylenes to 2323 μg/L trichloroethene (Solomon et al. 1988). MCLs for 1,1-dichloroethene, 1,2-dichloroethane, trichloroethene, and benzene were exceeded. Groundwater from Well 648 may be influenced by materials released from the high-activity trench area in addition to the releases from the solvent auger holes. Only three VOCs (tetrachloroethene, 1,2-dichloroethene, and acetone) were detected in Well 1225. The highest concentration detected was 120 μg/L of acetone.

Known vertical distribution of VOCs in this area is presented in Figs. 4.26 and 4.27. Although VOCs have not been detected to the east of the high-activity trench area in Well 847, this well is screened in the shallow bedrock approximately 15 ft below the base of Well 1225 and does not intersect the water table (Fig. 4.26). Its screen may be too deep to intercept VOCs migrating to the east. The VOC 4-methyl-2-pentanone has been detected in upgradient Well 846; however, the concentration at which it was detected, 2 μg/L, is very low and has not been detected elsewhere on WAG 6.

A cluster of three wells (HHMS-7A, HHMS-7B, and HHMS-7C) are south of the high-activity trench area and downgradient, but these have not been sampled for VOCs (Fig. 4.27). VOCs are known to occur at least as deep as Well 648 extends, which is 45 ft below the surface (778 ft above mean sea level). Based on the understanding of site hydrogeology (described in Sect. 3), groundwater is expected to flow towards Drainage FB. Surface water samples collected southwest of the solvent auger hole area in Drainage FB have had 1,2-dichloroethene, tetrachloroethene, and trichloroethene detected (Sect. 4.3.2.3).

- **Drainage FB—49 Trench area.** VOCs have been detected in groundwater samples from wells located at the 49 Trench area of the central waste disposal area. Maximum concentration levels ranged from a low of 1 μg/L for chloroform, 1,1-dichloroethene, trichloromethane, and ethyl chloride to a high of 6800 μg/L tetrachloroethene. MCLs were exceeded for trichloroethene and vinyl chloride in Wells 849 and 1233. Well 1232 has been dry since installation. The MCL for benzene was exceeded in Well 849. MCLs that become effective January 30, 1992, for tetrachloroethene and 1,2-dichloroethene were also exceeded.

  Lateral extent of VOCs detected in these wells appears to be bounded by Drainage FB. Surface water samples collected adjacent to Well 1233 have had several VOCs detected. Wells on the western side of drainage FB, while having VOCs detected in groundwater samples from them, contain different suites of VOCs except for the occurrence of trichloroethene at very low levels.

  Eastern migration of VOCs from the 49 Trench area in groundwater is most likely intercepted by the French Drain. Water samples from the outfall of the French Drain have contained relatively high concentrations of similar VOCs detected in upgradient wells and surface water locations (Sect. 4.3.3.3). Known vertical distribution of VOCs at the 49 Trench area is presented in Figs. 4.28 and 4.29.

- **Drainage FB—west waste disposal area.** The west waste disposal area can be divided into the northern section, consisting of low-activity trenches, and a southern section consisting of biological trenches (Fig. 4.25). Five ORNL RCRA compliance and RFI monitoring wells are adjacent to the low-activity trenches. Wells 1226, 1237, and 1238 are located upgradient. Well 1226 has been dry since installation. Except for carbon disulfide, which was detected once at 22 μg/L in Well 1237, no other VOCs were detected in these wells. Downgradient to the low activity trenches, Wells 853 and 854 have had no other VOCs detected in groundwater samples, except for one occurrence of methylene chloride in each well at 25 and 20 μg/L, respectively.

Analyses of groundwater samples collected from wells 851, 852, 1228, and 1229 indicate that the biological trenches of the west waste disposal area have released components of scintillation fluids (benzene, toluene, xylenes, 1,4-dioxane, and diethyl ether) along with acetone and minor amounts of 1,2-dichloroethene, tetrachloroethene, chloroform, methylene chloride, vinyl chloride, and possibly 2-butanone (Fig. 4.25). The greater concentrations were detected in Well 852 (393 μg/L maximum summed VOC concentration) and Well 1228 (294 μg/L maximum summed VOC concentration) located at the southwestern tip of the biological trench area.

Low concentrations of VOCs have been detected in groundwater samples from wells 851 and 1229, which are located in close proximity to each other. Well 851 contained the scintillation fluid components 1,4-dioxane (160 μg/L) and ethyl ether (8 μg/L), while 1229 contained tetrachloroethene and possibly 2-butanone, each at 2 μg/L (which is below sample quantification limits for these compounds). Of the VOCs detected, only benzene in Well 852 exceeded any primary MCL (detected at 110 μg/L). Benzene was also detected in Well 1228 at a maximum concentration of 1 μg/L. Groundwater samples from monitoring wells 745 and 833, located approximately 300 ft south of wells 1228 and 852, do not appear to be affected by VOC releases from the biological trench area. Except for a one-time detection of 1,1-dichloroethene in Well 833 at an estimated level of 1 μg/L (Fig. 4.30), no other VOCs have been detected. However, these wells are completed in bedrock and not the saturated overburden. They are positioned on the west side of the FA drainage system, possibly precluding them from intercepting any VOC plume migrating southward from the west waste disposal area. It is probable that VOCs are migrating towards the FB or FA drainage systems, which surface water sampling results suggest.

- **South waste disposal area.** The south waste disposal area consists of biological trenches to the west and low-level trenches to the east (Fig. 4.25). This waste disposal area is located near the southern boundary of WAG 6 just north of WOL. Three compliance wells (835, 836, and 837) monitor groundwater conditions in the unconsolidated zone. Groundwater samples from Well 837 have not contained detectable concentrations of VOCs. Groundwater samples from Well 835 have contained levels of trichloroethene and benzene below the MCL in addition to other components of scintillation fluids (1,4-dioxane and ethyl ether). Well 836 has contained low to moderate concentrations of chloroform and tetrachloroethene, which on one occasion was detected at 5 μg/L, equal to the MCL effective January 30, 1992 (Fig. 4.31). Trench leachate analysis has indicated the presence of various VOCs, of which components of scintillation fluids were detected at highest concentrations (i.e., xylenes 3696 μg/L, toluene 2,500 μg/L). No primary MCLs have been exceeded. Considering the wells' proximity to the WAG boundary, it is probable that VOCs are migrating off-WAG.

- **Southwestern area of WAG 6.** VOCs have been detected sporadically and at low concentrations in Well 859 in the extreme southwest corner of WAG 6 (Fig. 4.25). Well 859 is completed in the overburden and is paired with 860, which monitors the bedrock. Ethyl benzene, toluene, and xylenes—the typical suite of VOCs associated with biological trench leachate—were detected occasionally during January and May 1989. Occurrences of these volatiles were not repeated in a sampling event that occurred one year later (May 1990).

- **19 Trench area.** The 19 Trench area is located in the northeastern part of WAG 6 on high ground between two small drainages: Drainage A to the northwest and Drainage B to the southeast (Fig. 4.25). In this area, relatively low levels of VOCs have been detected in groundwater samples from compliance wells 843 and 844. Between June 1988 and May 1990, ORNL sampled and analyzed groundwater from these wells five times. Maximum summed VOC concentrations were 20.9 μg/L for Well 843 and 14 μg/L for Well 844. No primary MCLs were exceeded in these wells.

  Low levels (4 to 11 μg/L) of 1,2-dichloroethene were detected in shallow Well 843 for each sampling event except the last (in May 1990). In April 1989, four other VOCs were detected in Well 843, but at levels below the sample quantification limit. Results from the May 1990 sampling indicate a presence of low levels of chlorinated organics in deeper Well 844 for the first time. VOCs have not been detected north of the 19 Trench area in Well 1241. Three sides of the 19 Trench area are bounded by streams, and it is likely that contaminated groundwater from the area flows toward these drainages and discharges groundwater to them. Samples from RFI monitoring Well 1242, located approximately 300 ft southwest of Wells 843 and 844 and across a drainage divide (Drainage B), did not reveal detectable levels of VOCs the one time it was sampled (Fig. 4.32).

- **East waste disposal area.** South of the 19 Trench area is the east waste disposal area, which is bounded by Drainage B to the north, Drainage DB to the southwest, and the unnamed tributary to the east (Fig. 4.25). This waste disposal area is divided into a northern section that consists primarily of high-activity auger holes (including a group of solvent auger hole areas) and a southern section consisting of asbestos waste trenches. Separating the two major areas is a narrow zone of biological trenches and additional auger holes.

  As shown in Fig. 4.32, VOCs at elevated concentrations have been detected in groundwater samples from several compliance monitoring wells located immediately downslope of the solvent auger hole area. Summed maximum VOC concentration is greatest in shallow Well 842 (808.5 μg/L), compared with 54 μg/L in the deeper adjacent Well 841. Both of these wells are completed in bedrock. MCLs for carbon tetrachloride, 1,2-dichloroethane, and trichloroethene have been exceeded in Well 842. The MCL for trichloroethene was also exceeded in Well 841. The MCL for tetrachloroethene, effective January 30, 1992, was exceeded in both wells. Well 841 exhibits a subset of the suite of VOCs detected in Well 842 and generally in lesser concentrations. Well 845, located immediately southwest of the high-activity auger hole area, has shown low levels of 1,2-dichloroethene in groundwater samples. Wells to the south and east of the waste disposal area (1243, 1249, 1234, and 840), have not exhibited VOCs in groundwater samples except for Well 840, in which trichloroethene was detected once in five sampling events at a level of 1 μg/L and chloroform at 2 μg/L (Fig. 4.32).

It appears that the plume emanating from the solvent auger hole area is migrating southeastward toward WOC. Off-WAG migration to the east of Well 841 and 842 is likely considering the length of time VOCs have been detected in these wells (over 2 years) and the concentrations at which they have been detected. Wells 1244 and 1245 were installed east of the solvent auger hole area on the eastern side of the unnamed tributary (Fig. 4.25) to aid in defining the extent of VOC contamination. With the exception of chloroform, detected at an estimated 3 μg/L in Well 1244, no other VOCs have been detected. VOC-contaminated groundwater is likely discharging to the unnamed tributary in this area.

- **Central waste disposal area—eastern section.** Varying concentrations of VOCs have been detected in RCRA compliance wells located at the eastern part of the central waste disposal area east of the 49 Trench area (Fig. 4.25). This area consists primarily of low-level trenches with an area of biological trenches to the south, followed by a thin zone of more low-activity trenches. This area is bounded to the west by the French Drain, to the east by Drainage DB, and to the south by Drainage DA. Near the northern low-level trenches, low concentrations of VOCs consisting of acetone, 1,1-dichloroethane, 1,2-dichloroethene, 1,1,1-trichloroethene, and 1,4-dioxane have been detected in Well 848, which monitors the bedrock/unconsolidated zone. Maximum summed VOC concentration was 59.7 μg/L. It is possible that the VOC plume coming from these trenches could be discharging to Drainage DB to the east, the French Drain to the west, or may be attenuated by the time water from this reaches other monitoring wells. Figure 4.33 illustrates the known vertical distribution of VOCs in the low-activity trench area.

  VOCs have been detected at high concentrations (up to 3800 μg/L) in ORNL RCRA compliance Well 850, located southeast of the biological trenches of the central waste disposal area. Groundwater samples collected from Well 850 contain typical components of scintillation fluids (toluene, xylenes, ethylbenzene, and benzene) with occurrences of chloroform, acetone, and trichloroethene. The second highest summed total VOC concentration for the site (6778 μg/L) was obtained from this well. MCLs for benzene and trichloroethene were exceeded. VOCs that exceeded MCLs that become effective January 30, 1992, include chloroform, ethyl benzene, and toluene.

  Wells 1231, 838, and 839 are located downgradient of the biological trenches and the thin zone of low-level trenches. VOCs have been detected in each of these wells. VOCs detected in Well 1231, located due south of the biological trenches, include trichloroethene and tetrachloroethene. The MCL for trichloroethene has been exceeded in groundwater samples from this well. Tetrachloroethene was also detected at levels greater than the MCL effective January 30, 1992, in Well 1231. However, the components of scintillation fluid that were detected in upgradient Well 850 have not been detected in Well 1231. It is possible that outfall from the French Drain (located to the west) is the source of VOC contamination in this well.

Wells 838 and 839 are southeast of the biological trenches and south of the low-activity trenches. Relatively low concentrations (15 μg/L or less) of VOCs (chloromethane and tetrachloroethene) have been detected (Fig. 4.34). If migration of VOCs in this region is primarily along strike, as may be indicated by the presence of VOCs in Well 850 and a different suite of VOCs in Well 1231, discharge of contaminated groundwater may be to Drainage DB. This creek is dry, however, for most of the year. During these times, discharge may occur to WOC, which is off-WAG.

Semivolatile organic compounds. SVOC contamination of the groundwater is not extensive at WAG 6. The occurrences of SVOCs have been associated with releases of VOCs and are apparently confined to the biological trench areas of the west waste disposal area and the central waste disposal area. Groundwater samples from Wells 1228 and 852, located at the biological trenches of the west waste disposal area, have contained low concentrations of dimethyl benzene and naphthalene. Concentration levels have been below detection limits and are therefore estimated values. Groundwater samples from Well 850, located east of the biological trenches of the central waste disposal area, contain higher concentrations of SVOCs. Naphthalene is the most prevalent, having been detected during several sampling events. The maximum detected concentration of naphthalene, for which there is no MCL, was 1000 μg/L. Other SVOCs detected include 2-methyl phenol and total recoverable phenolics, seen at levels below 30 μg/L. SVOCs as a group are generally less mobile than VOCs, and therefore probably have not migrated far from source areas.

Herbicides. Groundwater samples from 16 wells have contained detectable concentrations of herbicides (Table 4.18). The herbicide 2,4-D was detected most frequently in these wells; concentrations ranged from 0.3 to 12 μg/L. The herbicides 2,4,5-T and 2,4,5-TP were detected in one and four wells, respectively. Concentrations did not exceed any MCLs for any of the herbicides detected.

Metals. Occurrences of metals in groundwater samples from WAG 6 in concentrations exceeding established MCLs, as determined by total metal analyses, are not extensive and are confined to eight wells adjacent to four waste disposal areas (Fig. 4.35): (1) the high-activity trench area of the central waste disposal area, (2) the 49 Trench area of the central waste disposal area, (3) the biological trench area of the west waste disposal area, and (4) the biological trench area of the east disposal area. Lead was detected at levels exceeding the MCL in one groundwater sample from Well 857, located in the westernmost section of WAG 6 and not associated with any known waste disposal areas.

Barium has been detected exceeding 1000 μg/L at all the above-mentioned sites except the high-activity trench area. It appears to be associated with possible releases from the biological trench areas of these sites. Lead and cadmium have also been detected in these disposal sites. Mercury and silver were detected above their respective MCLs at one location each.

Comparison of corresponding filtered metals analyses to the unfiltered results and with other results collected during the RFI or RCRA compliance monitoring suggest that some unfiltered (total) results are anomalous and may be a result of analyzing turbid samples (Table 4.19). Only barium, in Well 852, has consistently exceeded the MCL in both filtered and unfiltered analyses. Filtered results for barium exceeding the MCL have also been detected in wells 850, 1228, and 1233; however, other results have not, generally suggesting anomalous results. During one sampling event of five, a filtered cadmium result exceeded the MCL in Well 1225. For all other corresponding filtered results, the MCL was not exceeded.

4.3.1.4 Summary of groundwater contamination

Radionuclides. Both man-made and naturally occurring radionuclides were detected in groundwater samples at WAG 6. Of these radionuclides, tritium was the most prevalent. Except for wells located along the western boundary, tritium has been detected at levels exceeding the 20,000 pCi/L MCL in groundwater samples from wells adjacent to all major waste disposal areas. Concentrations have exceeded 1,000,000 pCi/L in a few wells. The highest concentration detected has been over 4,000,000 pCi/L. Off-WAG migration of tritium in the groundwater system is probable because this radionuclide has been detected in close proximity to WAG 6 boundaries. Tritium is most likely migrating off-WAG along the northern and eastern boundaries.

Additional man-made radionuclides detected in WAG 6 groundwater samples include cobalt-60, strontium-90, cesium-137, and the transuranic radionuclides americium-241, curium-244, plutonium-238, and plutonium-239/240. These radionuclides have been detected at relatively low concentrations (typically less than 20 pCi/L) compared to tritium, and have not been detected in many groundwater samples. Waste disposal areas in the northeastern section of WAG 6 appear to have released cobalt-60 to the groundwater system. Releases of strontium-90 have been detected in groundwater samples from the northern-most waste disposal area. There have been other isolated occurrences of strontium-90 and cesium-137; however, detection of these radionuclides has been sporadic. All four transuranic radionuclides have been detected in groundwater samples from one well located in the northwestern part of WAG 6. Americium-241 has been detected in four other on-WAG wells but in low concentrations (2.39 pCi/L or less).

Chemicals. Organic compounds, both VOCs and SVOCs, have been detected in groundwater samples from WAG 6. SDWA MCLs for [illegible] VOCs have been exceeded in various wells. The distribution of VOCs is similar to that of tritium, having been detected in wells adjacent to all major waste disposal areas. Highest concentrations of VOCs have been associated with solvent auger hole areas and biological trench areas. VOCs have also been detected in groundwater samples from some upgradient wells; however, concentrations have been low (22 μg/L or less), are usually estimated values below 5 μg/L, and in two instances have been common laboratory contaminants. Few SVOCs have been detected (limited to three compounds), and these have been detected in groundwater samples in which VOCs have also been detected. Off-WAG migration of organic compounds in groundwater is likely along the northern, eastern, and southeastern boundaries of WAG 6.

Metals have been detected exceeding various criteria in a limited number of wells at WAG 6. Of these metals, lead and barium are the most widely detected. Wells in which metals concentrations exceeded various criteria are located in the interior of WAG 6. It appears that migration of metals in the groundwater system has not advanced far beyond the boundaries of the waste disposal areas near which they were detected.

4.3.2 Surface Water

Approximately 70% of the surface area of WAG 6 lies within four main basin areas with associated intermittent streams (FA, FB, DA, and DB, as shown in Fig. 4.36). In addition, three small, ephemeral drainages (A, B, and C) drain the northeastern portion of the site. All of the surface water runoff eventually flows off WAG 6 to WOC and WOL.

As described in Sect. 2 during the WAG 6 RFI, surface water grab samples were collected from all drainages in WAG 6 in six separate sampling events over an 8-month period in 1989. In addition, the outfall from the 49 Trench area French Drain and the EWB were sampled. Table 4.20 summarizes these sampling events. Samples were generally analyzed for radionuclides, VOCs, SVOCs, and metals.

Continuous surface water sampling was performed during two storm events in April-May 1990 at flumes constructed in Drainages DA, DB, FA, and FB (Fig. 4.36). The purpose of this sampling was to characterize contaminant flux off-WAG and for calibration of the surface water and subsurface water contaminant transport model. This sampling effort and its results are discussed in greater detail in Sect. 5.

4.3.2.1 Previous investigations

This section provides an overview of previous investigations conducted at WAG 6 that yielded surface water quality analytical data. The actual data and sampling locations are presented in Appendix 1B. Table 4.21 lists WAG 6 surface water investigations, the locations sampled, and the constituents that were analyzed for during each of the studies. As appropriate, specific data from these investigations are integrated into the discussion on the nature and extent of contamination.

Surface water samples were collected from Drainages FAA and FAB between October 1980 and March 1983 by Davis et al. (1984) as part of the performance assessment of selected site characterization techniques. Samples were analyzed for tritium, cobalt-60, strontium-90, and cesium-137.

Between February and April 1987, pre-ICM, storm flow surface water samples were collected by Davis et al. (1987) from Drainages DA, DB, FA, and FB. These samples were analyzed for tritium.

During a performance assessment of the 49 Trench area French Drain, conducted by Davis and Marshall (1988), samples were collected from the southern outflow of the French

Drain and from a seep adjacent to the 49 Trench area, in Drainage FB. Samples were analyzed for tritium, cobalt-60, strontium-90, and cesium-137.

Miller, Black, and Craig (1989) performed an extensive surface water sampling effort for major drainages in WAG 6. Samples were collected from October 1988 to June 1989 and analyzed for tritium, cobalt-60, strontium-90, and cesium-137.

4.3.2.2 Radiological contamination in surface water

4.3.2.2.1 Nature of radiological contamination

The radionuclides detected in surface water samples from WAG 6 are listed in Table 4.6. Tritium and strontium-90 were detected more frequently and at higher levels than any other radionuclide. Tritium and strontium-90 occurred at levels above MCLs at 12 of 15 and 5 of 12 locations sampled, respectively. The highest detected level of tritium was 14,000,000 pCi/L, which occurred at the outfall of the 49 Trench area French Drain. All sampling locations in Drainage FB had strontium-90 levels over the MCL. The highest level of strontium-90 in WAG 6 surface water, 10,222 pCi/L, occurred at the headwaters of Drainage FB. The distribution of tritium, cobalt-60, and strontium-90 is shown in Fig. 4.37. The transuranic radionuclides plutonium-239/240, curium-242, and curium-244 have also been detected.

4.3.2.2.2 Extent of radiological contamination

Drainages A and B. Radionuclide concentrations in Drainages A and B are low compared to concentrations elsewhere at WAG 6. Tritium concentrations in these drainages ranged from 2550 pCi/L to 30,000 pCi/L—approximately 1% to 10% of levels in other surface water samples from WAG 6. The MCL for tritium (20,000 pCi/L) was exceeded only in Drainage B. Strontium-90 was detected at 11.2 pCi/L in one sample from Drainage A. However, this was the only time strontium-90 or any other radionuclide other than tritium was detected in either of these drainages.

Emergency Waste Basin. Both times the EWB was sampled during the RFI, tritium was detected in surface water at concentrations exceeding the MCL. Concentrations were 190,000 pCi/L and 160,000 pCi/L. Strontium-90 was also detected (7.5 and 5.1 pCi/L); however, these concentrations are below the strontium-90 MCL of 8 pCi/L. Surface water sampled below the EWB in Drainage A indicated tritium concentrations below the MCL; strontium-90 was not detected.

Drainage C. Only one sample was collected from Drainage C, this during a storm flow event when the water table was high. The drainage was dry during all other sampling events. Tritium was detected above the MCL at 190,000 pCi/L. Wells adjacent to Drainage C have also contained comparable levels of tritium exceeding the MCL (Fig. 4.18). No strontium-90 was detected in this sample; however, cobalt-60 was detected at 550 pCi/L—the only time cobalt-60 was detected in any WAG 6 surface water sample. Cobalt-60 was also detected in Well 842 (267-351 pCi/L) and Well 1243 (1880 pCi/L) located to the north and south, respectively, of Drainage C (Fig. 4.19). Groundwater in this area flows in a south/southeasterly direction; therefore, the most probable source of the cobalt-60 is the northeast auger hole area, which is known to contain cobalt-60.

Drainage DA. Surface water samples were collected at sampling point WDA1 in Drainage DA (Fig. 4.37). Samples were also collected at the effluent point, W49TS, for the 49 Trench area French Drain that flows into Drainage DA. Tritium was detected at concentrations of 4,270,000 pCi/L to 14,000,000 pCi/L from samples collected from the 49 Trench area French Drain. These levels greatly exceed the MCL. Davis and Marshall (1988) also detected comparable concentrations of tritium from samples collected at the outfall. Similar concentrations of tritium were detected at sampling station WDA1 in Drainage DA. Well 1231, located east of the outfall of the French Drain (Fig. 4.18), has shown tritium concentrations exceeding the MCL (51,000-83,000 pCi/L). Comparable concentrations near WDA1 were also found by Davis et al. (1987) and Miller, Black, and Craig (1989). The tritium levels found during the RFI at the WDA1 location were approximately an order of magnitude below the levels found at the W49TS location.

Strontium-90 was detected only once in four sampling events at both the French Drain outfall and in Drainage DA samples. Concentrations have not exceeded the MCL. Davis and Marshall (1988) have detected strontium-90 in French Drain outfall samples.

It appears that the 49 Trench area French Drain is providing a direct conduit for the migration of contaminants leaching from the low-activity trenches of the central waste disposal area to the DA drainage, WOC, and WOL.

Drainage DB. This drainage is frequently dry; therefore, few samples were collected during the RFI. Surface water samples collected during a high groundwater storm event contained tritium at a concentration of 73,000 pCi/L, which exceeds the MCL (Fig. 4.37). Strontium-90, detected at 6 pCi/L, was below the MCL. Strontium-90 was also detected in Well 848 located west of WDB2 adjacent to the central waste disposal area (Fig. 4.19).

Drainage FA. Drainages FAA and FAB combine to form Drainage FA. Drainage FAA has had enough flow to sample only during periods of high groundwater after storm events. Its one sampling location was located near the confluence of drainages FAA and FAB. Tritium was detected at 1,500 pCi/L—which is well below the MCL. Plutonium-239/240 was detected in a surface water sample collected from WFAB2 at a concentration of 1.1 pCi/L.

Tritium concentrations in Drainage FAB are about two orders of magnitude higher than in FAA near the meeting point of the two drainages (WFAB1) (Fig. 4.37). Davis et al. (1984) also found the same relationship of tritium concentrations in these two drainages. Tritium levels near the head of Drainage FAB (WFAB2) are low and did not exceed 19.7 pCi/L. This fact indicates that FAB is gaining tritium-contaminated groundwater between WFAB2 and WFAB1. The source of the tritium contamination is most likely the west waste disposal area. This is supported by the fact that the wells directly downgradient of the west waste disposal area also have elevated levels of tritium (Fig. 4.18).

Samples collected beyond the point where drainages FAA and FAB meet to form Drainage FA (WFBA1 sampling location) indicate concentrations of tritium that are slightly less than those seen upstream at WFAB1 (Fig. 4.37). However, the concentration levels still exceed the MCL.

Drainage FB. Samples from surface water sampling point WFBB3, located at the headwaters of Drainage FB and west of a high-activity trench area, have had concentrations of tritium and strontium-90 exceeding both MCLs. Tritium was detected in concentrations up to 190,000 pCi/L. Strontium-90 concentrations were as high as 10,222 pCi/L. Well 1225 is located between WFBB3 and the high-activity trench area and shows tritium and strontium-90 (Fig. 4.19) at similar concentrations. Curium-242 and curium-244 were detected at WFBB3 at 7.88 and 134.7 pCi/g, respectively. These radionuclides were not detected in any downstream sampling locations. Curium-244 was also detected in Well 1225. It appears that radionuclides leaching from the high-activity trench area are being discharged to Drainage FB in the vicinity of WFBB3.

Although the results from downstream sampling location WFBB2 show lower levels of tritium and strontium-90 than found at WFBB3, levels still exceed the MCL. Tritium levels at this sampling point increase generally by a factor of 2 to 3 times from those detected in upstream location WFBB2—which implies a new source area. Groundwater wells in the vicinity of WSP1, located east toward the 49 Trench area (Fig. 4.18) have been shown to contain tritium in concentrations up to 770,000 pCi/L. Strontium-90 concentrations at WSP1 are not as high as upstream concentrations, possibly suggesting that no additional sources are originating from the 49 Trench area. Drainage FB appears to be losing water from sampling point WFBB2 to downstream location WSP1. This loss is evident because surface water levels have not been adequate to collect samples at WSP1 during any of the base flow sampling events; however, water levels were adequate to collect a sample at the WFBB2 point that lies approximately 350 ft upstream from WSP1.

Sampling point WFBB1 has consistently shown order of magnitude higher concentrations of tritium (up to 3,910,000 pCi/L) when compared with upstream samples. Strontium-90 levels have remained fairly consistent with upstream levels. This difference in tritium concentrations appears to be due to a seep directly west of Well 849. Sampling of Well 849 have confirmed comparable levels of tritium in the groundwater. This area is underlain by the geologic contact between the Nolichucky Shale and Maryville Limestone, which may provide a conduit for east/west groundwater flow instead of the southerly flow that might be expected. The most probable source of the tritium contamination is the 49 Trench area. Sampling of a seep adjacent to the 49 Trench area was performed by Davis and Marshall (1988). Results of this sampling indicated tritium and strontium-90 entering Drainage FB from this seep. The Miller, Black, and Craig (1989) study confirmed tritium entering Drainage FB in this area.

4.3.2.3 Chemical contamination in surface water

4.3.2.3.1 Nature of chemical contamination

VOCs were detected in surface water samples collected from WAG 6 (Table 4.2). Trichloroethene, 1,2-dichloroethene, and tetrachloroethene represent 98% of the VOCs detected. The distribution of these VOCs is shown in Fig. 4.38. Drinking water MCLs were exceeded by trichloroethene, 1,2-dichloroethene, tetrachloroethene (MCL effective 1/30/92), and vinyl chloride.

SVOCs were detected in very few surface water samples (Table 4.2). Most of the SVOCs were detected at extremely low levels and were either phthalates or phthalate derivatives, both of which are common laboratory contaminants.

Results from metals analyses were compared to the following water quality criteria: MCLs found in the SDWA, RFI Guidance Health Based Criteria, and Tennessee Water Quality Control Board standards. Results from only two surface water samples exceeded water quality limits for metals. The barium MCL of 1000 μg/L was exceeded by a 2400 μg/L concentration level at WFAB1 in Drainage FAB. The cadmium MCL of 10 μg/L was exceeded by a factor of three from a sample collected at WFAA1 in Drainage FAA. Concentrations of these metals did not exceed MCLs during any other sampling events at these locations. Therefore, the high concentrations are believed to be outliers and do not accurately represent metal concentrations at these locations.

4.3.2.3.2 Extent of chemical contamination

Drainages A, B, C, and DB. No VOCs were detected in samples from Drainages A, B, C, and DB, with the exception of 2 μg/L of trichloroethene detected in one sample from SWA1 in the A drainage. Therefore, none of these drainages currently represent a significant pathway for off-WAG migration of VOCs.

Emergency Waste Basin. VOCs were not detected in samples collected from the EWB, except for 2-hexanone, which was detected once at an estimated 1 μg/L.

Drainage DA. Surface water samples collected at sampling point WDA1 indicated virtually no VOCs. However, upstream samples collected at W49TS, the outfall of the 49 Trench area French Drain, showed highly elevated levels of VOCs. As pointed out in Sect. 4.3.2.2.2, dilution appears to occur between W49TS and WDA1. A combination of dilution and air stripping would account for VOCs not appearing in the WDA1 samples.

Surface water samples collected from Drainage DA contained few VOCs (chloroform at 15 μg/L and 2-butanone at 2 μg/L) and one SVOC, bis(2-ethylhexyl)phthalate, detected at 3 μg/L. These concentrations are low and all three compounds are common laboratory contaminants.

In contrast to samples from Drainage DA, samples collected from the outfall of the 49 Trench area French Drain have contained various VOCs. VOCs detected that have exceeded MCLs include 1,2-dichloroethene and trichloroethene. The MCL for tetrachloroethene (5 μg/L) will be effective January 30, 1992, and it was exceeded in samples from the outfall of the French Drain. Although 1,1-dichloroethane and 1,2-dichloroethane were detected, they were present in relatively lower concentrations.

Drainage FA. Tetrachloroethene, trichloroethene, and 1,2 dichloroethene were detected at sampling point WFAA1 in Drainage FAA. The MCL for trichloroethene was equalled in the one sampling event of Drainage FAA. The concentration at which tetrachloroethene was detected (70 μg/L) exceeds the MCL that will be effective in 1992. Except for tetrachloroethene, which was detected once at an estimated concentration of 1 g/L, no VOCs have been detected in Drainage FAB.

Although VOCs were detected in Drainage FA, none were detected upstream in samples from FAA and FAB. The compounds—acetone, toluene, 2-butanone, methylisobutyl ketone, and 1,1,2,2-tetrachloroethane—all were detected at estimated concentrations. Except for 1,1,2,2-tetrachloroethane, all other compounds are common laboratory contaminants.

Drainage FB. Drainage FB exhibits the greatest diversity in VOCs of all drainages at WAG 6, and the MCLs for trichloroethene and vinyl chloride were exceeded. The MCLs for 1,2-dichloroethene and tetrachloroethene (effective January 30, 1992) were also exceeded at various sampling points along the drainage. The MCL for trichloroethene was equalled at the most downstream sampling point.

Upstream sampling point WFBB3 contained the VOCs trichloroethene and 1,2-dichloroethene in relatively low concentrations compared with elsewhere in the drainage. Downstream sampling locations WFBB2 and WSP1 each contained trichloroethene, 1,2-dichloroethene, and tetrachloroethene, all in excess of their respective MCLs. Vinyl chloride, chlorobenzene, toluene, benzene, and 1,1-dichloroethene also were detected. The MCL for vinyl chloride was exceeded at sampling location WPBB2. Vinyl chloride was detected at a maximum concentration of 11 μg/L. Downstream sampling point WFBB1 contained trichloroethene, 1,2-dichloroethene, and tetrachloroethene, and exhibited one detect of 2-butanone.

Surface water sampling point WFBB2 is located in a position that may be intercepting water that has flowed through the solvent auger hole area. Elevated levels of chlorinated hydrocarbons and radionuclides were detected during several sampling events. A comparison between points WFBB2 and (upstream location) WFBB3 (Fig. 4.39) indicates a net loss of radionuclides from WFBB3 to WFBB2 and a net gain of chlorinated hydrocarbons. Well 1225 has a high concentration of radionuclides and a low concentration of organics and Well 648 displays the opposite concentrations of contaminants. One can assume that the surface water, as it moves away from its radionuclide source near WFBB3 and toward a primarily organic source near Well 648, should show lower values of radionuclides and higher values of organics (contamination originating from one source is diluted and the contamination from the other source begins to contribute).

Once the VOCs enter Drainage FB between WFBB3 and WFBB2, concentrations downstream decrease. This indicates that there are not any other VOC sources contaminating Drainage FB.

4.3.2.4 Summary of surface water contamination

Radionuclides. Both man-made and naturally occurring radionuclides have been detected in surface water samples from WAG 6. Naturally occurring radionuclides do not exceed reference (background) concentrations. Of the man-made radionuclides detected, tritium and strontium-90 are the most prevalent and have been detected at concentrations exceeding their respective MCLs. The highest concentration of tritium detected on-WAG (14,000,000 pCi/L) was collected from the outfall of the 49 Trench area French Drain. Tritium has been detected in all drainages at WAG 6, and in the majority of samples the MCL has been exceeded.

Cobalt-60 was detected only once—in a surface water sample from Drainage C, located in the northeastern section of WAG 6 adjacent to where cobalt-60 was detected in groundwater and soil samples. The concentration at which cobalt-60 was detected exceeded the MCL. Strontium-90 was detected at concentrations exceeding the MCL in several drainages at WAG 6. Highest concentrations were detected in surface water samples collected from Drainage FB, which flows nearly the whole length of WAG 6 from north to south. Strontium-90 was also detected in groundwater from Well 1225 located near the headwaters of Drainage FB. Transuranic radionuclides were detected, but in only two samples.

Chemicals. VOCs and SVOCs were detected in surface water samples from WAG 6. MCLs were exceeded for some VOCs. Of the VOCs detected the most common were 1,2-dichloroethene, tetrachloroethene, and trichloroethene. Concentrations were generally less than 200 μg/L, except in samples collected from the outfall of the 49 Trench area French Drain, where tetrachloroethene was detected at 2,200 μg/L. Organic compounds detected in downstream samples were generally lower in concentration than upstream samples.

The SVOCs that were detected were generally at low concentrations and are common laboratory contaminants. The SVOCs are not considered to be significant contaminants in surface water at WAG 6.

Barium and cadmium concentrations exceeded reference or MCL values but in only two samples.

4.3.3 Sediments

Thirty sediment samples were collected from drainages in WAG 6 and the EWB area during the RFI. These sample locations are shown in Figs. 4.40 and 4.41. The WAG 6 samples were collected in mid-February 1990 and the EWB samples were collected in mid-February through early-March 1990. These samples were analyzed for volatile and semivolatile organics, metals and radionuclides. From these analyses an assessment of the nature and extent of sediment contamination is presented below. These results are grouped by drainages from which the samples were collected.

4.3.3.1 Previous investigations

The following paragraphs provide an overview of previous investigations conducted at WAG 6 that yielded sediment data. The actual data and sampling locations are presented in Appendix 1B.

Cerling and Spalding (1981) completed a comprehensive assessment of cobalt-60, strontium-90, and cesium-137 concentrations in streambed gravel from contaminated drainages to WOC. This study included sampling locations in Drainages DA, DB, and FB drainages in WAG 6 and locations along the unnamed tributary along the east side of WAG 6.

A study performed by Cerling and Huff (1985) measured cobalt-60, strontium-90, and cesium-137 concentrations in streambed gravel for major tributaries to WOC. Samples were collected during July and August 1985 from: Drainage F, downstream from where Drainages FA and FB combine; Drainage D, downstream from where the DA and DB drainages combine; and in the unnamed tributary along the east side of WAG 6.

Sediment samples were collected by Huff (1986) from Drainages DA, DB, and F in WAG 6, from the unnamed tributary to the east of WAG 6, and from the marshy area near where the unnamed tributary enters WOC. These samples were analyzed for cobalt-60, strontium-90, and cesium-137. The samples from the unnamed tributary were collected from a location due east of the northeast auger holes.

The most extensive sediment sampling of the WAG 6 drainages for cobalt-60, cesium-37, and strontium-90 was performed by Miller, Black, and Craig (1989). Sampling was conducted in Drainages DA, DB, F, FA, and FB from October 1988 to April 1989.

Table 4.22 lists WAG 6 sediment investigators, the locations sampled, and the dates the investigations occurred. Table 4.23 presents the constituents analyzed for during the studies.

4.3.3.2 Radiological contamination in sediments

4.3.3.2.1 Nature of radiological contamination

Both naturally occurring and man-made radionuclides have been detected in sediment samples collected at WAG 6 (Table 4.7). Naturally occurring radionuclides include potassium-40, radionuclides belonging to the uranium series (uranium-238, thorium-234, uranium-234, thorium-230, radium-226), and the thorium series (thorium-232, radium-238, thorium-228, radium-224). No transuranic radionuclides were detected. Naturally occurring radionuclides were within background ranges and were analyzed to account for gross radiation concentrations. These radionuclides are not addressed in extent.

Man-made radionuclides detected in sediment samples include tritium, cobalt-60, strontium-90, and cesium-137. Tritium was detected in 13 of 30 sediment samples. The highest level of tritium detected was 840 pCi/g in Drainage FB at a point just before it combines with Drainage FA. Cesium-137 was the most frequently detected radionuclide in WAG 6 sediment samples, detected at 22 of 30 locations sampled. Concentrations of cesium-137 ranged from 0.244 to 2.14 pCi/g. Cobalt-60 and strontium-90 were detected at three locations each. Cobalt-60 was analyzed for at all 30 locations. Strontium-90 was only analyzed for three times.

4.3.3.2.2 Extent of radiological contamination

Analytical results for radionuclides detected in the EWB and WAG 6 drainages are presented in Figs. 4.40 and 4.41, respectively.

Drainage A and the Emergency Waste Basin. Tritium was detected only in sediment samples EWB02, EWB05, and EWB08 from the EWB. The EWB is both fed and drained by Drainage A. The first two of these three samples were upgradient to EWB, while the third sample was collected below the point where Drainage A is intercepted by EWB (Fig. 4.40). Tritium concentrations ranged from 197 pCi/g in the first sample to 518 pCi/g and 453 pCi/G in the other two samples. Tritium was not detected in sample SWA1 collected from Drainage A.

Cesium-137 was detected in 11 of the 15 samples collected from the EWB and in the one sample from Drainage A. In these samples, cesium-137 was detected at concentrations levels ranging from just above the detection limit (0.2 pCi/g) to 4.57 pCi/g (in sample EWB18 from the northeast side of EWB). There was no apparent pattern to the distribution of cesium-137 contamination in the EWB.

Cobalt-60 was detected once at 0.0775 pCi/g in sample EWB05. Strontium-90 was detected at EWB14 (1.49 pCi/g) and SWA1 (0.79 pCi/g).

Drainage B. Tritium and cesium-137 were the only man-made radionuclides detected in Drainage B. The concentration of tritium detected was 153 pCi/g. [Tritium was also detected in surface water samples (2700 to 30,000 pCi/L)].

Drainage C. Analysis of sediment sample SWC1 collected from Drainage C near the edge of WAG 6 detected tritium at 71 pCi/g, cesium-137 at 2.14 pCi/g, and cobalt-60 at 53.3 pCi/g—the highest concentration of cobalt-60 detected in sediment samples during the RFI. Tritium and cobalt-60 were also detected in the only surface water sample collected from Drainage C. As stated in Sect. 4.3.2.2.2, the most likely source for this contamination is the northeast auger hole area. Miller, Black, and Craig (1989) collected one sediment sample from the unnamed tributary just north of Drainage C and detected cobalt-60 in their analysis. Cerling and Spalding (1981) also found high levels of cobalt-60 in the same area.

Drainage DA. Cesium-137 was the only man-made radionuclide detected in sediment sample W49TS, collected from the southern end of the 49 Trench area French Drain. It is unusual that no tritium was detected in this sample because the surface water samples collected from this location showed the highest tritium levels (up to 14,000,000 pCi/L) found on WAG 6. However, in sediment sample WDA1, collected near the flume on Drainage DA downgradient from the 49 Trench area, tritium was detected at 414 pCi/g and cesium-137 was detected at 1.12 pCi/g. Miller, Black, and Craig (1989) detected cesium-137 and cobalt-60 near the WDA1 sampling location but did not analyze for tritium in sediment samples.

Drainage DB. The analysis of sediment sample WDB2 from Drainage DB detected tritium at 66.7 pCi/g and cesium-137 at 0.797 pCi/g, just above detection limits. Surface water samples also indicated tritium levels in this drainage to be only slightly higher than at other locations in WAG 6. Miller, Black, and Craig (1989) detected higher concentrations of strontium-90 and cesium-137 in this drainage than detected during the RFI and also detected cobalt-60.

Drainage FA. Four sediment samples were collected along Drainage FA at WBAB2, WBAB1, WBAA1, and WFBA1. These samples were collected in the order listed from the upper reach of Drainage FA to the point where Drainage FA combines with Drainage FB near the southern boundary of WAG 6.

Tritium was detected in sediment sample WBAB2 at 312 pCi/g and at 78.8 pCi/g in sample WFBA1. However, it was not detected in downstream samples WBAB1 and WBAA1. Cesium-137 was detected in samples WBAB1 and WBAA1 at 0.368 pCi/g and 0.748 pCi/g, respectively, but was not detected in upstream sample WBAB2 or downstream sample WFBA1. Cesium-137 was also detected near WFBA1 by Miller, Black, and Craig (1989). However, the highest level detected was low (0.3 pCi/g).

Drainage FB. Four sediment samples were collected along Drainage FB. These samples, WFBB3, WFBB2, WSP1, and WFBB1, were collected in the order listed—from the upper reach of Drainage FB near the northern boundary of WAG 6, to the lower end of Drainage FB near the southern boundary of WAG 6.

Tritium levels in upstream samples WFBB3 and WFBB2 were 70.6 pCi/g and below the detection limit, respectively. However, in samples from the lower portion of drainage FB (WSP1 and WFBB1), tritium was detected at 551 pCi/g and 840 pCi/g, respectively. This trend of decreasing tritium levels from WFBB3 to WFBB2 and increasing levels from WFBB2 to WFBB3 is consistent with the same trend observed in surface water samples.

Cesium-137 was detected at 1.13 pCi/g in sample WFBB2, at 0.652 pCi/g in sample WSP1, and at 0.46 pCi/g in sample WFBB1, but was below detection in sample WFBB3 from the upper reach of Drainage FB. Sample WFBB3 indicated a gross beta concentration of 508 pCi/g. The FOF confirmed a high gross beta count on this sample. Strontium-90 was not analyzed for in this sample; however, WFBB3 is located downgradient from the high-activity trench area, which is believed to be leaching strontium-90. Strontium-90 was detected in the farthest downstream sample, WFBB1, at 25.5 pCi/g, which exceeds HEAST ingestion limits. Relatively high levels of strontium-90 were detected in surface water samples collected at WFBB3. Miller et al. (1989) detected high levels of strontium-90 entering Drainage FB in this area.

4.3.3.3 Chemical contamination in sediments

4.3.3.3.1 Nature of chemical contamination

The most frequently detected VOCs in sediment samples collected from WAG 6 are trichloroethene, chloroform, and toluene (Table 4.3). All other VOCs were detected in 20 or fewer samples. No occurrence exceeded any environmental or health-based criteria. The highest detected concentrations of trichloroethene, chloroform, and toluene were 17 μg/kg, 64 μg/kg, and 5 μg/kg, respectively.

TCL SVOCs were detected in very few sediment samples with the exception of bis(2-ethylhexyl)phthalate (Table 4.3). This SVOC is a common laboratory contaminant and is not believed to be associated with detected contaminants found at WAG 6. Due to the low number of SVOCs in sediment samples, they are not discussed further.

Reference samples for sediments and soils were combined to establish a statistically appropriate range for metals concentrations in sediment samples. When sediment sampling results were compared to this range, no sample indicated any metal concentrations significantly above reference levels.

4.3.3.3.2 Extent of chemical contamination

Because of the lack of diversity in VOCs detected in WAG 6 sediments, a general WAG-wide overview of extent will be presented. Figures 4.42 and 4.43 illustrate VOC occurrence in sediment at the EWB and WAG 6.

In nearly all drainages, either trichloroethene, chloroform, or toluene were detected. Combinations of the three were also very common. Only in Drainage B, in the northeast section of the WAG 6, were no VOCs detected. Sediment samples from drainages on the eastern side of WAG 6 did not contain any VOCs other than trichloroethene, chloroform, or toluene, except for a single occurrence of xylenes (2 μg/kg) detected in Drainage DA. Highest concentration for any VOC did not exceed 17 μg/kg, which is approximately four orders of magnitude below any environmental or health-based criteria.

Sediment samples from drainages on the west side of WAG 6 revealed the same VOCs as the east side, with the addition of tetrachloroethene, propylene dichloride, 1,2-dichloroethene, ethylbenzene, and methyl bromide. Except for methyl bromide, these additional VOCs have been detected in sediments adjacent to the 49 Trench area. In samples of other media (groundwater and surface water) from the same area, VOCs also have been detected, strongly implicating this region as a VOC source.

Toluene and chloroform have been detected in sediment samples from the EWB. However, trichloroethene was detected in one sample only. Acetone was also detected in sediment samples in concentrations not exceeding 11 μg/kg. Although acetone was not detected in laboratory blanks associated with analyses of the sediment samples, acetone is a common laboratory contaminant and occurrence of low concentrations should be considered tenuous.

4.3.3.4 Summary of sediment contamination

Radionuclides. Both man-made and naturally occurring radionuclides were detected in sediment samples from WAG 6. Naturally occurring radionuclides were detected at concentrations within reference (background) ranges and are not considered site contaminants. Of the man-made radionuclides detected (cobalt-60, strontium-90, cesium-137, and tritium), cesium-137 was detected most frequently. Except for one detection of strontium-90 and cobalt-60, which exceeded HEAST soil ingestion limits, concentrations of the other radionuclides in other samples did not exceed any action levels (Table 4.7). Transuranic radionuclides were not detected in any sediment samples that were analyzed. The potential for off-WAG migration of strontium-90 in sediment from Drainage FB does exist. Strontium-90 was detected in the sample location closest to the southern boundary of WAG 6.

Chemicals. Except for one sampling location, organic compounds—primarily VOCS—were detected in every sediment sample collected at WAG 6. The most frequently detected VOCs include chloroform, acetone, and toluene in sediments from the EWB, and trichloroethene, toluene, and chloroform from sediment samples collected in drainages from WAG 6. The SVOCs that were detected were few and consisted of common laboratory contaminants. HEAST soil ingestion concentrations for VOCs were not exceeded in any sample. Surface water detects for VOCs have been adjacent to locations where VOCs have been detected in relatively high concentrations in groundwater.

Sediment sampling results indicated no metals present in concentrations significantly above reference concentrations.

4.3.4 Soil

The objectives of the WAG 6 RFI soils investigation were: (1) to identify areas of gross soil contamination that might represent potential secondary source areas, (2) to aid in location of areas of contaminant release, and (3) to provide geological and soil engineering data to aid in understanding site physical characteristics. The scope of the soils investigation is described in greater detail in Sect. 2. (Secondary source areas are defined as areas having contaminants in sufficient quantity and concentration to act as a source reservoir for contaminant migration. The presence of a secondary source may require the extension of a cap or other remedial action targeted for the primary source areas—trenches and auger holes.)

For collection of soil samples, boring locations were chosen along the perimeter of disposal areas to define the extent of the secondary sources. As illustrated in Fig. 4.44, soil borings were generally arranged around the perimeter of the major waste disposal areas, and were frequently within 10 to 15 ft of waste disposal trenches. Specific areas were the 19 Trench area and the EWB, the central waste disposal area, the western waste disposal area, the southwestern region of WAG 6, the southern waste disposal area, and the eastern waste disposal area (as shown in Fig. 4.44 and listed on Table 4.24). The Tumulus area is presented as a separate area. No RFI soil investigations were conducted for the Tumulus area; however, previous investigations have characterized this site and are discussed below.

Eighty soil samples were analyzed from 61 soil borings completed to auger refusal at WAG 6 as part of the RFI effort between December 1988 and March 1990 (Fig. 4.44). Borings were drilled to auger refusal (typically the top of bedrock), from which one to three samples were usually collected for analysis. Field screening and RFI CSL screening resulted in some samples from some borings requiring no further analysis due to a lack of identifiable contamination. Soil samples were also collected at off-WAG locations and analyzed to develop background data. The results of these analyses are presented in Technical Memorandum 06-15. The details of the methods and procedures for radionuclide analysis of soil samples and detailed soil boring logs are presented in Technical Memorandums 06-12 and 06-12A. The results of the investigation for radiological and chemical contamination in soils are discussed below and are presented in Figs. 4.44 through 4.50.

4.3.4.1 Previous investigations

Previous investigations conducted at WAG 6 have yielded soil sample analytical data. Principal sources of analytical data from previous investigations are reported in Davis et al. (1989) and Yager and Craig (1989). Table 4.25 summarizes the scope and timing of these soil investigations and the analytes of concern.

As part of the site characterization investigation of the TARA area in the extreme northeastern part of WAG 6, Davis et al. (1989) collected soil samples from 22 soil borings around the waste disposal trenches. Soil samples were collected from the soil borings every 18 in. to a total depth of 15 ft. Soil samples were also collected from the bottoms of trenches at 15 locations. All soil samples were analyzed for radionuclides, and selected samples were analyzed for VOCs.

Yager and Craig (1989) presented quarterly analytical data for soils collected as part of the environmental monitoring for the Tumulus Demonstration Project. Soil samples were collected from the uppermost 7 to 10 in. of soil at 11 sampling locations around the Tumulus pad between February 1987 and July 1988. These soil samples were collected and analyzed for radionuclides to evaluate the potential spread of contaminants as a result of the Tumulus project.

4.3.4.2 Radiological contamination in soils

4.3.4.2.1 Nature of radiological contamination

Table 4.7 lists the radionuclides detected during RFI sampling at WAG 6. Both naturally occurring and man-made radionuclides were detected. Naturally occurring radionuclides detected were within concentration ranges detected in reference samples and therefore are not discussed further. No transuranic radionuclides were detected. Man-made radionuclides that were detected include cobalt-60 and strontium-90. These occurrences are discussed below.

4.3.4.2.2 Extent of radiological contamination

Because man-made radionuclides were detected in only a few soil samples, an area-by-area discussion, as presented for other media, will not be presented here. Instead, the discussion will focus directly on the occurrence of the specific man-made radionuclide.

Strontium-90. Strontium-90 was detected at concentrations exceeding background at the following locations: (1) SB07 located north of the EWB (Fig. 4.45); (2) SCA19 located west of the solvent auger hole area of the central waste disposal area (Fig. 4.46); and (3) location 1243 located east of the biological trenches of the eastern disposal area (Fig. 4.49). Maximum detected concentration above backgrounds was 8.54 pCi/g at SB07.

A region of suspected radiological contamination was identified during a surface radiological investigation conducted in the vicinity of the EWB during December 1989 (Technical Memorandum No. 06-04A). Soil sample SB07 was collected and the 2 to 4 ft interval was analyzed. Only strontium-90 was detected at concentrations above referenced. This concentration (8.54 pCi/g) is below HEAST soil ingestion limits (Table 4.7).

At location 1243 (Fig. 4.49), strontium-90 was detected at 6.22 pCi/g from the 0 to 2 ft interval. Soil samples to the west and south (SCCO5 and 1234) did not have strontium-90 detected. Strontium-90 was also detected in the 20 to 22 ft interval from location SCA19 (Fig. 4.46). The concentration (1.10 pCi/g) is slightly higher than the maximum detected reference concentration of 0.8pCi/g.

Cobalt-60. Cobalt-60 has been detected in the following soil borings: (1) SCA17 located west of the low-activity trenches of the central waste disposal area; (2) SCC4 located at the high-activity auger holes of the east disposal area; and (3) SCC9 located west of the low-activity trenches of the east waste disposal area.

The concentration of cobalt-60 at SCA17 (0.06 pCi/g), from the 6 to 12 ft interval, is low and barely above detectable concentrations. Other media sampled in the area did not have cobalt-60 detected. In contrast, a sample from the 12 to 18 ft interval of SCC4 at the east waste disposal area had cobalt-60 detected at 7.96 pCi/g. Cobalt-60 has been detected in both groundwater and surface water and sediments in the area. Soil sample location SCC9, located southwest of SCC4, also had cobalt-60 detected in a sample from the 0 to 6 ft interval. Cobalt-60 has not been detected in other media from this area; however, groundwater samples from Well 848 (Fig. 4.19) contained strontium-90.

Yager and Craig (1989) report low levels of cesium-137 (up to 0.4 pCi/g), cobalt-60 (up to 0.08 pCi/g), gross alpha (up to 37 pCi/g), and gross beta (up to 46 pCi/g) in shallow soils (0 to 6 in.) in the vicinity of the Tumulus. Yager and Craig further report that the concentration of radionuclides in shallow surficial soils in the Tumulus Pad area are comparable to other soils within SWSA 6. The results indicate no difference between preoperational and operational periods for the Tumulus I facility; data show a great deal of random variation with no discernible trends or patterns in radionuclide concentration.

4.3.4.3 Chemical contamination in soils

4.3.4.3.1 Nature of chemical contamination

Analyses were performed on soil samples for VOCs, SVOCs, and metals. Table 4.3 lists the chemicals detected in soil samples.

Volatile organic compounds. VOCs detected include acetone, toluene, 2-butanone, and methylene chloride, which are common laboratory contaminants. While acetone and toluene have been detected in trench leachate samples and in groundwater samples from wells adjacent to biological trenches, the nearly ubiquitous detection of these two VOCs in soil samples is very suspect except where they have been detected in other media within the vicinity.

Methylene chloride and 2-butanone must also be considered suspect in WAG 6 soils primarily because they have been detected frequently and in no obvious correlation with releases detected in other media and because they are commonly induced contaminants.

Various SVOCs were also detected, among which bis(2-ethylhexyl)phthalate is the most common. Its detection should also be considered suspect because it is widely distributed vertically and laterally in WAG 6 soils and is a common laboratory contaminant.

Metals. Metals detected in WAG 6 soils are listed in Table 4.5. Concentrations vary in some instances by orders of magnitude for both on-WAG and reference soil samples. Variations both vertically and laterally must also be considered in evaluating metals due to different soil types developed in various strata at WAG 6. Taking into account these considerations and the analytical procedure used (which can produce anomalous results), the metals that have been detected at WAG 6 that are discussed in detail include arsenic, cobalt, lead, and mercury.

4.3.4.3.2 Extent of chemical contamination

Emergency Waste Basin and 19 Trench area. Relatively low levels of VOCs and SVOCs were detected in soil samples collected from the EWB and the 19 Trench area (Fig. 4.45). Other media sampled in the area indicate comparable levels of organics. VOC concentrations did not exceed environmental or health-based criteria. Lead was detected at an estimated concentration of 125 μg/kg at soil sample location 1241. This occurrence should be considered suspect due to laboratory quality control problems encountered during the analysis.

Central waste disposal area—northern region. This area includes the high-activity trench area and the solvent auger hole area to the south. VOCs and some SVOCs have been detected in soil samples at relatively higher concentrations than other waste areas. Soil samples collected from the east and west sides of the high-activity and low-activity trenches (SCA20 and SCA02) contained greater concentrations of VOCs and SVOCs than soil samples to the north and south. The relatively high levels of VOCs detected in boring location SCA20 are consistent with concentrations detected in groundwater in Well 1225 (Fig. 4.25).

This supports the contention that the high-activity trench area is a source for organic contamination. Soil samples from boring location SCA02, located east of the high-activity trenches and adjacent to low-activity trenches, contained a greater diversity of VOCs than boring location SCA20. Acetone, common to samples from both borings, was detected in SCA02 at concentrations generally an order of magnitude less than SCA20. Ethylbenzene, trichloroethene, and 1,2-dichloroethene were detected in SCA02 and not in SCA20, suggesting an additional source. Groundwater sampling in Well 847 (Fig. 4.25), located near boring SCA02 revealed no VOCs. This well is screened in the bedrock, however, and not across the water table, where VOCs are more likely to be detected if present.

Samples from other borings indicate lower levels of VOCs than detected in SCA20 and SCA02. Concentrations did not exceed levels presented in proposed RCRA Subpart S.

Central disposal area—southern region. Analyses of soil samples from borings in this area indicated VOCs; however, concentrations were relatively low and did not exceed levels presented in proposed RCRA Subpart S.

Higher concentrations of VOCs have been detected at the western boundary of the low-activity trenches (Fig. 4.47). VOCs were also detected in groundwater samples along the western boundary surface water samples from Drainage FB.

The highest concentration of arsenic (195 μg/kg) was detected in a sample from location SCA18. This concentration exceeds the next highest concentration by an order of magnitude and also exceeds the proposed RCRA Subpart S (80 μg/kg) concentration criteria. In addition to arsenic, lead and cobalt were detected at the highest concentrations observed at WAG 6 (200 and 120 μg/kg, respectively) at soil sample location SCA12. Both of these concentrations are estimated and should be considered suspect due to laboratory quality control problems encountered during analysis.

Western waste disposal area. VOCs have been detected in soil samples from borings from the northern and southern sections of the west waste disposal area (Fig. 4.48). VOCs have also been detected in groundwater and surface water samples in the western waste disposal area (Figs. 4.25 and 4.38, respectively). Ethylbenzene and acetone have been detected most frequently. Concentrations of other VOCs are frequently below the detection limit and estimated values. Naphthalene, an SVOC, has also been detected in groundwater samples from wells located at the biological trenches.

VOCs and SVOCs detected in samples from borings SDMS1 and 1236, located in the southwestern part of WAG 6 (Fig. 4.48) are anomalous. No waste disposal activities have been conducted in these areas. Trichloroethene, detected in samples throughout boring 1236 (down to 48 ft), suggests a localized problem possibly associated with the sample collection. Trichloroethene has not been commonly detected in soil samples closer to known waste disposal areas, although it has been detected in groundwater samples. The VOC 1,1,2,2-trichloroethene was detected at an estimated 1700 μg/L. The occurrences of acetone, toluene, and xylenes are also anomalous because SDMS1 is located in an area not known to have received organic wastes.

East waste disposal area. VOCs have been detected in soil samples collected from borings adjacent to the east waste disposal area (Fig. 4.49). Acetone is the most widely distributed. Concentrations for all detected compounds are below proposed RCRA Subpart S concentration criteria.

Soil samples from boring SCC03, located southeast of a solvent auger hole area in the northern section of the waste disposal area, has had a greater diversity of VOCs detected in it than other borings at this disposal area. Concentrations do not exceed 65 μg/L for any compound detected, however. Groundwater samples from wells located to the east (Fig. 4.25) of the solvent auger hole area have also had VOCs detected.

South waste disposal area. VOCs have been detected in soil samples collected from borings adjacent to the south waste disposal area. These occurred at relatively low concentrations and diversity compared with other borings at the site (Fig. 4.50). Acetone is the most widely distributed VOC detected. Concentrations of all VOCs detected did not exceed proposed RCRA Subpart S concentration criteria. Mercury was detected at the highest concentration on-WAG (252 μg/kg) at soil sample location 1235, located northeast of the south waste disposal area. The next highest mercury concentration detected on-WAG was 0.7 μg/kg. The proposed RCRA Subpart S concentration criteria for mercury (20 μg/kg) was exceeded by an order of magnitude. Laboratory quality control protocols appear to have been in order during the analysis. The occurrence of mercury at location 1235 is anomalous because of its distance from any identified disposal area and the depth from which it was detected (30-36 ft).

4.3.4.4 Summary of soils contamination

Contaminant concentrations vary considerably from one soil horizon to another and from one location to another. Results indicate that maximum concentrations typically occur at depths in the range of 12 to 20 ft. This is to be expected since the waste is buried to similar depths.

Radionuclides. Radiological contamination detected in RFI soil samples is limited to the man-made radionuclides strontium-90 and cobalt-60. Areal extent of each radionuclide was limited to three soil borings each. The maximum concentration detected for strontium-90 and cobalt-60 did not exceed HEAST soil ingestion limits (Table 4.7).

Chemicals. Although various organic compounds (both VOCs and SVOCs) have been detected in soil samples from WAG 6, some of the compounds are suspected to be sampling- or laboratory-induced contaminants. These compounds are acetone, 2-butanone, methylene chloride, toluene, and bis(2-ethylhexyl) phthalate. Concentrations detected for these compounds were typically below 100 μg/L. Proposed RCRA Subpart S concentration criteria were not exceeded for any organic compound detected in soil.

Metal contamination in soils at WAG 6 appears to be very localized. Arsenic, lead, mercury, and cobalt have been detected in a few samples at concentrations above background.

4.4 OFF-WAG MIGRATION

The preceding "medium by medium" discussion focused primarily on describing on-WAG releases and contamination. The following conclusions regarding off-WAG migration of contamination are determined primarily from inferences based on on-WAG sampling and are summarized to support assessment of impacts to off-WAG receptors.

4.4.1 Summary of Off-WAG Transport Mechanisms

The primary media by which contaminants are transported off-WAG are groundwater and surface water. There is some transport off-WAG via sediments, soils, and air; however, these appear to be relatively minor pathways. Sediments, although contaminated, are not transported off-WAG at significant rates due to the intermittent nature of the streams and erosion control features implemented in conjunction with the RCRA ICM caps. Erosion of surface soils off-WAG via surface water runoff from waste disposal areas is not considered significant. The radiological walkover survey revealed minimal surface soil contamination, and erosion is controlled by the regularly maintained vegetative cover over the waste disposal areas and by the RCRA ICM caps. Air, although not sampled during the RFI, is not expected to be a significant pathway due to the vegetative cover over the waste areas (controlling suspension of soil particles) and the general absence of surface soil contamination. Volatilization of contaminants from surface water may be a significant pathway and is included in the fate and transport analysis (Sect. 5.)

4.4.2 Off-WAG Migration Patterns

Figure 4.51 and associated Table 4.26 identify areas of the site perimeter where off-WAG migation of radionuclides is likely. Figure 4.52 and Table 4.27 present the same information for chemicals. The figures divide the perimeter of the site into three classifications:

- Areas where off-WAG migration is likely (designated by H)
- Areas where off-WAG migration is not likely (designated L)
- Areas of uncertainty for off-WAG migration

Areas where off-WAG migration is likely are defined as regions along the boundary of WAG 6 where groundwater and/or surface water features in proximity to the boundary have been shown to contain various contaminants. Unless downstream sampling indicated otherwise, when contamination was detected upstream in any of the on-WAG drainages, it was assumed that these contaminants are likely migrating off-WAG. If wells close to any boundary contained contaminants and groundwater flow directions indicated possible off-WAG flow, these areas were also labeled as areas of likely off-WAG migration.

Areas not likely for off-WAG migration are defined as those areas that lack any waste disposal facility in the vicinity (such as the southwestern region of WAG 6); those that are upgradient to any waste disposal facility and where samples collected from the area show little, if any, contamination; and areas downgradient to waste disposal areas where sampling and an understanding of flow characteristics suggest that this area is not a possible exit point from the WAG.

Areas of uncertainty for off-WAG migration of contaminants include regions along the perimeter of WAG 6 downgradient of waste areas but where no direct evidence to support off-WAG migration exists.

Section 4 Tables

Table 4.1. WAG 6 RFI technical memorandums in which environmental data are recorded

Media sampled	TM number and title	
Groundwater	06-09; 06-09A	Groundwater Sampling at SWSA 6;
Surface water and sediment	06-05A	Groundwater Sampling at WAG 6 Surface Water and Sediment Sampling for the ORNL WAG 6 RFI
Soil	06-12; 06-12A	Phase I Soil Sampling for the ORNL WAG 6 RFI; Soil Sampling for the ORNL WAG 6 RFI (Activity 1, Activity 2)

Table 4.2. Organic constituent concentration ranges in groundwater and surface water samples

		Comparison criteria^{a}			Groundwater							Surface water					
Analyte	CAS no.	Proposed MCLs^{b} (ug/L)	SDWA MCLs (ug/L)	TN secondary drinking water regulation^{d} (ug/L)	Proposed RCRA subpart S water^{e} (ug/L)	Reference sampling minimum (ug/L)	Reference sampling maximum (ug/L)	Site sampling minimum (ug/L)	Site sampling maximum (ug/L)	Total samples	Total detects	Reference sampling minimum^{h} (ug/L)	Reference sampling maximum^{h} (ug/L)	Site sampling minimum (ug/L)	Site sampling maximum (ug/L)	Total samples	Total detects
Volatile Organic Compounds																	
1,1,2,2-Tetrachloroethane	79-34-5				2			5 U	53 J	64	3			5 U	1 J	62	1
1,1,1-Trichloroethane	71-55-6		200		3000			5 U	4 J	159	10			5 U	2 J	62	1
1,1,2-Trichloroethane	79-00-5	5			6			1 U	3 J	64	12						
1,1-Dichloroethane	75-34-3							5 U	17	159	22			5 U	6	62	5
1,1-Dichloroethene	75-35-4		7		MCL			5 U	6	159	8			5 U	4 J	62	3
1,2-Dichloroethane	107-06-2		5		MCL			5 U	44	159	23			5 U	2 J	62	1
1,2-Dichloroethene	540-59-0		70^{f}					5 U	680	157	35			5 U	180	62	28
1,4-Dioxane	123-91-1				3			5 J	160 J	12	12						
2-Butanone	78-93-3				2000			3 UJ	2 J	64	3			5 U	10 J	56	3
2-Hexanone	591-78-6													10 U	1 J	62	1
4-Methyl-2-Pentanone	108-10-1					10 U	2 J							10 U	1 J	62	1
Acetone	67-64-1				4000	10 U	4700 B	1 UJ	280 J	159	34			1 U	4 J	62	2
Benzene	71-43-2		5					5 U	110	159	26			5 U	3 J		
Carbon Disulfide	75-15-0				4000	5 U	1 J	5 U	22	64	1			2 UJ	3 J		
Carbon Tetrachloride	56-23-5		5		0.3			5 U	96 J	159	7						
Chlorobenzene	108-90-7		100^{f}		700									5 U	3 J	62	2
Chloroethane	75-00-3							10 U	1 J	64	2						
Chloroform	67-66-3				6			5 U	190 J	174	28			5 U	15	62	1
Chloromethane	74-87-3							10 U	15	95	3						
Dichlorodifluoromethane	75-71-8				7000									20 U	17 J	2	1
Ethylbenzene	100-41-4		700^{f}	30^{g}	4000	5 U	2 J	5 U	220	159	11						
Isobutyl Alcohol	78-83-1				10000												
Methylene Chloride	75-09-2	5			5	5 U	5 J	1 U	25 J	159	84						
Styrene	100-42-5		100^{f}	10^{g}	1.2												
Tetrachloroethene	127-18-4		5		0.7	5 U	4 J	5 U	6800	159	23			5 U	83 J	62	16
Toluene	108-88-3		1000^{f}	40^{g}	7000	5 U	3 J	2 UJ	2000	159	31			3 U	3 J	62	3
Trichloroethene	71-55-6		5	2700	MCL			5 U	490	159	17			5 U	2200	62	29
Trichlorofluoromethane	75-69-4				10000			3 J	12 J	6	6						

Table 4.2. (continued)

		Comparison criteria[a]				Groundwater						Surface water					
Analyte	CAS no.	Proposed MCLs[b] (ug/L)	SDWA MCLs (ug/L)	TN secondary drinking water regulation[d] (ug/L)	Proposed RCRA subpart S water[e] (ug/L)	Reference sampling minimum (ug/L)	Reference sampling maximum (ug/L)	Site sampling minimum (ug/L)	Site sampling maximum (ug/L)	Total samples	Total detects	Reference sampling minimum[h] (ug/L)	Reference sampling maximum[h] (ug/L)	Site sampling minimum (ug/L)	Site sampling maximum (ug/L)	Total samples	Total detects
Vinyl Acetate	108-05-4													10 U	1 U	62	1
Vinyl Chloride	75-01-4		2					10 U	88	159	10			10 U	11	62	3
Xylene, total	1330-20-7		10,000[f]	20[g]	70000			5 U	3200	159	13						
Semivolatile Organics																	
2-Methyl-Phenol	95-48-7							5.5 U	22								
Bis(2-Ethylhexyl)Phthalate	117-81-7	4			3	10 U	19 B	5.5 U	16 B	61	4			10 U	4 J	54	3
Di-N-Butyl Phthalate	84-74-2					10 U	2 J	2 U	19	61	3			10 U	3 J	54	2
Di-N-Octyl Phthalate	117-84-0							10 U	39	61	3						
Diethyl Phthalate	84-66-2				30000	10 U	0.5 J	5.5 U	48								
Naphthalene	91-20-3				20000			5.5 U	1000	61	5			10 U	2 J	54	1
Phthalic Acid	84-74-2									61	3						
PCBs/Herbicides																	
2,4,5-T	93-76-5							0.1 U	0.3	42	1						
2,4,5-TP (Silvex)	93-72-1		50			0.1 U	0.2	0.1 U	0.3	80	2						
2,4-D	120-83-2		70			0.2 U	1.6	0.2 U	11.7	80	1						
AROCHLOR-1254	11097-69-1		2[f]					1 U	1 J	82	2						

[a] Proposed Safe Drinking Water Act (SDWA) Maximum Contaminant Levels (MCLs), SWDA MCLs, and Tennessee Secondary Drinking Water Regulations are taken from Applicable or Relevant and Appropriate Requirements (ARAR) and To-Be-Considered (TBC) Guidance, Biomedical and Environmental Information Analysis Section, Health and Safety Research Division, Oak Ridge National Laboratory, April 1991.

[b] 55 Federal Register 30370 (July 25, 1990) unless otherwise indicated.

[c] 52 Federal Register 25690 (July 8, 1987).

[d] Chapter 1200-5-1.12 of the Rules of the Tennessee Department of Health and Environment.

[e] 55 Federal Register 30865 (July 27, 1990) modified as described in text.

[f] 56 Federal Register 3526 (January 30, 1991). Effective January 30, 1992.

[g] 54 Federal Register (May 22, 1989). Proposed Secondary Drinking Water Standard.

[h] No VOCs or SVOCs were detected above detection limits from surface water reference samples.

J — Estimated concentration
U — Not detected at indicated concentration
B — Compound also found in laboratory blank

Table 4.3. Organic constituent concentration ranges in soil and sediment samples

		Concentration criteria	Sediments						Soils					
Analyte	CAS no.	Proposed RCRA Subpart S soil ^{a} (μg/kg)	Reference sampling minimum (μg/kg)	Reference sampling maximum (μg/kg)	Site sampling minimum (ug/kg)	Site sampling maximum (ug/kg)	Total samples	Total detects	Reference sampling minimum (ug/kg)	Reference sampling maximum (μg/kg)	Site sampling minimum (ug/kg)	Site sampling maximum (ug/kg)	Total samples	Total detects
Semivolatile Organic Compounds														
1,1-Dichloroethene	75-35-4	1,000							5 U	2 J				
1,2-Dichloroethene	540-59-0				6 U	15	30	1			5 U	39	76	3
1,2-Dichloropropane	78-87-5				6 U	3 J	30	1						
1,3-Dichloropropene	10061-01-5	20,000			6 U	1 J	30	1						
2-Butanone	78-93-3	4,000,000			10 U	340 J	24	14	4 UJ	46 J	10 U	150 J	75	11
2-Hexanone	591-78-6										11 U	1 J	76	1
Acetone	67-64-1	8,000,000			10 U	18	30	5			8 UJ	2700 J	76	43
Benzene	71-43-2								5 U	2 J				
Carbon disulfide	75-15-0	8,000,000									5 U	8	76	1
Chlorobenzene	108-90-7	2,000,000									5 U	1 J	76	1
Chloroform	67-66-3	100,000			6 U	64	32	13	5 U	9	5 U	34	76	3
Ethylbenzene	100-41-4	8,000,000									5 U	6 J	77	11
Isobutyl alcohol	78-83-1	20,000,000			25 UJ	45	3	1						
Methylene bromide	74-95-3				11 U	10 J	30	1						
Methylene chloride	75-09-2	90,000	7 U	68	6 U	200 J	30	9	6 UJ	12 J	3 U	42 J	76	11
Styrene	100-42-5	23,000			6 U	8	30	1						
Tetrachloroethene	127-18-4	10,000			6 U	3 J	30	2			5 U	420 J	76	2
Toluene	108-88-3	16,000,000			6 U	5 J	30	17	5 U	3 J	5 U	190	76	11
Trichloroethene	79-01-6	60,000			6 U	17	30	13	5 U	55	5 U	150		
Trichlorofluoromethane	75-69-4	20,000,000			45 U	72	3	2	6 U	340 J				
Xylenes, total	1330-20-7	200,000,000			6 U	2 J	29	2			5 U	7	76	16
Semivolatile Organics														
2-Chlorophenol	95-57-8	400,000									350 U	56 J	74	1

Table 4.3. (continued)

		Concentration criteria	Sediments						Soils					
Analyte	CAS no.	Proposed RCRA Subpart S soil [a] (μg/kg)	Reference sampling minimum (μg/kg)	Reference sampling maximum (μg/kg)	Site sampling minimum (ug/kg)	Site sampling maximum (ug/kg)	Total samples	Total detects	Reference sampling minimum (ug/kg)	Reference sampling maximum (μg/kg)	Site sampling minimum (ug/kg)	Site sampling maximum (ug/kg)	Total samples	Total detects
2-Methyl-1,3 DIN Benzene	606-20-2				370 U	140 J	30	3						
Benzo(a)Anthracene	56-55-3										350 U	51 J	74	1
Benzo(a)Pyrene	50-32-8										350 U	37,000 J	74	1
Benzoic acid	65-85-0				1600 U	67 J	30	4	510 U	110 J	1700 U	67 J	74	2
Benzyl alcohol	100-51-6													
Bis(2-Ethylhexyl)Phthalate	117-81-7	50,000	410 U	490	270 UJ	1700	30	17			180 U	470,000 J	75	19
Butyl Benzyl Phthalate	85-68-7	20,000,000			370 U	68 J	30	1			150 U	1500 J	74	1
Chrysene	218-01-9										350 U	140 J	74	2
Di-N-Butyl Phthalate	84-74-2		410 U	47 J							360 UJ	3400 J	74	4
Di-N-Octyl Phthalate	117-84-0										350 U	60 J	74	2
N-Nitroso N-Phenyl Benzamine	86-30-6		410 U	150 J	49 UJ	59 J	30	3	120 UJ	150 J	350 U	61 J	74	1
N-Phenyl Benzamine	122-39-4			73 J					82 U	94 J				
Naphthalene	91-20-3										350 U	760 J	74	2
Pentachlorophenol	87-86-5	5,800									1700 U	78 J	74	1
Phenol	108-95-2	50,000,000			370 U	89 J	30	2			350 U	100 J	74	8

[a] 55 Federal Register 30370 (July 27, 1990) modified as described in text.

J—Estimated concentration.
U—Not detected at indicated concentration.

		Concentration criteria[a]							
Analytes (unfiltered)	CAS no.	Proposed MCLs[b] (ug/L)	SDWA MCLs[c] (ug/L)	TN Secondary Drinking Water Regulations[d] (ug/L)	Freshwater Organism 24-h max.[e] (ug/L)	Freshwater Organism max.[f] (ug/L)	Proposed RCRA Subpart S water[g] (ug/L)	Reference sampling minimum (ug/L)	Refe samp maxi (ug
Total Metals									
Aluminium	7429-90-5			50[h,i]	87	750		50 U	7
Antimony	7440-36-0	10/5[j]					10		
Arsenic	7440-38-2		50		190	360	MCL		
Barium	7440-39-3	1000[k]	1000				MCL		
Beryllium	7440-41-7	1					0.008	0.3 U	
Boron	7440-42-8								
Cadmium	7440-43-9		51		0.66	1.8	MCL	2 U	
Calcium	7440-70-2						–	100 U	1800
Chromium VI/III	7440-47-3		100 (total)		11/117	16/984	MCL/		
Cobalt	7440-48-4							1.8 U	
Copper	7440-50-8	1300[m]		1000	6.54	9.22		6 U	
Iron	7439-89-6			300	1000			50 U	32
Lead	7439-92-1	5	50		1.32	33.8	MCL	20 U	
Magnesium	7439-95-4							990	54
Manganese	7439-96-5			50				10 U	1
Mercury	7439-97-6		2		0.012	2.4	MCL	0.1 U	
Nickel	7440-02-0	100			88	789	700	3.6 U	
Potassium	7440-09-7								
Selenium	7782-49-2		50		5	20		5 U	
Silicon	7440-21-3							1620	14
Silver	7440-22-4		50	90[h]		1.23	MCL		
Sodium	7440-23-5							690	187
Strontium	7440-26-6							5 U	
Titanium	7440-32-6							20 U	
Vanadium	7440-62-2							44	
Zinc	7440-66-6			5000	59	65		74	

[a] Proposed Safe Drinking Water Act (SDWA) Maximum Contaminant Levels (MCLs), SWDA MCLs, and Tennessee Secondary Drinking Water Regulations are taken Safety Research Division, Oak Ridge National Laboratory, April 1991.

[b] 55 Federal Register 30370 (July 25, 1990) unless otherwise indicated.

[c] 52 Federal Register 25690 (July 8, 1987).

[d] Chapter 1200-5-1.12 of the Rules of the Tennessee Department of Health and Environment.

[e] Federal Ambient Water Quality Criteria for protection of freshwater organisms, one-hour average concentration not to be exceeded more than once every 3 years, E

[f] Federal Ambient Water Quality Criteria for protection of freshwater organisms, four-day average concentration not to be exceeded more than once every 3 years, EP

[g] 55 Federal Register 30865 (July 27, 1990) modified as described in text.

[h] 56 Federal Register 3526 (January 30, 1991). Final rule effective July 30, 1992.

[i] Level recommended to prevent post treatment precipitation in distribution system.

[j] EPA proposes two MCLs for public comment based on two practical quantitation limits.

[k] 56 Federal Register 3526 (January 30, 1990). The MCL has been reproposed.

[l] 56 Federal Register 3526 (January 30, 1990). The final MCL has been reproposed and supercedes the original MCL.

[m] 53 Federal Register 31516 (August 18, 1988).

J—Estimated concentration.

U—Not detected at indicated concentration.

Table 4.4. Inorganic constituent concentration ranges in groundwater and surface water samples

	Groundwater				Surface water					
ice ling um L)	Site sampling minimum (ug/L)	Site sampling maximum (ug/L)	Total samples	Total detects	Reference sampling minimum (ug/L)	Reference sampling maximum (ug/L)	Site sampling minimum (ug/L)	Site sampling maximum (ug/L)	Total samples	Total detects
0	40 U	115,000	161	135	53	286	40 U	40,700	119	113
	30 U	91.50	82	4						
	2 U	66.5 J	162	24			2 U	15.9 J	118	10
	1000 U	27,100	162	88	40	43	16.2	2600	120	120
5	0.30 U	87.10	161	91			1 U	3.6 J	118	22
	80 U	3810	81	3						
3	2 U	94.50 J	82	30			2 U	30	120	25
0	100 U	649,000	170	169	34930	38200	102 U	107,000	119	117
	10 U	216 J	161	55			10 U	58.3	120	61
3.1	2 U	169	162	32			10 U	37	120	3
0	10 U	139	162	40	10 U	13	10 U	88.3 J	120	20
0	14 U	110,000	161	154	359 U	447	10 U	57,900 J	119	113
0	2 U	100	81	45	7	7	2 U	31.6	120	60
0	10 U	57,500	170	169	5023	5110	11,400 U	14,900	119	118
0	2 U	17,400 J	161	153	92.70	113	2 U	3440	119	115
0.7	0.1 U	2.40 J	158	17			0.2 U	0.67 J	119	15
0	5 U	244	162	70			20 U	49.7	120	12
	844 U	86300	90	64			1000 U	27,600	119	94
J							2 U	2.3 J	99	2
0	200 U	14,000	80	79						
	5 U	72.60	154	18			5 U	47 J	112	16
0	1010	237,000	170	170	3460	3578	361 U	13,200	119	111
0	5 U	1200	57	53						
7	20 U	33	80	23						
1	4 U	164	102	162			10 U	55.7	120	39
5	2 UJ	2590 J	162	80	100	115	5 U	1260 J	120	66

from Applicable or Relevant and Appropriate Regulrements (ARAR) and To-Be-Considered (TBC) Guidance, Biomedical and Environmental Information Analysis Section, Health and

. Region IV Criteria Chart, January 1991.
Region IV Criteria Chart, January 1991.

Table 4.5. Inorganic constituent concentration ranges in soil and sediment samples

		Concentration criteria	Sediment						Soil					
Analyte	CAS no.	Proposed RCRA Subpt. S soil[a] (mg/kg)	Reference sampling minimum (mg/kg)	Reference sampling maximum (mg/kg)	Site sampling minimum (mg/kg)	Site sampling maximum (mg/kg)	Total samples	Total detects	Reference sampling minimum (mg/kg)	Reference sampling maximum (mg/kg)	Site sampling minimum (mg/kg)	Site sampling maximum (mg/kg)	Total samples	Total detects
Aluminum	7429-90-5		3660	18,500	7380 J	24,600 J	30	30	10,500	22,600	8710	31,700	63	63
Antimony	7440-36-0	30							4.7 UJ	5.7 J	0.1 UJ	1.2 J	45	5
Arsenic	7440-38-2	80	2.3 J	13.6 J	0.37 U	4.2	30	25	0.35 U	5.3	1.0 UJ	195	72	63
Barium	7440-39-3	5600	37.7 J	385 J	70.6	228	30	30	75.1	250	48.9	420	74	74
Beryllium	7440-41-7	0.2	0.2	2.2	1.2 U	2.4	30	23	1 U	2.1	1.7 U	2	74	72
Cadmium	7440-43-9	40	0.6 U	3.3	0.58 U	4.0	30	26	2.5	4.4	0.5 U	6.1	74	49
Calcium	7440-70-2		194 J	3090 J	447	16900	30	30	820	14,400 J	42.9	54,700	71	71
Chromium (total)	7440-47-3	400 (Cr+6)	5.0 J	87 J	13.1	34.8	30	30	40.9 U	36.9 J	16.7 J	52.3	74	74
Cobalt	7440-48-4		3	40	9 J	39.8	30	30	20 U	33.5 J	11 J	120 J	74	74
Copper	7440-50-8		6 J	32 J	12.5 U	27.5	30	26	18.5 U	26.2	6	86.7	74	74
Iron	7439-89-6		5410	86,520	2040	41700	30	30	34,900 U	41,200	21,300	45,300 J	63	63
Lead	7439-92-1		6.2 J	34.7 J	0.54 UJ	46.2	29	28	19 U	47.3	3.2	200 J	74	74
Magnesium	7439-95-4		528 J	4283	1320	8500	30	30	2850	8420	65.3	45,700	74	74
Manganese	7439-96-5		146	3606	54.1	3530	30	30	426	2380	90.1 J	723 J	74	74
Mercury	7439-97-6	20			0 UJ	0.06	28	4	0.11 U	0.2 J	0 U	252	73	10
Nickel	7440-02-0	2000	4	41	18.7	59.7	30	30	28.6	76	24 J	86.6	74	74
Osmium	7440-04-2				318 J	421	3	3	122 U	1558				
Potassium	7440-09-7		566	3480	494	3330 J	30	30	2320 U	2420	1170 UJ	4570 J	74	73
Selenium	7782-49-2				0.28 UJ	0.37	13	1	0.32 UJ	0.47 5	0.3 UJ	0.9 J	56	3
Silver	7440-22-4	200	0.6 UJ	1.7 J	0.77 UJ	3.6 J	26	6	1.7 U	2.9	0.1 U	2.5	67	4
Sodium	7440-23-5		55	96	30.8 U	93.30	30	29	53.2 U	118	40 U	194	74	64
Thallium	7440-28-0								0.16 UJ	0.4	0.2 U	1.0	69	4
Tin	7440-31-5		16 J	16 J	25.7	71.6	3	3	5.4 UJ	136				
Vanadium	7440-62-2		5	48	5.7 J	32.[illegible]	30	30	10.6	21	8.5	34 J	74	74
Zinc	7440-66-6		17.4	117	23.5 J	103	30	30	62.2 U	86.8	78.8	143 J	74	73

[a] 55 Federal Register 30370 (July 27, 1990) modified as described in text.

J—Estimated concentration

U—Not detected at indicated concentration

Table 4.6. Radionuclide concentration ranges in groundwater and surface water samples

		Concentration criteria			Groundwater						Surface water					
Analytes	CAS no.	HEAST drinking water ingestion[a] (pCi/L)	EPA 4 mrem/yr gross beta[b] (pCi/L)	SDWA MCLs[c] (pCi/L)	Reference sampling minimum (pCi/L)	Reference sampling maximum (pCi/L)	Site sampling minimum (pCi/L)	Site sampling maximum (pCi/L)	Total samples	Total detects	Reference sampling minimum (pCi/L)	Reference sampling maximum (pCi/L)	Site sampling minimum (pCi/L)	Site sampling maximum (pCi/L)	Total samples	Total detects
Gross Alpha and Beta Particle Activity																
Gross Alpha	12587-46-1			15[d]	1.03 U	6.75	0.2703	335	141	121	3.74	4.84	1 U	106	54	37
Gross Beta	12587-47-2				0.81	51.3	0.05406	9450	141	120			4 U	16,937	54	51
Man-Made Radionuclides																
Tritium	10028-17-8	357.1	85,544.5	20,000	864	1944	500 U	4,324,800	158	151	749	877	1080	1.4E+7	102	102
Cobalt-60	10198-40-0	1.3	203.3		2.7 U	5.4 U U	5.40 U	1880	86	84			550	550	1	1
Strontium-90	10098-97-2	0.6	38.4	8			5 U	5400	62	9			5 U	10,222	91	43
Total Strontium	7440-26-6				4.06 U	15.13	2.703	51.357	78	78						
Cesium-137	10045-97-3	0.7	109.6		2.7 U	18.36	2.703 U	21.08	80	78			7.36 U	11.2	63	1
Americium-241	14596-10-2	0.1	1.5				1 U	23.90	6	44						
Plutonium-238	13981-16-3	0.1					1 U	4.29	42	2			1 U	3.73	64	2
Plutonium-239/240	0-013	0.6					1 U	1.90	42	2			1 U	3.16	64	2
Curium-244	13981-15-2						1 U	19.7	2	1			134.7	134.7	1	1
Curium-242	15510-73-3												7.88	7.88	1	1
Naturally Occurring Radionuclides																
Potassium-40	13966-00-2	1.8	294.8				11.52 U	252	9	8			20 U	276	5	4
Total Radium					1.16 U	1.54	0.29	2.703	78	78						
Uranium Series																
Uranium-238	7440-61-1	0.1					1 U	7.83	59	12						
Thorium-234	15065-10-8	5.0	401.1				87.9	268	5	5	263[e]	263[e]	143	272	5	5
Uranium-234	13966-29-5	0.1					1 U	15.8	59	22						
Thorium-230	14269-63-7	0.8					1 U	13.6	60	23	1 U	3.29	1 U	1.20	63	1

Table 4.6. (continued)

		Concentration criteria			Groundwater						Surface water					
Analytes	CAS no.	HEAST drinking water ingestion[a] (pCi/L)	EPA 4 mrem/yr gross beta[b] (pCi/L)	SDWA MCLs[c] (pCi/L)	Reference sampling minimum (pCi/L)	Reference sampling maximum (pCi/L)	Site sampling minimum (pCi/L)	Site sampling maximum (pCi/L)	Total samples	Total detects	Reference sampling mimimum (pCi/L)	Reference sampling maximum (pCi/L)	Site sampling minimum (pCi/L)	Site sampling maximum (pCi/L)	Total samples	Total detects
Radium-226	13982-63-3	0.2		5[f]			1 U	113	42	18						
Thorium Series																
Thorium-232	7440-29-1	0.9					1 U	9.13	60	16			1 U	1.14	63	1
Radium-228	15262-20-1	0.2	3.8	5[f]			3 U	860	41	6			3 U	55	12	2
Thorium-228	14274-82-9	1.3					1 U	8.21	60	27			1 U	1.92	63	7
Radium-224	13233-32-4	0.4					12.50	23.3	15	8	16 U	18.70	15.5	27.2	5	5

[a] Represents concentration in drinking water which yields a 1 x 10^{-6} lifetime excess cancer risk based on 2 L per day drinking water consumption for 70 years. Health Effects Assessment Summary Tables, Fourth Quarter, FY 1990, OERR 9200 6-3-3 (90-4), September 1990.

[b] Represents 4 mrem per year total body or organ dose equivalent exposure from consumption of 2 L per day of drinking water (40 Code of Federal Regulations 141.16) and the dose conversion factors in Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion, Federal Guidance Report No. 11, EPA-520/1-88-020, September 1988.

[c] 40 Code of Federal Regulations 141.15 and 141.16.

[d] Includes radium-226 but excludes radon and uranium.

[e] Not confirmed by beta isotopic analysis. Corresponding gross beta analyses did not detect activity above 4 pCi/L.

[f] The sum of radium-226 and radium-228 cannot exceed 5 pCi/L.

U—Not detected at indicated concentration.

Table 4.7. Radionuclide concentration ranges in soil and sediment samples

		Concentration criteria		Sediment						Soil					
Analytes	CAS no.	HEAST soil ingestion[a] (pCi/g)	HEAST soil direct exposure[b] (pCi/g)	Reference sampling minimum (pCi/g)	Reference sampling maximum (pCi/g)	Site sampling minimum (pCi/g)	Site sampling maximum (pCi/g)	Total samples	Total detects	Reference sampling minimum (pCi/g)	Reference sampling maximum (pCi/g)	Site sampling minimum (pCi/g)	Site sampling maximum (pCi/g)	Total samples	Total detects
Gross Alpha and Beta Particle Activity															
Gross Alpha	12587-46-1			13.2	37.7	10.7	27.3	30	30	4.0	27	6	26.7	73	73
Gross Beta	12587-47-2			15.2	44.9	13	508	30	30	11.0	55	11	43	73	73
Man-Made Radionuclides															
Tritium	10028-17-8	6666.7				0.50 U	840	30	13						
Cobalt-60	10198-40-0	24.4	7.7E-04			0.08	53.3	3	3			0.06	7.96	3	3
Strontium-89	14158-27-1			3 U	8.16							10 U	7.81	5	1
Strontium-90	10098-97-2	11.2	–	0.52 U	0.92	0.79	25.5	3	3	0.5 U	0.8	0.5 U	8.54	22	4
Cesium-137	10045-97-3	13.2	–	0.2 U	0.04	0.2 U	4.57	30	22	0.2	0.38	0.02 U	0.4	26	11
Lead-210	14255-04-0											1.31	1.31	1	1
Actinium	14952-40-0											0.33	0.33	1	1
Naturally Occurring Radionuclides															
Potassium-40	13966-00-2	33.3	0.01	9.54	31.8	1.79	27.7	25	25	16.9	41.5	7.48	39.2	69	69
Uranium Series															
Uranium-238	7440-61-1	2.9	2.22	0.62	1.69	0.75	1.05	3	3	0.6 U	1.12	0.6 U	1.26	62	45
Thorium-234	15065-10-8	90.9	0.18	1.09	1.16	0.81	1.63	15	15	0.79	1.76	0.33	1.36	53	53
Uranium-234	13966-29-5	2.6	1.79	0.6 U	2.15	0.89	2.04	3	3	0.6 U	5.68	0.6 U	1.25	62	43
Thorium-230	14269-63-7	15.4	1.69	0.6 U	1.29	1.09	1.59	3	3	0.6 U	1.34	0.6 U	1.37	62	49
Radium-226	13982-53-3	3.1	0.24	0.63	0.94	0.61	0.99	29	29	0.68	65	0.3 U	0.91	72	71
Thorium Series															
Thorium-232	7440-29-1	16.9	2.22	0.6 U	1.57	0.94	1.18	15	15	0.79	2.15	0.6 U	2.06	62	60
Radium-228	15262-20-1	3.7	1.8E+06	1.30	1.37	0.86	1.63	29	29	0.96	1.67	0.5 U	1.81	72	71
Thorium-228	14274-82-9	24.4	0.63	0.6 U	1.72	0.97	1.42	3	3	0.69	2.13	0.6 U	2.69	62	61
Radium-224	13233-32-4	7.7	0.16	1.1	1.37	0.81	13.1	30	30	0.94	1.56	0.39	2.42	73	73

[a] Represents concentration in soil which yields a 1 x 10^{-6} lifetime excess cancer risk based on 0.2 g/day childhood ingestion (6 year period) and 0.1 g/day adult ingestion (64 year period). Health Effects Assessment Summary Tables, Fourth Quarter, FY 1990, OERR 9200 6-3-3(90-4), September 1990.

[b] Represents exposure from gamma-emitting radionuclides uniformly mixed in soil (effective surface density = 143 kg/m^2*103[g/kg] for 70 years. Health Effects Assessment Summary Tables, Fourth Quarter, FY 1990, OERR 9200 6-303 (90-4), September 1990.

U—Not detec[illegible] indicated concentration.

Table 4.8. Constituent screening for nature and extent analysis

Analyte	CAS no.	Reference comparison performed	Statistically greater than reference	Greater than comparison criteria	Maximum concentration comparison	Include for nature and extent consideration
Volatile Organic Compounds						
1,1,2,2-Tetrachloroethane	79-34-5	gw,so		gw^{f}		gw
1,1,1-Trichloroethane	71-55-6	gw,so				
1,1,2-Trichloroethane	79-00-5	gw,so				
1,1-Dichloroethane	75-34-3	gw,so				
1,1-Dichloroethene	75-35-4	gw,so				
1,2-Dichloroethane	107-06-2	gw,so		gw^{b}		gw
1,2-Dichloroethene	540-59-0	gw,sw,so		gw^{b},sw^{b}		gw,sw
1,2-Dichloropropane	78-87-5	so				
1,3-Dichloropropene	10061-01-5	so				
1,4-Dioxane	123-91-1			gw^{f}		gw
2-Butanone	78-93-3	gw,sw,so,sed	sed		sed	sed
2-Hexanone	591-78-6	sw,so				
4-Methyl-2-Pentanone	108-10-1	so				
Acetone	67-64-1	gw,sw,so,sed			sed	
Benzene	71-43-2	gw,so		gw^{b}		gw
Carbon Disulfide	75-15-0	gw,sw,so				
Carbon Tetrachloride	56-23-5	gw,so		gw^{bf}		gw
Chlorobenzene	108-90-7	so				
Chloroform	67-66-3	gw,so,sed	sed	gw^{f},sw^{f}	gw,sed	gw,sw,sed
Chloromethane	74-87-3	gw,so				
Dibromochloropropane	96-12-8	so				
Dichlorodifluoromethane	75-71-8	so				
Ethylbenzene	100-41-4	gw,so	gw	gw^{c}	gw	gw
Ethyl Methacrylate	97-63-2	so				
Isobutyl Alcohol	78-83-1	so				
Methylene Bromide	74-95-3	so				
Methylene Chloride	75-09-2	gw,so,sed		gw^{af}		gw

Table 4.8. (continued)

Analyte	CAS no.	Reference comparison performed	Statistically greater than reference	Greater than comparison criteria	Maximum concentration comparison	Include for nature and extent consideration
Styrene	100-42-5	so				
Tetrachloroethene	127-18-4	gw,sw,so		gw^{b,f},sw^{b,f}	gw	gw,sw
Toluene	108-88-3	gw,so,sed		gw^{b,c}	gw,sed	gw, sed
Trichloroethene	71-55-6	gw,sw,so,sed		gw^{b},sw^{b}	gw,sed	gw,sw, sed
Trichlorofluoromethane	75-69-4	so,sed				
Vinyl Acetate	108-05-4	so				
Vinyl Chloride	75-01-4	gw,so		gw^{b},sw^{b}	gw	gw, sw
Xylene, total	1330-20-7	gw,so			gw	gw
Semivolatile Organics						
2-Chlorophenol	95-57-8	so				
2-Methyl-Phenol	95-48-7	gw				
Benzoic Acid	65-85-0	so,sed			sed	sed
Bis(2-Ethylhexyl)Phthalate	117-81-7	gw,so,sed		gw^{a,f},sw^{a,f},so^{g}	so,sed	gw,sw,so,sed
Chrysene	218-01-9	so				
Di-N-Butyl Phthalate	84-74-2	gw,sw,sed				
Diethyl Phthalate	84-66-2	gw,so				
N-Phenyl Benzamine	122-39-4	so,sed				
Naphthalene	91-20-3	gw				
Pentachlorophenol	87-86-5	so				
Phenol	108-95-2	sed				
PCBs/Herbicides						
2,4,5-T	93-76-5	gw,so				
2,4,5-TP (Silvex)	93-72-1	gw,so				
2,4-D	120-83-2	gw				
AROCHLOR-1254	11097-69-1	gw				

Table 4.8. (continued)

Analyte	CAS no.	Reference comparison performed	Statistically greater than reference	Greater than comparison criteria	Maximum concentration comparison	Include for nature and extent consideration
METALS						
Aluminium	7429-90-5	gw,sw,so,sed	so	$gw^{c,d,e}$,$sw^{c,d,e}$	sw	gw,sw,so
Antimony	7440-36-0	gw,so		gw^{a}		gw
Arsenic	7440-38-2	gw,sw,so,sed		gw^{b},so^{g}	sw,so	gw,sw,so
Barium	7440-39-3	gw,sw,so,sed		$gw^{a,b}$,$sw^{a,b}$		gw,sw
Beryllium	7440-41-7	gw,sw,so,sed		$gw^{a,f}$,$sw^{a,f}$,so^{g}	sw	gw,sw,so
Boron	7440-42-8	gw				
Cadmium	7440-43-9	gw,sw,so,sed	gw	$gw^{b,d,e}$,$sw^{b,d,e}$	gw,sw	gw,sw
Calcium[l]	7440-70-2	gw,sw,so,sed	gw			
Chromium	7440-47-3	gw,so,sed	gw,so	$gw^{b,d,f}$,$sw^{d,f}$	sw	gw,sw,so
Cobalt	7440-48-4	gw,so,sed				
Copper	7440-50-8	gw,sw,so,sed		$gw^{d,e}$,$sw^{d,e}$		gw,sw
Iron[l]	7439-89-6	gw,sw,so,sed	gw,so	$gw^{c,d}$,$sw^{c,d}$		
Lead	7439-92-1	gw,sw,so,sed	sw	$gw^{a,b,d,f}$,$sw^{a,d}$		gw,sw
Magnesium[l]	7439-95-4	gw,sw,so,sed	gw			
Manganese	7439-96-5	gw,sw,so,sed	gw	gw^{c},sw^{c}	sw	gw,sw
Mercury	7439-97-6	gw,sw,so,sed	gw	$gw^{b,d,f}$,sw^{d},so^{g}	sw,so	gw,sw,so
Nickel	7440-02-0	gw,sw,so,sed		$gw^{a,d}$		gw
Osmium	7440-04-2	so,sed			sed	
Potassium[l]	7440-09-7	gw,sw,so,sed	so		sw	
Selenium	7782-49-2	so,sed	so			so
Silicon	7440-21-3	gw				
Silver	7440-22-4	gw,sw,so,sed	gw,sw,sed	$gw^{a,e}$,sw^{e}		gw,sw,sed
Sodium[l]	7440-23-5	gw,sw,so,sed				
Strontium	7440-26-6	gw	gw		gw	gw
Thallium	7440-28-0	gw,so	so			so
Titanium	7440-32-6	gw,so	gw			gw
Tin	7440-31-5	so,sed				

Table 4.8. (continued)

Analyte	CAS no.	Reference comparison performed	Statistically greater than reference	Greater than comparison criteria	Maximum concentration comparison	Include for nature and extent consideration
Vanadium	7440-62-2	gw,sw,so,sed	gw,sw,so		sw	gw,sw,so
Zinc	7440-66-6	gw,sw,so,sed		$gw^{d,e}$,$sw^{d,e}$		gw,sw
Gross alpha and beta						
Gross Alpha	12587-46-1	gw,sw,so,sed	gw	gw^{b},sw^{b}		gw,sw
Gross Beta	12587-47-2	gw,sw,so,sed	gw,sw		sed	gw,sw,sed
Man-Made Radionuclides						
Tritium	10028-17-8	gw,sw,so,sed	gw,sw	$gw^{b,h,i}$,$sw^{b,h,i}$	sw,sed	gw,sw,sed
Cobalt-60	10198-40-0	gw,so,sed		$gw^{h,i}$,$sw^{h,i}$,so^{k},$sed^{j,k}$	so,sed	gw,sw,so,sed
Strontium-89	14158-27-1	so,sed	so			so
Strontium-90	10098-97-2	gw,sw,so,sed	so, gw	$gw^{b,h,i}$,$sw^{b,h,i}$,sed^{j}	sw,so,sed	gw,sw,so,sed
Total Strontium	7440-26-6	gw				
Cesium-137	10045-97-3	gw,sw,so,sed	gw	gw^{h},sw^{h}		gw,sw
Americium-241	14596-10-2	gw,sw,so,sed		$gw^{h,i}$		gw
Plutonium-238	13981-16-3	gw,sw,so,sed		gw^{h},sw^{h}	sw	gw,sw
Plutonium-239/240	0-013	gw,sw,so,sed		gw^{h},sw^{h}	gw,sw	gw,sw
Actinium	14952-40-0	so				
Curium-244	13981-15-2	gw,sw			gw	gw
Curium-242	15510-73-3	gw,sw				
Naturally Occurring Radionuclides						
Potassium-40	13966-00-2	gw,so,sed		gw^{h},sw^{h},$so^{j,k}$,sed^{k}		gw,sw,so,sed
Total Radium		gw				
Uranium Series						
Uranium-238	7440-61-1	gw,sw,so,sed		gw^{h}		gw

Table 4.8. (continued)

Analyte	CAS no.	Reference comparison performed	Statistically greater than reference	Greater than comparison criteria	Maximum concentration comparison	Include for nature and extent consideration
Thorium-234	15065-10-8	gw,sw,so,sed		gw^h,sw^h,so^k,sed^k		gw,sw,so,sed
Uranium-234	13966-29-5	gw,sw,so,sed		gw^h,sed^k	gw	gw,sed
Thorium-230	14269-63-7	gw,sw,so,sed	sw	gw^h,sw^h	gw	gw,sw
Radium-226	13982-63-3	gw,sw,so,sed		gw^h,so^k,sed^k		gw,so,sed
Thorium Series						
Thorium-232	7440-29-1	gw,sw,so,sed		gw^h,sw^h	sw	gw,sw
Radium-228	15262-20-1	gw,sw,so,sed		gw^{h,l},sw^{h,l}		gw,sw
Thorium-228	14274-82-9	gw,sw,so,sed		gw^h,sw^h,so^k,sed^k	sw	gw,sw,so,sed
Radium-224	13233-32-4	gw,sw,so,sed		gw^h,sw^h,so^k,sed^{j,k}	sed	gw,sw,so,sed

[a] Proposed SDWA MCL
[b] SDWA MCL
[c] Tennessee Secondary Drinking Water Regulations
[d] EPA Ambient Water Quality Standard for Protection of Freshwater Organisms, 24 h value
[e] EPA Ambient Water Quality Standard for Protection of Freshwater Organisms, Maximum Value
[f] Proposed RCRA Subpart S Action Levels, Water
[g] Proposed RCRA Subpart S Action Levels, Soil
[h] HEAST drinking water ingestion
[i] EPA four mrem/year gross beta
[j] HEAST soil ingestion
[k] HEAST soil direct exposure
[l] Essential human nutrient

gw=groundwater
sw=surface water
so=soil
sed=sediment

Table 4.9. Risk assessment sample locations by matrix

Area	Sample locations
GROUNDWATER (41)[a]	
Reference (7)	831, 832, 846, 855, 857, 858
WAG 6 (29)	1225, 1227, 1228, 1229, 1231, 1233, 1237, 1241, 1249, 745, 833, 835, 836, 837, 838, 839, 840, 843, 844, 847, 848, 849, 850, 851, 852, 853, 854, 859, 860
Auger (5)	1242, 1243, 841, 842, 845
SURFACE WATER (15)	
Reference (2)	MBBA, MBBG
WAG 6 (12)	SWA1, SWB1, SWC1, WBAB1, WBAB2, WDA1, WDB1, WFBA1, WFBB1, WFBB2, WFBB3, WSP1
Emergency Waste Basin (1)	WEWB
SOIL (27)	
Reference (14)	CML1, CML2, CMR1, CMR2, MBBA1, MBBA2, MBBA3, MBBA4, MBBA5, MBBG1, MBBG2, MBBG3, MBBG4, MBBG5
WAG 6 (10)	1236, 1239, 1240, 1241, SCA02, SCA06, SCA12, SCA20, SCC09, SCD05
Auger Area (3)	1234, 1242, 1243
SEDIMENT (18)	
Reference (4)	BG01, BG02, BG03, BG04
Emergency Waste Basin (14)	EWB01, EWB02, EWB05, EWB06, EWB07, EWB08, EWB09, EWB14, EWB16, EWB17, EWB18, EWB19

[a] Numbers in parentheses indicate number of samples.

Table 4.10. Mann-Whitney comparisons: WAG 6 groundwater

Parameter	CAS no.	Probability reference/WAG 6
METALS/INORGANICS		
Vanadium	7440-62-2	0.003
Manganese	7439-96-5	0.003
Cobalt	7440-48-4	0.097
Nickel	7440-02-0	0.118
Sodium	7440-23-5	0.131
Chromium	7440-47-3	0.348
Iron	7439-89-6	0.138
Magnesium	7439-95-4	0.369
Strontium	7440-24-6	0.033
Calcium	7440-70-2	0.358
Mercury	7439-97-6	0.710
Thallium	7440-28-0	0.010
Cadmium	7440-43-9	0.679
Silver	7440-22-4	0.404
Titanium	7440-32-6	0.014
Aluminum	7429-90-5	0.434
Copper	7440-50-8	0.117
Arsenic	7440-38-2	0.624
Silicon	7440-21-3	0.172
Barium	7440-39-3	0.373
Beryllium	7440-41-7	0.214
Potassium	7440-09-7	0.147
Lead	7439-92-1	0.760
Zinc	7440-66-6	0.214
Boron	7440-42-8	0.573
Antimony	7440-36-0	0.931
Selenium	7782-49-2	0.778
RADIOISOTOPES		
Tritium	10028-17-8	0.000
Gross beta	12587-47-2	0.004
Gross alpha	12587-46-1	0.005
Cesium-137	10045-97-3	0.003
Cobalt-60	10198-40-0	0.561
Strontium-90	10097-97-2	0.006
Total radium		0.801
Radium-224	13233-32-4	NA

Table 4.10. (continued)

Parameter	CAS No.	Probability reference/WAG 6
RADIOISOTOPES (continued)		
Americium-241	14596-10-2	NA
Curium-244	13981-15-2	NA
Potassium-40	13966-00-2	NA
Plutonium-238	13981-16-3	NA
Plutonium-239/240	0-013	NA
Radium-226	13982-63-3	NA
Radium-228	15262-20-1	NA
Thorium-228	14274-82-9	NA
Thorium-230	14269-63-7	NA
Thorium-232	7440-29-1	NA
Thorium-234	15065-10-8	NA
Uranium-234	13966-29-5	NA
Uranium-238	7440-61-1	NA
VOLATILE ORGANICS		
Ethylbenzene	100-41-4	0.013
1,2-Dichloroethene	540-59-0	0.272
1,1-Dichloroethane	75-34-3	0.172
Trichloroethene	79-01-6	0.154
Methylene chloride	75-09-2	0.231
Toluene	108-88-3	0.595
Acetone	67-64-1	0.762
1,1,1-Trichloroethene	71-55-6	0.128
Vinyl chloride	75-01-4	0.439
Tetrachloroethene	127-18-4	0.315
Carbon tetrachloride	56-23-5	0.216
1,1,2-Trichloroethane	79-00-5	0.395
Total xylenes	1330-20-7	0.272
Chloromethane	74-87-3	1.000
Chloroethane	75-00-3	0.621
1,1,2,2-Tetrachloroethane	79-34-5	0.621
Carbon disulfide	75-15-0	0.565
1,2-dichloroethane	107-06-2	0.172
Chloroform	67-66-3	0.568
2-Butanone	78-93-3	0.841
Benzene	71-43-2	0.494
1,1-Dichloroethene	75-35-4	1.000
Trans-1,2-dichloroethene	156-60-5	0.044
2-Methyl naphthalene	91-57-6	0.125
Diethyl phthalate	84-66-2	0.673

Table 4.10. (continued)

Parameter	CAS No.	Probability reference/WAG 6
SEMIVOLATILE ORGANICS		
Di-n-butyl,phthalate	84-74-2	0.052
Naphthalene	91-20-3	0.332
HERBICIDES/PCBs		
2,4,5-TP	93-72-1	01.000
2,4,5-T	93-76-5	0.578
2,4-D	94-75-7	1.000

NA = Not applicable, analyte not detected in either reference or site samples.

Table 4.11. Mann-Whitney comparisons: WAG surface water

Analytical parameter	CAS no.	Probability reference/WAG 6
BASE NEUTRAL ORGANICS		
Fluoranthene	206-44-0	0.008
Di-n-butyl phthalate	84-74-2	1.000
INORGANIC/METAL		
Lead	7439-92-1	0.020
Silver	7440-22-4	0.029
Vanadium	7440-62-2	0.076
Barium	7440-39-3	0.081
Cadmium	7440-43-9	0.096
Zinc	7440-66-6	0.057
Aluminum	7429-90-5	0.105
Magnesium	7439-95-4	0.107
Potassium	7440-09-7	0.132
Calcium	7440-70-2	0.165
Beryllium	7440-41-7	0.220
Iron	7439-89-6	0.355
Sodium	7440-23-5	0.355
Nickel	7440-02-0	0.566
Manganese	7439-96-5	0.643
Copper	7440-50-8	0.927
Arsenic	7440-38-2	1.000
Mercury	7439-97-6	NA
RADIONUCLIDES		
Tritium	10028-17-8	0.008
Gross beta	12587-47-2	0.008
Thorium-230	14269-63-7	0.031
Cesium-137	10045-97-3	0.242
Radium-228	15262-20-1	0.364
Gross alpha	12587-46-1	0.611
Uranium-234	13966-29-5	1.000
Radium-226	13982-63-3	1.000
Thorium-228	14274-82-9	1.000
Thorium-232	7440-29-1	1.000
Uranium-235/236	0-012	1.000
Uranium-238	7440-61-1	1.000
Uranium-234	13966-29-5	1.000
Radium-224	13233-32-4	NA
Thorium-234	15065-10-8	NA
Plutonium-238	13981-16-3	NA

Table 4.11. (continued)

Analytical parameter	CAS no.	Probability reference/WAG 6
RADIONUCLIDES cont'd		
Plutonium-238/239	0-013	NA
Strontium-90	10098-97-2	NA
Curium-244	13981-15-2	NA
Americium-241	14596-10-2	NA
Curium-242	15262-20-1	NA
VOLATILE ORGANICS		
2-Butanone	78-93-3	0.268
1,2-Dichloroethene	540-59-0	0.465
Trichloroethene	79-01-6	0.562
Carbon disulfide	75-15-0	0.705
Acetone	67-64-1	1.000
Tetrachloroethene	127-18-4	1.000
2-Hexanone	591-78-6	NA

NA = Not applicable, analyte not detected in either reference or site samples.

Table 4.12. Mann-Whitney comparisons: WAG 6 soils

Analytical parameter	CAS no.	Reference/WAG 6 comparison
BASE NEUTRAL		
4-Nitrophenol	100-02-7	0.000
Benzyl alcohol	100-51-6	0.000
Phenol	108-95-2	0.000
Pyrene	129-00-0	0.000
Benzo(k)fluoranthene	207-08-9	0.000
Chrysene	218-01-9	0.000
Benzo(a)anthracene	56-55-3	0.000
4-Chloro-3-methyl phenol	59-50-7	0.000
Diethyl phthalate	84-66-2	0.000
Pentachlorophenol	87-86-5	0.000
2-Chlorophenol	95-57-8	0.000
Bis(2-ethyl hexyl) phthalate	117-81-7	0.002
Benzoic acid	65-85-0	0.186
N-nitrosodiphenylamine	86-30-6	0.511
Diphenylamine	122-39-4	NA
HERBICIDE		
2,4,5-TP	93-72-1	NA
2,4,5-T	93-76-5	NA
ORGANICS/METALS		
Selenium	7782-49-2	0.004
Aluminum	7429-90-5	0.006
Chromium	7740-47-3	0.006
Thallium	7440-28-0	0.010
Vanadium	7440-62-2	0.013
Iron	7439-89-6	0.020
Potassium	7440-09-7	0.036
Arsenic	7440-38-2	0.051
Lead	7439-92-1	0.080
Sodium	7440-23-5	0.100
Cadmium	7440-43-9	0.111
Nickel	7440-02-0	0.147
Copper	7440-50-8	0.182
Zinc	7440-66-6	0.222
Tin	7440-36-0	0.351
Manganese	7439-96-5	0.381
Magnesium	7439-95-4	0.394

Table 4.12. (continued)

Analytical parameter	CAS no.	Reference/WAG 6 comparison
ORGANICS/METALS (continued)		
Beryllium	7440-41-7	0.472
Silver	7440-22-4	0.564
Cobalt	7440-48-4	0.730
Mercury	7439-97-6	0.754
Barium	7440-39-3	0.927
Calcium	7440-70-2	0.980
Osmium	7440-04-2	NA
Tin	7440-31-5	NA
RADIONUCLIDES		
Strontium-89	14158-27-1	0.000
Strontium-90	10098-97-2	0.030
Radium-226	13982-63-3	0.004
Cesium-137	10045-97-3	0.016
Thorium-232	7440-29-1	0.063
Thorium-234	15065-10-8	0.171
Uranium-238	7440-61-1	0.254
Thorium-228	14274-82-9	0.356
Potassium-40	13966-00-2	0.383
Uranium-234	13966-29-5	0.428
Gross alpha	12587-46-1	0.464
Gross beta	12587-47-2	0.593
Thorium-230	14269-63-7	0.615
Radium-228	15262-20-1	0.757
Radium-224	13233-32-4	0.859
Uranium-235/236	0-012	1.000
Tritium	10028-17-8	1.000
Plutonium 239/240	0-013	NA
Cobalt-60	10198-40-0	NA
Plutonium-238	13981-16-3	NA
Nickel-63	13981-37-8	NA
Technetium-99	14133-76-7	NA
Promethium-147	14380-75-7	NA
Americium-241	14596-10-2	NA
Iron-55	14681-59-5	NA
Carbon-14	14762-75-5	NA
Actinium	14952-40-0	NA
Iodine-129	15046-84-1	NA

Table 4.12. (continued)

Analytical parameter	CAS no.	Reference/WAG 6 comparison
VOLATILE ORGANICS		
1,2-Dichloroethene	540-59-0	0.132
Carbon disulfide	75-15-0	0.132
2-Butanone	78-93-3	0.137
Acetone	67-64-1	0.145
Tetrachloroethene	127-18-4	0.173
Ethylbenzene	100-41-4	0.278
Styrene	100-42-5	0.278
Cis-1,3-dichloropropene	10061-01-5	0.278
Trans-1,3-dichloropropene	10061-02-6	0.278
1,2-Dichloroethane	107-06-2	0.278
Chlorobenzene	108-90-7	0.278
Dibromochloromethane	124-48-1	0.278
Carbon tetrachloride	56-23-5	0.278
Benzene	71-43-2	0.278
1,1,1-Trichloroethane	71-55-6	0.278
Bromoform	75-25-2	0.2782
Chloromethane	75-27-4	0.278
1,1-Dichloroethane	75-34-3	0.278
1,1-Dichloroethene	75-35-4	0.278
Propylene dichloride	78-87-5	0.278
1,1,2-Trichloroethane	79-00-5	0.278
1,1,2,2-Tetrachloroethane	79-34-5	0.278
Toluene	108-88-3	0.278
2-Chloromethane	75-90-2	0.573
Trichloroethane	79-01-6	0.645
Chloroform	67-66-3	0.667
Total xylenes	1330-20-7	0.789
Acetic acid	108-05-4	0.936
4-Methyl-2-pentanone	108-10-1	0.936
2-Hexanone	591-78-6	0.936
Bromomethane	74-83-9	0.936
Methylene chloride	74-87-3	0.936
Chloroethane	75-00-3	0.936
Vinyl chloride	75-01-4	0.936
Ethylene dibromide	106-93-4	NA
Acrolein	107-02-8	NA
Allylchloride	107-05-1	NA
Propionitrile	107-12-0	NA

Table 4.12. (continued)

Analytical parameter	CAS no.	Reference/WAG 6 comparison
VOLATILE ORGANICS (continued)		
Acrylonitrile	107-13-1	NA
1,4-Dichloro-(e)-2-butene	110-57-6	NA
Pyridine	110-86-1	NA
1,4-Dioxane	123-91-1	NA
Methacrylonitrile	126-98-7	NA
Chloropropene	126-99-8	NA
1,1,1,2-Tetrachloroethane	630-20-6	NA
Methyl iodide	74-88-4	NA
Methylene bromide	74-95-3	NA
Acetonitrite	75-05-8	NA
Trichlorofluoromethane	75-69-4	NA
Dichlorodifluoromethane	75-71-8	NA
2-Methyl-1-propanol	78-83-1	NA
Methylmethacrylate	80-62-6	NA
1,2-Dibromo-3-chloropane	96-12-8	NA
1,2,3-Trichloropropane	96-18-4	NA
Ethylmethacrylate	97-63-2	NA

NA = Not applicable, analyte not detected in either reference or site samples.

Table 4.13. Mann-Whitney comparisons: WAG 6 sediments

Analytical parameter	CAS No.	Reference/WAG 6 comparison
BASE NEUTRAL		
Bis(2-ethylhexyl)phthalate	117-81-7	0.149
Di-n-butylphthalate	84-74-2	0.180
Diphenylamine	122-39-4	0.317
N-nitrosodiphenylamine	86-30-6	0.392
2,6-Dinitrotoluene	606-20-2	0.453
Carboxylic acid	108-95-2	0.591
Benzoic acid	65-85-0	0.826
METALS/INORGANICS		
Silver	7440-22-4	0.008
Arsenic	7440-38-2	0.004
Sodium	7440-23-5	0.013
Nickel	7440-02-0	0.071
Beryllium	7440-41-7	0.134
Cadmium	7440-43-9	0.201
Potassium	7440-09-7	0.202
Magnesium	7439-95-4	0.203
Chromium	7440-47-3	0.203
Cobalt	7440-48-4	0.288
Tin	7440-31-5	0.317
Manganese	7439-96-5	0.396
Lead	7439-92-1	0.428
Iron	7439-89-6	0.524
Barium	7440-39-3	0.524
Zinc	7440-66-6	0.524
Aluminum	7429-90-5	0.595
Copper	7440-50-8	0.750
Vanadium	7440-62-2	0.873
Calcium	7440-70-2	0.915
Selenium	7782-49-2	1.000
Mercury	7439-97-6	NA
Osmium	7440-04-2	NA
RADIONUCLIDES		
Radium-224	13233-32-4	0.089
Strontium-90	100-98-97-2	0.147
Thorium-230	14269-63-7	0.157
Cesium-137	10045-97-3	0.279
Thorium-234	15065-10-8	0.380
Uranium-234	13966-29-5	0.480

Table 4.13. (continued)

Analytical parameter	CAS No.	Reference/WAG 6 comparison
RADIONUCLIDES cont'd		
Thorium-228	14274-82-9	0.480
Thorium-232	7440-29-1	0.480
Uranium-238	7440-61-1	0.480
Radium-228	15262-20-1	0.489
Gross alpha	12587-46-1	0.595
Tritium	10028-17-8	0.560
Radium-226	13982-63-3	0.671
Gross beta	12587-47-2	0.915
Potassium-40	13966-00-2	0.952
Uranium-235/236	0-012	1.000
Strontium-89	14158-27-1	1.000
Plutonium-239/240	0-013	NA
Cobalt-60	10198-40-0	NA
Curium-244	13981-15-2	NA
Plutonium-238	13981-16-3	NA
Nickel-63	13981-37-8	NA
Technetium-99	14133-76-7	NA
Promethium-147	14380-75-7	NA
Americium-241	14596-10-2	NA
Iron-55	14681-59-5	NA
Carbon-14	14762-75-5	NA
Iodine-129	15046-84-1	NA
Curium-242	15510-73-3	NA
VOLATILE ORGANICS		
2-Butanone	78-93-3	0.007
Chloroform	67-66-3	0.013
Trichloroethene	79-01-6	0.274
Toluene	108-88-3	0.303
Trichlorofluoromethane	75-69-4	0.317
Methylene chloride	75-09-2	0.790
Acetone	67-64-1	0.956

NA = Not applicable, analyte not detected in either reference or site samples.

Table 4.14. Constituents for consideration in nature and extent

Analyte	CAS no.	Groundwater	Surface water	Soil	Sediment
VOLATILE ORGANIC COMPOUNDS					
1,2-Dichloroethane	107-06-2	X			
1,2-Dichloroethene	540-59-0	X	X		
2-Butanone	78-93-3				X
Acetone	67-64-1	X			X
Benzene	71-43-2	X			
Carbon Tetrachloride	56-23-5	X			
Chloroform	67-66-3	X	X		X
Ethylbenzene	100-41-4	X			
Methylene Chloride	75-09-2	X			
Tetrachloroethene	127-18-4	X	X		
Toluene	108-88-3	X			
Trichloroethene	71-55-6	X	X		
Vinyl Chloride	75-01-4	X			
SEMIVOLATILE ORGANIC COMPOUNDS					
Benzoic acid	65-85-0			X	
Bis(2-ethylhexyl)phthalate	117-81-7	X	X		
INORGANICS					
Aluminum	7429-90-5	X	X	X	
Antimony	7440-36-0	X			
Arsenic	7440-38-2			X	
Barium	7440-39-3	X	X		
Beryllium	7440-41-7	X	X	X	X
Cadmium	7440-43-9	X	X		
Chromium	7440-47-3	X	X	X	
Copper	7440-50-8	X	X		
Lead	7439-92-1	X	X		
Manganese	7439-96-5	X	X		
Mercury	7439-97-6	X	X	X	
Nickel	7440-02-0	X			
Osmium	7440-04-2				X
Selenium	7782-49-2			X	
Silver	7440-22-4	X	X		X
Strontium	7440-26-6	X			
Thallium	7440-28-0	X		X	
Titanium	7440-32-6	X			
Vanadium	7440-62-2	X	X	X	
Zinc	7440-66-6	X	X		

Table 4.14. (continued)

Analyte	CAS No.	Groundwater	Surface water	Soil	Sediment
GROSS ALPHA AND GROSS BETA					
Gross alpha	12587-46-1	X	X		
Gross beta	12587-47-2	X	X		X
MAN-MADE RADIONUCLIDES					
Tritium	10028-17-8	X	X		
Cobalt 60	10198-40-0	X	X	X	X
Strontium-89	14158-27-1			X	
Strontium-90	10098-97-2	X	X	X	X
Cesium-137	10045-97-3	X	X		
Americium-241	14596-10-2	X			
Plutonium-238	13981-16-3	X	X		
Plutonium-239/240	0-013	X	X		
Curium-244	13981-15-2	X			
NATURALLY OCCURRING RADIONUCLIDES					
Potassium-40	13966-61-1	X	X	X	X
URANIUM SERIES					
Uranium-238	7440-61-1	X			
Thorium-234	15065-10-8	X	X	X	X
Uranium-234	13966-29-5	X			X
Thorium-230	14269-63-7	X			
Radium-226	13982-63-3	X		X	X
THORIUM SERIES					
Thorium-232	7440-29-1	X	X		
Radium-228	15262-20-1	X	X		
Thorium-228	14274-82-9	X	X	X	X
Radium-224	13233-32-4	X	X	X	X

Table 4.15. Summary of WAG 6 groundwater sampling by previous investigations

Source	Wells	Dates sampled
Yager and Craig 1989	1035-1040	6/87, 8/87, 12/87, 3/88, 6/88, 9/88, 1/89
Solomon 1988	741	5/88
Poreda, Cerling, and Solomon 1988	108, 109, 345, 347, 370, 371, 373, 374, 382, 383, 386, 381, 388	9/86
Sherwood & Borders 1987	739	3/86, 9/86
Vaughan et. al. 1982	ETF-1 through ETF-12 T-1, T-5, T-7, T-17, T-18, T-20	4/81, 8/81
Solomon et al. 1988	647	7/86
	648	1/87
	649, 650	3/87
	511	3/86
Doyle and Taylor 1986	279, 382, 371, 272, 380, 305	
Morrissey 1990	T1, T7, T10, T17, T18, T20, T22, T36	10/89
Boegly 1984	109, 284, 296, 356, 370, 372,	8/83
	381	8/83, 12/83
	313	3/82
	318	8/83, 12/83, 6/84
	345, 388	12/83, 3/84, 6/84
	355	6/84
	376	12/83, 6/84
	382	
Ashwood and Spalding 1990	1036, 1039	6/87, 8/87, 12/87, 3/88, 6/88, 9/88 1/89, 3/89, 6/89, 9/89, 9/89, 12/89, 5/90
Huff 1988a,b	T-1, T-5, T-7, T-017, T-18, T-20	10/88
Davis et al. 1989	T-A-R-A-1 through TARA-13, 367, 646	4/87, 11/87, 6/88, 8/88
Dreier & Toran 1989	27, 36, 41, 42, 123, 127, 129 135, 137, 139, 145, 150, 152, 153, 156, 160, 161, 163, 166, 168, 171, 173, 177, 177A, 279, 317	Spring 1976
Wickliff, Morrissey, and Ashwood 1990	4, 5, 6, 7, 8, 11, 12, 13	2/90

Table 4.15. (continued)

Source	Wells	Dates sampled
Ashwood 1991	8, 11, 15, 19, 25, 27, 28, 30, 32, 35, 38, 40, 42, 43, C520, C595, FISI, FIS, AUG4	12/90
Wickliff, Morrissey, and Ashwood 1991	4, 5, 6, 8, 9, 10, 11, 12, 15, 18, 19A, 198, 20, 22, 23, 24, 25, 26, 27, 28, 30, 32, 36, 38, 39, 40, 41, 42, 43, 44, 45, 46	2/91
Davis et al. 1987	49-Trench Area French Drain	9/83, 2/84, 4/84, 5/85, 10/85, 1/86, 3/86, 10/86

Table 4.16. Summary of WAG 6 groundwater analysis in previous investigations

Source[a]	Tritium	Gross Alpha	Gross Beta	Gross Gamma	Cs-137	Sr-90	Co-60	C-14	Tc-99	Total Radium	VOC	BNA	Metals	Cations/ Anions	TOC/ TOX
Yager and Craig 1989	X	X	X		X		X				X	X	X	X	XX
Solomon 1988	X														
Poreda, Craig and Soloman 1988	X														
Sherwood & Borders 1987	X	X	X		X	X	X						X	X	
Vaughn et al. 1989	X	X			X	X	X								
Solomon et al. 1988	X	X	X	X	X		X	X	X	X	X	X			
Doyle and Taylor 1986	X				X	X	X								
Morrisey 1990	X	X	X		X	X	X								
Boegly 1984	X	X			X	X	X						X		
Ashwood and Spalding 1990	X														
Huff 1988a,b	X			X	X										
Davis et al. 1989	X	X	X												
Dreier and Toran 1989	X	X	X		X	X	X								
Tamura et al. 1980					X	X	X								
Wickliff, Morrissey and Ashwood 1990		X	X		X		X								
Ashwood 1991		X	X		X		X								
Wickliff, Morrissey and Ashwood 1991		X	X		X	X									
Davis et al. 1987	X	X			X	X	X								

[a] For wells sampled by each investigation, see Table 4.15.

VOCS DETECTED WITH MCLs IN EFFECT	MCL (ug/L)	745	831	832	833	835	836	837	838	839	840	841	842	843	844	845	846	847	848	849	850	851	852
TRICHLOROETHENE	5					2					1	19	510	1	2					1300	5		
CARBON TETRACHLORIDE	5												96	0.5									
VINYL CHLORIDE	2																			63			
1,1,1-TRICHLOROETHANE	200													0.4					0.7	4			
1,2-DICHLOROETHANE	5												44							2			
BENZENE	5					3														7	20		110
1,1-DICHLOROETHENE	7			1									0.5							1			
*TOTAL TRIHALOMETHANES	100						6				2		97	2	2					1	190		
VOCS DETECTED, MCL EFFECTIVE JANUARY 30, 1992																							
*1,2-DICHLOROETHENE	70											18	27	11	8	10			15	260			
(CIS- AND TRANS-)																							
ETHYL BENZENE	700																				700		
TETRACHLOROETHENE	5		3				5			1		16	14							5			4
TOLUENE	1000																				2000		
XYLENES (TOTAL)	10,000																				3800		11
VOCS DETECTED, NO MCLs PROMULGATED																							
CHLOROMETHANE									15	5				4									
1,1-DICHLOROETHANE													9						17	15			
METHYLENE CHLORIDE												1	1	2	2								13
ACETONE																			22		28		
2-BUTANONE																							
CARBON DISULFIDE																							
1,1,1,1-TETRACHLOROETHANE																							
1,1,2-TRICHLOROETHANE																				3			
ETHYL CHLORIDE													4										
1,4-DIOXANE						63							6						5		6	160	75
ETHYL ETHER						99															29	8	180
4-METHYL-2-PENTANONE																	2						
MAXIMUM CONCENTRATION SUMMED		0	3	0	1	167	11	0	15	6	3	54	808.5	20.9	14	10	2	0	59.7	1661	6778	168	393

5.18G 4104.182

Table 4.17. Maximum concentrations of VOCs detected in groundwater at WAG 6 from ORNL compliance wells and RFI wells.

WELL NUMBER																													
853	854	855	856	857	858	859	860	1225	1226	1227	1228	1229	1230	1231	1232	1233	1234	1235	1236	1237	1238	1239	1240	1241	1242	1243	1244	1245	1249
														7		650													
										1						88													
																4													
																2													
										1						4													
																6													
										1						8											3		
								2			6					680													
						2																							
			4					11				2		69		6800													
						5					2																		
						11					3					2													
																13													
25	20															12													
		3						120			280																		
		2										2																	
																				22									
																53													
																3													
																1													
25	20	5	4	0	18	0	133	DRY 0	0	294	4	0	DRY 0	76	DRY 0	8326	0	DRY 0	0	22	0	0	0	0	0	0	3	0	0

Table 4.18. Herbicides detected in groundwater at WAG 6

Well number	Herbicides (μg/L)		
	2,4,5-TP	2,4,5-TP	2,4-D
745			2
831	2		2
833			12
835	0.3		0.8
836			0.6
838			2
839		0.3	1
841	0.3		0.8
842			1
843			0.4
844			4
855			0.3
856			2
857	0.2		0.3
858			2
860			9

Table 4.19. Historical comparison of filtered versus unfiltered results for selected metals detected in groundwater at WAG 6

WELL	METAL	MCL (UG/L)		SAMPLING EVENT									
				1	2	3	4	5	6	7	8	9	10
1225	CADMIUM	5	FILTERED			5.0 U	4.0 J	5.3 J	3.9 J	4.3 J			
			UNFILTERED			5.0 U	11.2	94.5	5.1	9.4			
	SILVER	50	FILTERED			5.0 U	14.7	13.5	11.6	14.5 J			
			UNFILTERED			5.0 UJ	64.9 J	72.6	12.1	14.5			
	LEAD	50	FILTERED			4.0 UJ	2.0 U	16.5	2.0 U	3.2 J			
			UNFILTERED			6.0	7.5	59.9	5.1	14.3			
	ANTIMONY		FILTERED			30.0 U	30.0 U	30.0 U	30.0 U	30.0 U			
			UNFILTERED			30.0 U	30.0 UJ	91.5	30.0 U	2.0 UJ			
1233	BARIUM	1000	FILTERED			232.0	314.0	3090.0	399.0	288.0			
			UNFILTERED			1130.0 J	3280.0	4120.0	969.0	969.0			
	CADMIUM	5	FILTERED			5.0 U	2.3 J	3.0 J	4.2 J	2.8 J			
			UNFILTERED			5.0 U	17.2	7.4 J	8.3	15.3			
	LEAD	50	FILTERED			3.0 UJ	2.0 U	8.6	5.0	2.3 J			
			UNFILTERED			52.0	25.2	35.3	31.4	27.3			
	ARSENIC	50	FILTERED			20.0 U	2.0 U	6.5 J	2.0 U	2.0 U			
			UNFILTERED			20.0 U	66.5 J	47.1	2.2 J	2.0 U			
1228	BARIUM	1000	FILTERED			446.0	822.0	600.0	1030.0	1330.0		735.0	
			UNFILTERED			1530.0	1190.0	4830.0	4580.0				
	CADMIUM	5	FILTERED			5.0 U	5.0 U	3.6.0 J	3.5.0 U	3.9 J		5 U	
			UNFILTERED			5.0 U	5.0 UJ	48.2 J	4.3 J				
	LEAD	50	FILTERED			2.0 UJ	2.0 U	3.9 J	2.0 U	2.0 UJ		2.8 UJ	
			UNFILTERED			22.0	7.0 J	72.1	14.6	9.8		56.6	
	ARSENIC	50	FILTERED			3.0 UJ	2.0 U	2.0 UJ	2.0 U	2.0 UJ		2.0 UJ	
			UNFILTERED			2.0 J	3.0	65.6	4.6 J	6.0 J		10.1	

Table 4.19. (continued)

WELL	METAL	MCL (UG/L)		SAMPLING EVENT 1	2	3	4	5	6	7	8	9	10
1231	BARIUM	1000	FILTERED			114 J	75.5 J		251	118 J			
			UNFILTERED			2290	443 J		203	131 J			
	CADMIUM	5	FILTERED				5 U	2 U		2 U	2 U		
			UNFILTERED				5 J	17.8		2 U	2.3 J		
	LEAD	50	FILTERED				2	2 U		2 UJ	2 UJ		
			UNFILTERED				100	98.4		13.6 J	8.3		
1227	BARIUM	1000	FILTERED				77 J	93 J	183 J		186 J		
			UNFILTERED				390	178 J	2940		246		
850	BARIUM	1000	FILTERED	1000 U	1100	1100	641			1010			
			UNFILTERED	1000 U	1200	1100	1030			1260			
851	BARIUM	1000	FILTERED	1000 U	1200	1000 U	510	646		652			
			UNFILTERED	1000 U	1400	1000 U	576	692		651			
	MERCURY	2	FILTERED	0.1 U	0.1 U	0.1 U		.2 UJ		.2 UJ			
			UNFILTERED	0.1 U	0.1 U	0.1 U	.2 U	2.4 J					
852	BARIUM	1000	FILTERED	2900	3400	2700	2370	2360		2690		735.0	
			UNFILTERED	2500	3300	2600	2490	2580		27100			
857	LEAD	50	FILTERED	20 U	20 U	20 U	20 U						3.1 J
			UNFILTERED	20 U	60	20 U	20 U						16.4

U = Undetected
J = Indicates estimated concentration below method detection limit

Table 4.20. Surface water sampling event conditions

Date	Groundwater table	Flow conditions	ICM cap status
2/8-9/89	High	Base	Under Construction
4/18-21/89	High	Base	Complete
5/9/89	High	Storm	Complete
6/9/89	High	Storm	Complete
9/5-6/89	Low	Base	Complete
9/26-27/89	Low	Storm	Complete

Table 4.21. Summary of historical surface water analysis

Data source	Sample locations	Gross alpha	Gross beta	Cobalt-60	Cesium-137	Strontium-90	Tritium
Davis et al. 1987	MS-1, MS-2, MS-3, MS-4						X
Boegly 1984	ETF Flume II	X		X	X	X	X
Davis and Marshall 1988	Stream FB SR-90 seep	X		X	X	X	X
Miller et al. 1989	102, 103, 104, 201, 303, 305, 306, 301, 302, 308, 322, 304, 321	X	X	X	X	X	X

Table 4.22. Summary of WAG 6 sediment sampling by previous investigations

Source	Site/Sample	Dates Sampled
Cerling and Huff 1985	15, 16, 17	7/85, 8/85
Cerling and Spalding 1981	234, 235, 236, 237, 238, 239, 240, 241, 242	10/78
	341, 342, 343, 344, 345, 346, 347, 348, 349, 350, 351, 352, 353, 354, 355, 356, 357, 358, 359, 360, 361	11/78
Huff 1986	6, 7, 34, 35, 36, 37	1978, 1985
Miller, Black, and Craig 1989	PRE-1-C-101, PRE-1-F-102, PRE-1-C-102, PRE-1-C-103, PRE-1-C-104, PRE-1-F-201, PRE-1-C-202, PRE-1-C-301, PRE-1-C-302, PRE-1-F-303, PRE-1-C-303, PRE-1-C-304, PRE-1-C-305, PRE-1-C-306, PRE-1-C-307, PRE-1-C-308, PRE-1-C-309, PRE-1-C-310, PRE-1-C-321, PRE-1-C-403	10/88
	DUR-1-C-101, DUR-1-F-102, DUR-1-C-102, DUR-1-C-103, DUR-1-C-104, DUR-1-F-201, DUR-1-C-201, DUR-1-C-202, DUR-1-C-301, DUR-1-C-302, DUR-1-F-303, DUR-1-C-303, DUR-1-C-304, DUR-1-C-305, DUR-1-C-306, DUR-1-C-307, DUR-1-C-308, DUR-1-C-309, DUR-1-C-310, DUR-1-C-321	1/89
	DUR-2-C-101, DUR-2-F-102, DUR-2-C-102, DUR-2-C-103, DUR-2-C-104, DUR-2-F-201, DUR-2-C-201, DUR-2-C-202, DUR-2-C-301, DUR-2-C-302, DUR-2-F-303, DUR-2-C-303, DUR-2-C-304, DUR-2-C-305, DUR-2-C-306, DUR-2-C-307, DUR-2-C-308, DUR-2-C-309, DUR-2-C-310, DUR-2-C-321	3/89
	DUR-3-C-101, DUR-3-C-104, DUR-3-C-201, DUR-3-F-201, DUR-3-C-202, DUR-3-C-301, DUR-3-C-302, DUR-3-F-303, DUR-3-C-303, DUR-3-C-304, DUR-3-C-305, DUR-3-C-306, DUR-3-C-307, DUR-3-C-308, DUR-3-C-309, DUR-3-C-310	4/89
	POS-1-C-101, POS-1-C-104, POS-1-F-201, POS-1-C-201, POS-1-C-202, POS-1-C-301, POS-1-C-302, POS-1-F-303, POS-1-C-303, POS-1-C-304, POS-1-C-305, POS-1-C-306, POS-1-C-307, POS-1-C-308, POS-1-C-309, POS-1-C-310, POS-1-C-321	6/89

Table 4.23. Summary of WAG 6 sediment sampling analyses by previous investigations

	Co-60	Cs-137	Sr-90	Gross beta	Gross alpha	Zn
Cerling and Huff 1985	X	X	X			X
Cerling and Spalding 1981	X	X	X			
Huff 1986	X	X	X			
Miller, Black, and Craig 1989	X	X	X	X	X	X

Table 4.24. WAG 6 soils investigation areas

Area	Soil borings included	Waste areas included
1	SCB-02 through SCB-07, SB-07, 1240, 1241	Emergency Waste Basin, 19 Trench area
2	SCA-01 through SCA-06, SCA-08 through SCA-20	Central waste disposal area, low-activity silo area 1, 49 Trench area, high-activity silo area, high-activity trench area
3	SCE-01 through SCE-11, 1234, 1242, 1243	East waste disposal area, solvent auger hole area
4	SCD-01 through SCD-09, 1235	South waste disposal area
5	SDMS-01, 1236	Hill Cut Test Facility
6	SCE-01 through SCE-11, 1237, 1239	West waste disposal area
7	Yager and Craig (1989) soil borings	Tumulus Pad area

Table 4.25. Summary of WAG 6 soil sampling and analysis in previous investigations

Source	Sample Location	Dates Sampled	Gross alpha	Gross beta	K-40	Cs-137	Co-60	Sr-90	VOC
Davis et al. 1989	1 through 22, 2A, 2B, 2C, 3A, 3B, 3C, 4A, 4B, 4C, 5A, 5B, 5C, 6A, 6B, 6C	2/29/88 - 4/18/88				X	X	X	X
Yager and Craig 1989	3	2/87, 12/87, 3/88, 7/88, 9/88,	X	X	X	X	X	X	X
	4	1/89							
	5	2/87, 9/87, 12/87, 3/88, 7/88,							
	6	9/88, 1/89							
	7	9/87, 3/88, 7/88, 9/88, 1/89							
	9	9/87, 3/88, 7/88, 9/88, 1/89							
	12C	1/89							
	56C	9/87, 12/87, 3/88, 9/88, 1/89							
	78C	2/87, 9/87, 12/87							
	1011C	2/87, 12/87							
		12/87, 3/88, 7/88, 9/88							
		3/88, 7/88, 9/88, 1/89							

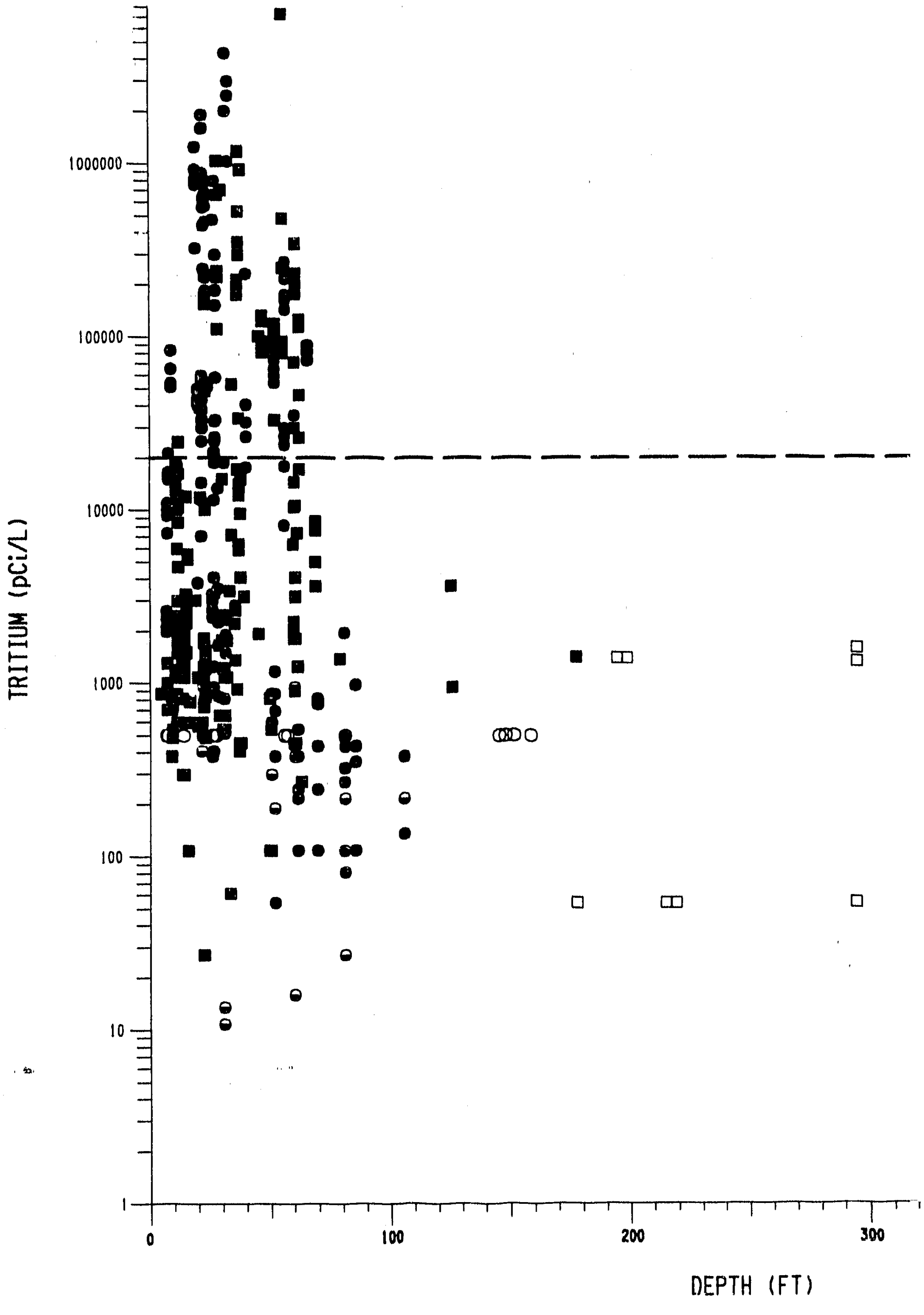
Table 4.26. Off-WAG migration of radionuclides

Perimeter segment	Medium/ radionuclides	Discussion
AREAS LIKELY FOR OFF-WAG MIGRATION OF CONTAMINANTS		
H1	<u>Surface water</u> Tritium Strontium-90	Storm flow sampling identified significant flux of contaminants in surface water.
	<u>Groundwater</u> Tritium Strontium-90	Wells upgradient are contaminated.
H2	<u>Surface water</u> Tritium	Storm flow sampling identified significant flux of tritium off site.
	<u>Groundwater</u> Tritium	Concentrations exceeding MCL detected in several upgradient wells.
H3	<u>Groundwater</u> Tritium Cobalt-60 Strontium-90	Compliance and RFI monitoring data indicate presence of radionuclides in groundwater.
AREAS NOT LIKELY FOR OFF-WAG MIGRATION OF CONTAMINANTS		
L		Upgradient to waste disposal areas.

Table 4.27. Off-WAG migration of chemicals

Perimeter segment	Medium/ chemicals	Discussion
AREAS LIKELY FOR OFF-WAG MIGRATION OF CONTAMINANTS		
H1	Surface water None	No significant VOC concentrations detected during storm flow sampling.
	Groundwater VOCs	Upgradient wells contaminated, monitor unconsolidated zone and bedrock zone. Groundwater wells in this vicinity monitor bedrock zone only.
H2	Surface water VOCs	Low concentration detected during storm flow.
	Groundwater None	Well positioned between creeks DA and DB have had relatively low concentrations of VOCs detected in it.
H3	Surface water None	No surface water features in this area.
	Groundwater VOCs	VOCs detected during compliance and RFI monitoring.
AREAS NOT LIKELY FOR OFF-WAG MIGRATION OF CONTAMINANTS		
L		Upgradient to waste disposal areas.

Section 4 Figures



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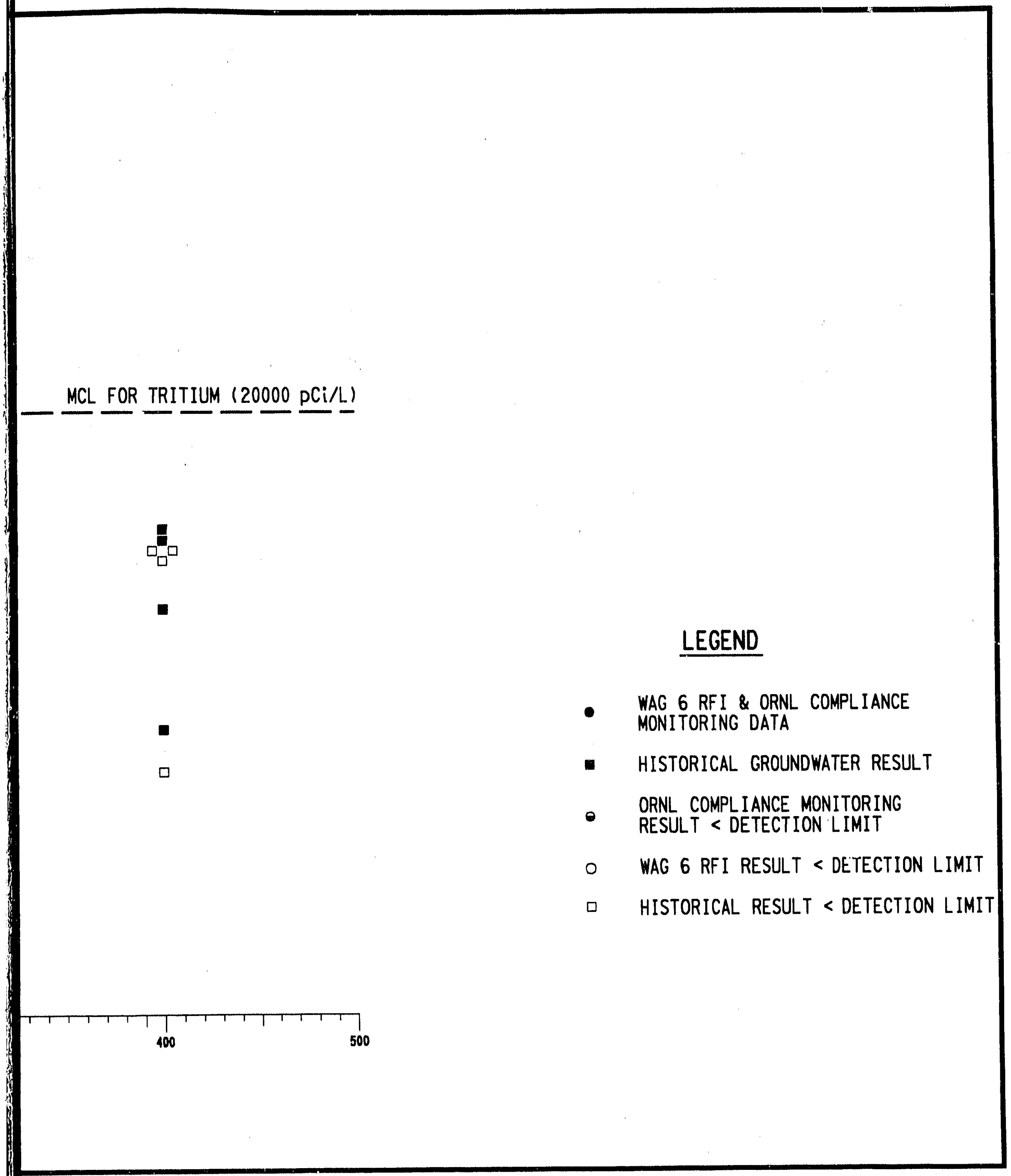
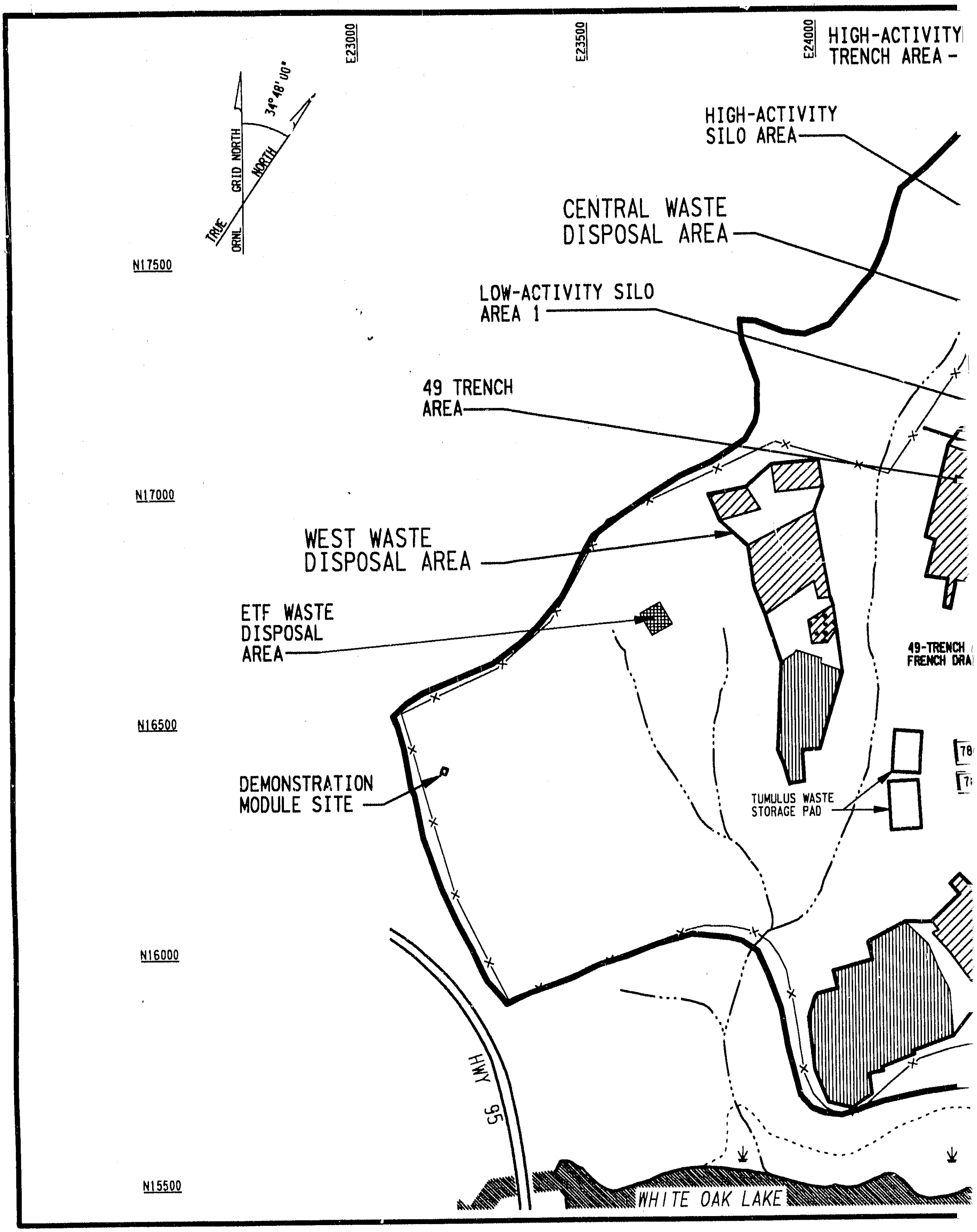


Fig. 4.1. Vertical distribution of tritium in WAG 6 groundwater.



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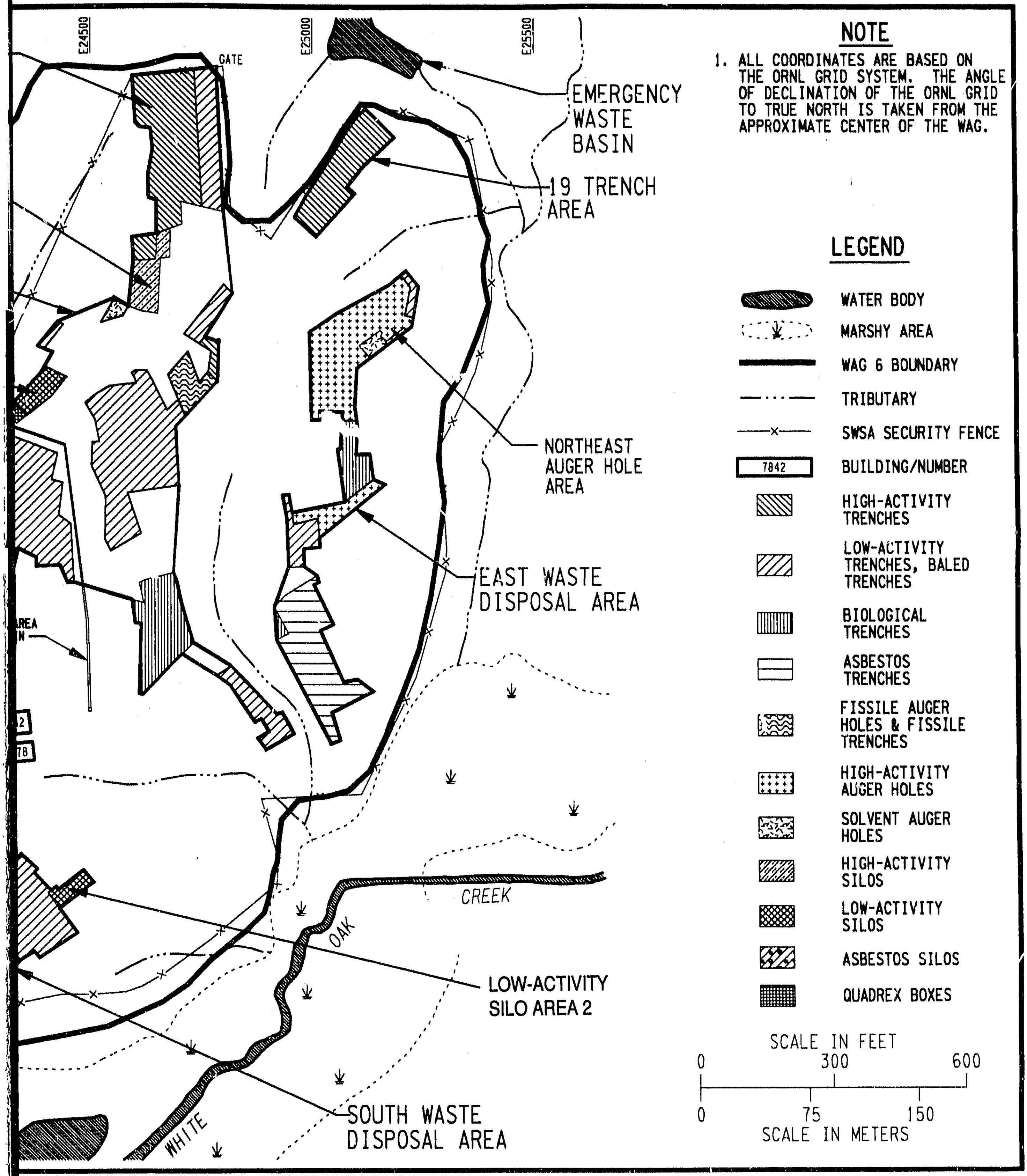
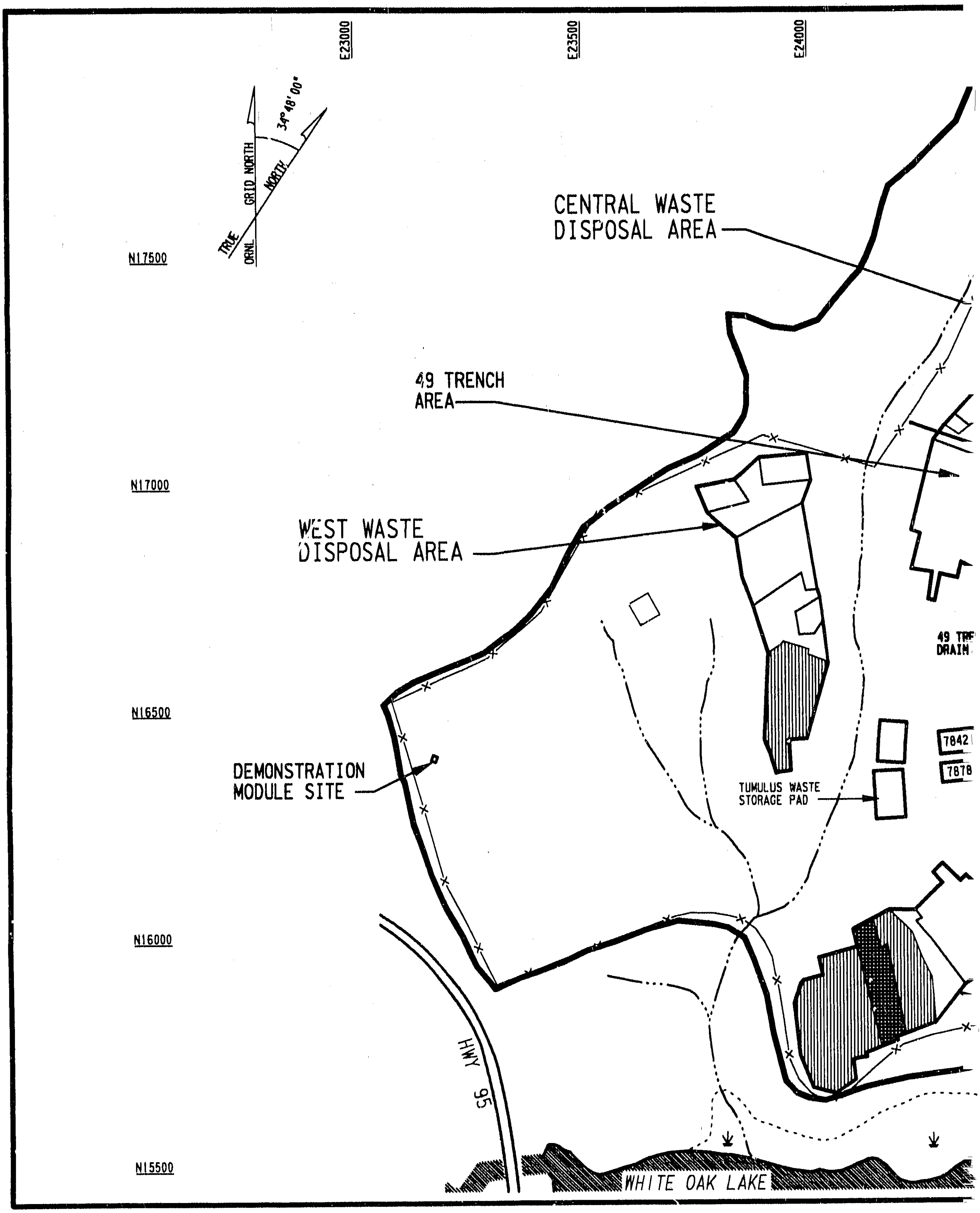
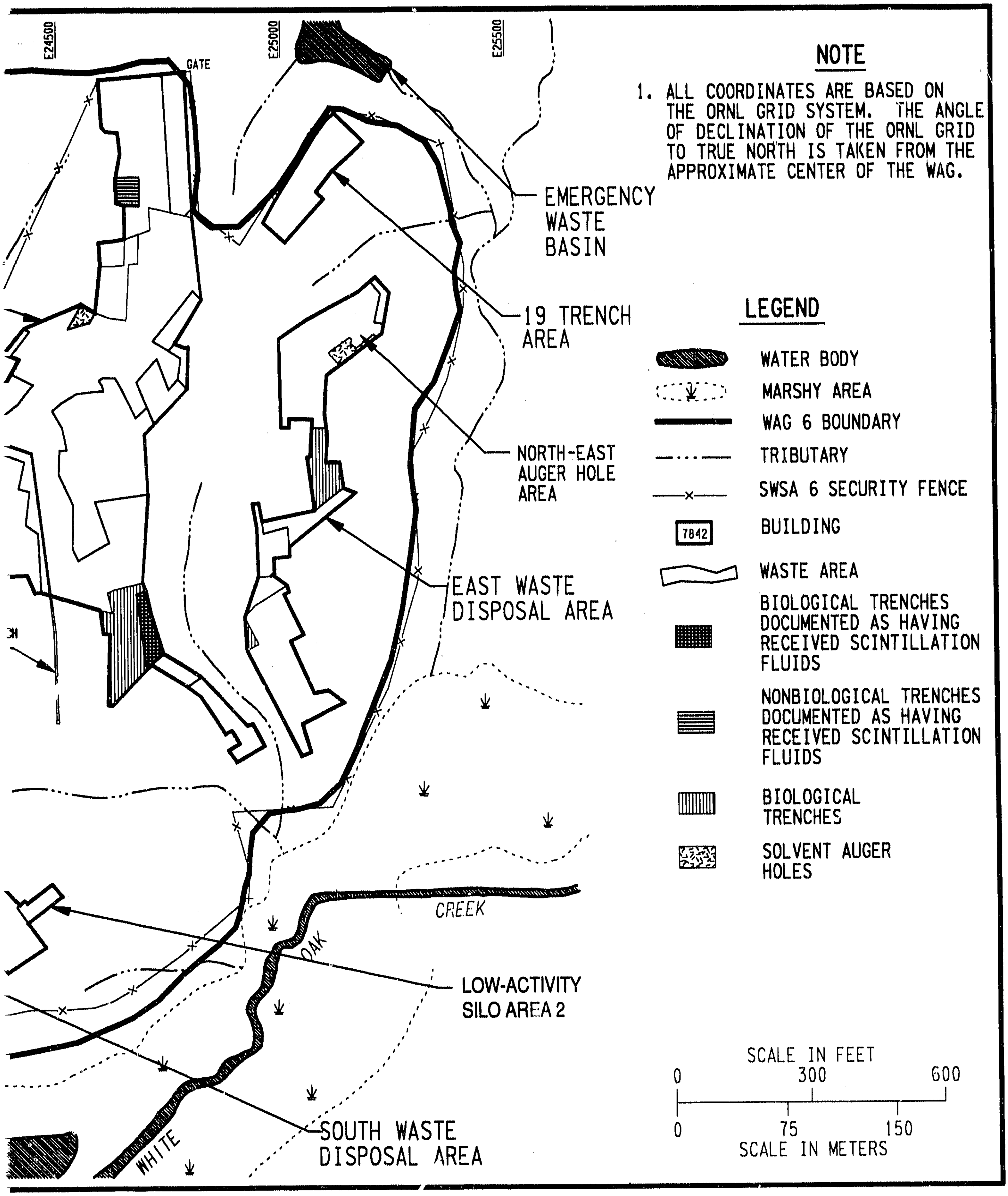


NOTE

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.

Fig. 4.2. WAG 6 waste disposal areas.


E23000
E23500
E24000
TRUE
ORNL
GRID NORTH
NORTH
34°48'00"
N17500
N17000
N16500
N16000
N15500
CENTRAL WASTE DISPOSAL AREA
49 TRENCH AREA
WEST WASTE DISPOSAL AREA
DEMONSTRATION MODULE SITE
49 TR
DRAIN
7842
7878
TUMULUS WASTE STORAGE PAD
HWY 95
WHITE OAK LAKE



NOTE

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.

Fig. 4.3. WAG 6 solvent auger hole and biological trench areas.

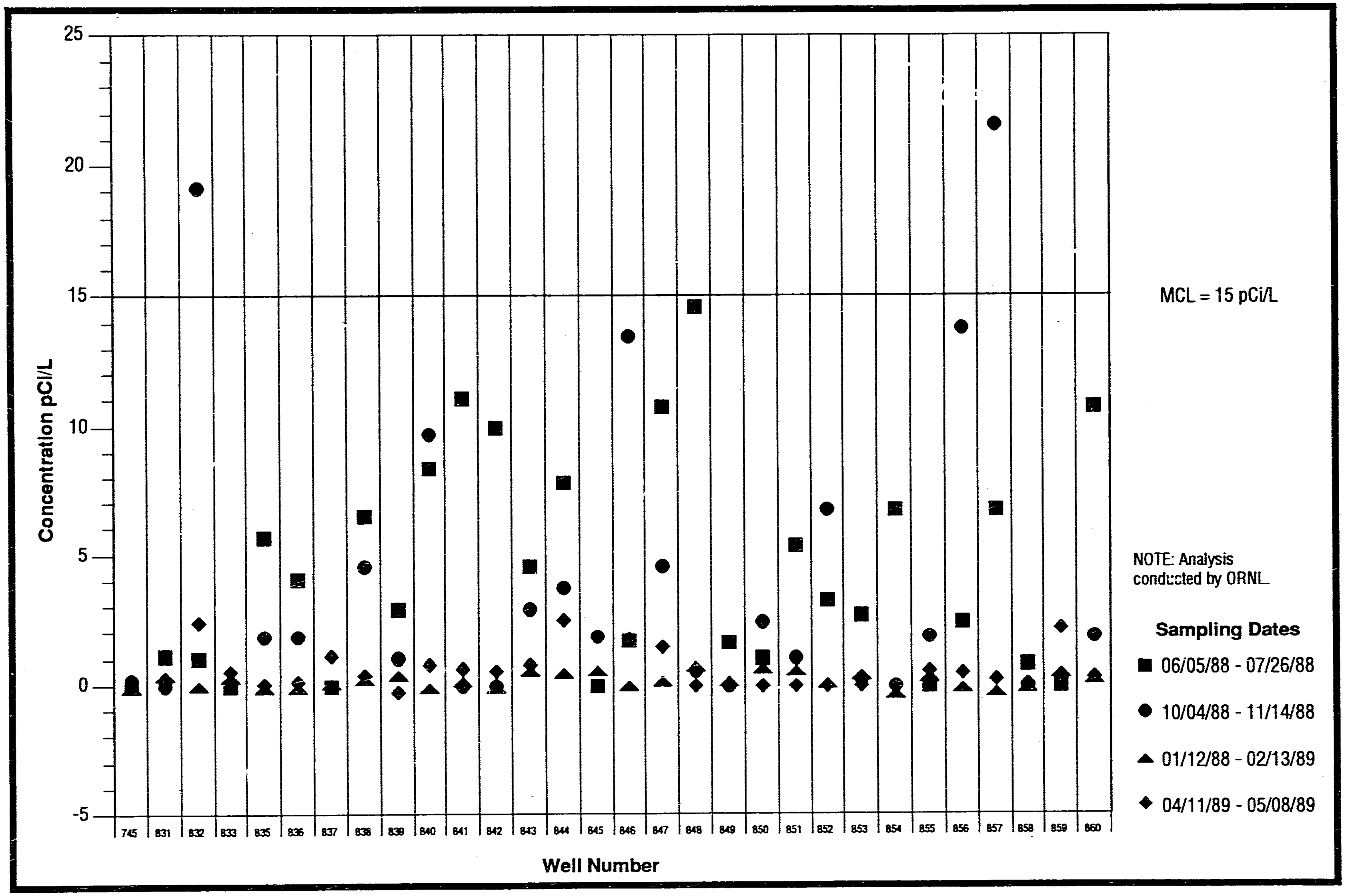


Fig. 4.4. Gross alpha concentrations detected in groundwater samples from ORNL RCRA compliance wells at WAG 6.

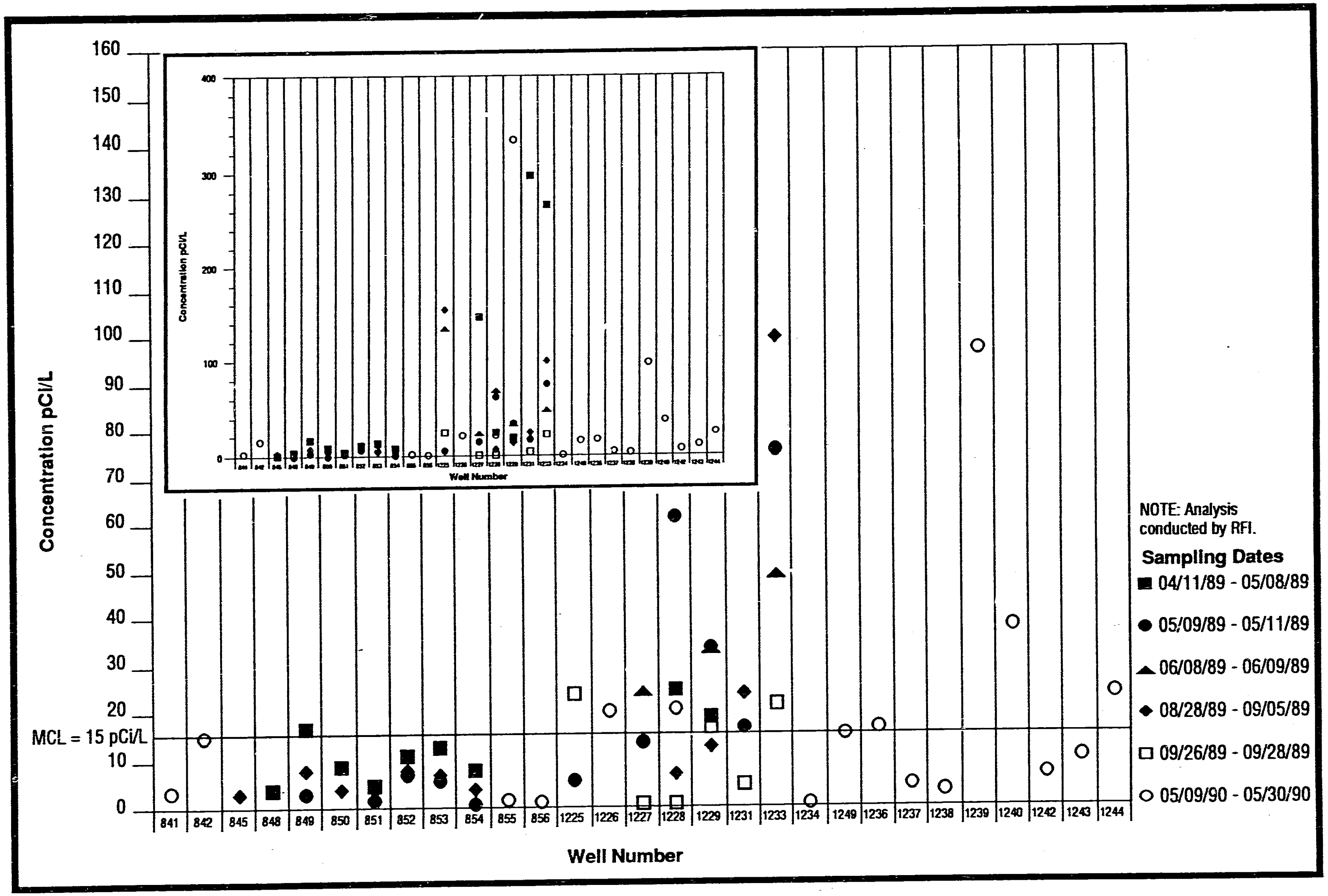


Fig. 4.5. Gross alpha concentrations detected in groundwater samples from RFI wells and some ORNL RCRA compliance wells at WAG 6.

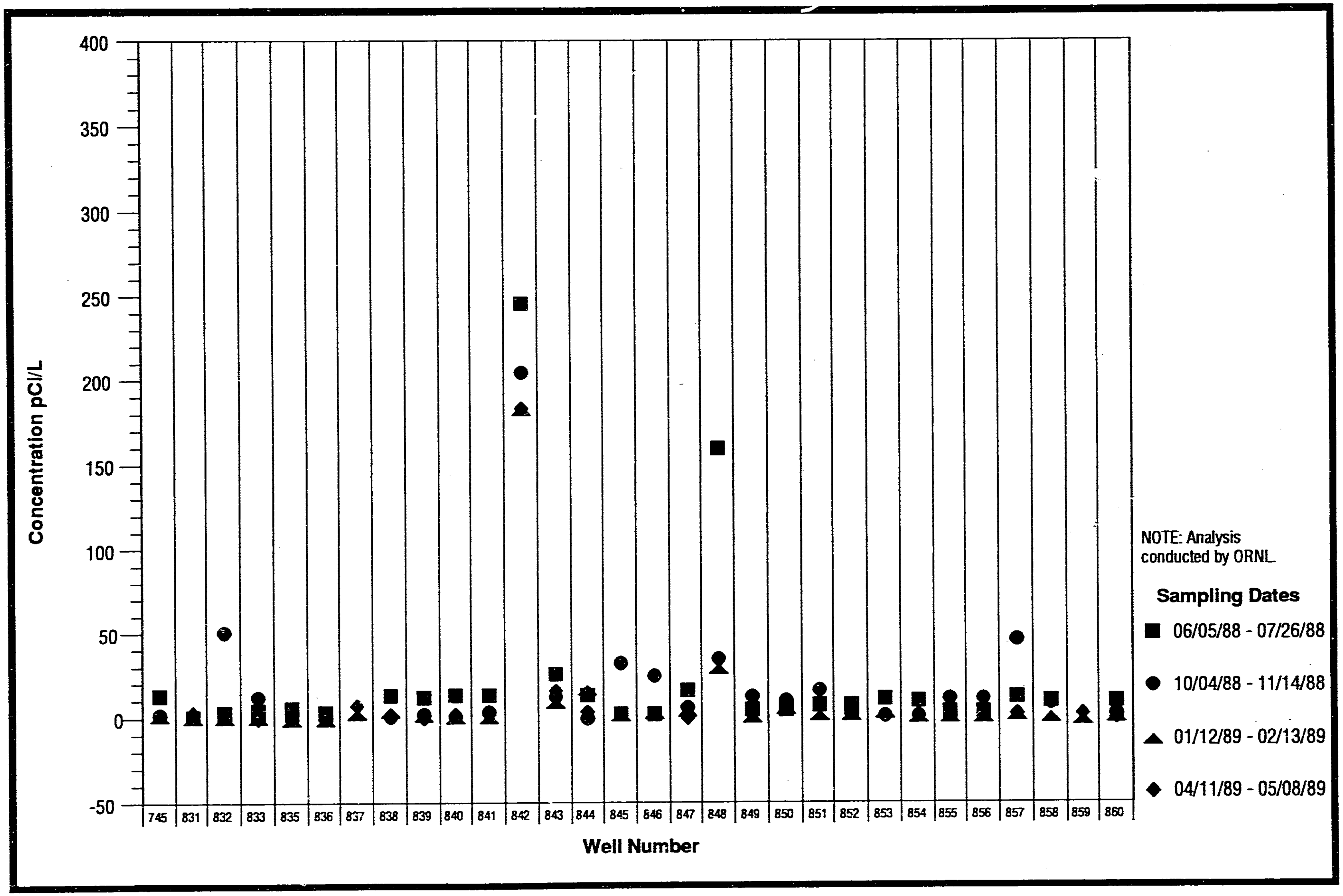


Fig. 4.6. Gross beta concentrations detected in groundwater samples from ORNL RCRA compliance wells at WAG 6.

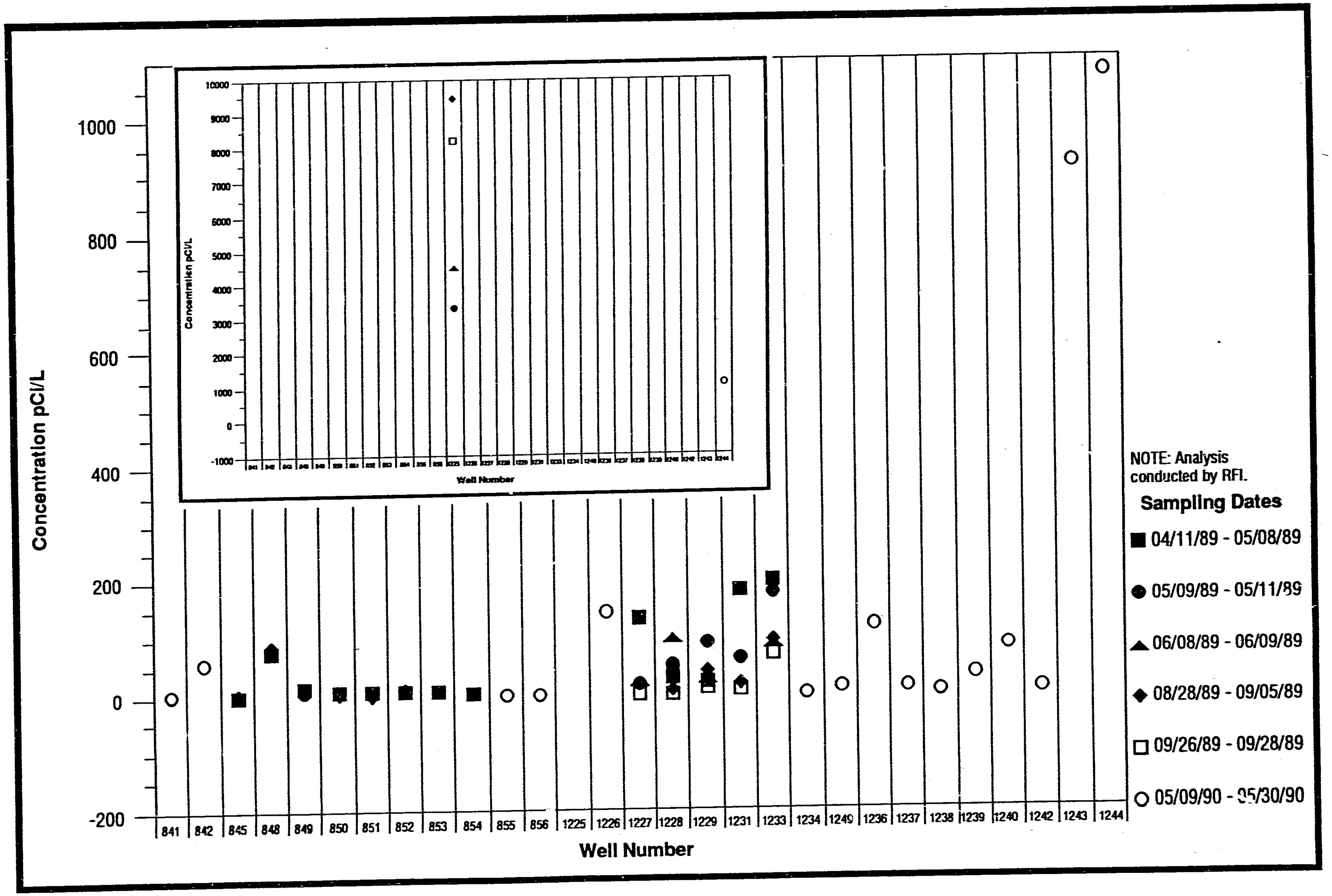


Fig. 4.7. Gross beta concentrations detected in groundwater samples from RFI wells and some ORNL RCRA compliance wells at WAG 6.

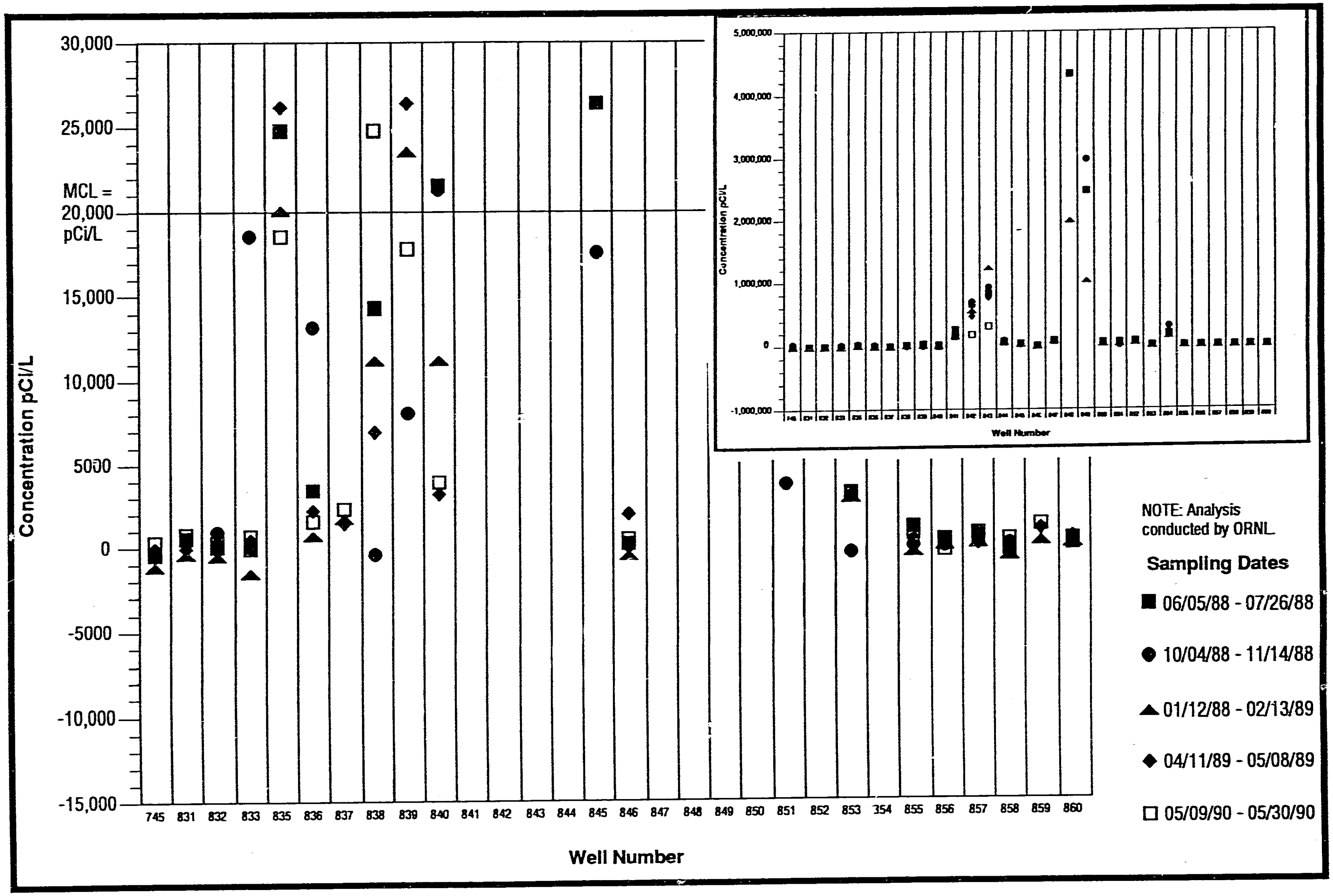


Fig. 4.8. Tritium concentrations detected in groundwater samples from ORNL RCRA compliance wells at WAG 6.

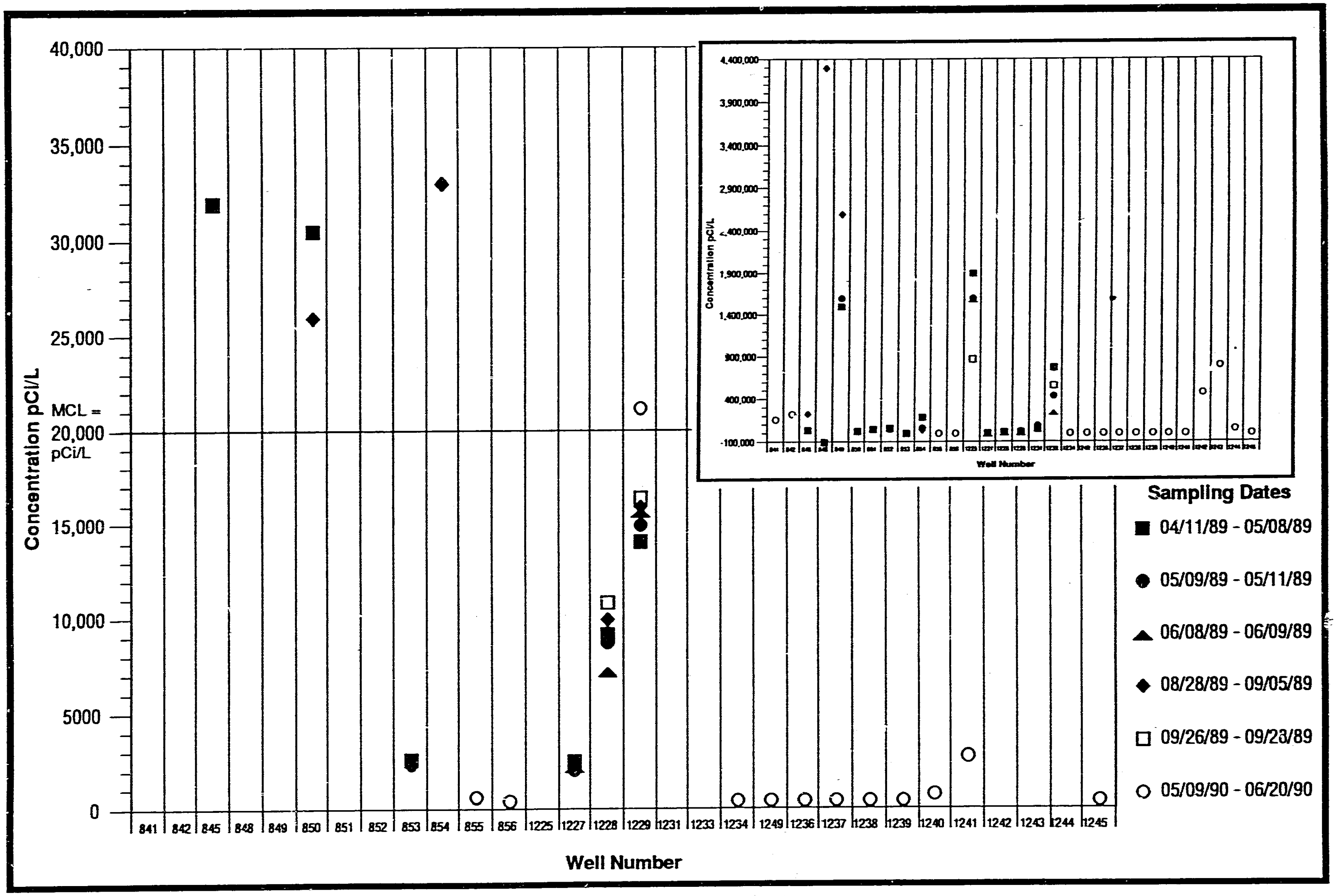


Fig. 4.9. Tritium concentrations detected in groundwater samples from RFI wells and some ORNL RCRA compliance wells at WAG 6.

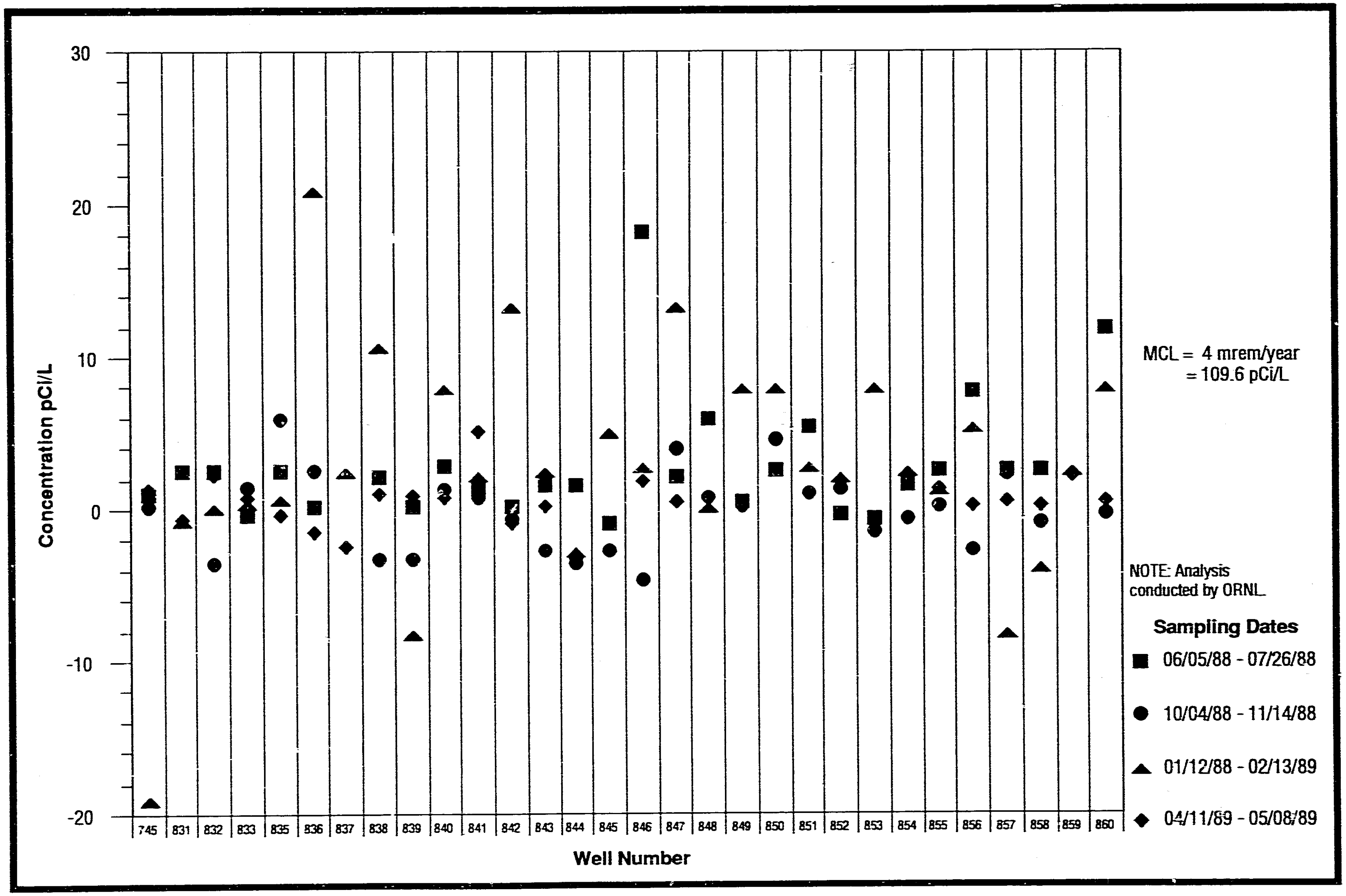


Fig. 4.10. Cesium-137 concentrations detected in groundwater samples from ORNL RCRA compliance wells at WAG 6.

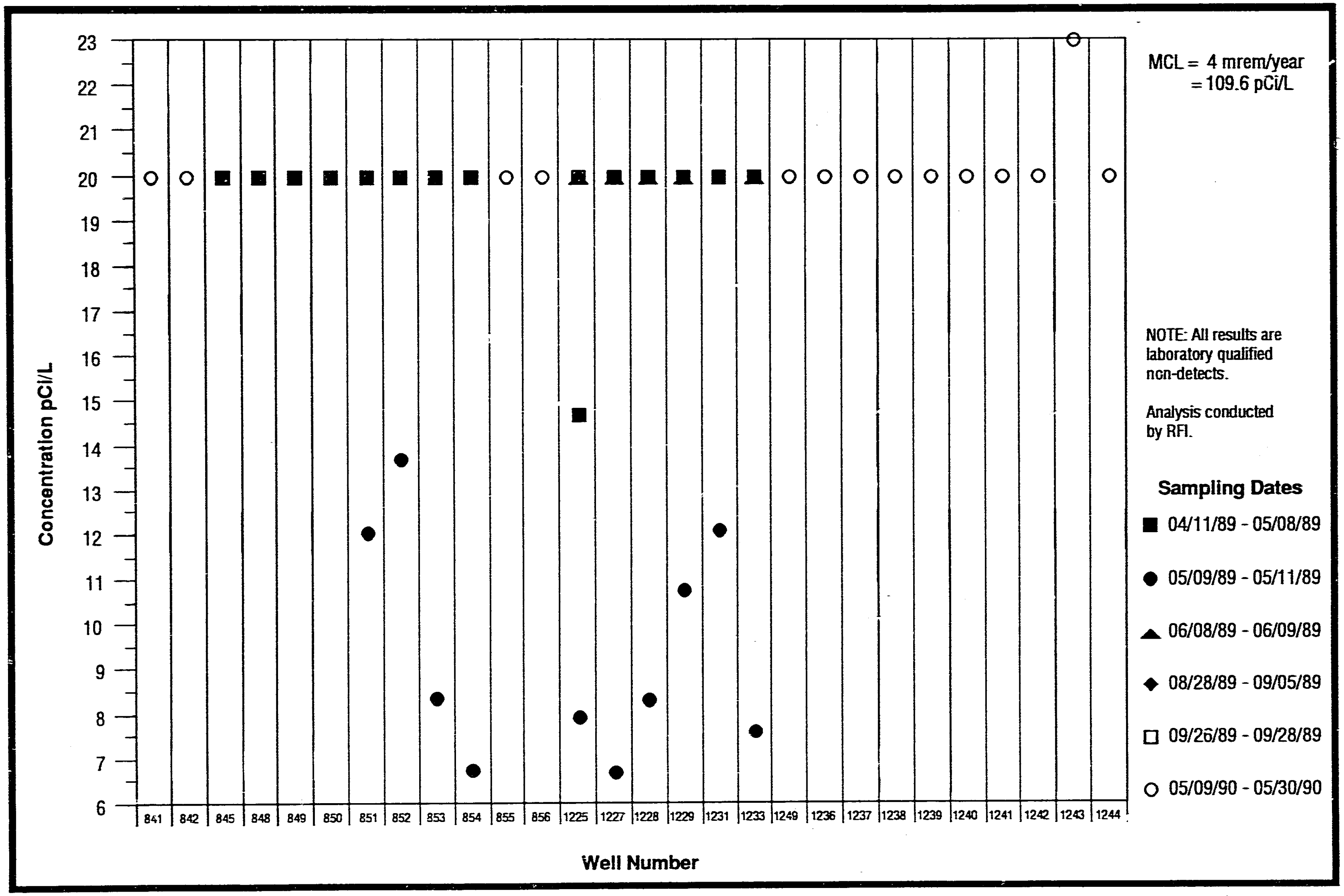


Fig. 4.11. Levels of nondetection of cesium-137 from groundwater samples from RFI wells and some ORNL RCRA compliance wells at WAG 6.

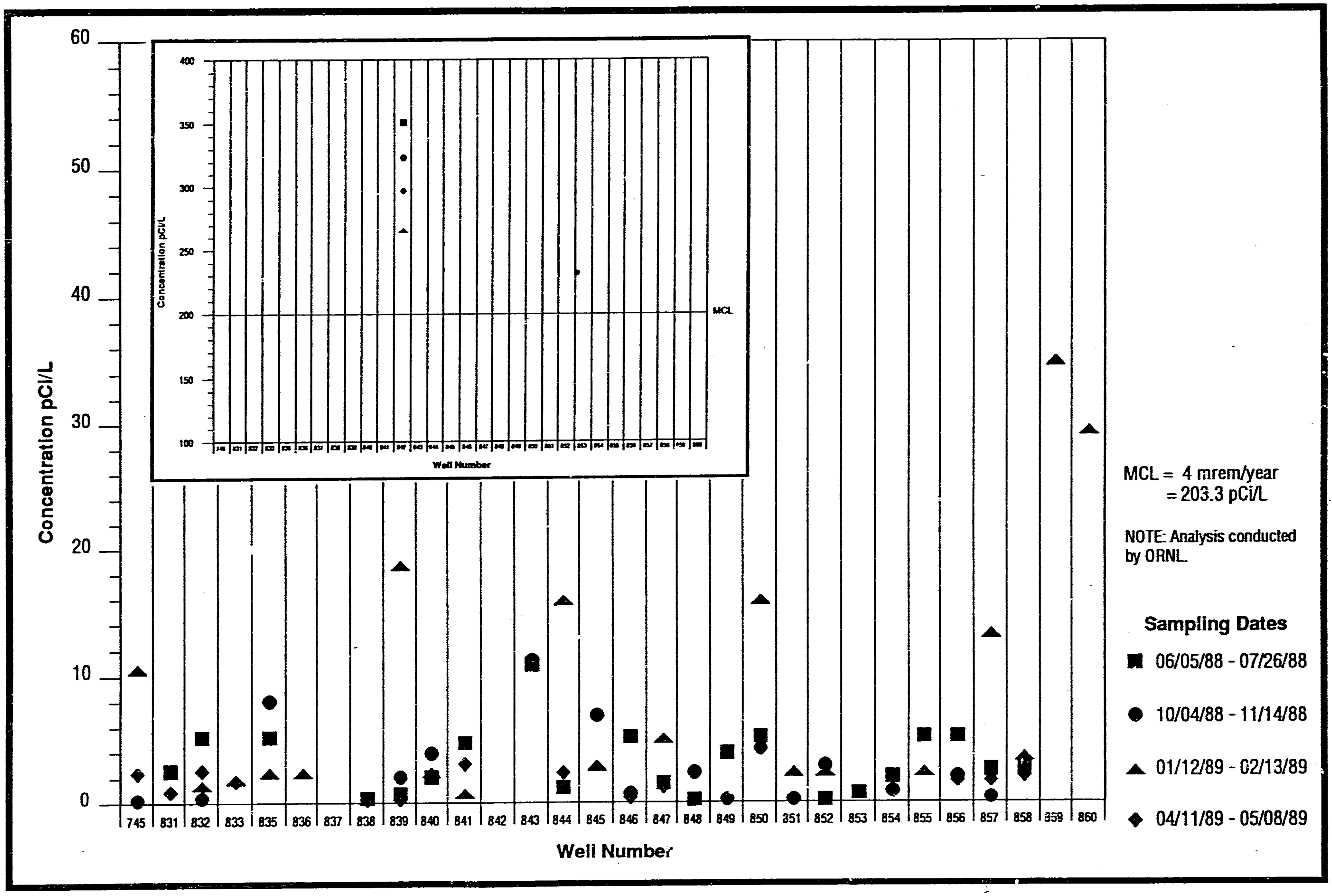


Fig. 4.12. Cobalt-60 concentrations detected in groundwater samples from ORNL RCRA compliance wells at WAG 6.

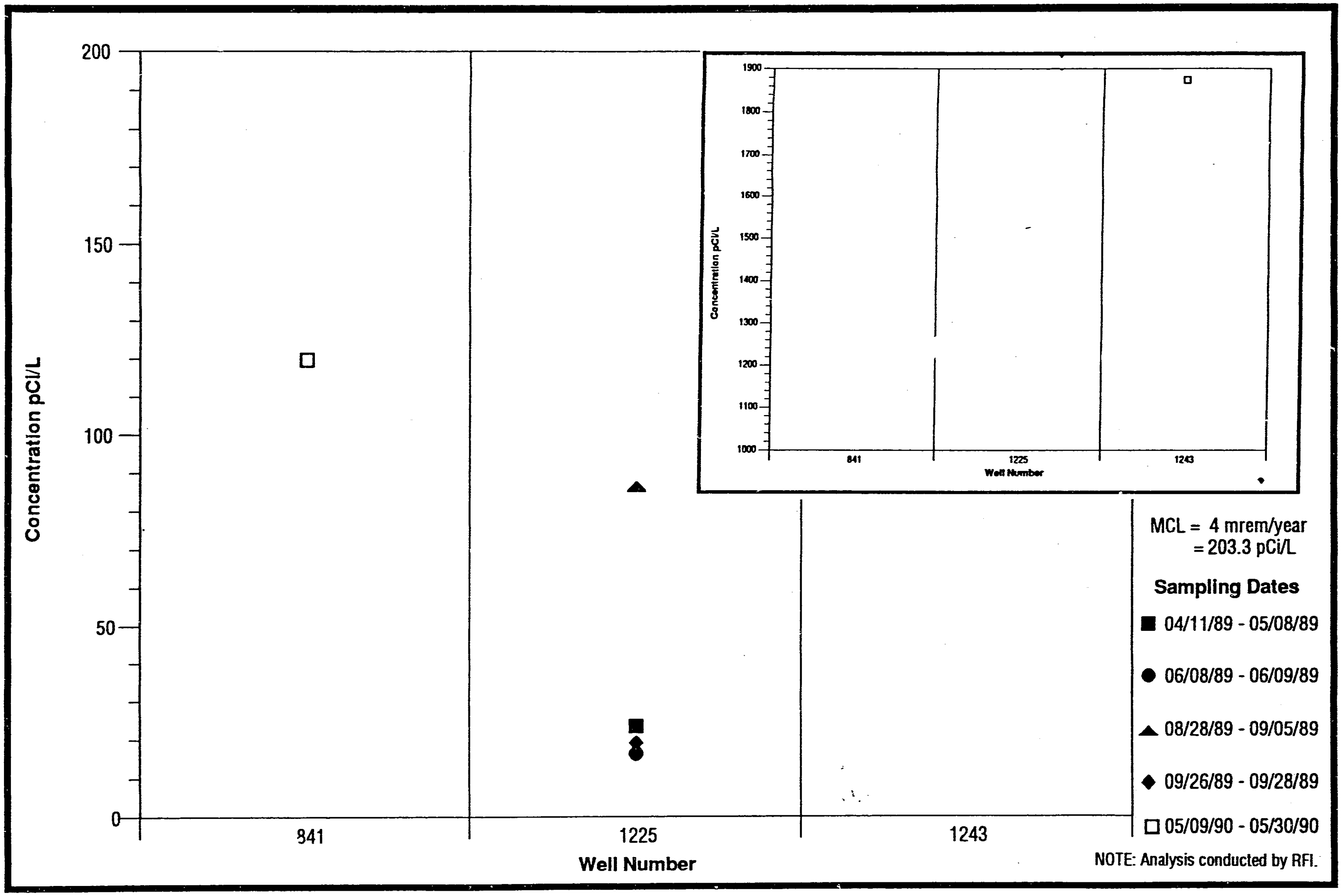


Fig. 4.13. Cobalt-60 concentrations detected in groundwater samples from RFI wells and some ORNL RCRA compliance wells at WAG 6.

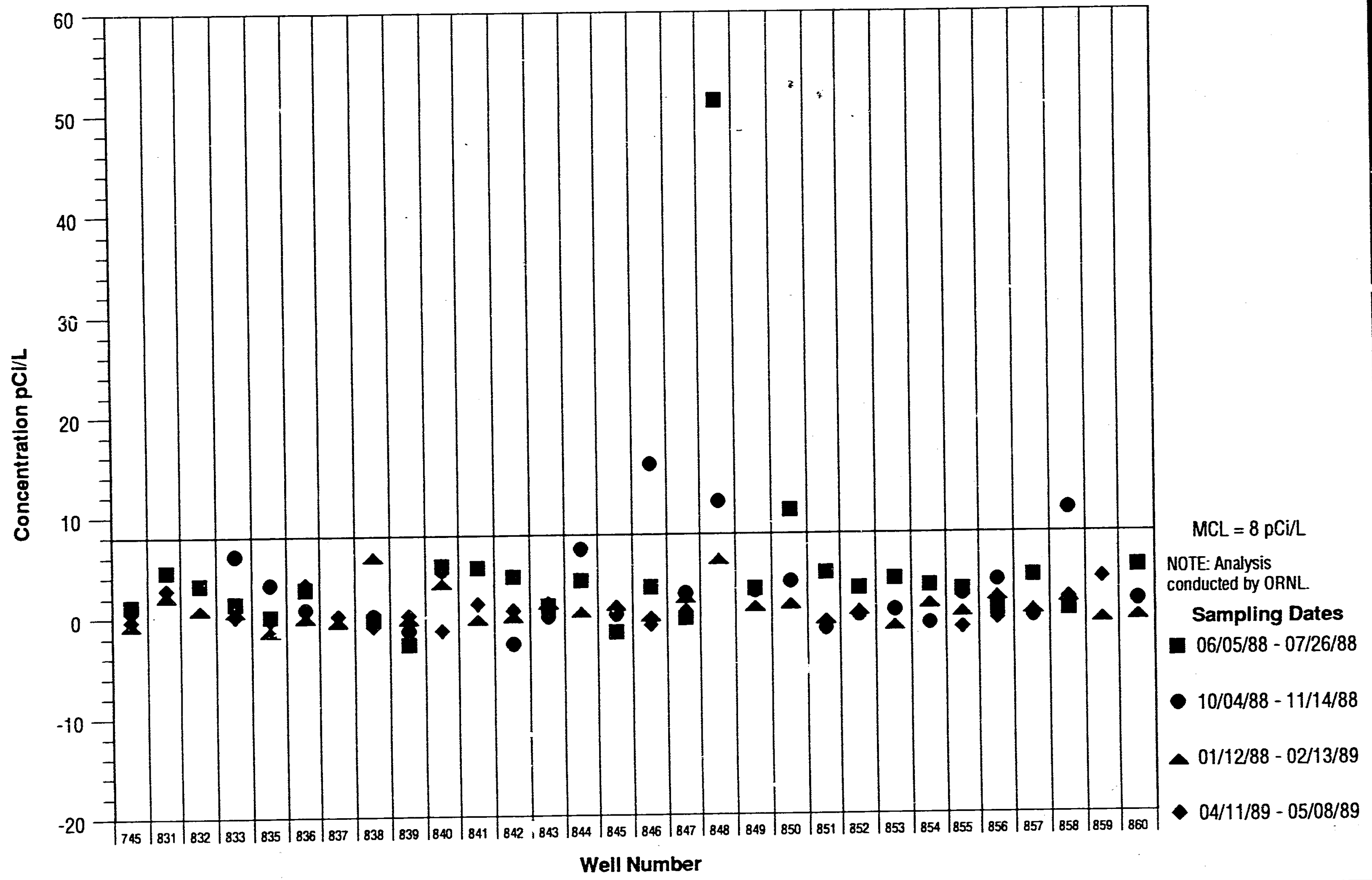


Fig. 4.14. Total strontium concentrations detected in groundwater samples from ORNL RCRA compliance wells at WAG 6.

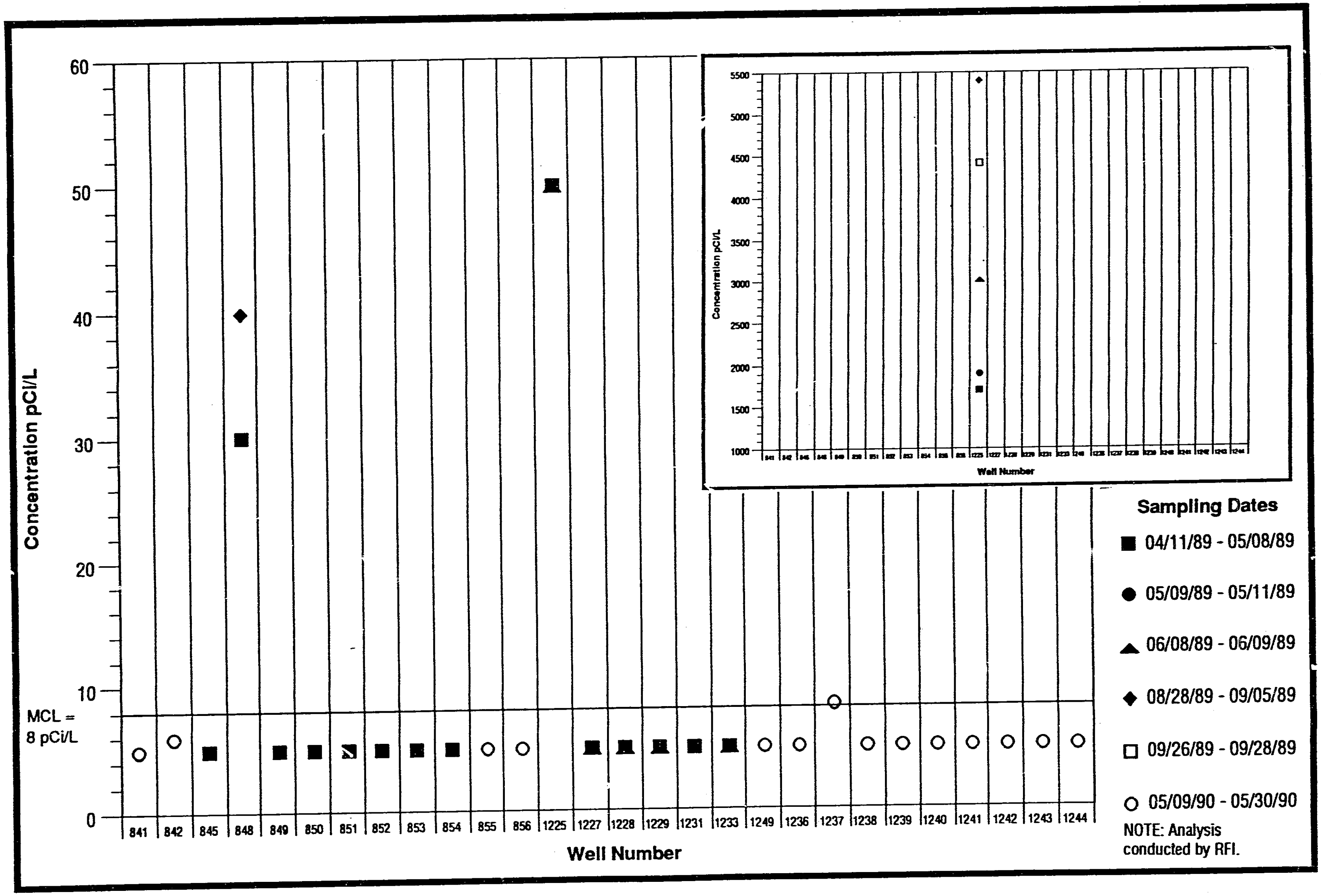


Fig. 4.15. Strontium-90 concentrations detected in groundwater samples from RFI wells and some ORNL RCRA compliance wells at WAG 6.

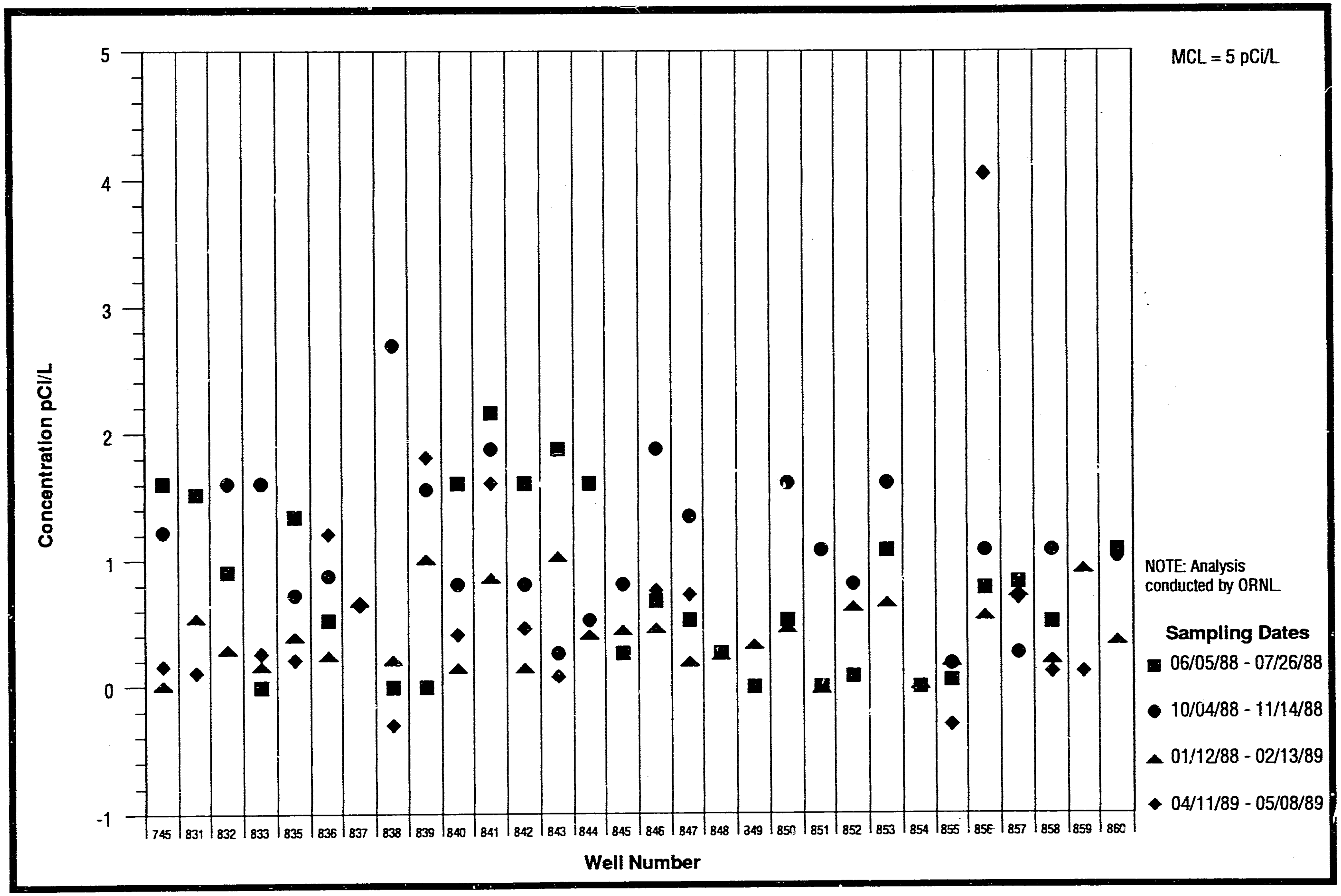
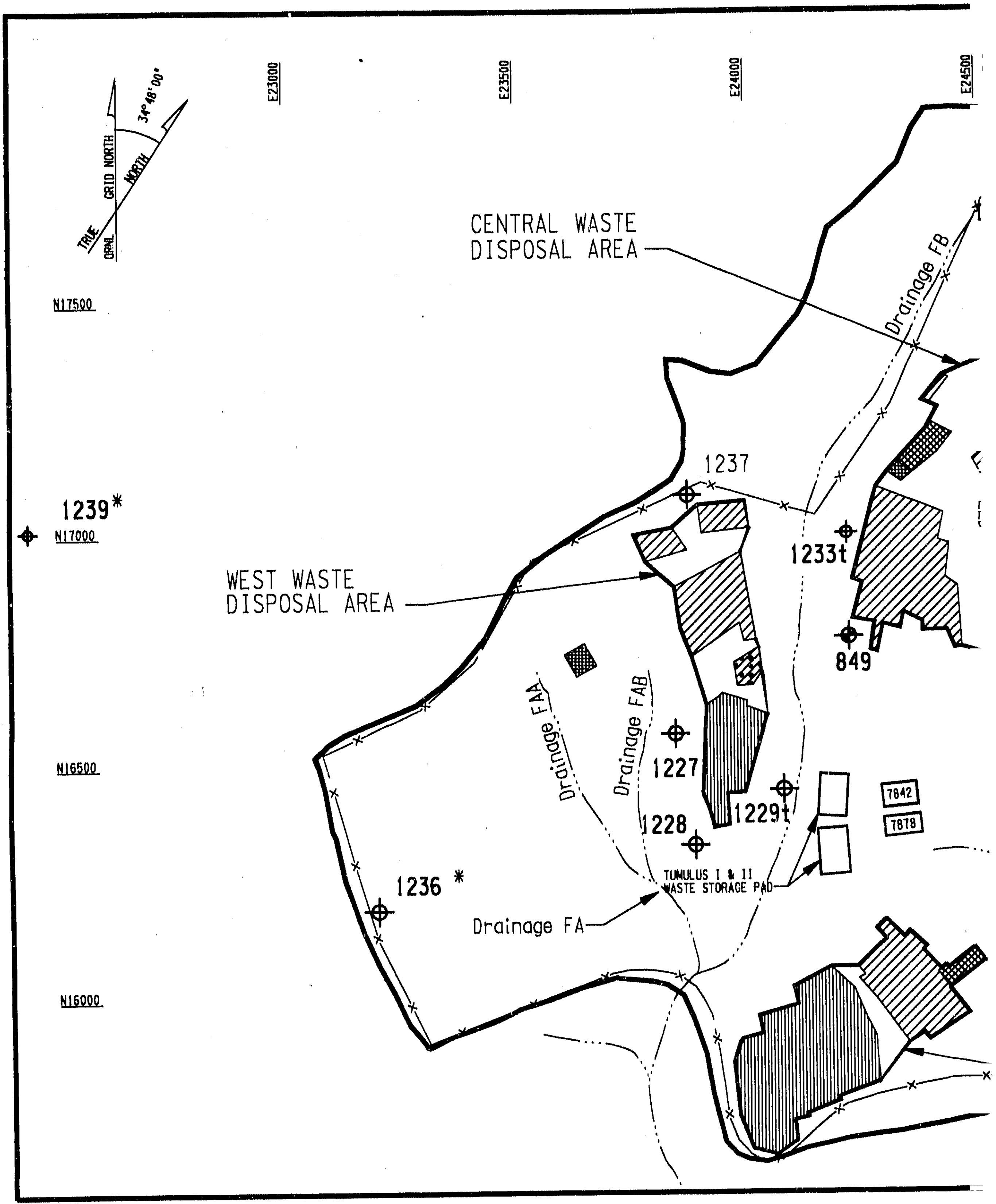


Fig. 4.16. Total radium concentrations detected in groundwater samples from ORNL RCRA compliance wells at WAG 6.



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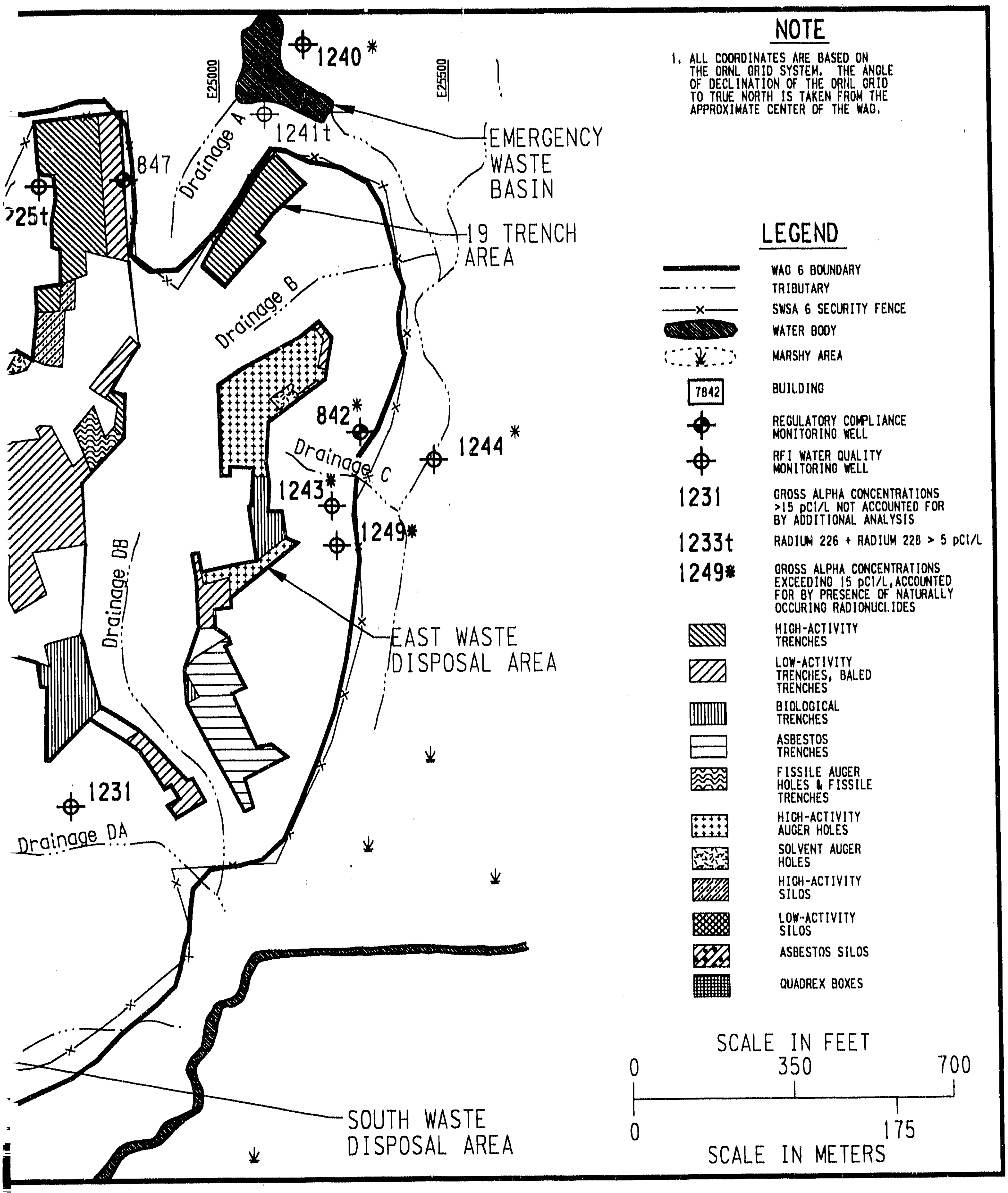
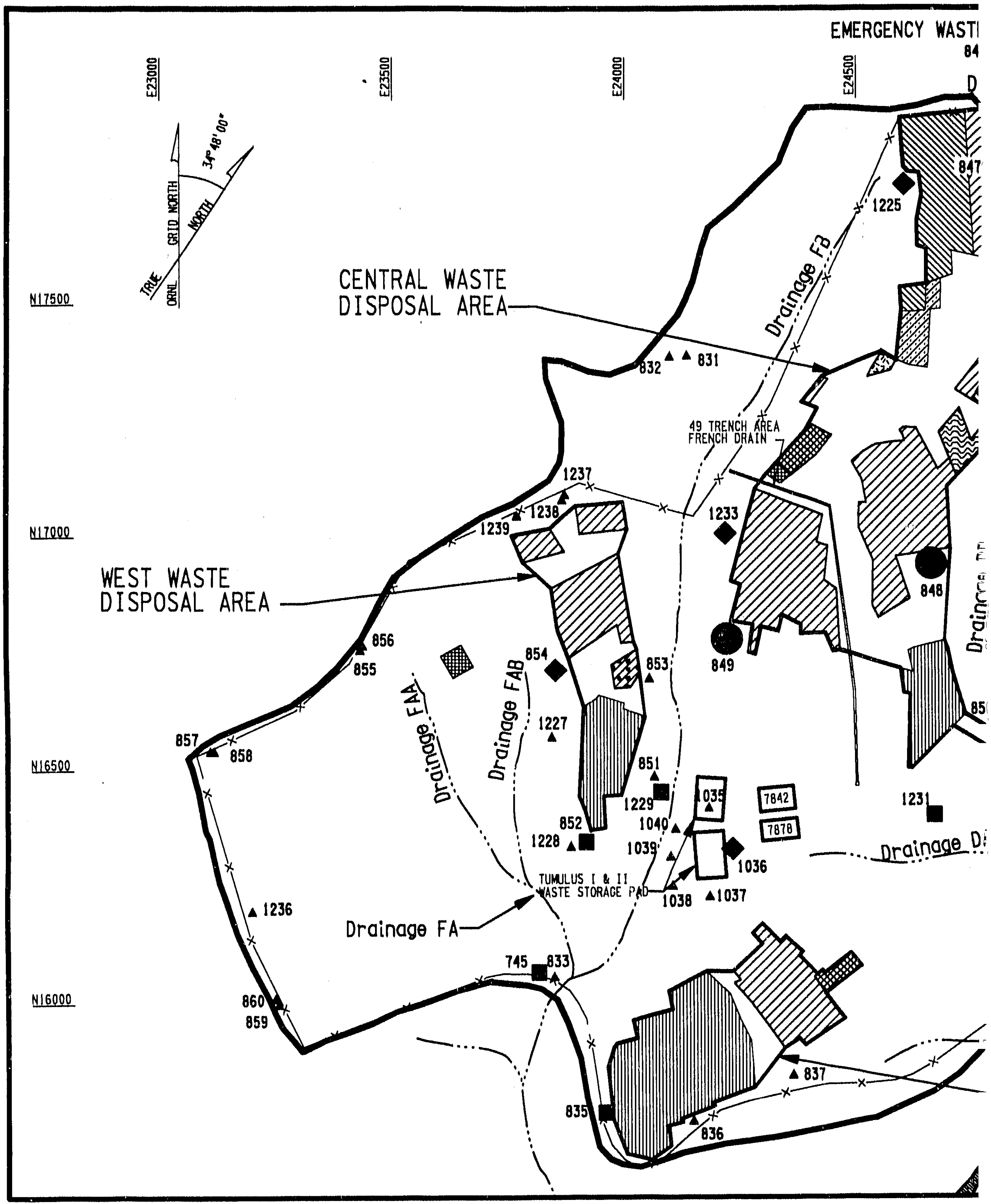


Fig. 4.17. Groundwater sampling locations where maximum gross alpha concentrations exceed 15 pCi/L and the sum of radium-226 and radium-228 exceed 5 pCi/L.



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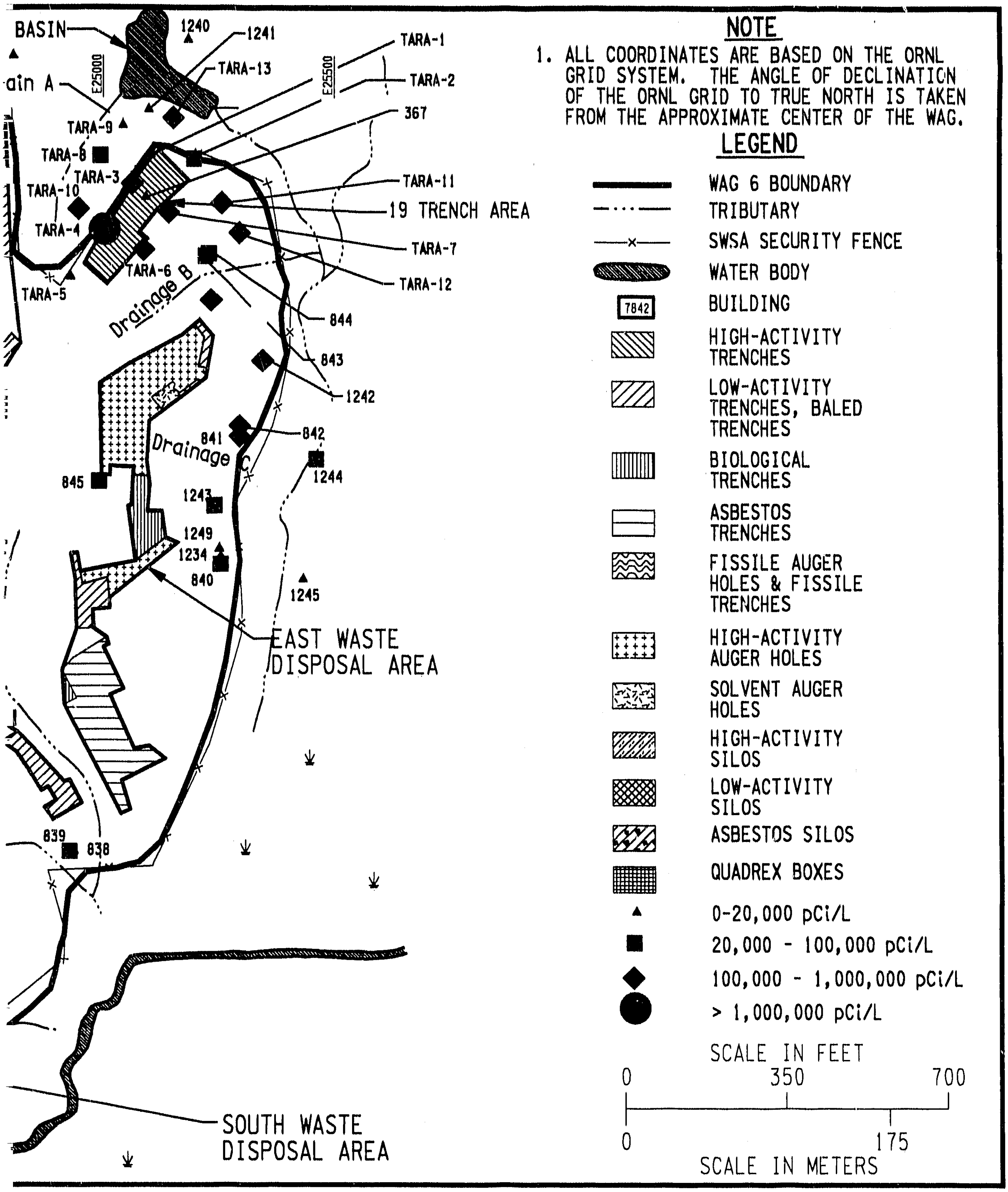
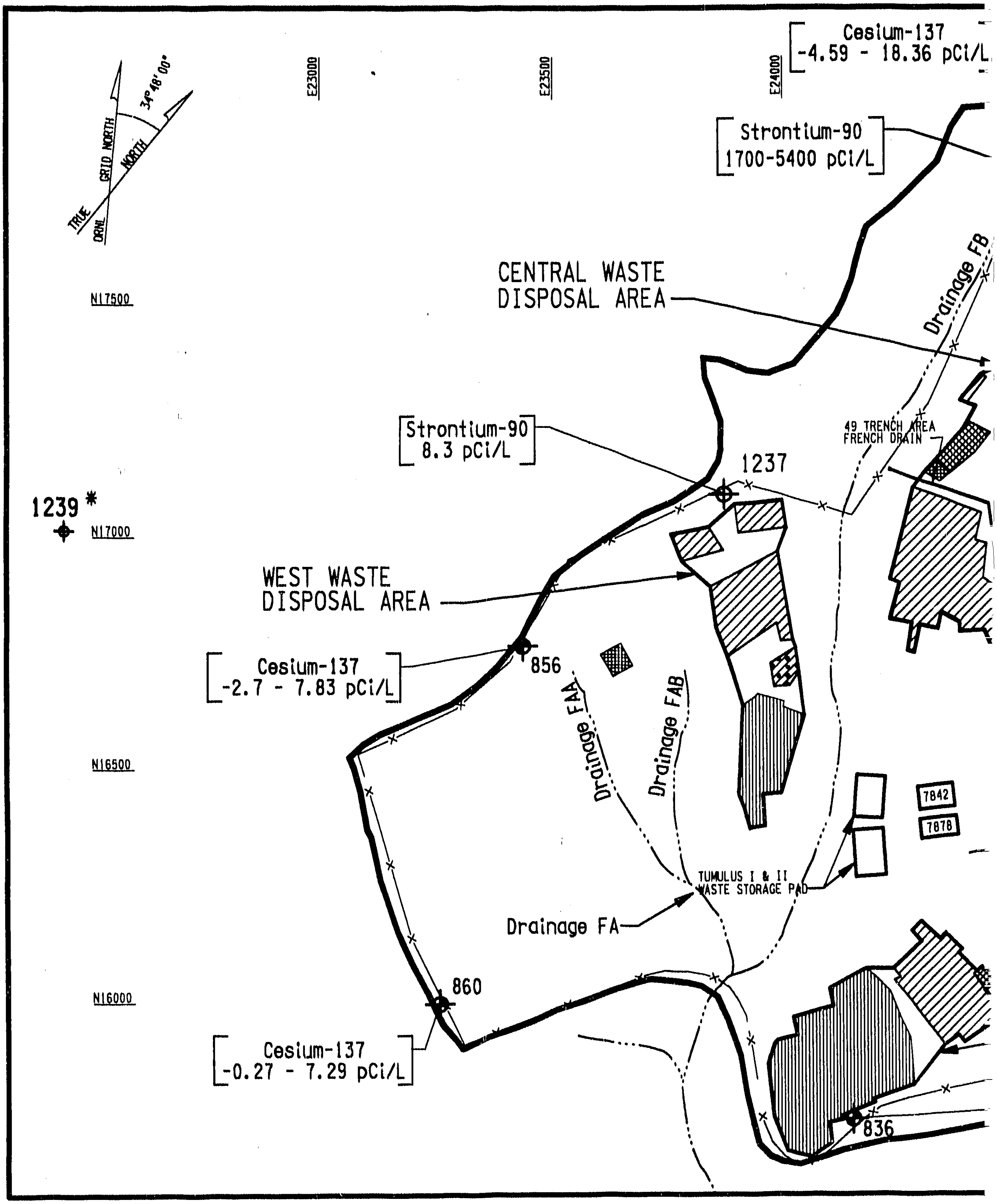


Fig. 4.18. Distribution of maximum detected tritium concentrations in groundwater at WAG 6.



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9-9

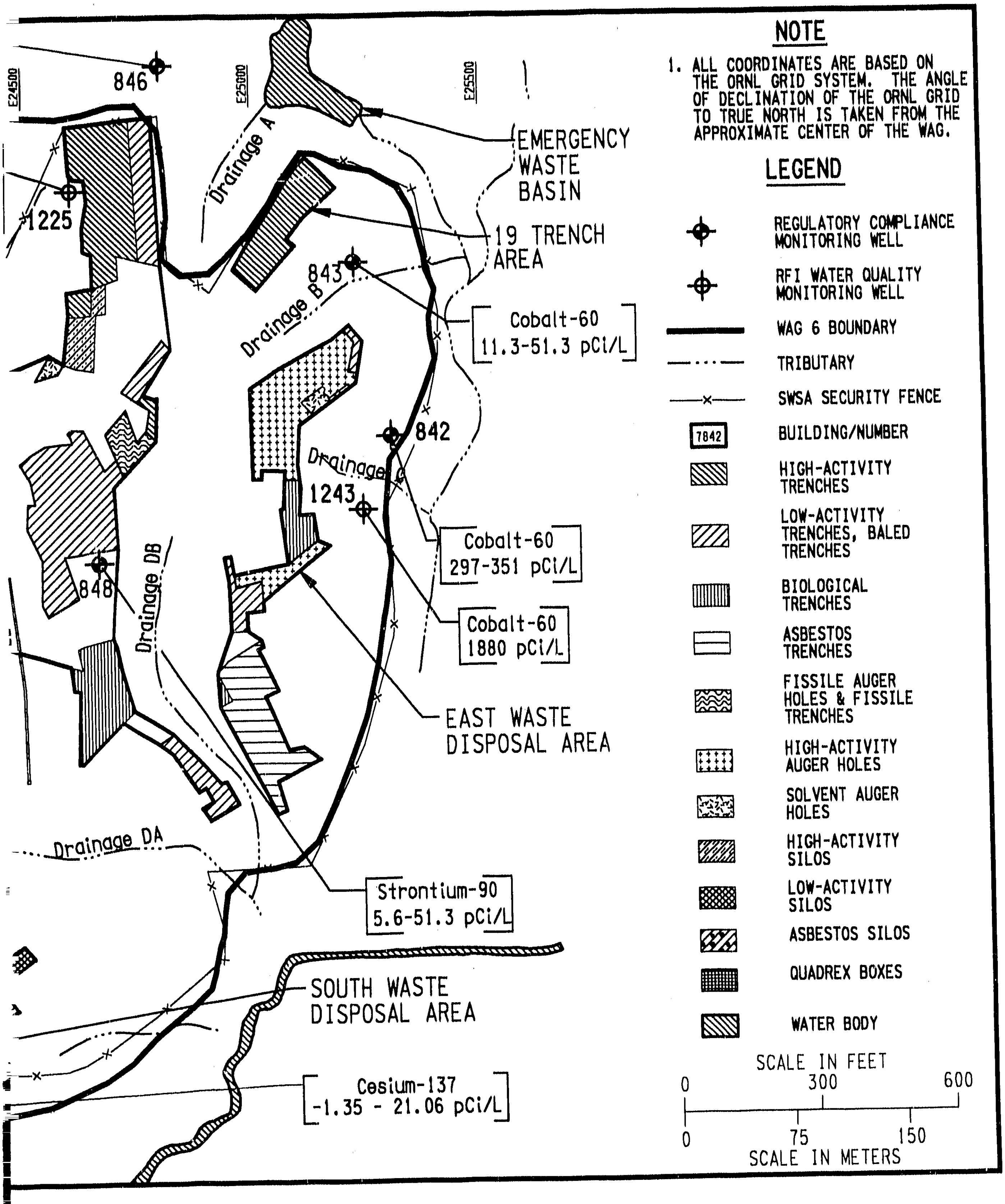
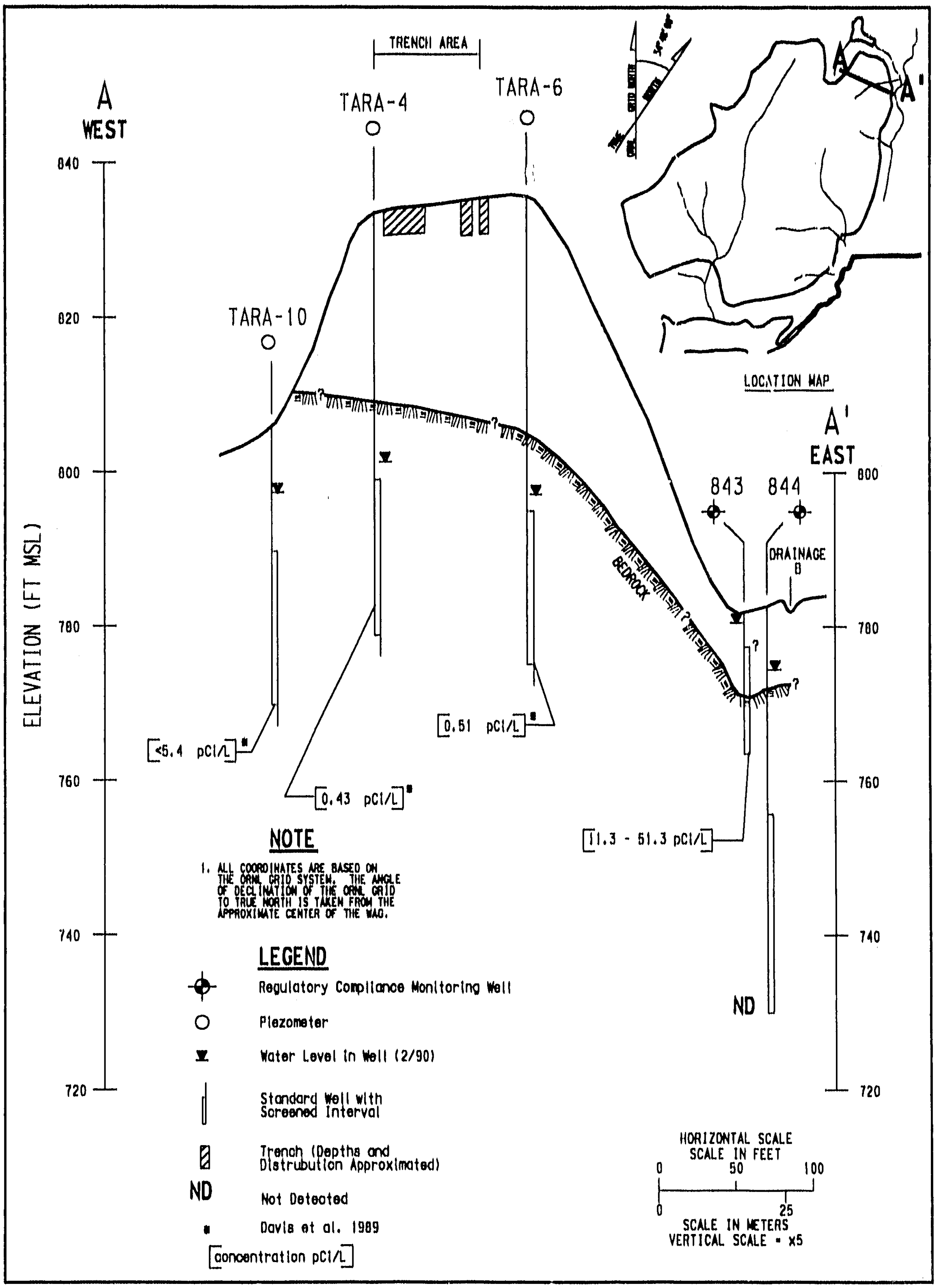
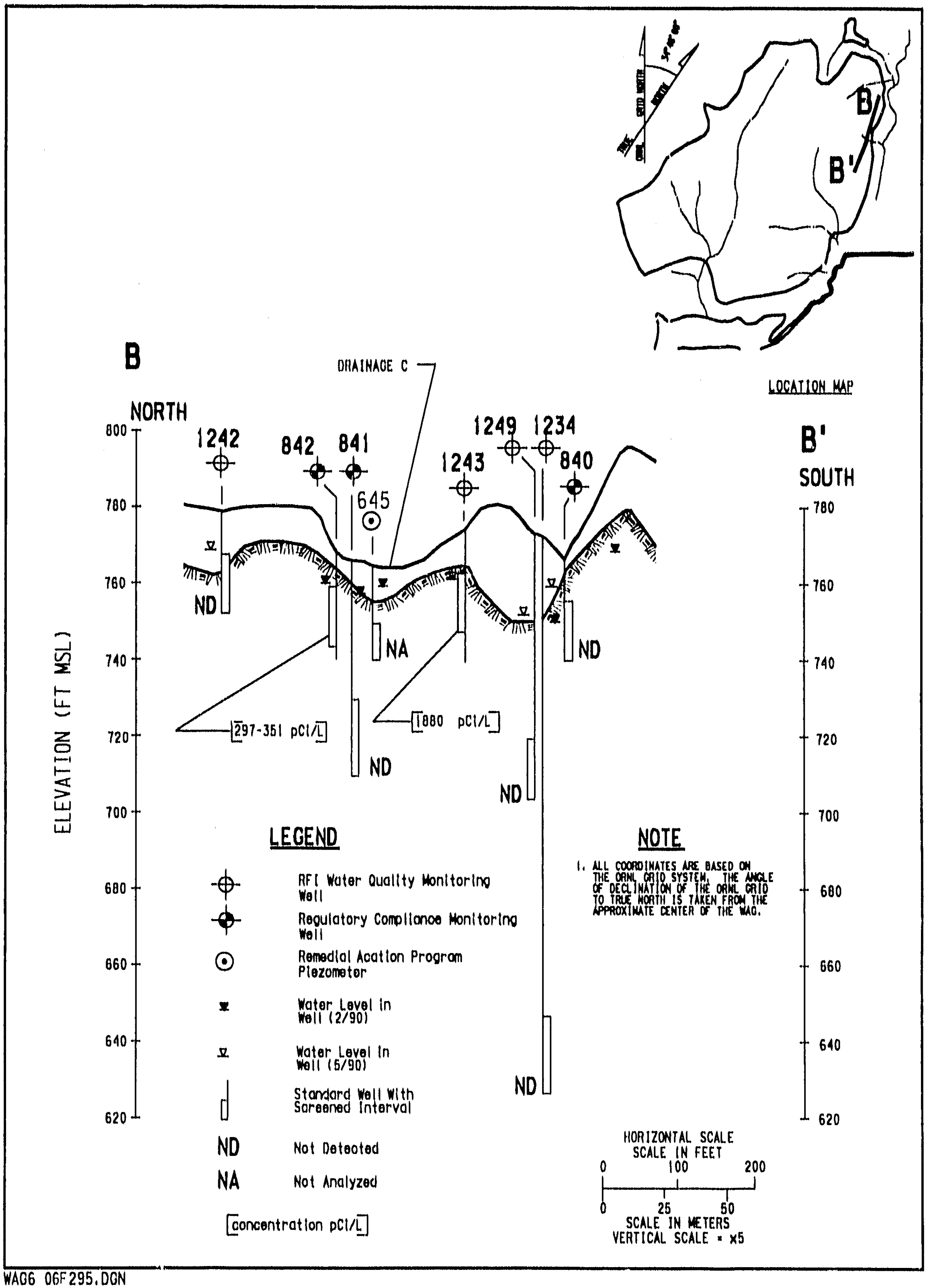


Fig. 4.19. Cesium-137, cobalt-60, and strontium-90 occurrences in groundwater at WAG 6.



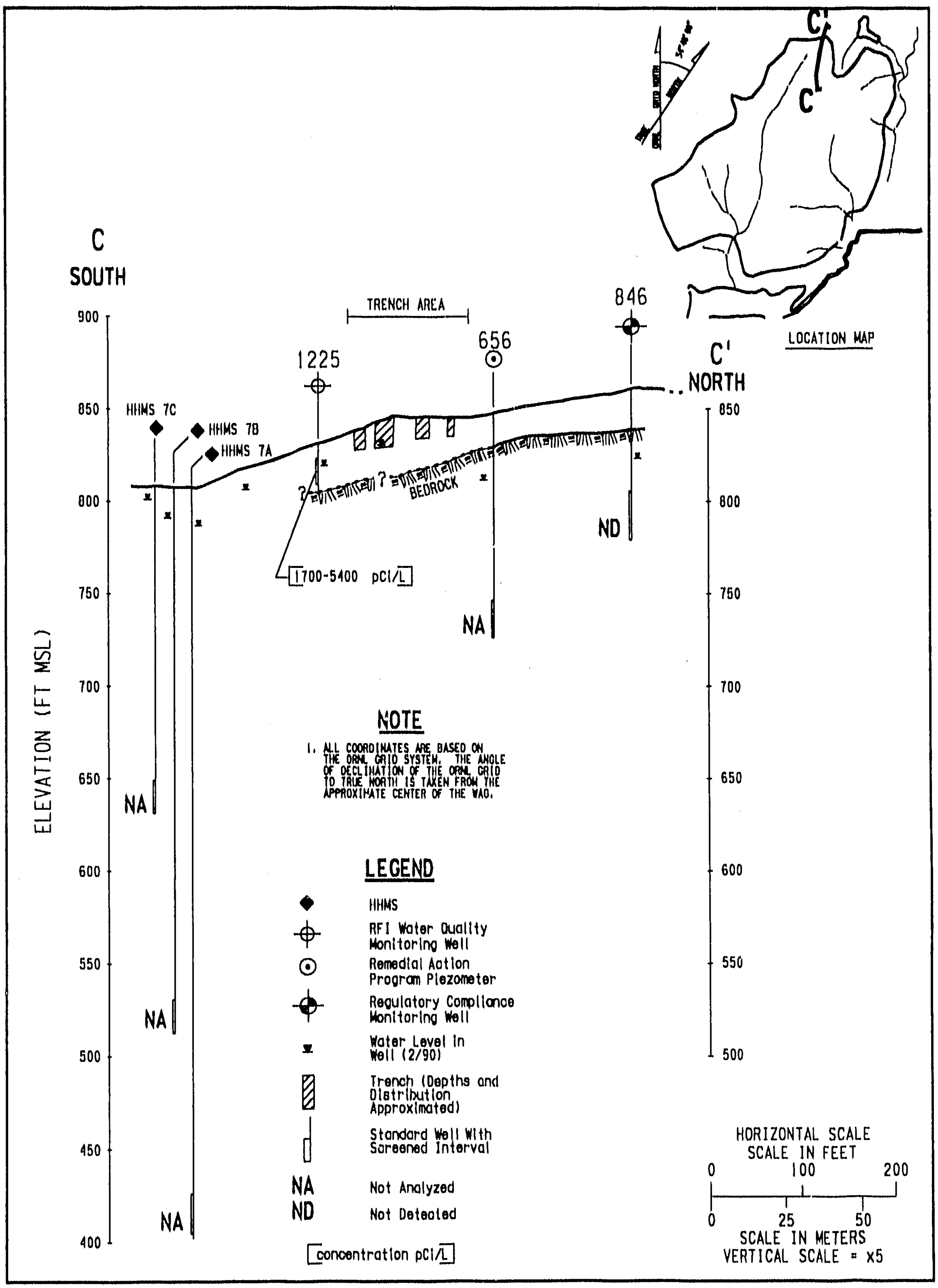
WAG6 06F296.DGN
9-1

Fig. 4.20. Occurrence of cobalt-60 in groundwater, cross section A-A', 19 Trench area.



WAG6 06F295.DGN
9-1

Fig. 4.21. Occurrence of cobalt-60 in groundwater, cross section B-B', east waste disposal area.

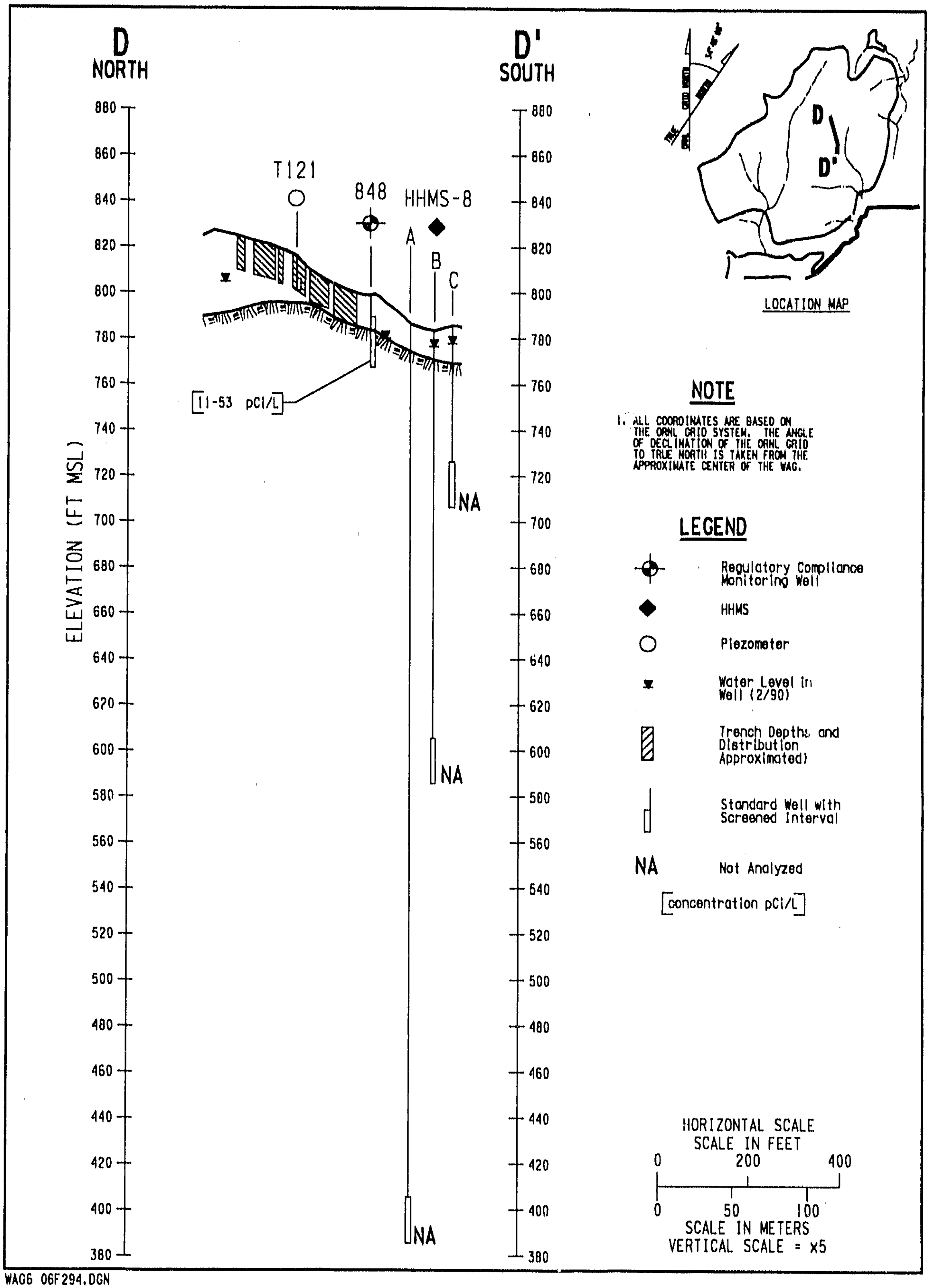


NOTE

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.

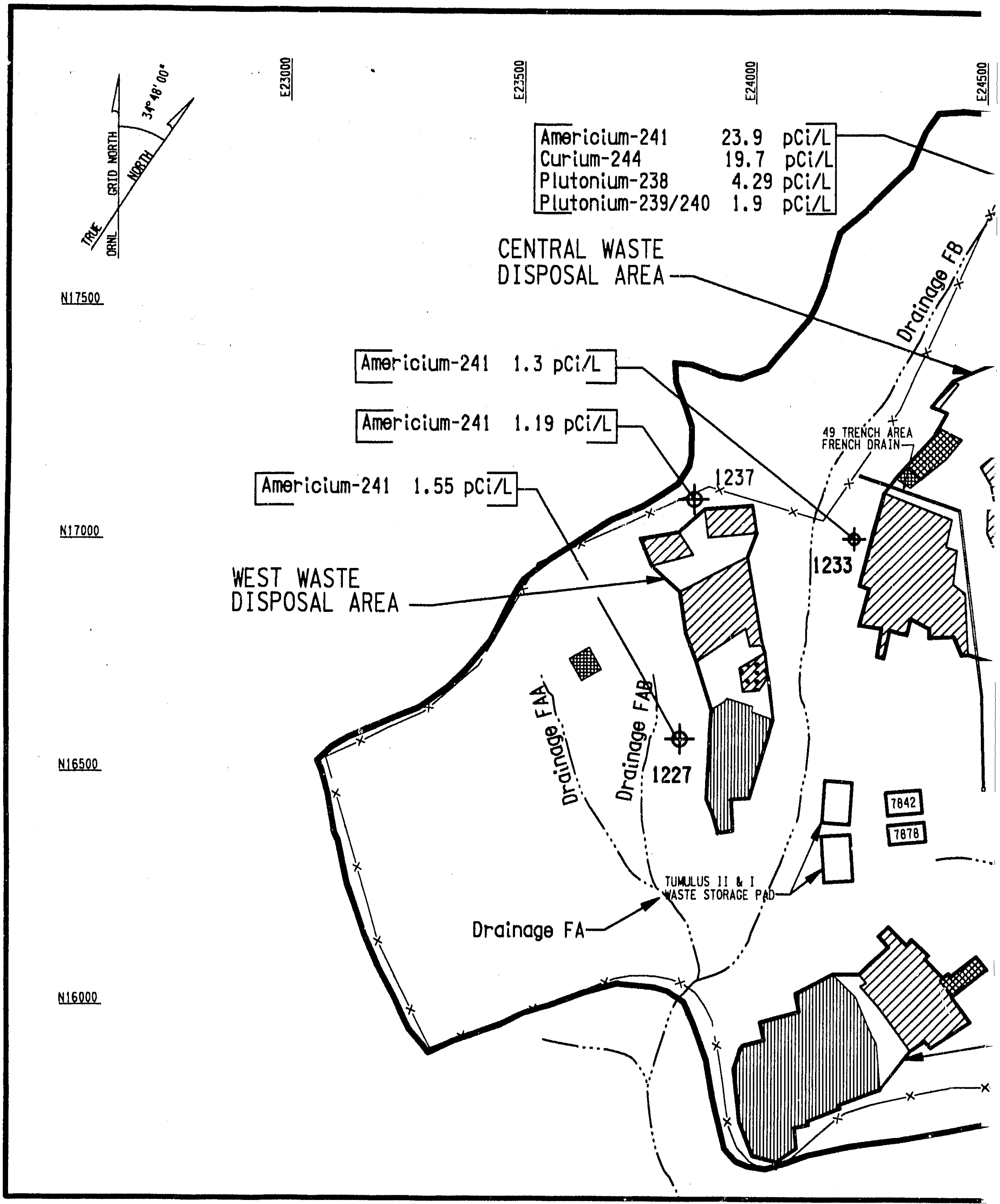
WAG6 06F293.DGN
8-31

Fig. 4.22. Occurrence of strontium-90 in groundwater, cross section C-C'; high-activity trench area of the central waste disposal area.



WAG6 06F294.DGN
8-31

Fig 4.23. Occurrence of strontium-90 in groundwater, cross section D-D', eastern low-activity trench area of the central waste disposal area.



WAG6 06F300.DGN
9-1

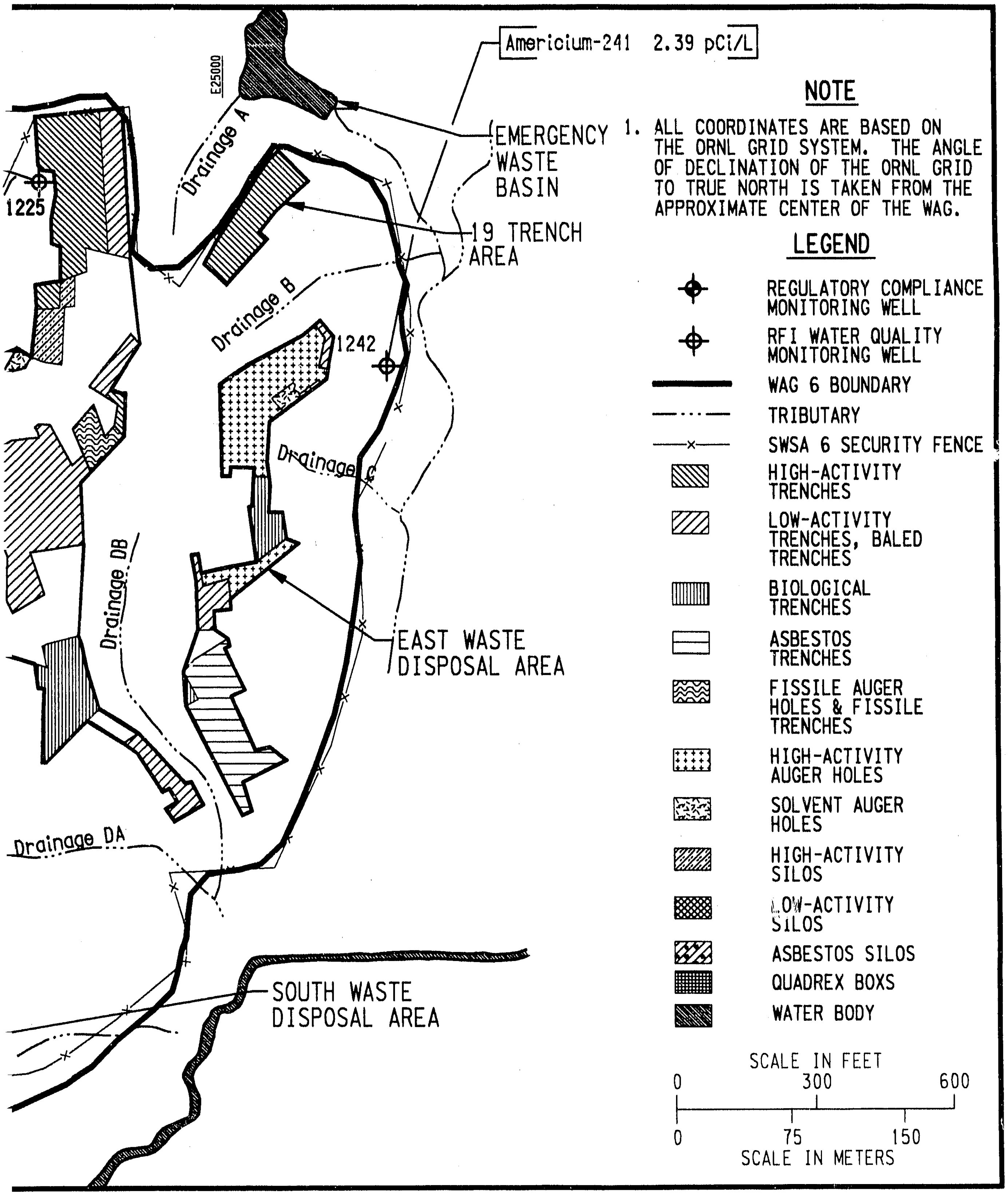
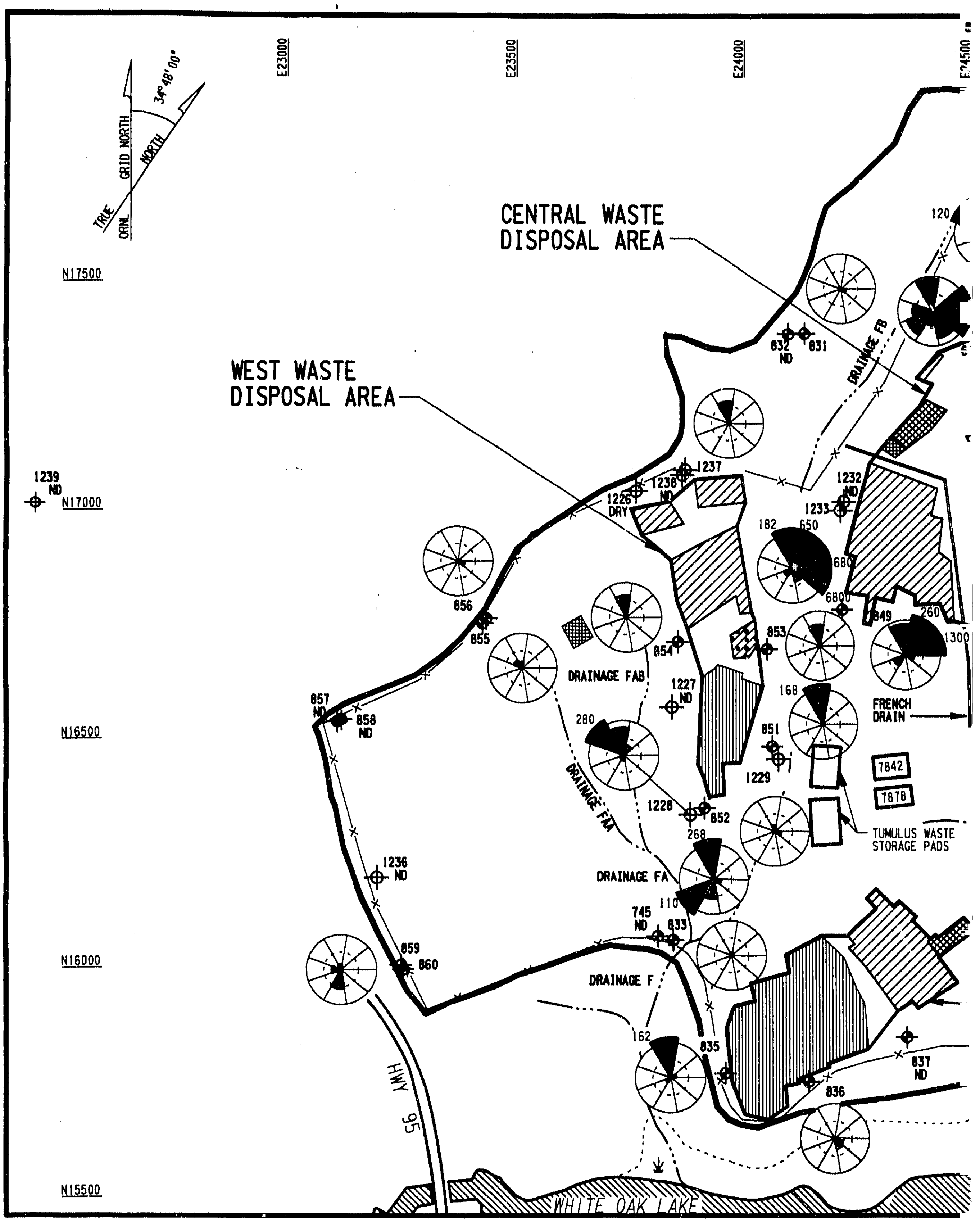
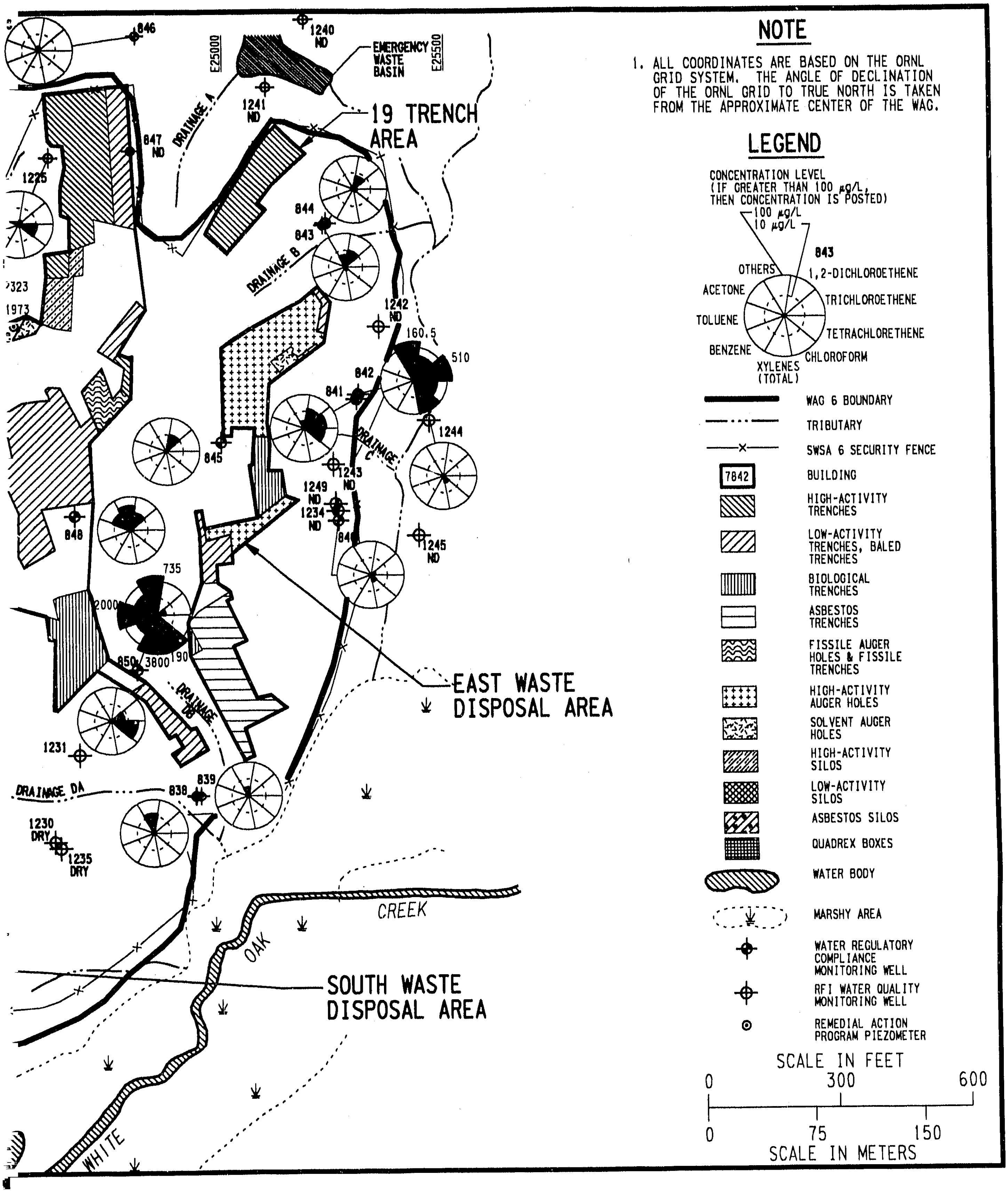


Fig. 4.24. Occurrence of transuranic radionuclides in groundwater samples at WAG 6.



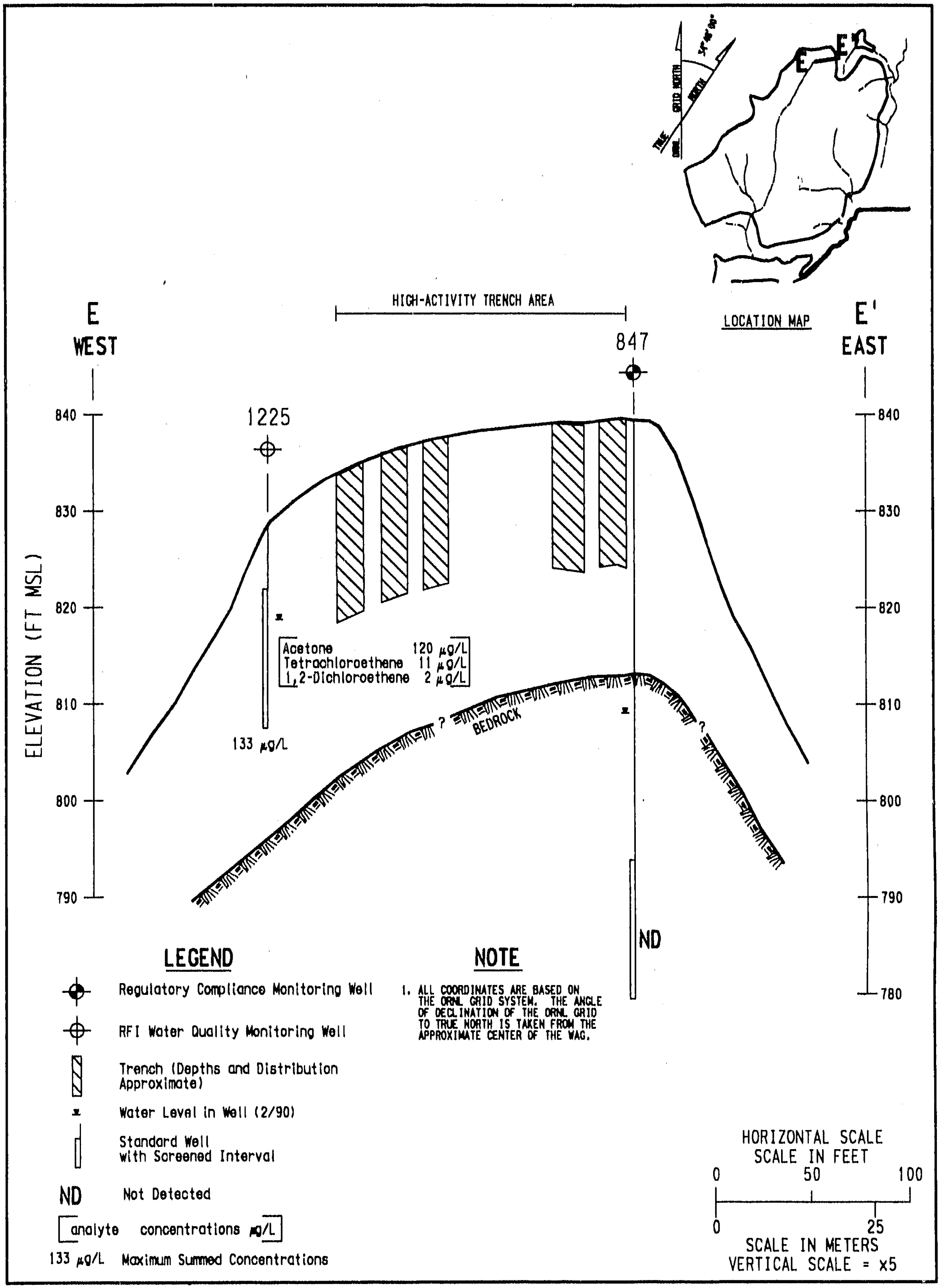
WAG6 06F286.DGN
9-9



NOTE

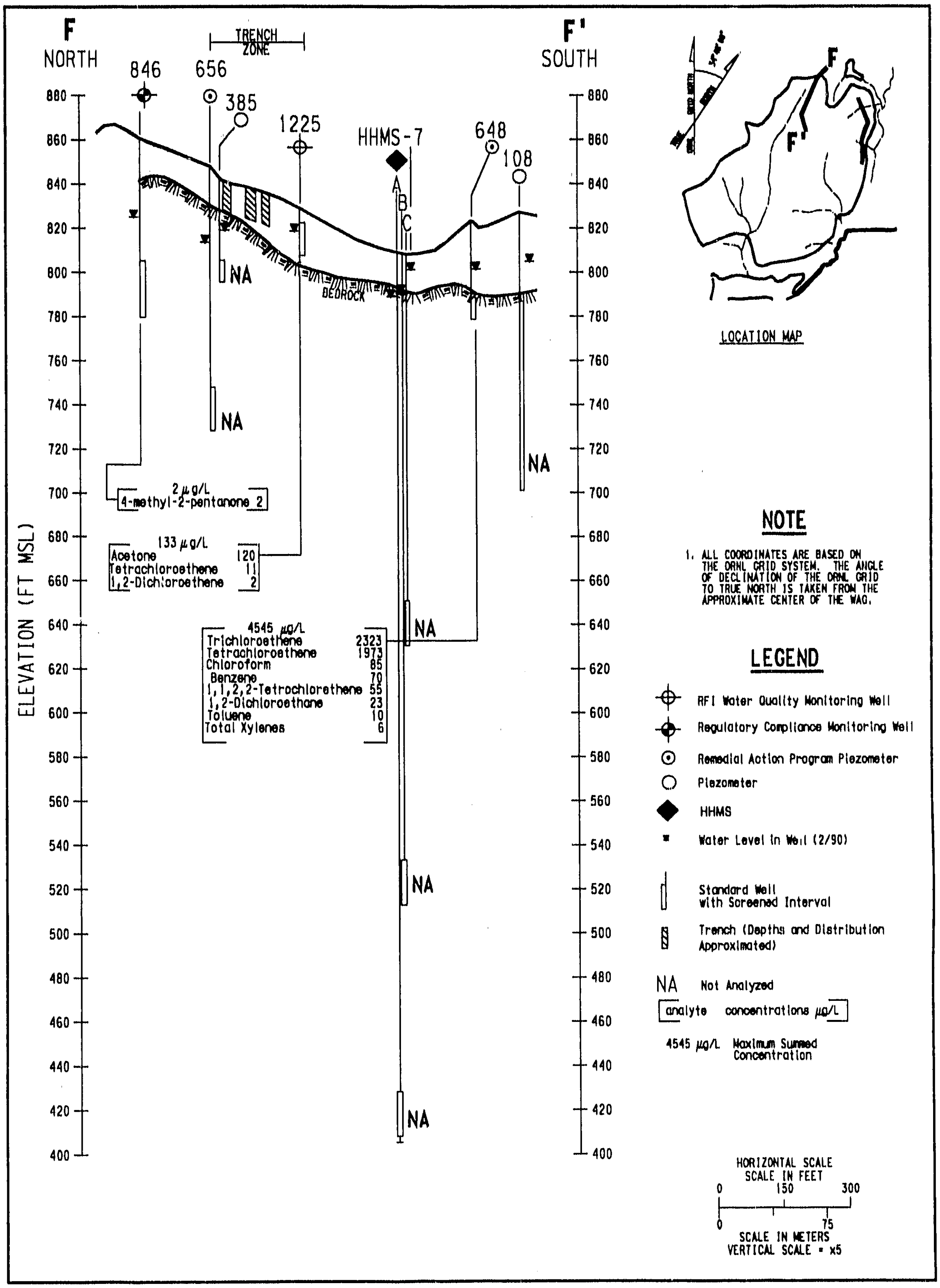
1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.

Fig. 4.25. Maximum concentration distribution of selected volatile organic compounds detected in groundwater at WAG 6, June 1988 to June 1990.



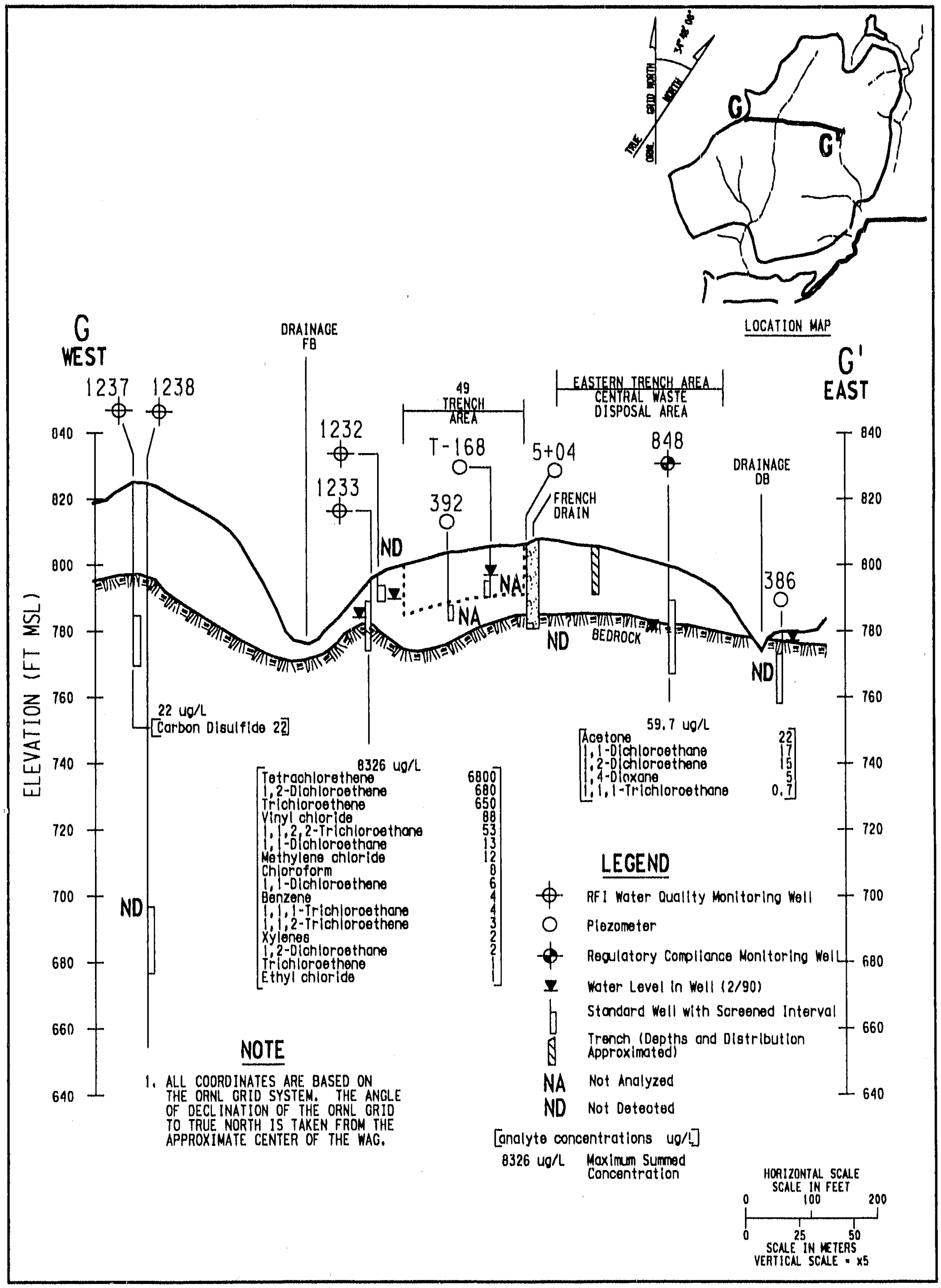
WAG6 06F226.DGN
8-31

Fig. 4.26. Maximum VOC concentrations in groundwater, cross section E-E', high-activity trench area.



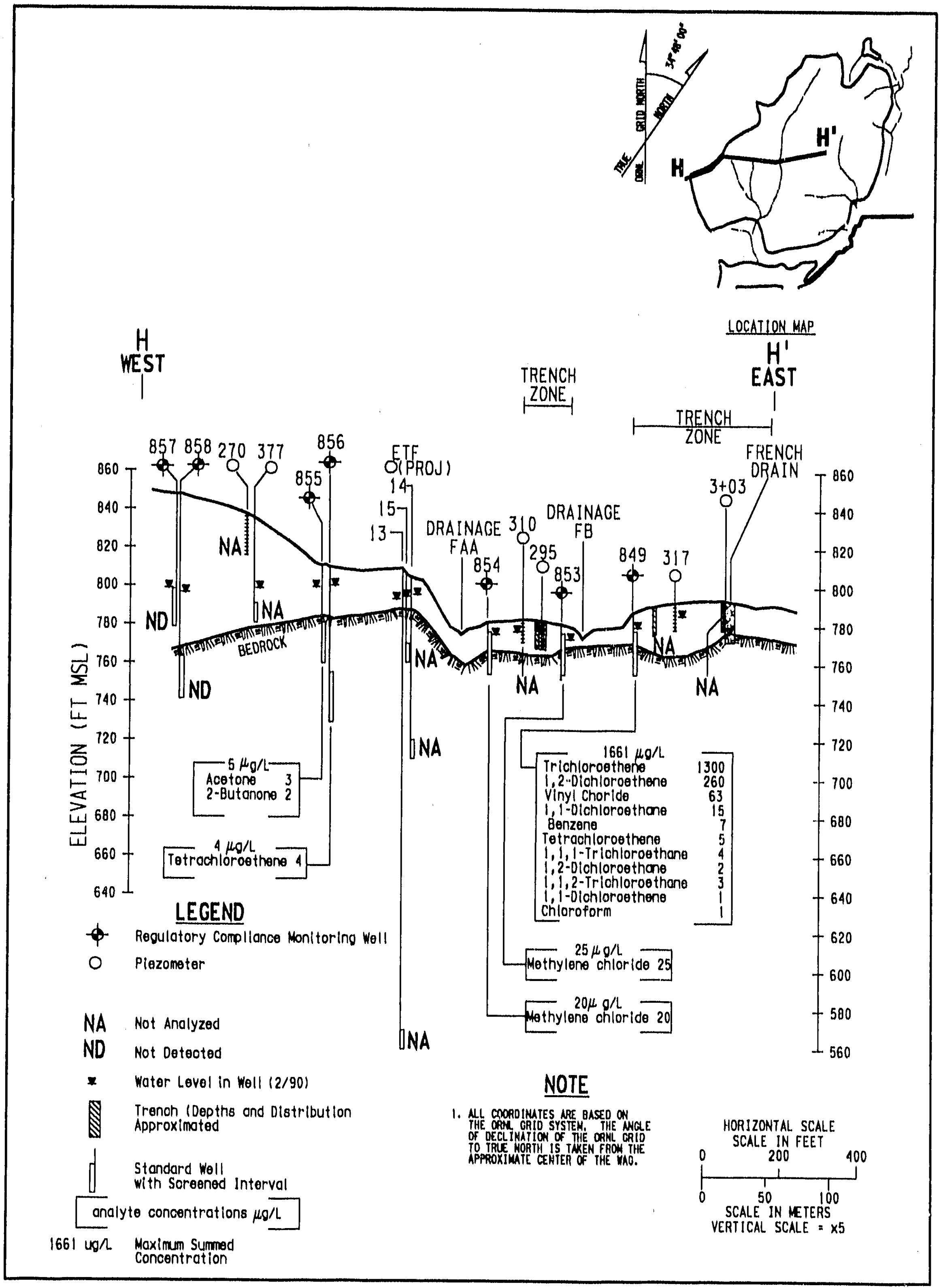
WAG6 06F297.DGN
8-31

Fig. 4.27. Maximum VOC concentrations in groundwater, cross section F-F', high-activity trench area.



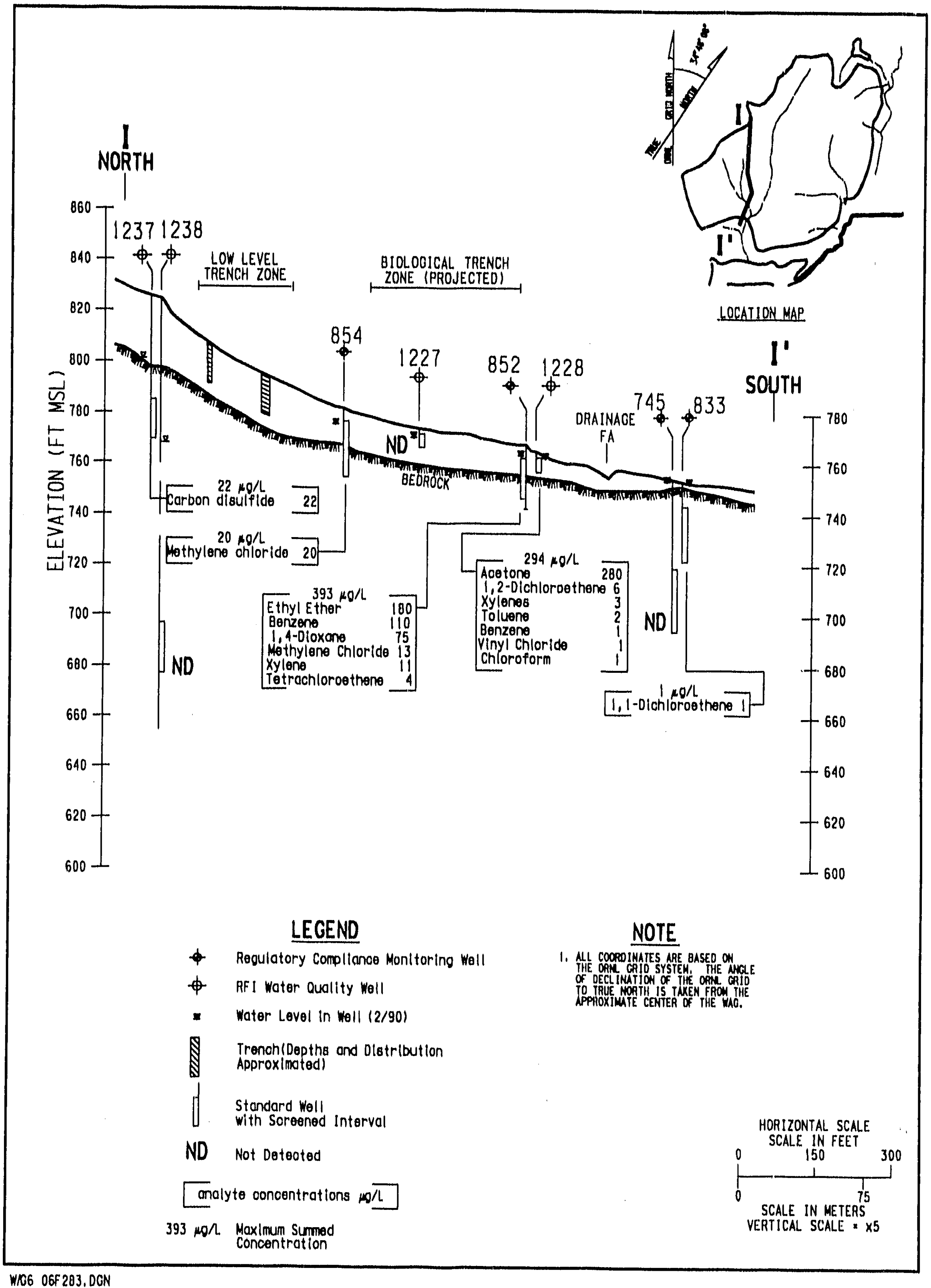
WAG6 06F180.DGN
8-31

Fig. 4.28. Maximum VOC concentrations in groundwater, cross section G-G', central waste disposal area.



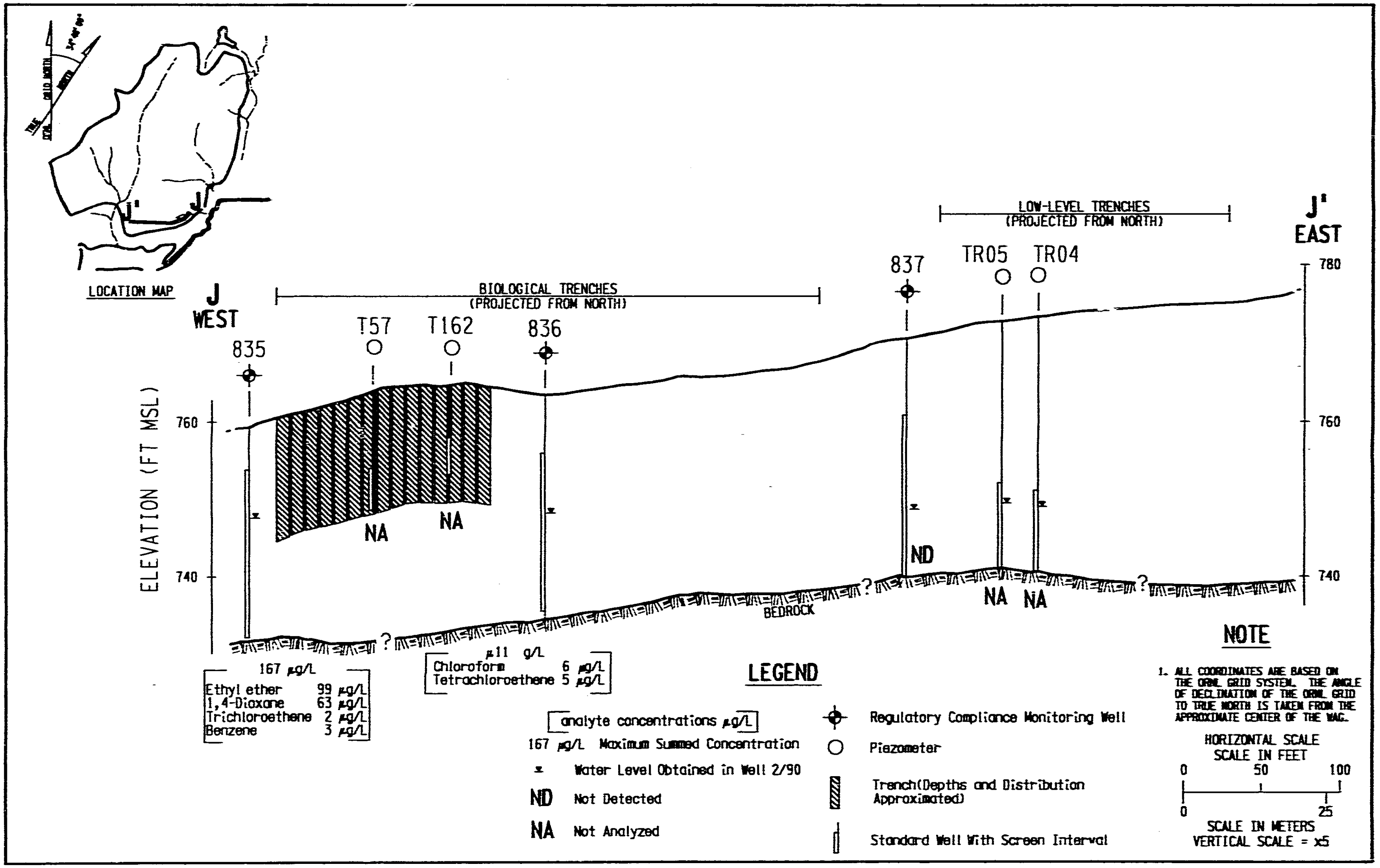
WAG6 06F282.DGN
8-31

Fig. 4.29. Maximum VOC concentrations in groundwater, cross section H-H', west waste disposal area including the 49 Trench area.



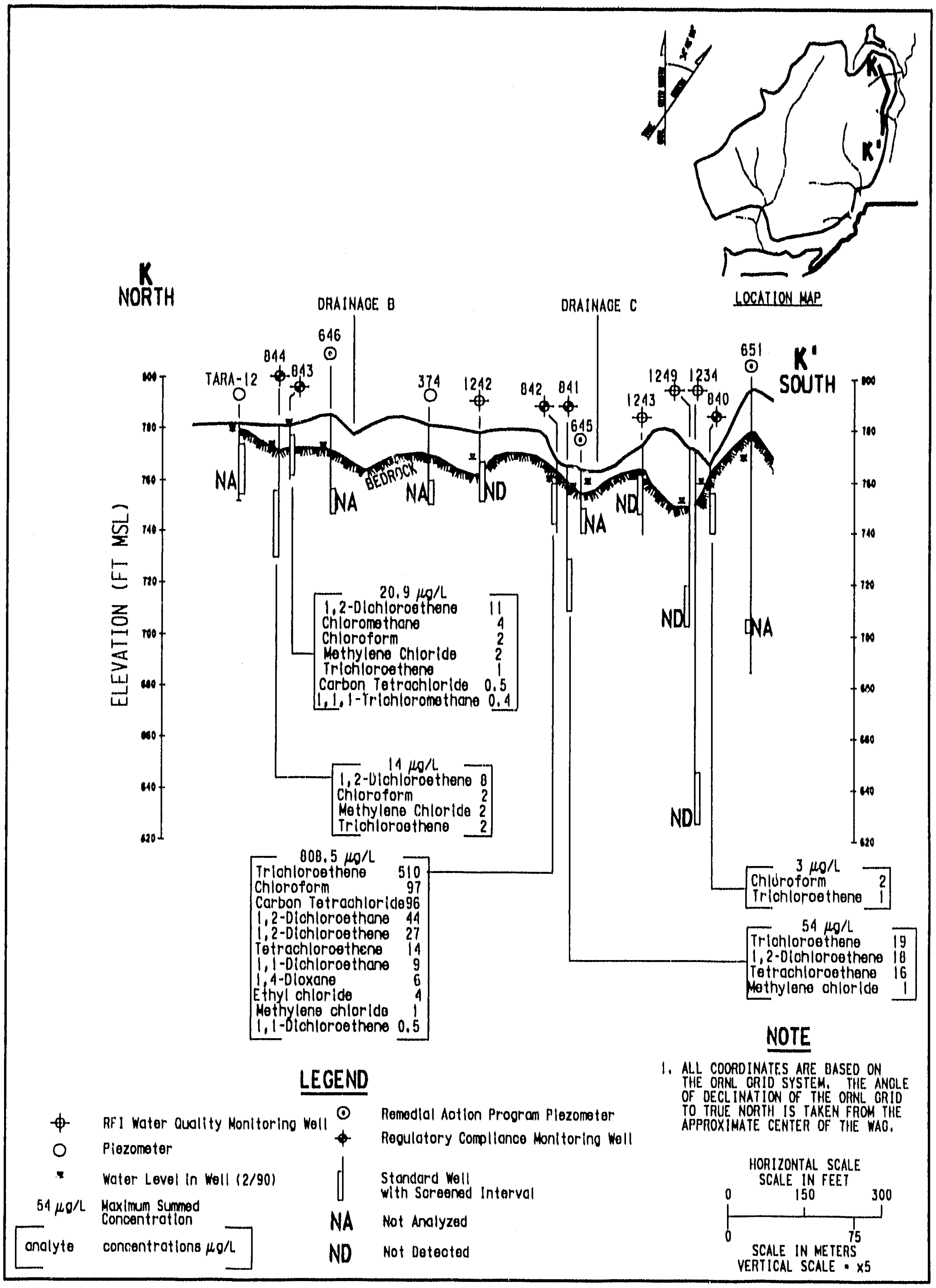
W/G6 06F283.DGN
8-31

Fig. 4.30. Maximum VOC concentrations in groundwater, cross section I-I', west waste disposal area.



WAG6 06F290.DGN
8-31

Fig. 4.31. Maximum VOC concentrations in groundwater, cross section J-J', south waste disposal area.



20.9 µg/L	
1,2-Dichloroethene	11
Chloromethane	4
Chloroform	2
Methylene Chloride	2
Trichloroethene	1
Carbon Tetrachloride	0.5
1,1,1-Trichloromethane	0.4

14 µg/L	
1,2-Dichloroethene	8
Chloroform	2
Methylene Chloride	2
Trichloroethene	2

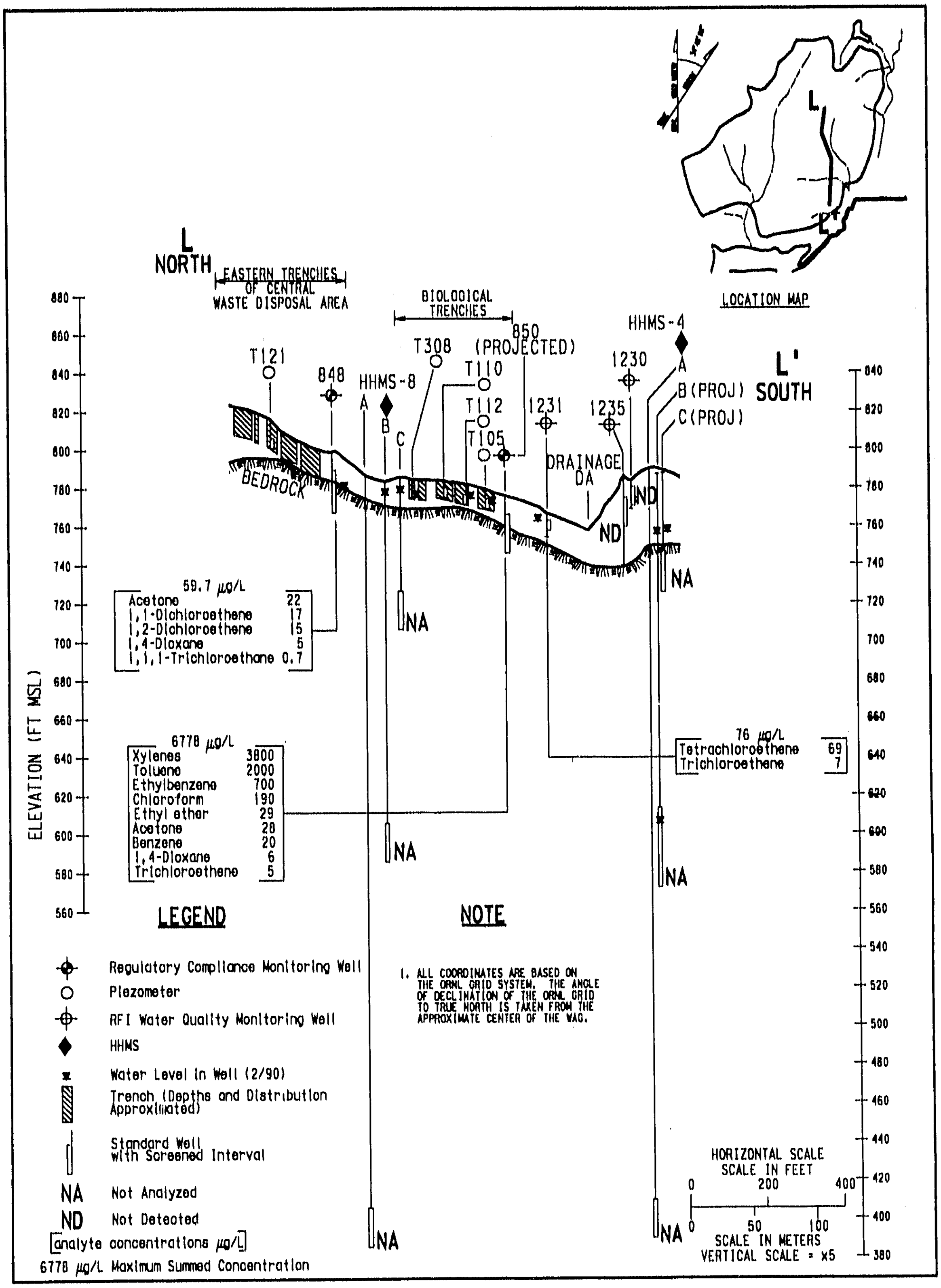
808.5 µg/L	
Trichloroethene	510
Chloroform	97
Carbon Tetrachloride	96
1,2-Dichloroethane	44
1,2-Dichloroethene	27
Tetrachloroethene	14
1,1-Dichloroethane	9
1,4-Dioxane	6
Ethyl chloride	4
Methylene chloride	1
1,1-Dichloroethene	0.5

3 µg/L	
Chloroform	2
Trichloroethene	1

54 µg/L	
Trichloroethene	19
1,2-Dichloroethene	18
Tetrachloroethene	16
Methylene chloride	1

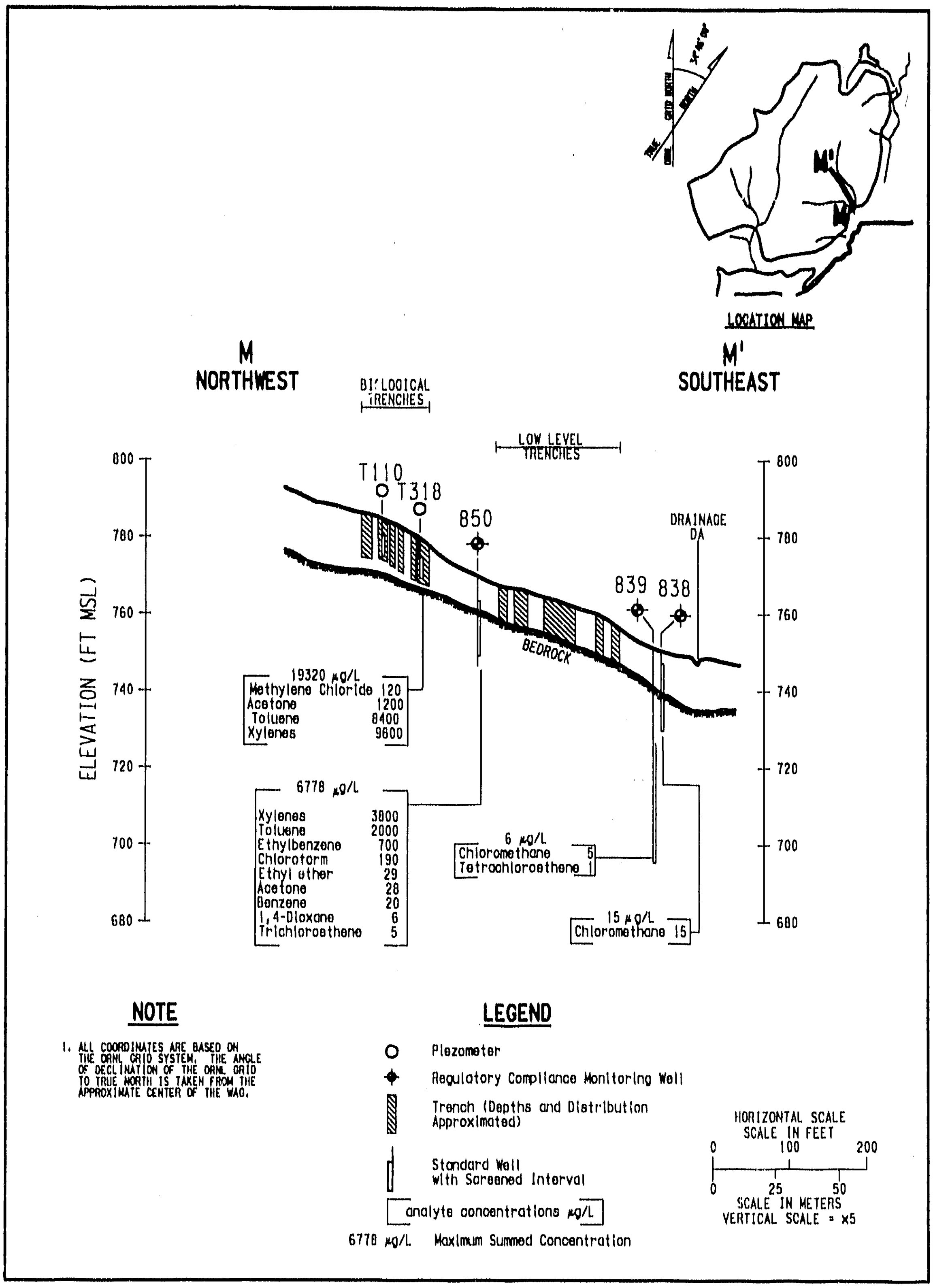
WAG6 06F284.DGN
8-31

Fig. 4.32. Maximum VOC concentrations in groundwater, cross section K-K', 19 Trench area and east waste disposal area.



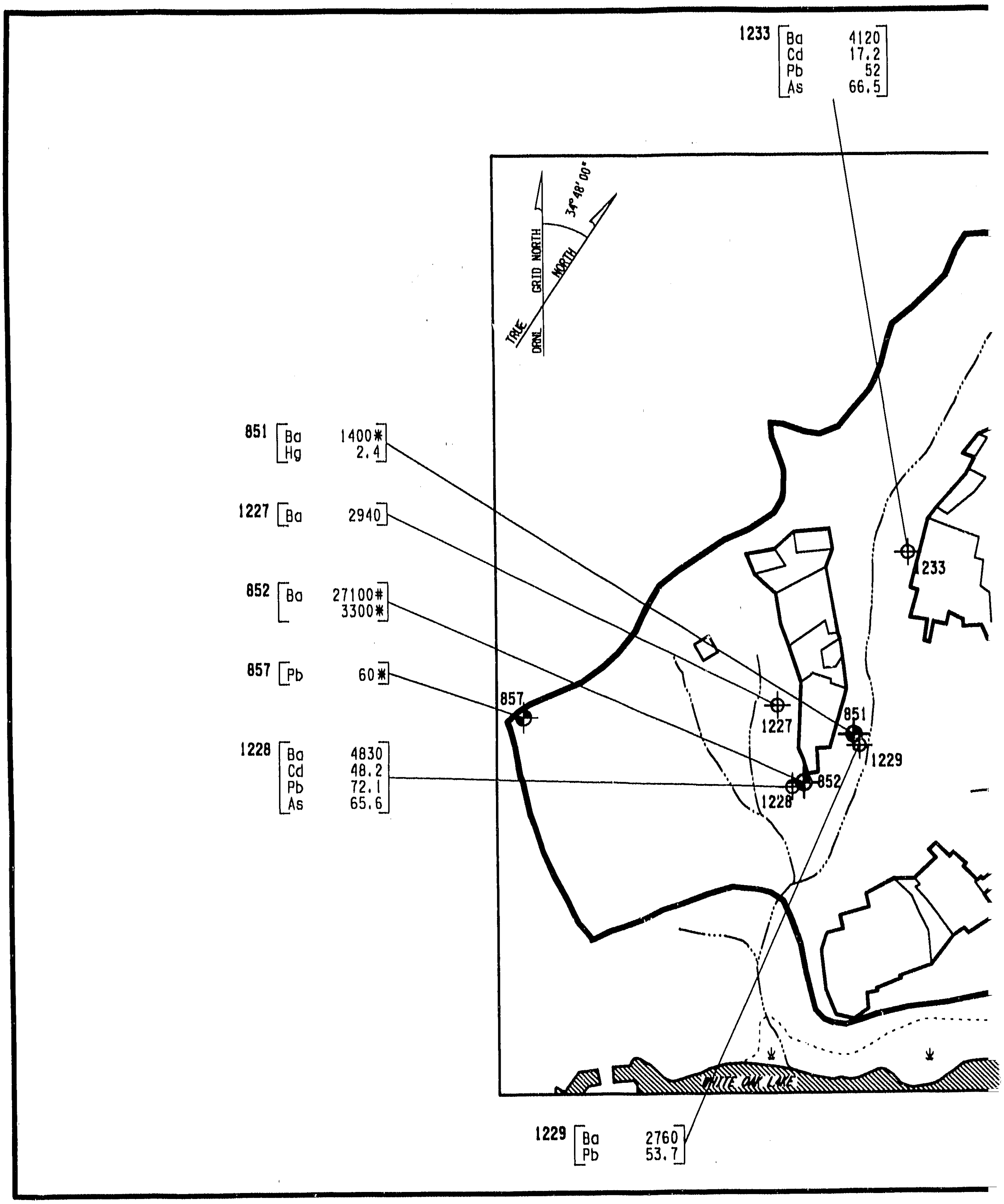
WAG6 06F285.DGN
8-31

Fig. 4.33. Maximum VOC concentrations in groundwater, cross section L-L', eastern trenches of the central waste disposal area.

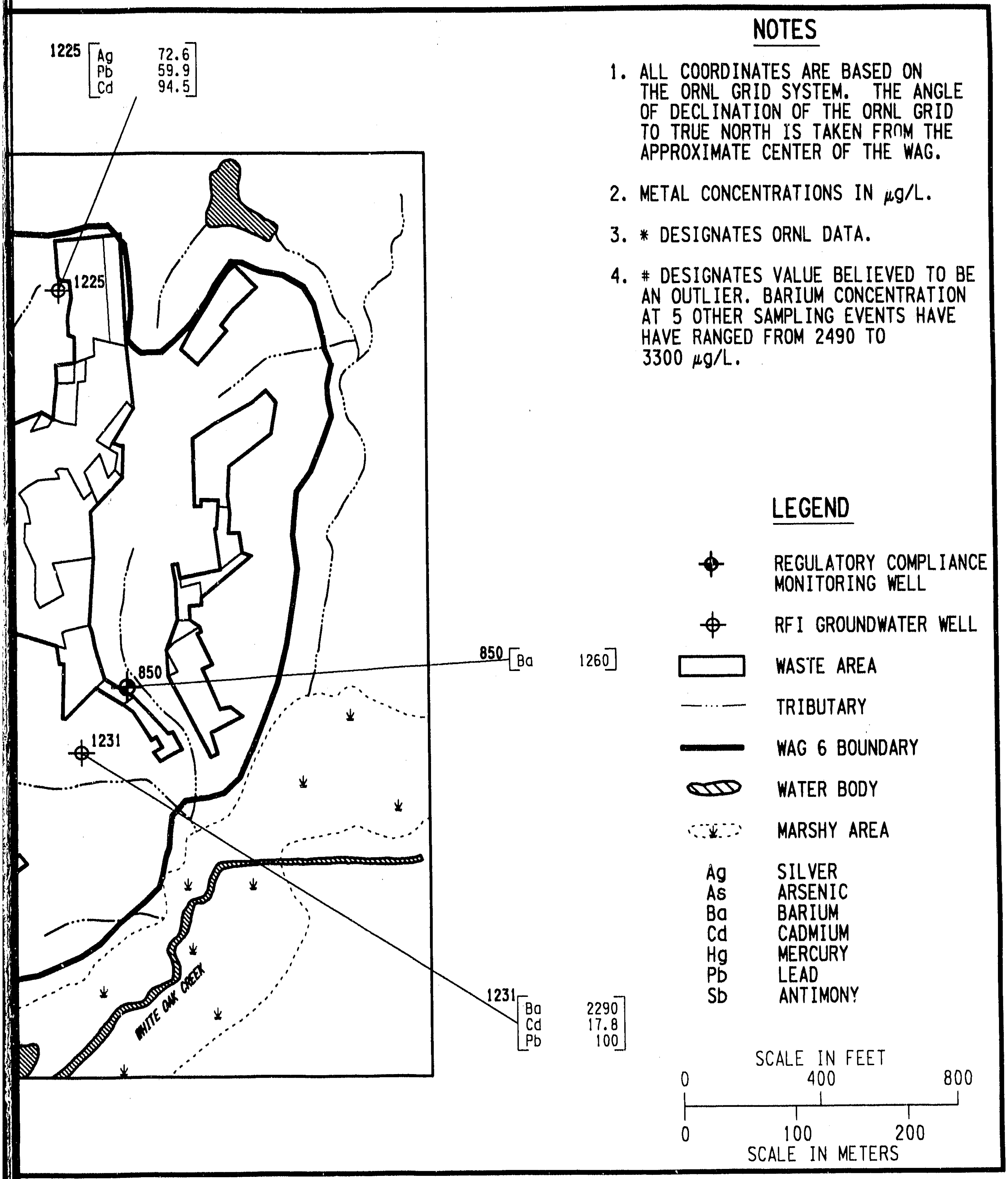


WAG6 06F289.DGN
9-9

Fig. 4.34. Maximum VOC concentrations in groundwater, cross section M-M', southeastern trenches of the central waste disposal area.



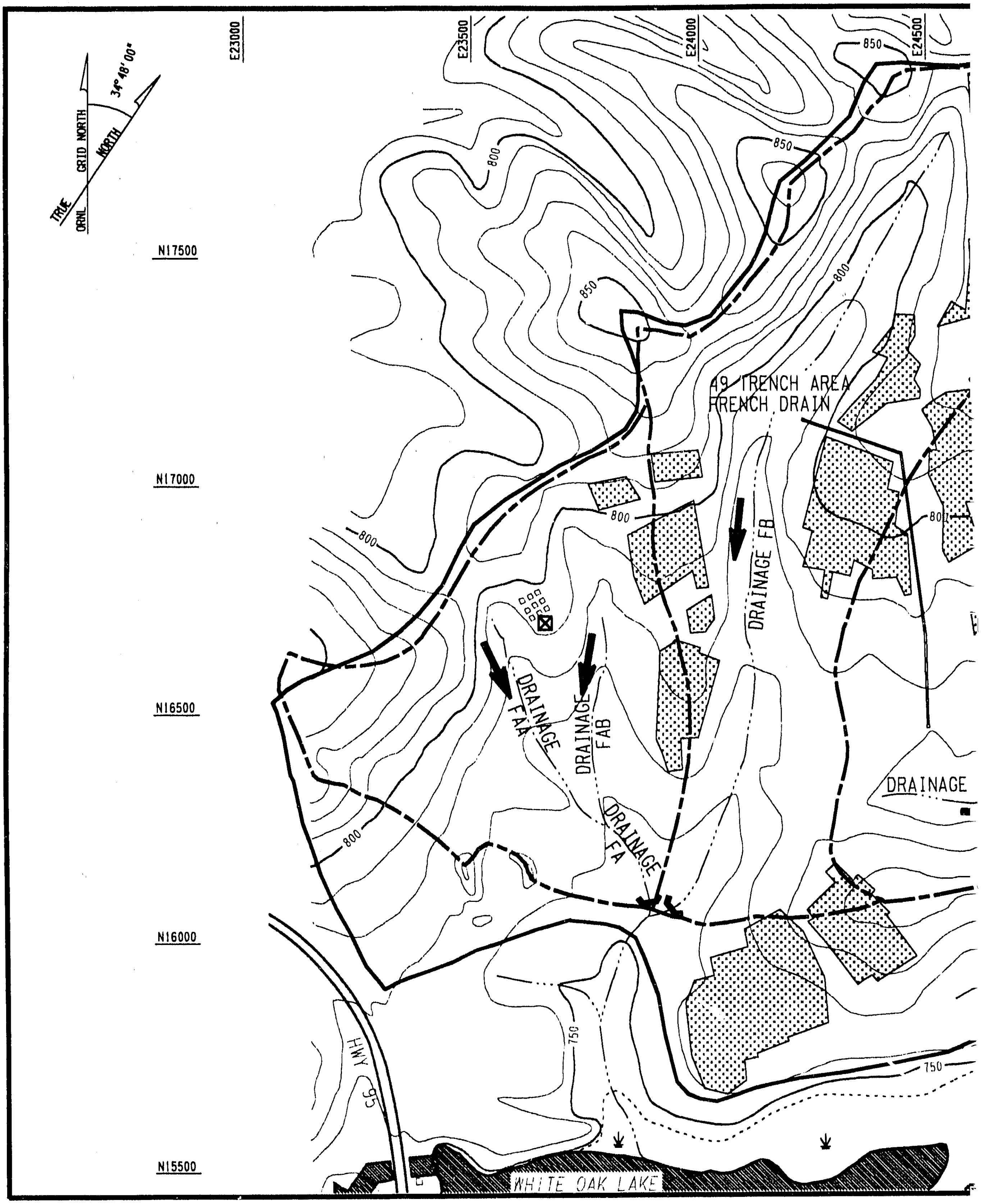
WAG6 06F281.DGN
9-1



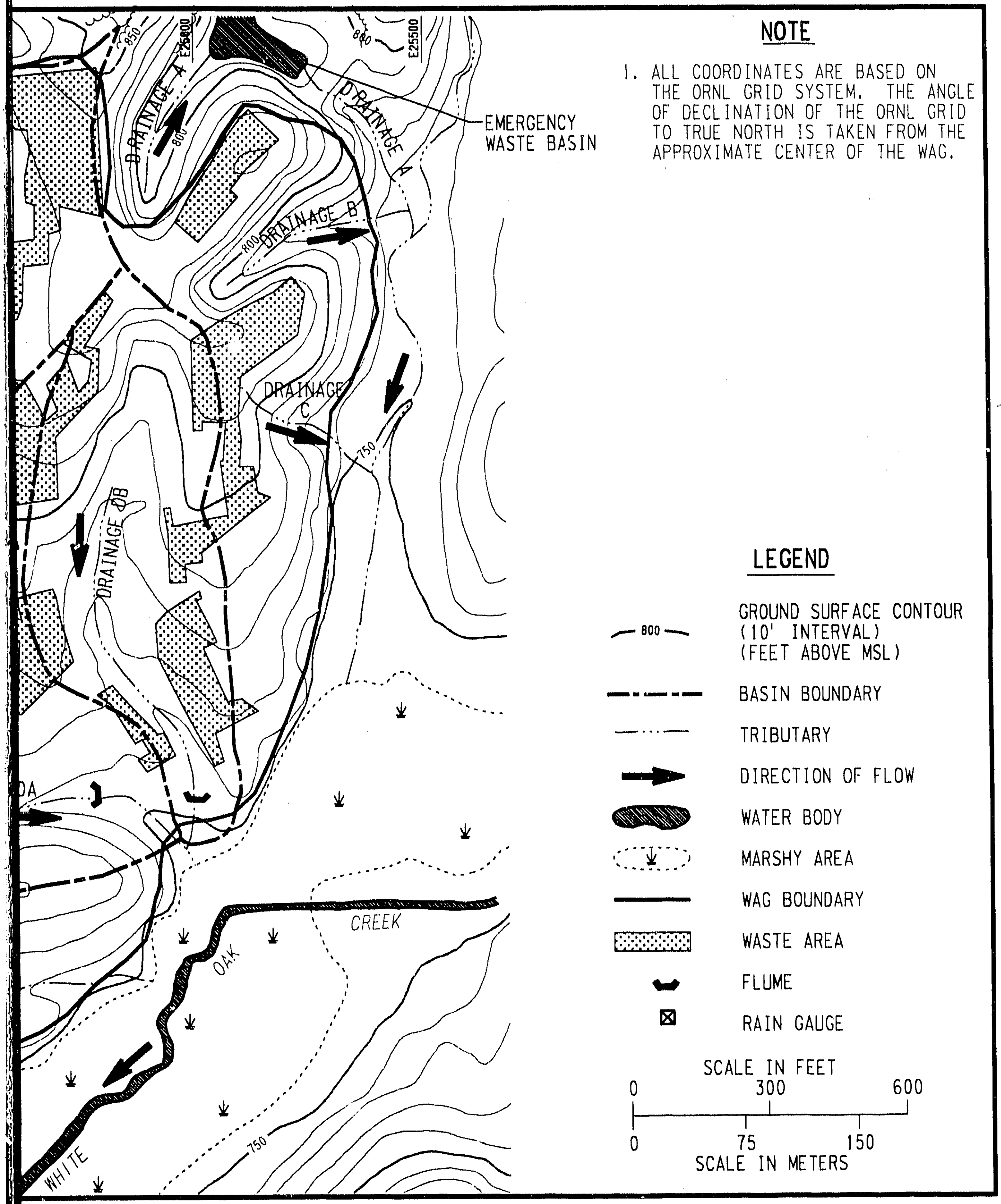
NOTES

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.
2. METAL CONCENTRATIONS IN μg/L.
3. * DESIGNATES ORNL DATA.
4. # DESIGNATES VALUE BELIEVED TO BE AN OUTLIER. BARIUM CONCENTRATION AT 5 OTHER SAMPLING EVENTS HAVE HAVE RANGED FROM 2490 TO 3300 μg/L.

Fig. 4.35. Maximum concentrations of selected unfiltered metals detected above MCLs in groundwater.



WAG6 06F203.DGN
8-31



NOTE

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.

Fig. 4.36. Topography and drainage patterns at WAG 6 and adjacent areas.

WFBB3	04/21/89	05/09/89	09/06/89	09/26/89
Flow Rate	No Flow	*	No Flow	*
Tritium		190,000		161,000
Co-60		--		--
Sr-90		5,900		10,222
Alpha		--		12
Beta		9,000		12,501

WFBB2	04/21/89	05/09/89	09/06/89	09/26/89
Flow Rate	*	1.3	No Flow	*
Tritium	60,800	62,000		59,000
Co-60	--	--		--
Sr-90	286	299		506
Alpha	20	8		--
Beta	500	78		815

WSP1	04/19/89	05/09/89	09/06/89	09/26/89
Flow Rate	No Flow	25.3	No Flow	1.3
Tritium		140,000		439,000
Co-60		--		--
Sr-90		126		252
Alpha		9		4
Beta		235		410

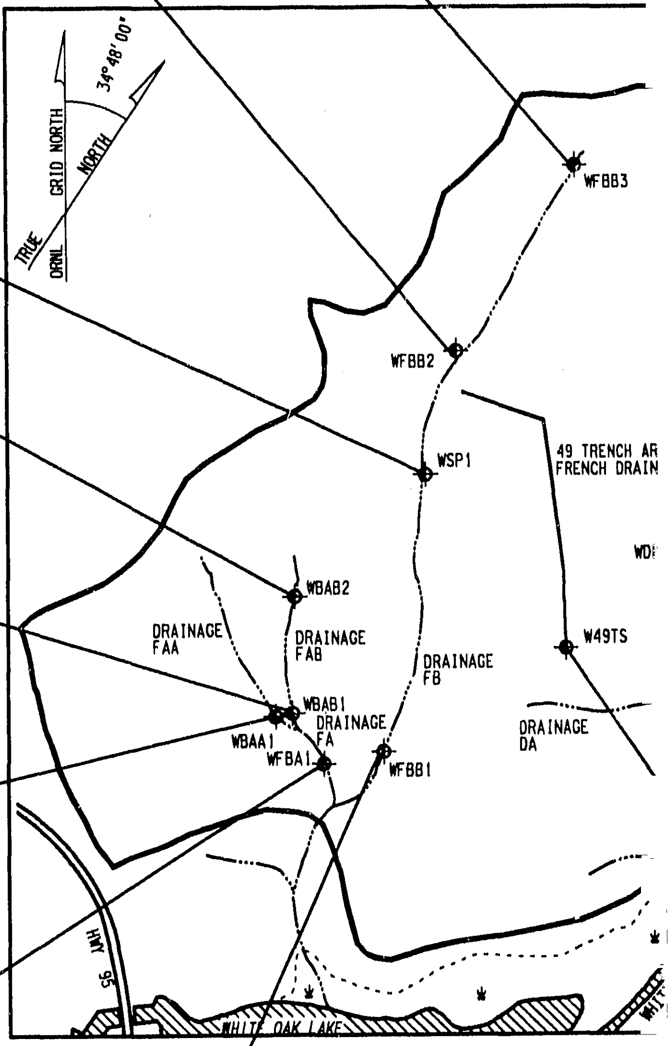
WBAB2	04/19/89	05/09/89	09/06/89	09/26/89
Flow Rate	No Flow	*	No Flow	*
Tritium		19.7		7.5
Co-60		--		--
Sr-90		--		--
Alpha		6		2
Beta		16		15

WBAB1	04/19/89	05/09/89	09/06/89	09/27/89
Flow Rate	No Flow	0.5	No Flow	*
Tritium		130,000		122,000
Co-60		--		--
Sr-90		--		--
Alpha		7		4
Beta		9		5

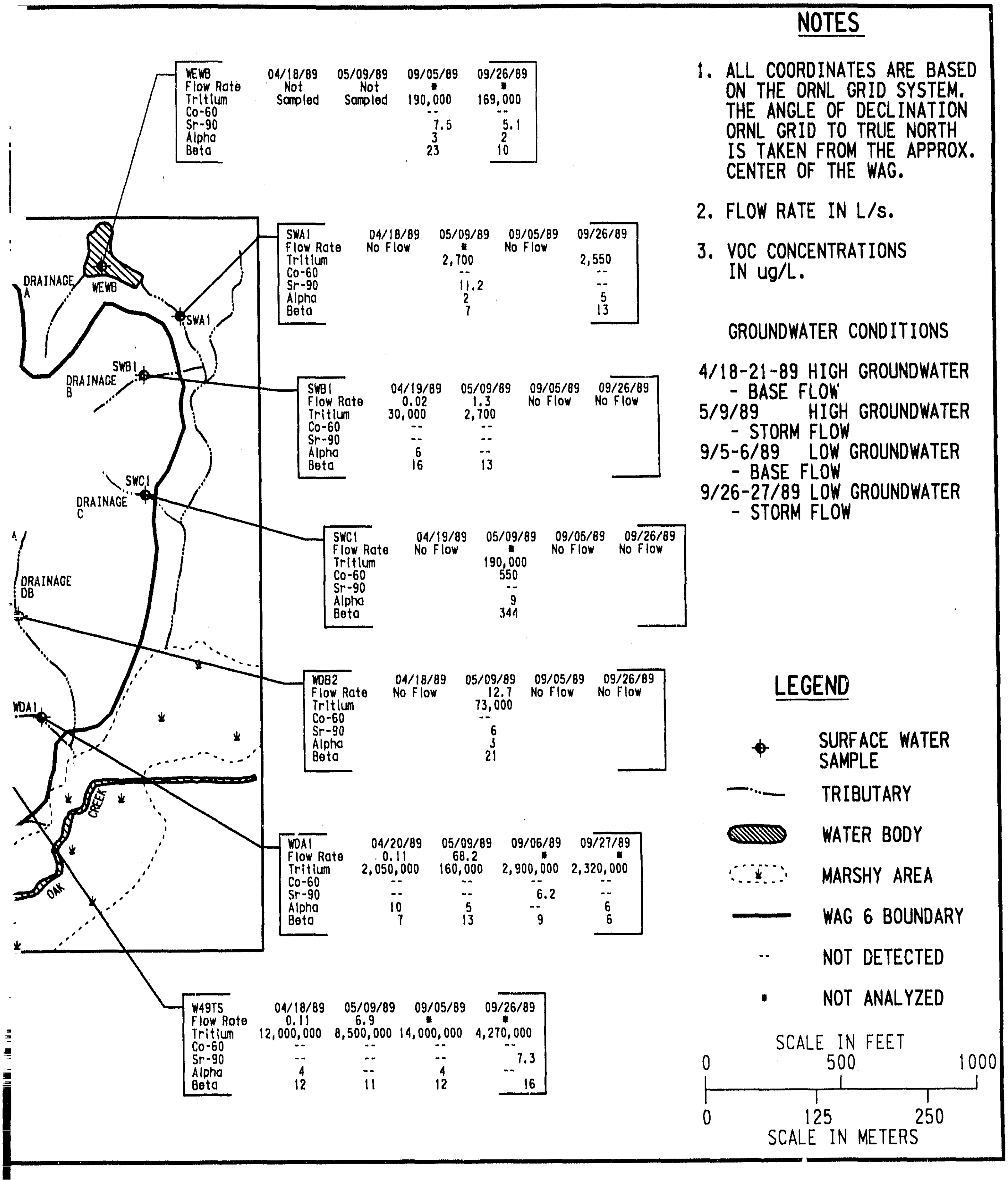
WBAA1	04/19/89	05/09/89	09/06/89	09/26/89
Flow Rate	No Flow	*	No Flow	No Flow
Tritium		1,500		
Co-60		--		
Sr-90		--		
Alpha		10		
Beta		12		

WFBA1	04/19/89	05/09/89	09/06/89	09/27/89
Flow Rate	*	5.1	No Flow	*
Tritium	86,000	96,000		103,000
Co-60	--	--		--
Sr-90	--	--		--
Alpha	3	--		2
Beta	--	--		5

WFBB1	04/19/89	05/09/89	09/06/89	09/27/89
Flow Rate	*	2.9		*
Tritium	3,100,000	2,200,000	3,600,000	3,910,000
Co-60	--	--	--	--
Sr-90	319	295	370	415
Alpha	10	6	6	57
Beta	575	479	490	639



WAG6 06F236.DGN
9-1



WEWB	04/18/89	05/09/89	09/05/89	09/26/89
Flow Rate	Not Sampled	Not Sampled	*	*
Tritium			190,000	169,000
Co-60			--	--
Sr-90			7.5	5.1
Alpha			3	2
Beta			23	10

SWA1	04/18/89	05/09/89	09/05/89	09/26/89
Flow Rate	No Flow	*	No Flow	
Tritium		2,700		2,550
Co-60		--		--
Sr-90		11.2		--
Alpha		2		5
Beta		7		13

SWB1	04/19/89	05/09/89	09/05/89	09/26/89
Flow Rate	0.02	1.3	No Flow	No Flow
Tritium	30,000	2,700		
Co-60	--	--		
Sr-90	--	--		
Alpha	6	--		
Beta	16	13		

SWC1	04/19/89	05/09/89	09/05/89	09/26/89
Flow Rate	No Flow	*	No Flow	No Flow
Tritium		190,000		
Co-60		550		
Sr-90		--		
Alpha		9		
Beta		344		

WDB2	04/18/89	05/09/89	09/05/89	09/26/89
Flow Rate	No Flow	12.7	No Flow	No Flow
Tritium		73,000		
Co-60		--		
Sr-90		6		
Alpha		3		
Beta		21		

WDA1	04/20/89	05/09/89	09/06/89	09/27/89
Flow Rate	0.11	68.2	*	*
Tritium	2,050,000	160,000	2,900,000	2,320,000
Co-60	--	--	--	--
Sr-90	--	--	6.2	--
Alpha	10	5	--	6
Beta	7	13	9	6

W49TS	04/18/89	05/09/89	09/05/89	09/26/89
Flow Rate	0.11	6.9	*	*
Tritium	12,000,000	8,500,000	14,000,000	4,270,000
Co-60	--	--	--	--
Sr-90	--	--	--	7.3
Alpha	4	--	4	--
Beta	12	11	12	16

NOTES

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2. FLOW RATE IN L/s.
3. VOC CONCENTRATIONS IN ug/L.

GROUNDWATER CONDITIONS

4/18-21-89 HIGH GROUNDWATER - BASE FLOW
5/9/89 HIGH GROUNDWATER - STORM FLOW
9/5-6/89 LOW GROUNDWATER - BASE FLOW
9/26-27/89 LOW GROUNDWATER - STORM FLOW

LEGEND

⊕ SURFACE WATER SAMPLE
-·-·- TRIBUTARY
▨ WATER BODY
MARSHY AREA
WAG 6 BOUNDARY
-- NOT DETECTED
* NOT ANALYZED

Fig. 4.37. Radionuclides detected in WAG 6 surface water.

WFBB3

Parameter	Units	04/21/89	05/9/89	06/09/89	09/06/89	09/26/89
						*
1,2-DCE	(ug/L)		11	5		4
PCE	(ug/L)			--		--
TCE	(ug/L)		7	3		1

WFBB2

Parameter	Units	04/21/89	06/09/89	09/06/89	09/26/89
Flow Rate	(L/s)	*	*	No Flow	*
1,2-DCE	(ug/L)	180	120		31
PCE	(ug/L)	1	2		1
TCE	(ug/L)	28	19		10

WSP1

Parameter	Units	04/19/89	06/09/89	09/06/89	09/26/89
Flow Rate	(L/s)	No Flow	*	No Flow	1.3
1,2-DCE	(ug/L)		30		11
PCE	(ug/L)		7		9
TCE	(ug/L)		10		6

WBAB2

Parameter	Units	04/19/89	06/09/89	09/06/89	09/26/89
Flow Rate	(L/s)	No Flow	*	No Flow	
1,2-DCE	(ug/L)				
PCE	(ug/L)				
TCE	(ug/L)				

WBAB1

Parameter	Units	04/19/89	06/09/89	09/06/89	09/27/89
Flow Rate	(L/s)	No Flow	*	No Flow	
1,2-DCE	(ug/L)		--		--
PCE	(ug/L)		1		--
TCE	(ug/L)		--		--

WBAA1

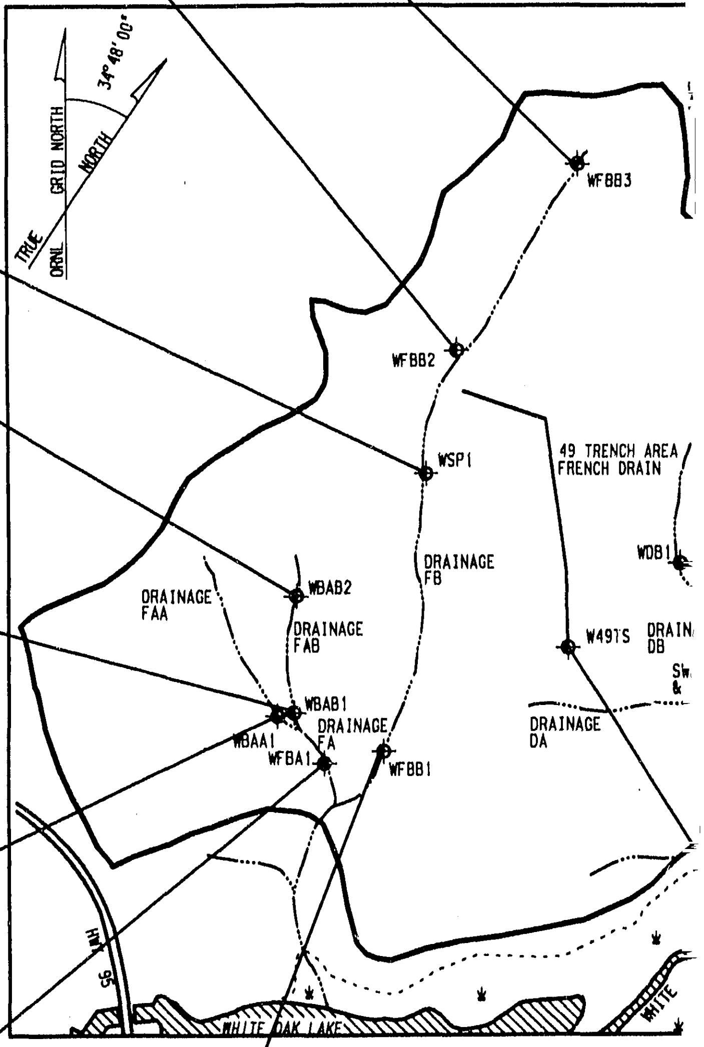
Parameter	Units	04/19/89	06/09/89	09/06/89	09/26/89
Flow Rate	(L/s)	No Flow	*	No Flow	No Flow
1,2-DCE	(ug/L)		6		
PCE	(ug/L)		70		
TCE	(ug/L)		5		

WFBA1

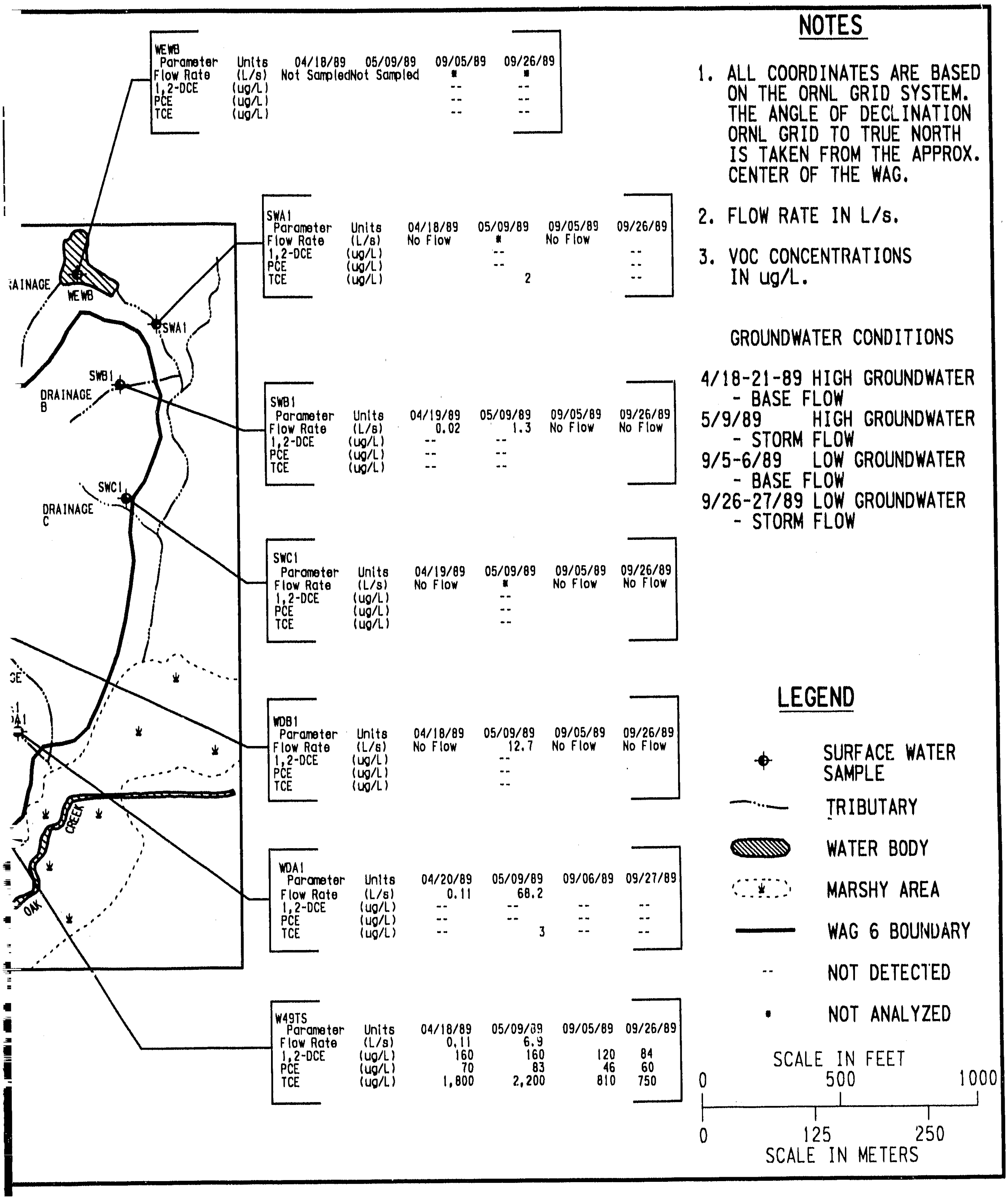
Parameter	Units	04/19/89	06/09/89	09/06/89	09/27/89
Flow Rate	(L/s)	*	*	No Flow	
1,2-DCE	(ug/L)	--	--		--
PCE	(ug/L)	--	--		--
TCE	(ug/L)	--	--		--

WFBB1

Parameter	Units	04/19/89	06/09/89	09/06/89	09/27/89
Flow Rate	(L/s)	*	*	*	*
1,2-DCE	(ug/L)	--	25	--	7
PCE	(ug/L)	--	5	--	5
TCE	(ug/L)	--	7	--	5



WAG6 SURFMODL.DGN
9-1



WEWB

Parameter	Units	04/18/89	05/09/89	09/05/89	09/26/89
Flow Rate	(L/s)	Not Sampled	Not Sampled	*	*
1,2-DCE	(ug/L)			--	--
PCE	(ug/L)			--	--
TCE	(ug/L)			--	--

SWA1

Parameter	Units	04/18/89	05/09/89	09/05/89	09/26/89
Flow Rate	(L/s)	No Flow	*	No Flow	
1,2-DCE	(ug/L)		--		--
PCE	(ug/L)		--		--
TCE	(ug/L)		2		--

SWB1

Parameter	Units	04/19/89	05/09/89	09/05/89	09/26/89
Flow Rate	(L/s)	0.02	1.3	No Flow	No Flow
1,2-DCE	(ug/L)	--	--		
PCE	(ug/L)	--	--		
TCE	(ug/L)	--	--		

SWC1

Parameter	Units	04/19/89	05/09/89	09/05/89	09/26/89
Flow Rate	(L/s)	No Flow	*	No Flow	No Flow
1,2-DCE	(ug/L)		--		
PCE	(ug/L)		--		
TCE	(ug/L)		--		

WDB1

Parameter	Units	04/18/89	05/09/89	09/05/89	09/26/89
Flow Rate	(L/s)	No Flow	12.7	No Flow	No Flow
1,2-DCE	(ug/L)		--		
PCE	(ug/L)		--		
TCE	(ug/L)		--		

WDA1

Parameter	Units	04/20/89	05/09/89	09/06/89	09/27/89
Flow Rate	(L/s)	0.11	68.2		
1,2-DCE	(ug/L)	--	--	--	--
PCE	(ug/L)	--	--	--	--
TCE	(ug/L)	--	3	--	--

W49TS

Parameter	Units	04/18/89	05/09/89	09/05/89	09/26/89
Flow Rate	(L/s)	0.11	6.9		
1,2-DCE	(ug/L)	160	160	120	84
PCE	(ug/L)	70	83	46	60
TCE	(ug/L)	1,800	2,200	810	750

NOTES

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2. FLOW RATE IN L/s.
3. VOC CONCENTRATIONS IN ug/L.

GROUNDWATER CONDITIONS

4/18-21-89 HIGH GROUNDWATER - BASE FLOW
5/9/89 HIGH GROUNDWATER - STORM FLOW
9/5-6/89 LOW GROUNDWATER - BASE FLOW
9/26-27/89 LOW GROUNDWATER - STORM FLOW

LEGEND

Fig. 4.38. VOCs detected in WAG 6 surface water on selected dates.

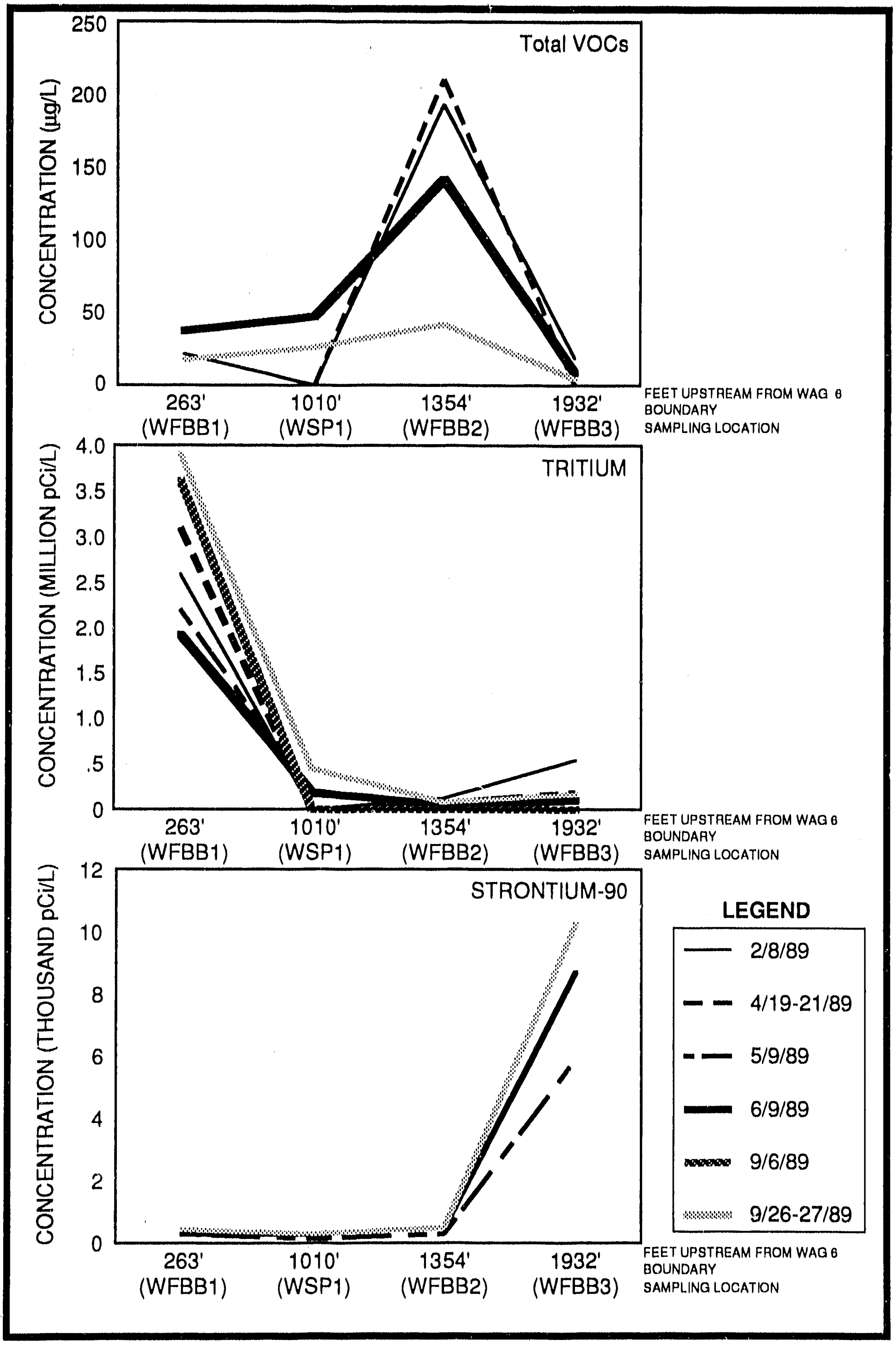
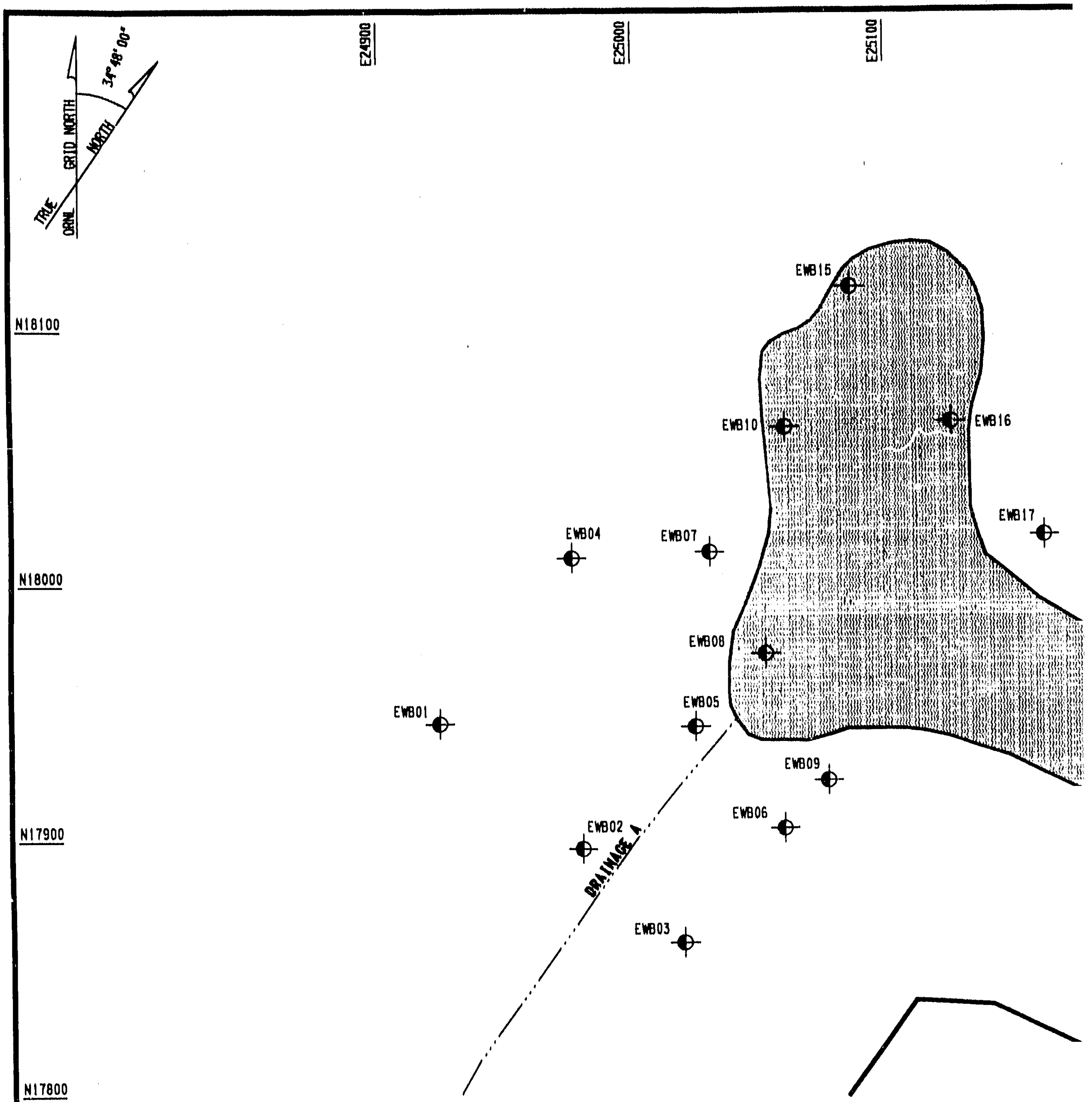


Fig. 4.39. Concentrations of total VOCs, tritium, and strontium-90 in FB drainage.

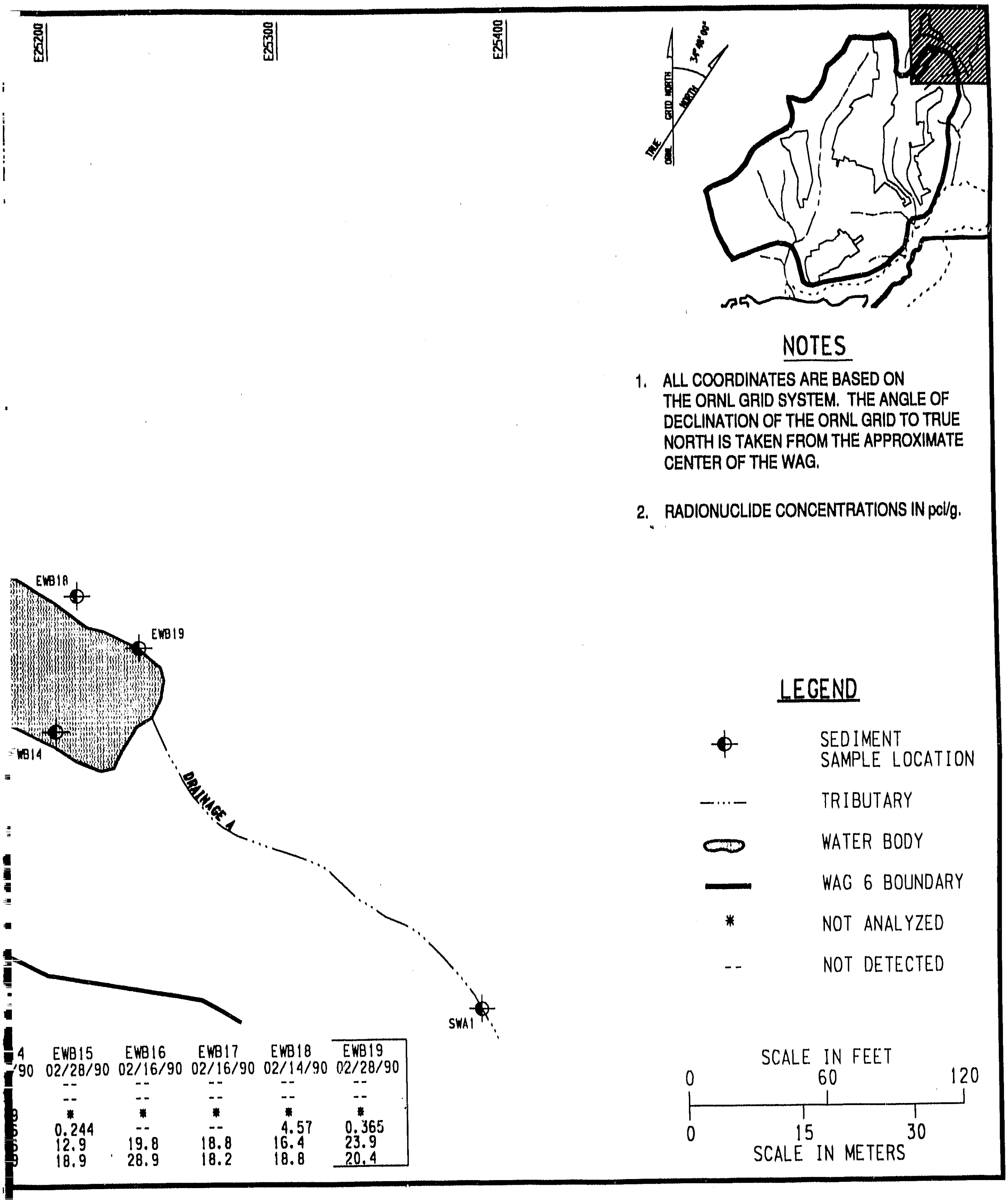
5.18 4104.1



Emergency Waste Basin Sediment Samples

Location	EWB01	EWB02	EWB04	EWB05	EWB06	EWB07	EWB08	EWB09	EWB10	EW
Date	02/20/90	02/20/90	02/20/90	02/20/90	02/20/90	03/01/90	03/02/90	02/20/90	03/01/90	02/1
Tritium	--	197	--	518	--	--	453	--	--	--
Co-60	--	--	--	0.0775	--	--	--	--	--	--
Sr-90	*	*	*	*	*	*	*	*	*	1.4
Cs-137	--	0.493	0.316	2.03	0.906	0.315	1.76	--	0.944	1.9
Alpha	22.9	24.3	15.7	16	11.7	10.7	24.3	14.9	23.8	1
Beta	24	20.7	22.2	21.6	23.3	22.2	21.6	32.8	22.6	24.

WAG6 06F249.DGN
08/30/91



NOTES

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.

2. RADIONUCLIDE CONCENTRATIONS IN pci/g.

LEGEND

⊕	SEDIMENT SAMPLE LOCATION
—···—	TRIBUTARY
⬭	WATER BODY
━━	WAG 6 BOUNDARY
*	NOT ANALYZED
--	NOT DETECTED

4	EWB15	EWB16	EWB17	EWB18	EWB19
/90	02/28/90	02/16/90	02/16/90	02/14/90	02/28/90
	--	--	--	--	--
	--	--	--	--	--
9	*	*	*	*	*
6	0.244	--	--	4.57	0.365
6	12.9	19.8	18.8	16.4	23.9
9	18.9	28.9	18.2	18.8	20.4

Fig. 4.40. Radionuclide concentrations in Emergency Waste Basin sediment samples.

WFBB2	02/14/90
Tritium	--
Co-60	0.106
Sr-90	*
Cs-137	1.13
Alpha	26
Beta	42

WSP1	02/13/90
Tritium	557
Co-60	--
Sr-90	*
Cs-137	0.652
Alpha	25
Beta	17

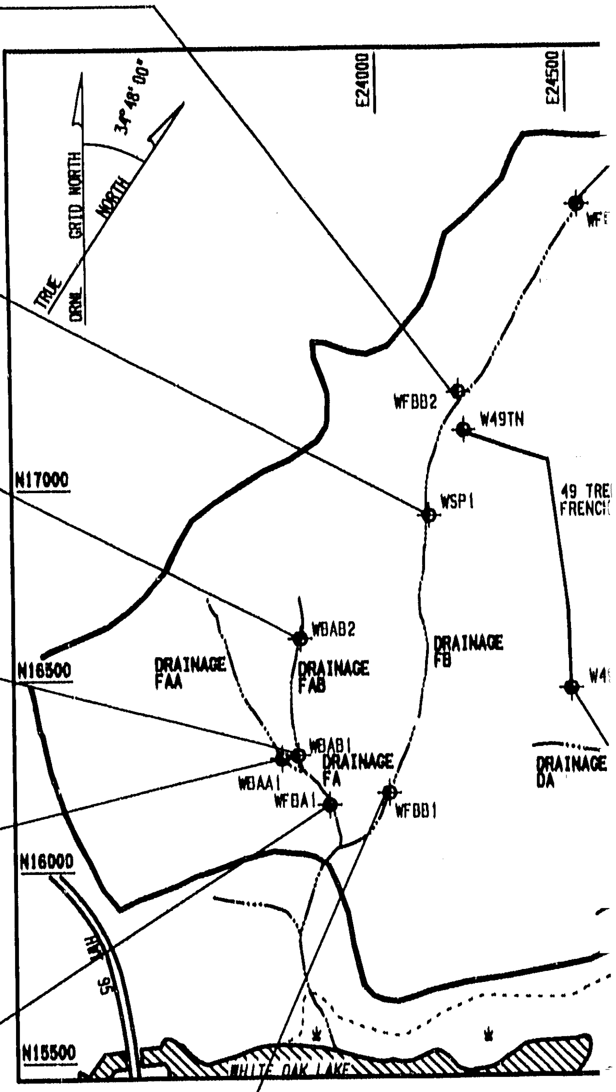
WBAB2	02/14/90
Tritium	312
Co-60	--
Sr-90	*
Cs-137	--
Alpha	14
Beta	24

WBAB1	02/13/90
Tritium	--
Co-60	--
Sr-90	*
Cs-137	0.368
Alpha	14
Beta	21

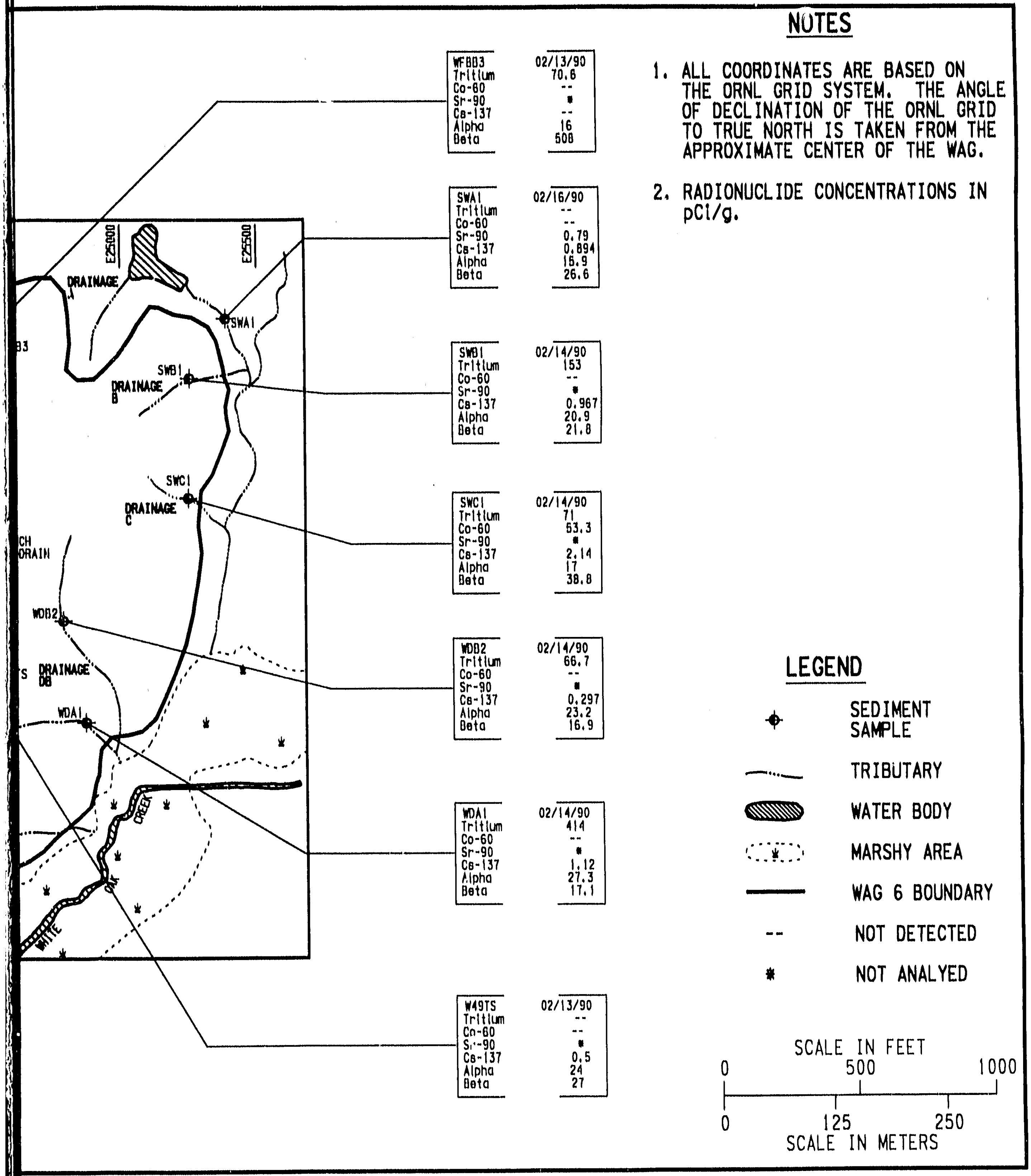
WBAA1	02/13/90
Tritium	--
Co-60	--
Sr-90	*
Cs-137	0.748
Alpha	20
Beta	19

WFBA1	02/13/90
Tritium	78.8
Co-60	--
Sr-90	*
Cs-137	--
Alpha	19
Beta	13

WFBB1	02/15/90
Tritium	840
Co-60	--
Sr-90	25.5
Cs-137	0.46
Alpha	15.6
Beta	46.1



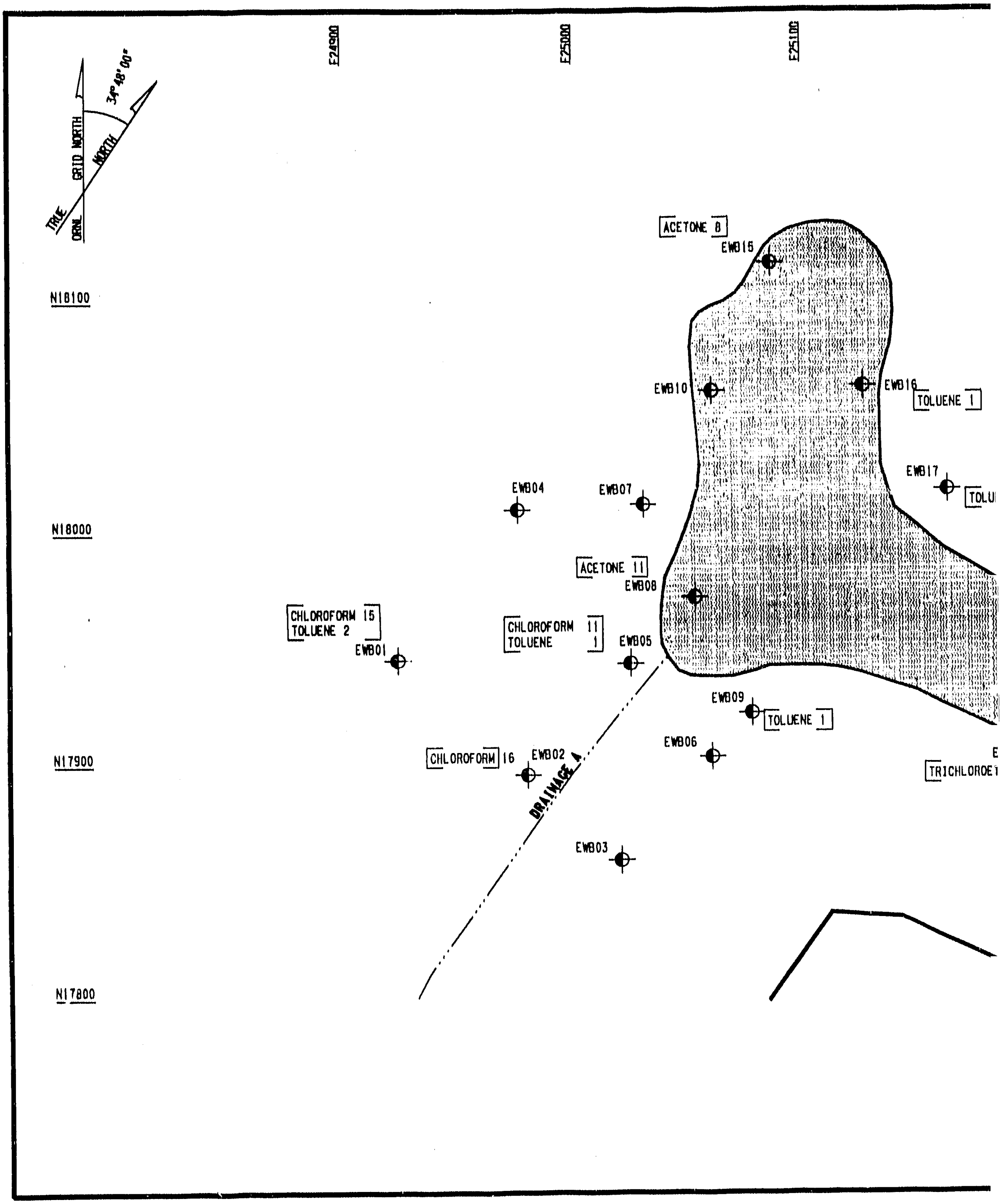
WAG6 06F248.DGN
8-31



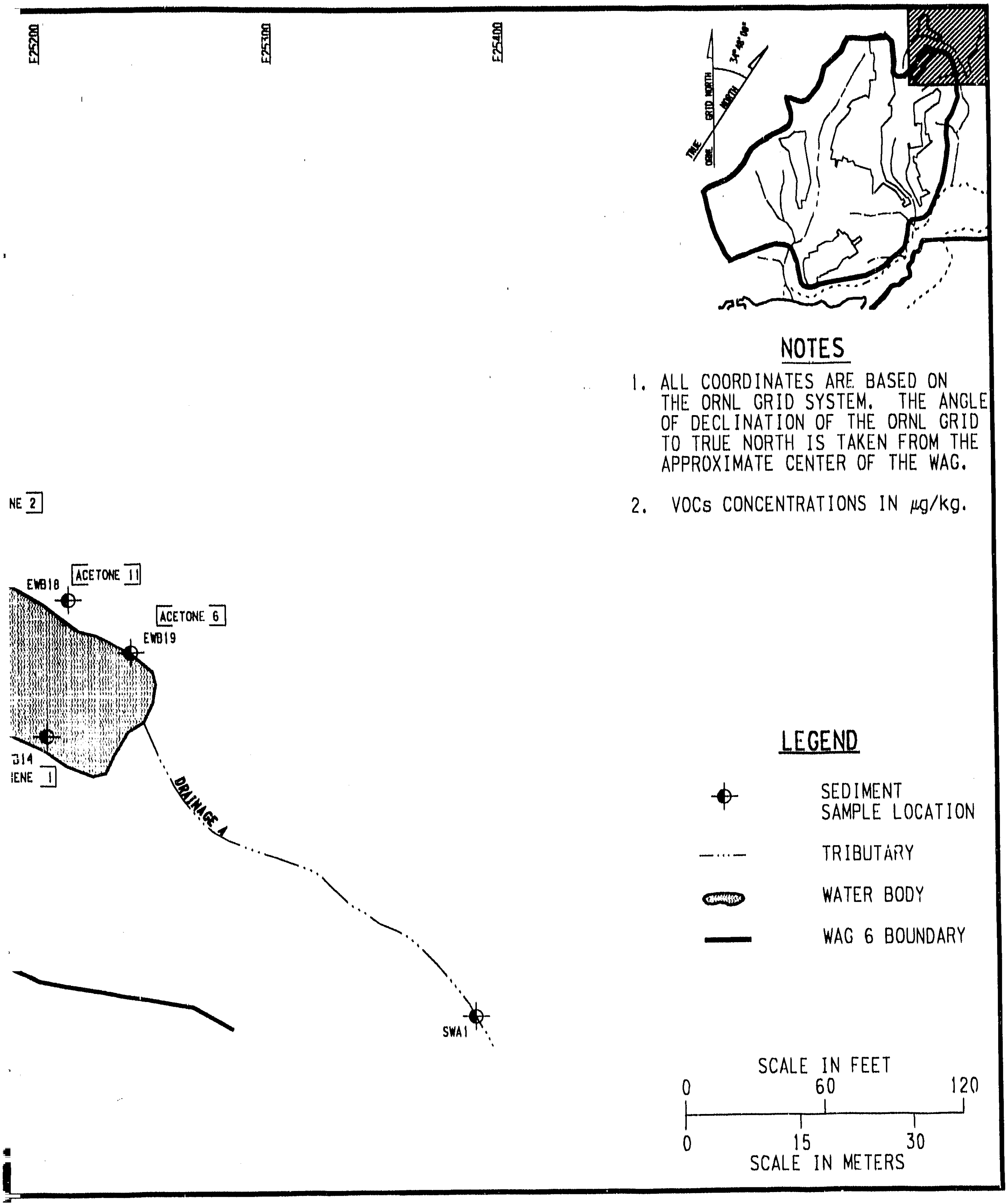
NOTES

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.
2. RADIONUCLIDE CONCENTRATIONS IN pCi/g.

Fig. 4.41. Radionuclides detected in WAG 6 sediment samples.



WAG6 06F306.DGN
08/30/91



NOTES

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.
2. VOCs CONCENTRATIONS IN µg/kg.

LEGEND

⌖	SEDIMENT SAMPLE LOCATION
—··—	TRIBUTARY
▨	WATER BODY
▬	WAG 6 BOUNDARY

Fig. 4.42. VOCs detected in Emergency Waste Basin sediment samples.

Trichloroethene	2
Chloroform	17
Toluene	2

Toluene	5
Chloroform	64
Trichloroethene	3
Propylene dichloride	3

Ethylbenzene	8
Trichloroethene	17
Tetrachloroethene	3
1,2-Dichloroethene	15
Chloroform	2

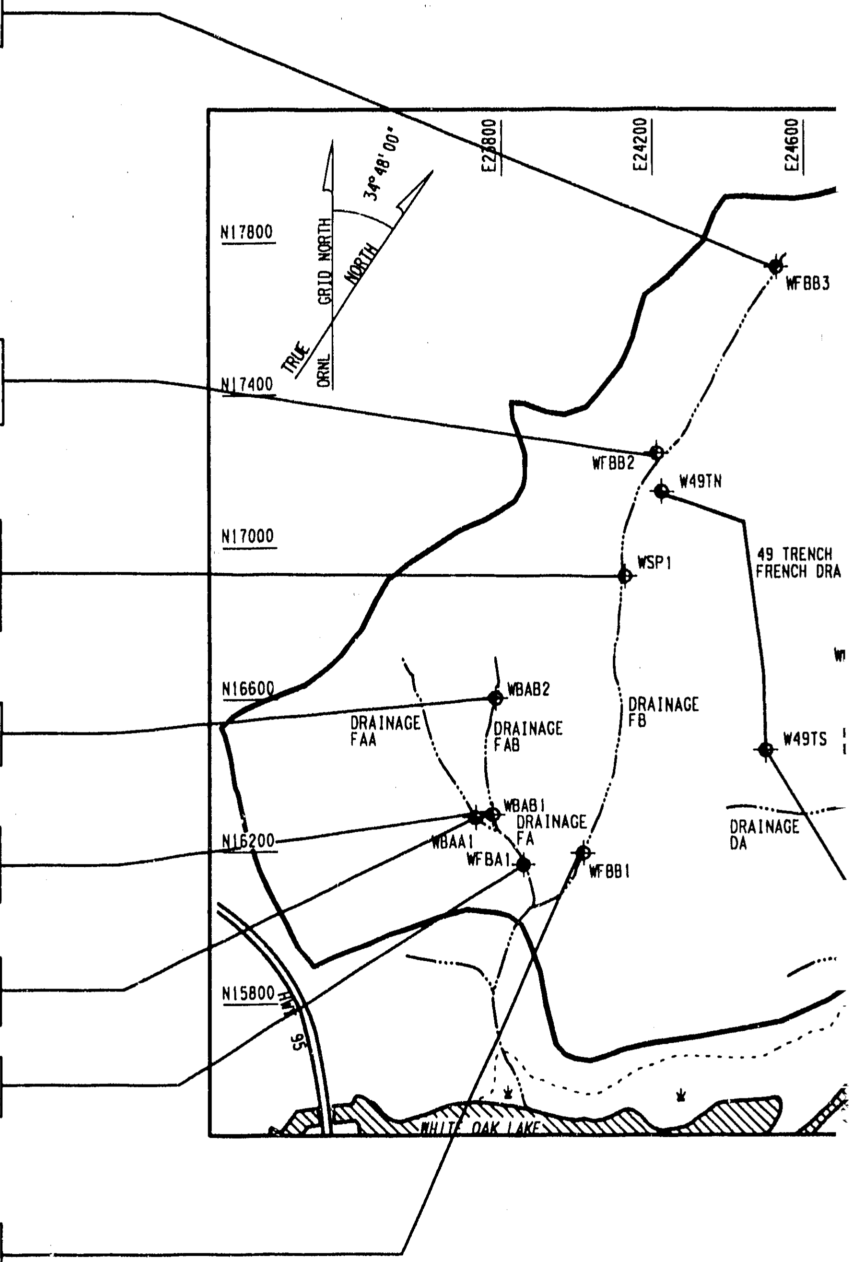
Toluene	3
Chloroform	23
Trichloroethene	2

Xylenes	2
Trichloroethene	3
Chloroform	29
Toluene	3

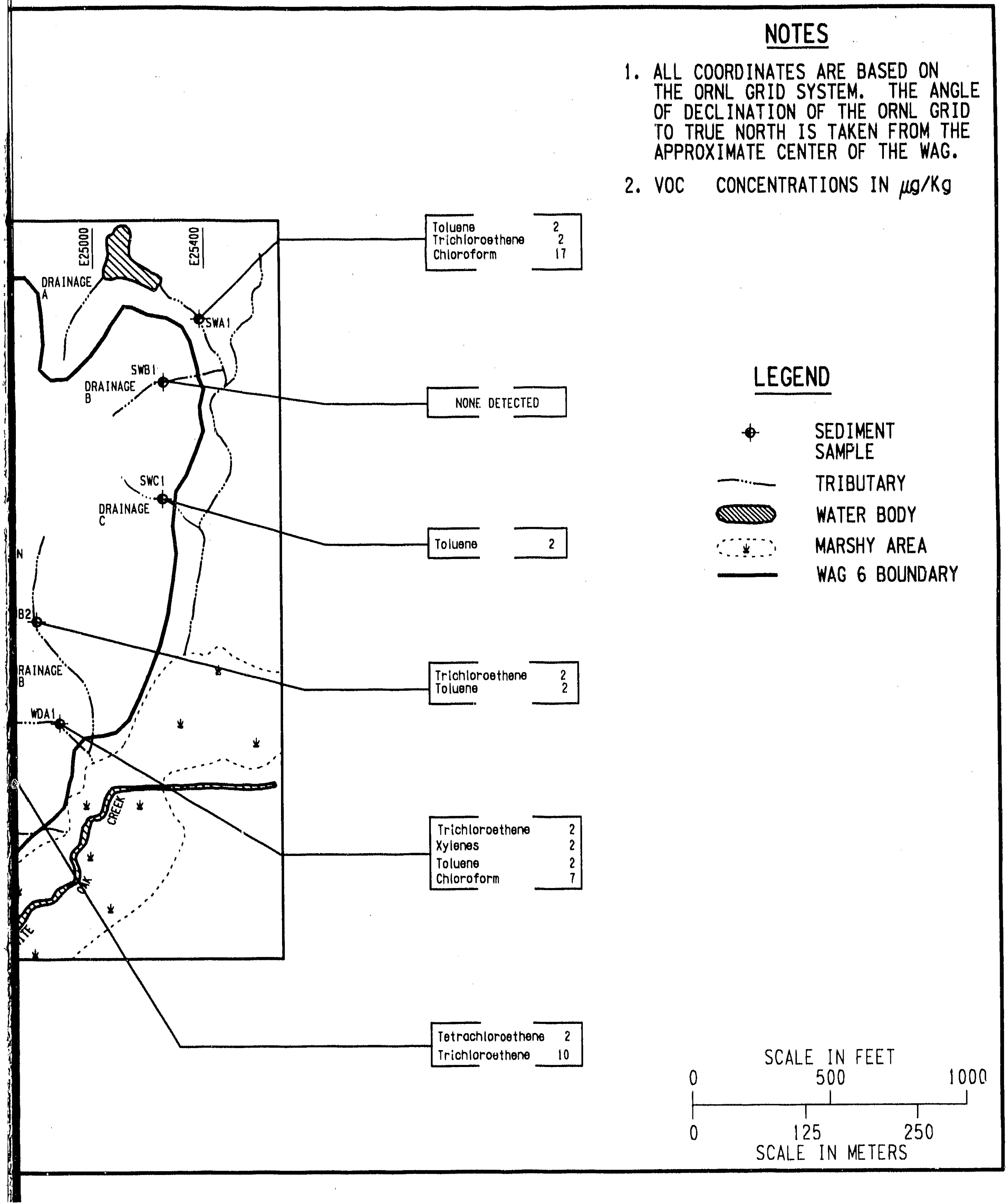
Trichloroethene	2
Chloroform	12
Toluene	2

Toluene	2
Chloroform	10
Trichloroethene	2

Trichloroethene	3
Methylbromide	10



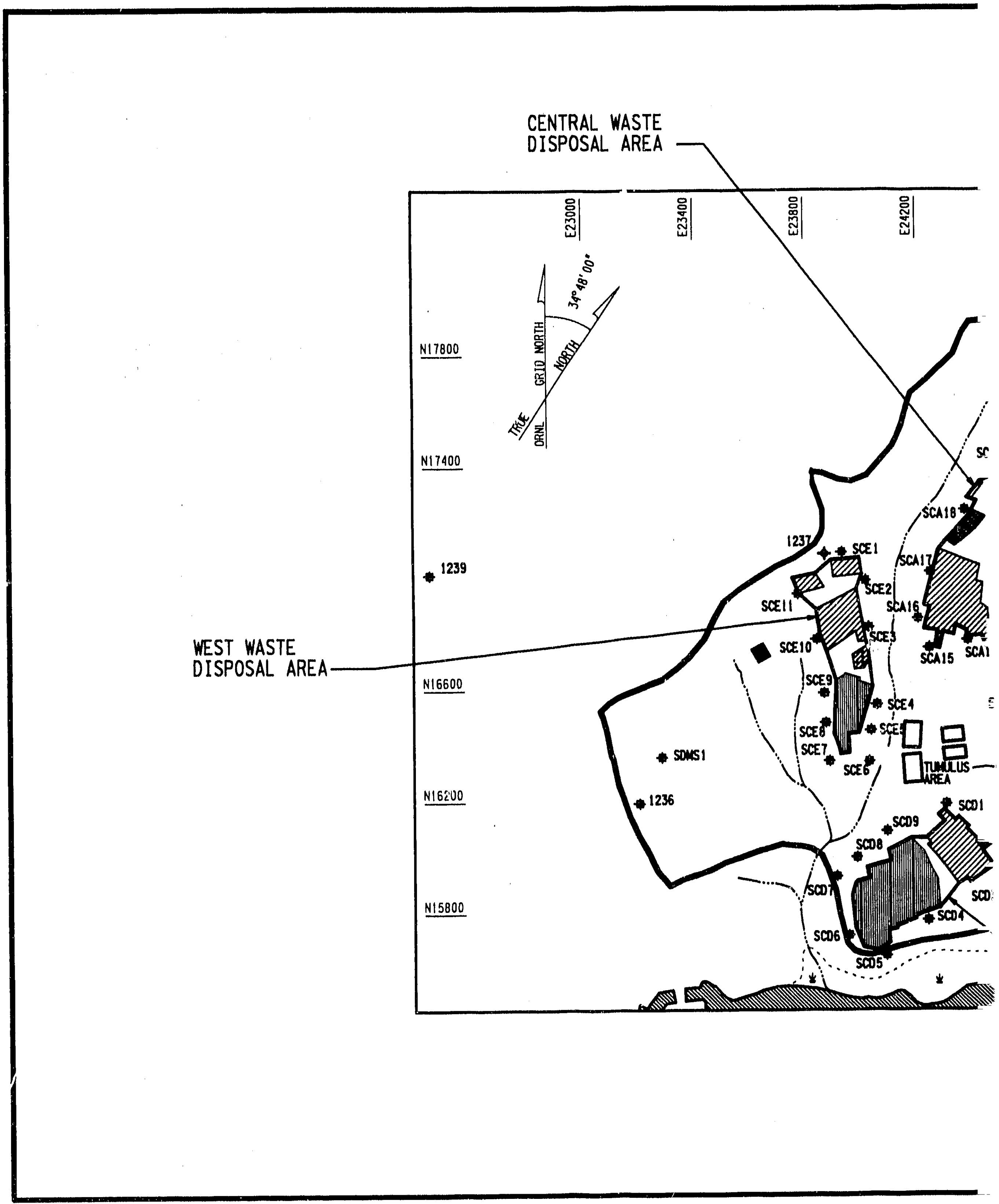
WAG6 06F307.DGN
9-1



NOTES

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.
2. VOC CONCENTRATIONS IN μg/Kg

Fig. 4.43. VOCs detected in WAG 6 sediment samples.



WAG6 06F237.DGN

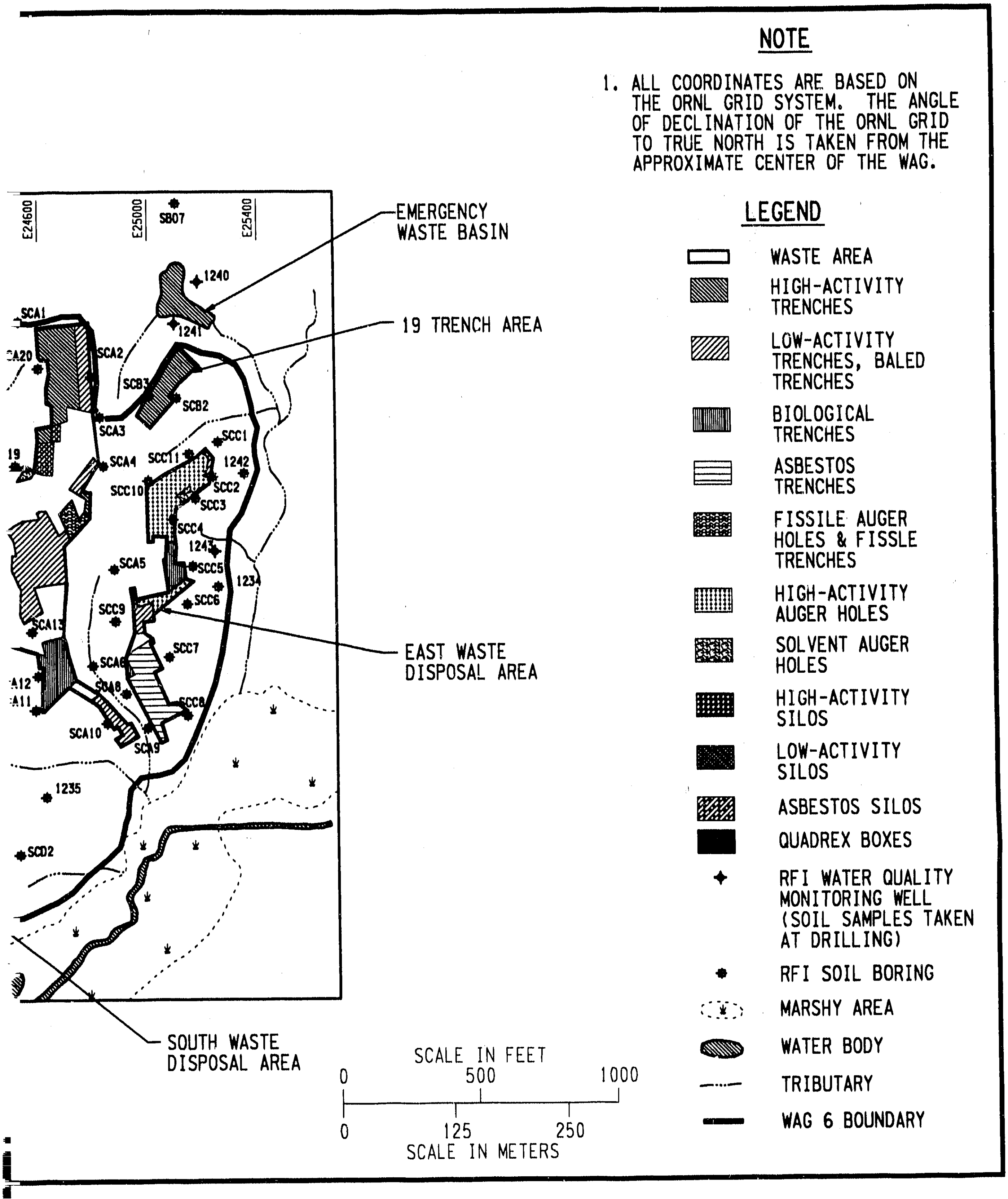
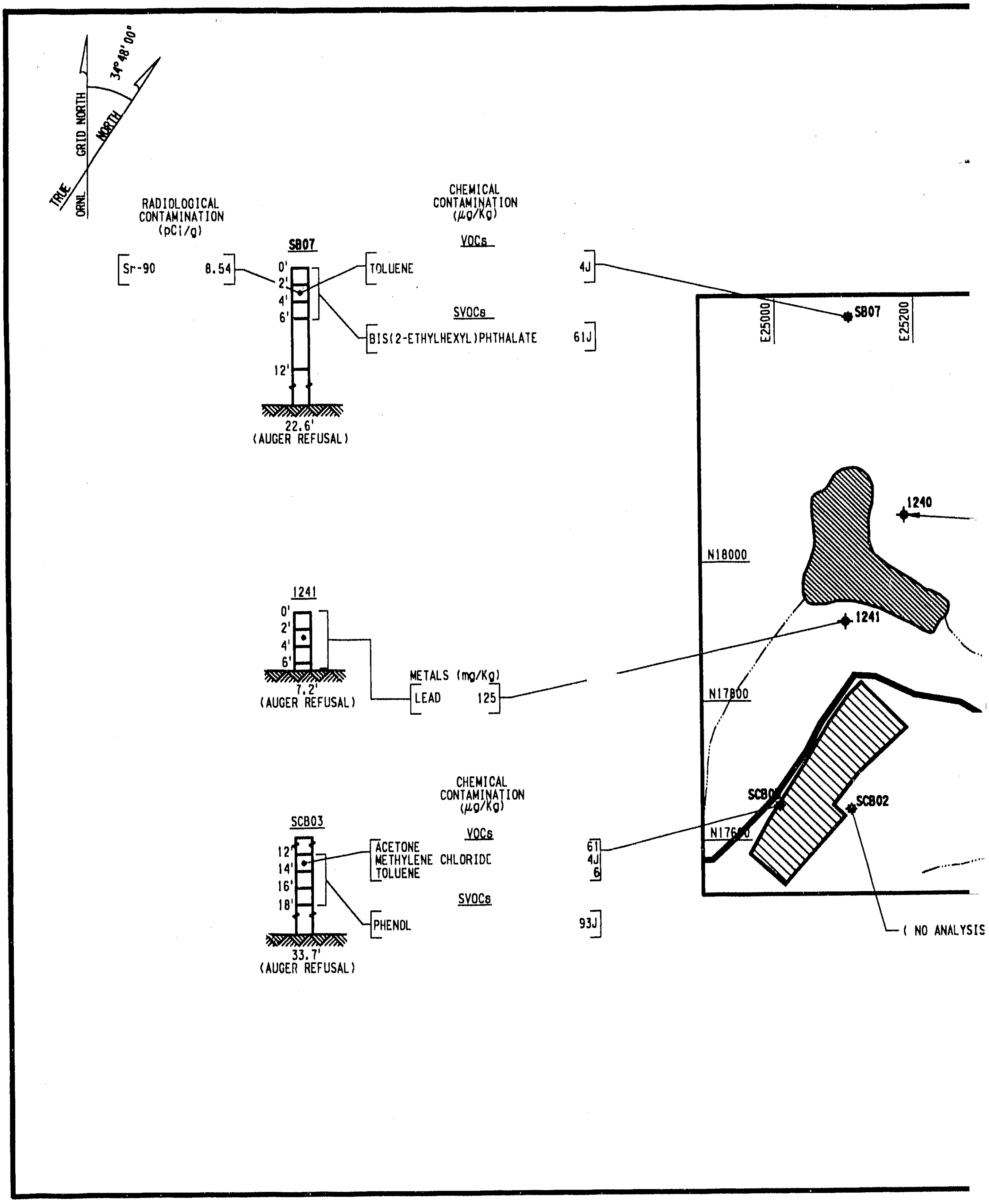


Fig. 4.44. Soil boring locations in WAG 6.



WAG6 06F241.DGN
9-1

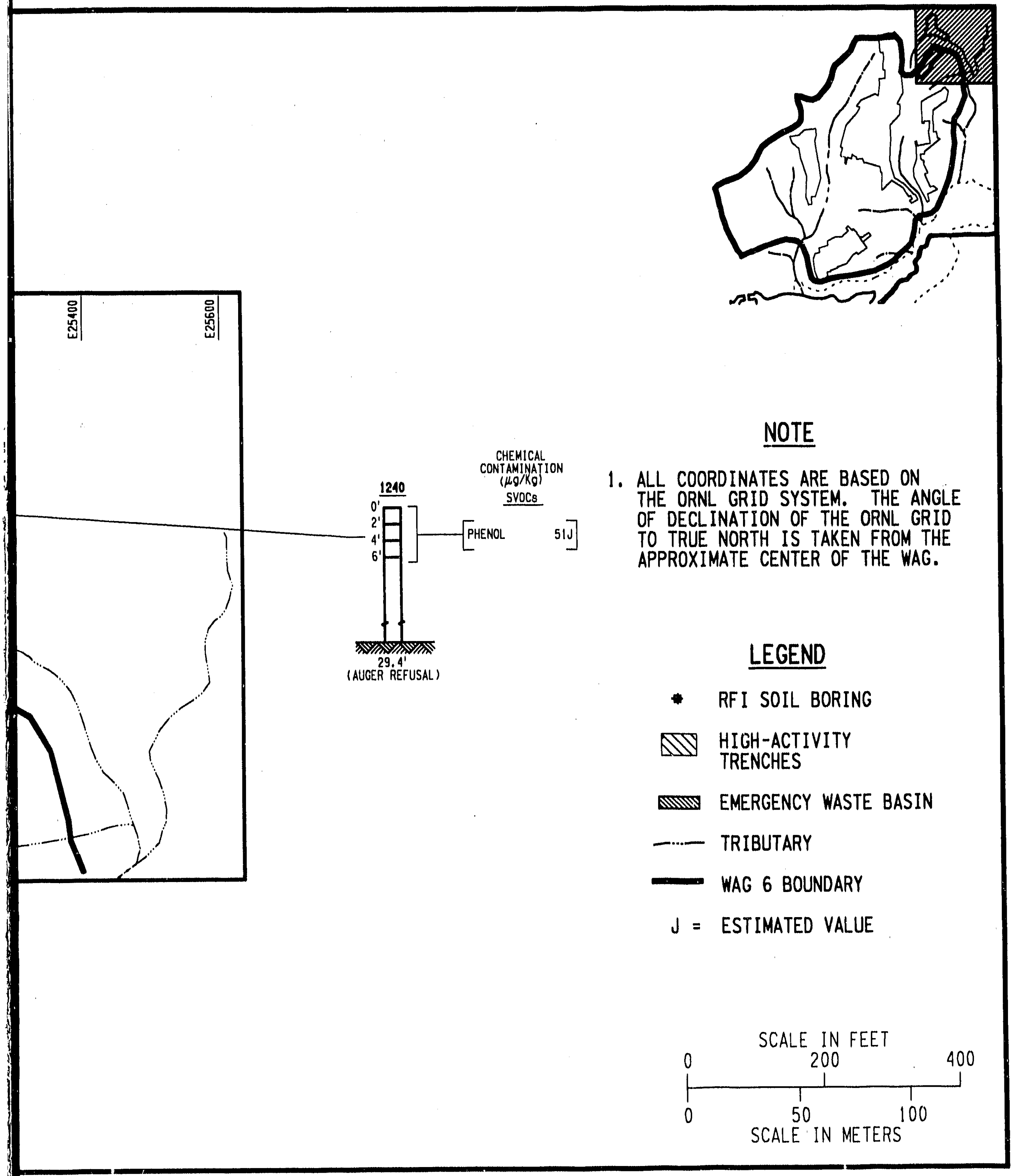
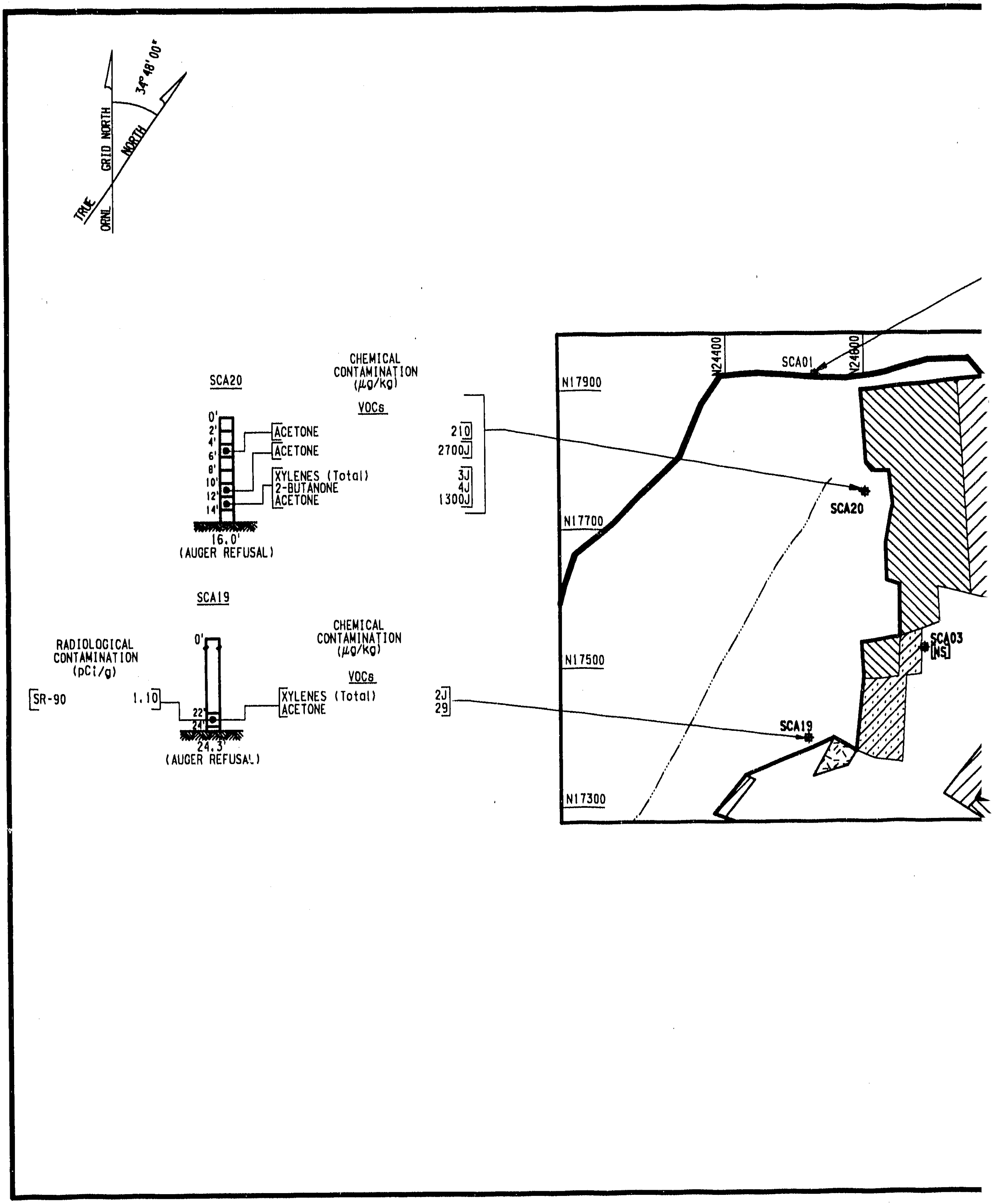


Fig. 4.45. Soil sample results from the 19 Trench area and the Emergency Waste Basin.



WAG6 06F240A.DGN
9-3

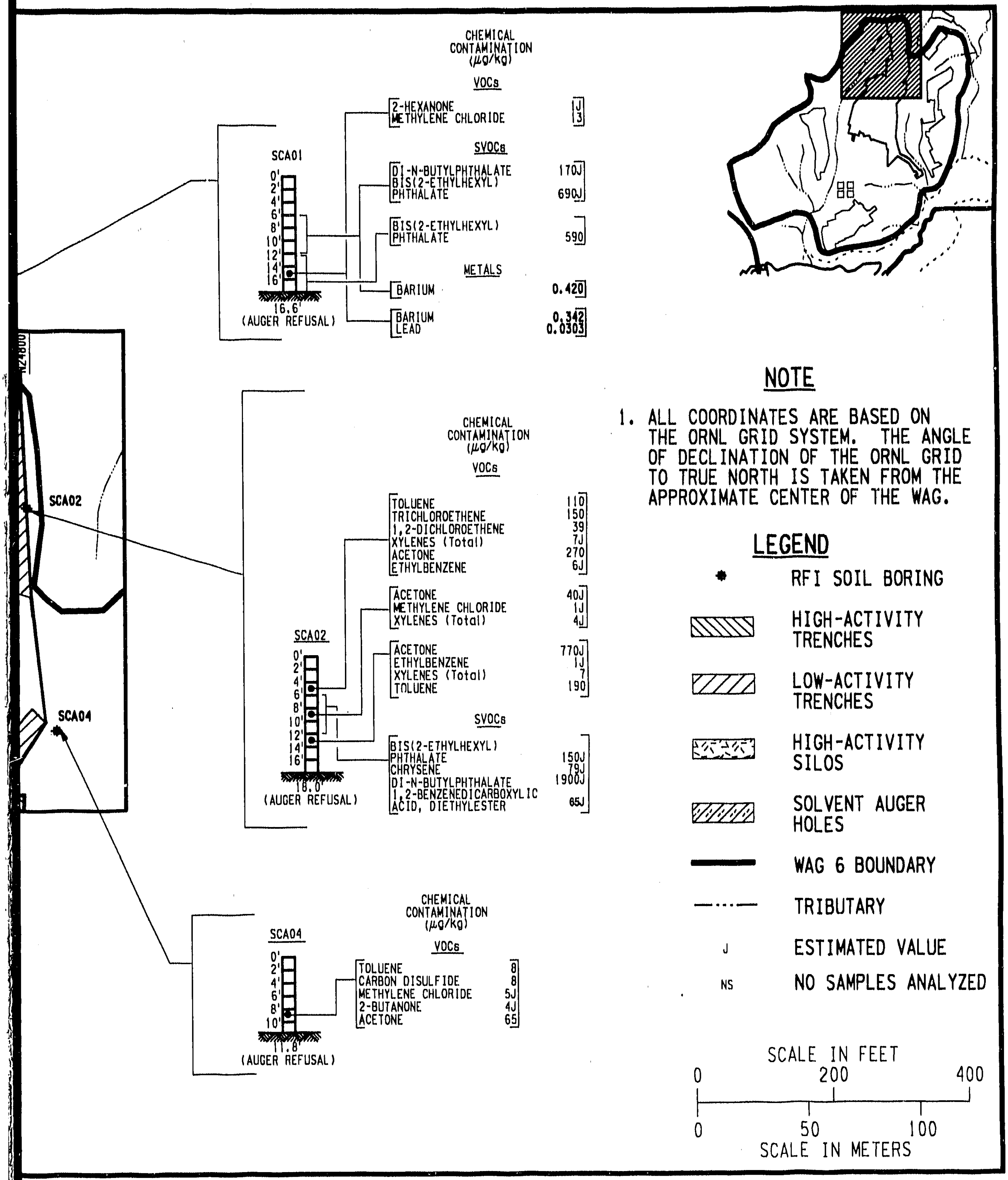
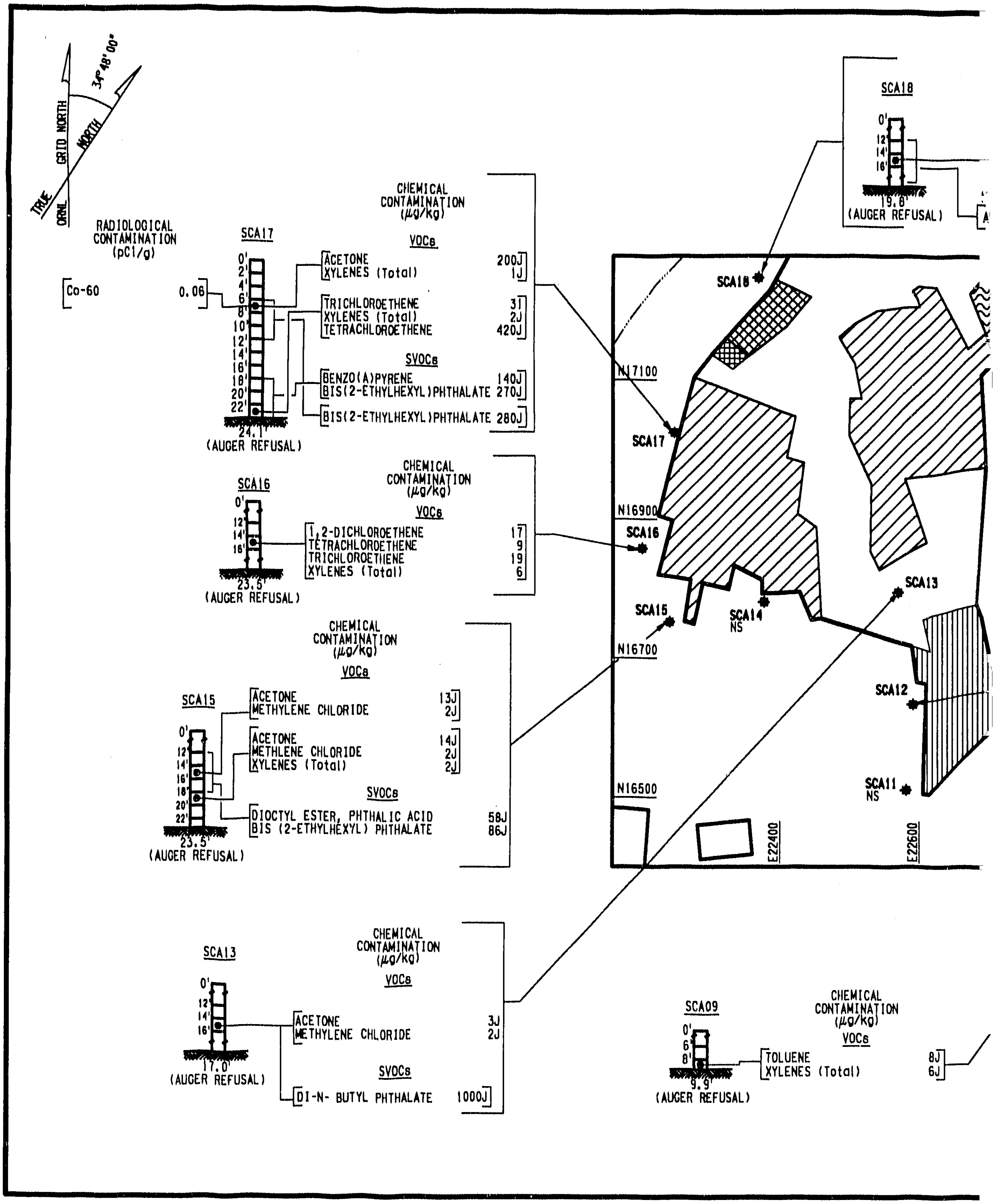
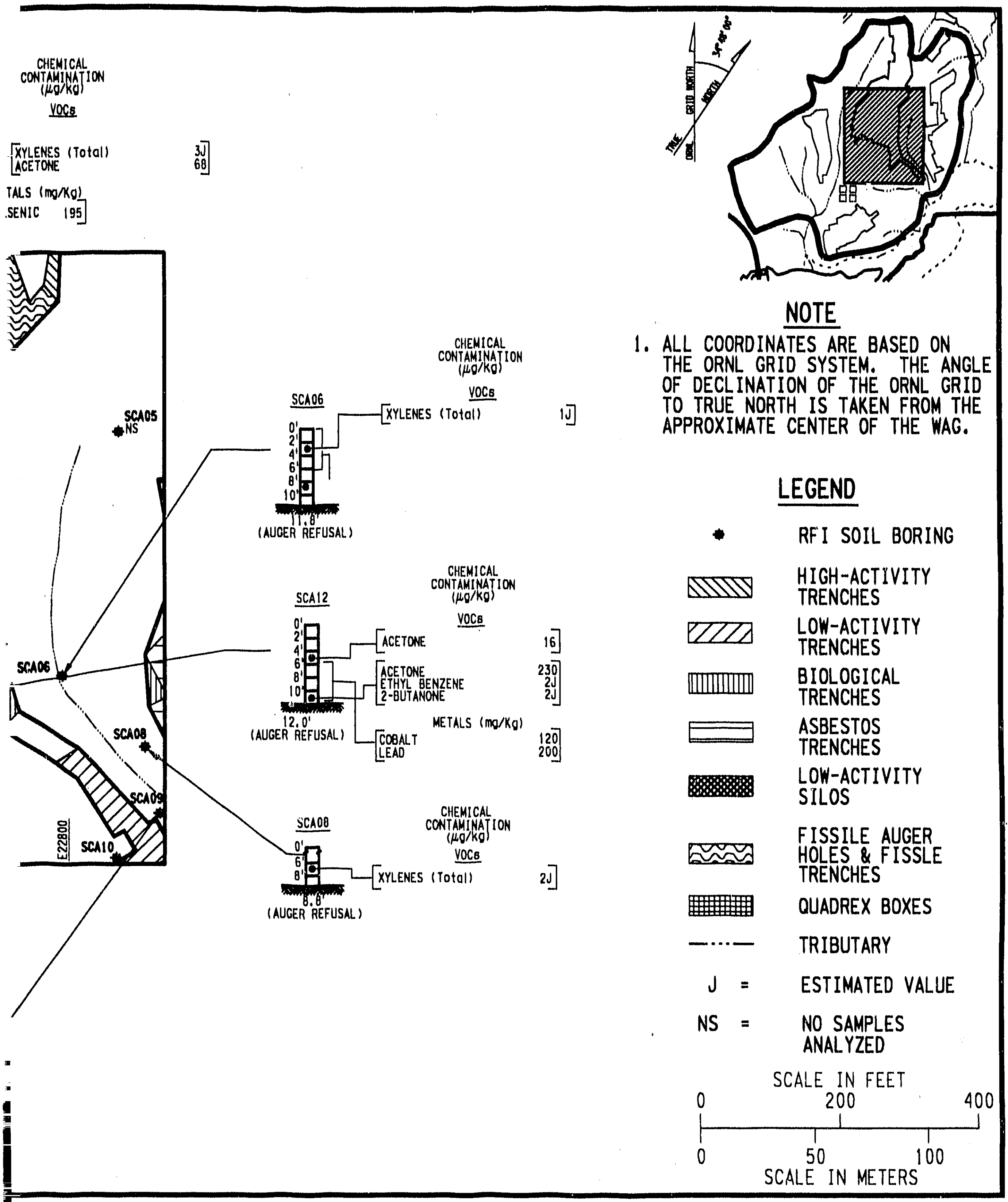


Fig. 4.46. Soil sample results from the northern region of the central waste disposal area of WAG 6.



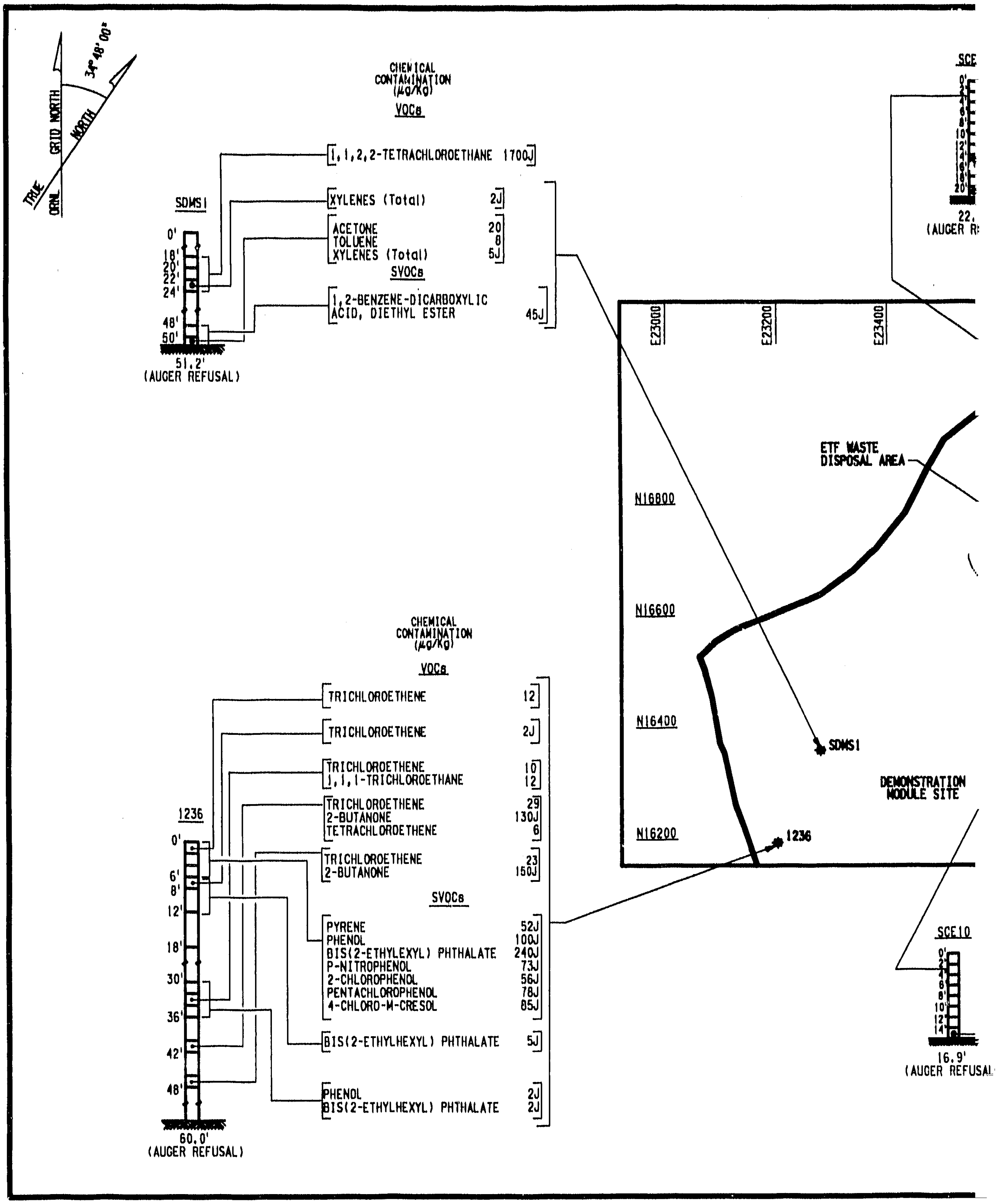
WAG6 06F240B.DGN
8-31



NOTE

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.

Fig. 4.47. Soil sample results from the southern region of the central waste disposal area.


TRUE
ORNL
GRID NORTH
NORTH
34° 48' 00"
CHEMICAL CONTAMINATION (µg/Kg)
VOCs
1,1,2,2-TETRACHLOROETHANE 1700J
XYLENES (Total) 2J
ACETONE 20
TOLUENE 8
XYLENES (Total) 5J
SVOCs
1,2-BENZENE-DICARBOXYLIC ACID, DIETHYL ESTER 45J
SDMS1
0'
18'
20'
22'
24'
48'
50'
51.2' (AUGER REFUSAL)
SCE
22.
(AUGER R
E23000
E23200
E23400
ETF WASTE DISPOSAL AREA
N16800
N16600
N16400
N16200
SDMS1
DEMONSTRATION MODULE SITE
1236
CHEMICAL CONTAMINATION (µg/Kg)
VOCs
TRICHLOROETHENE 12
TRICHLOROETHENE 2J
TRICHLOROETHENE 10
1,1,1-TRICHLOROETHANE 12
TRICHLOROETHENE 29
2-BUTANONE 130J
TETRACHLOROETHENE 6
TRICHLOROETHENE 23
2-BUTANONE 150J
SVOCs
PYRENE 52J
PHENOL 100J
BIS(2-ETHYLEXYL) PHTHALATE 240J
P-NITROPHENOL 73J
2-CHLOROPHENOL 56J
PENTACHLOROPHENOL 78J
4-CHLORO-M-CRESOL 85J
BIS(2-ETHYLHEXYL) PHTHALATE 5J
PHENOL 2J
BIS(2-ETHYLHEXYL) PHTHALATE 2J
1236
0'
6'
8'
12'
18'
30'
36'
42'
48'
60.0' (AUGER REFUSAL)
SCE10
16.9' (AUGER REFUSAL

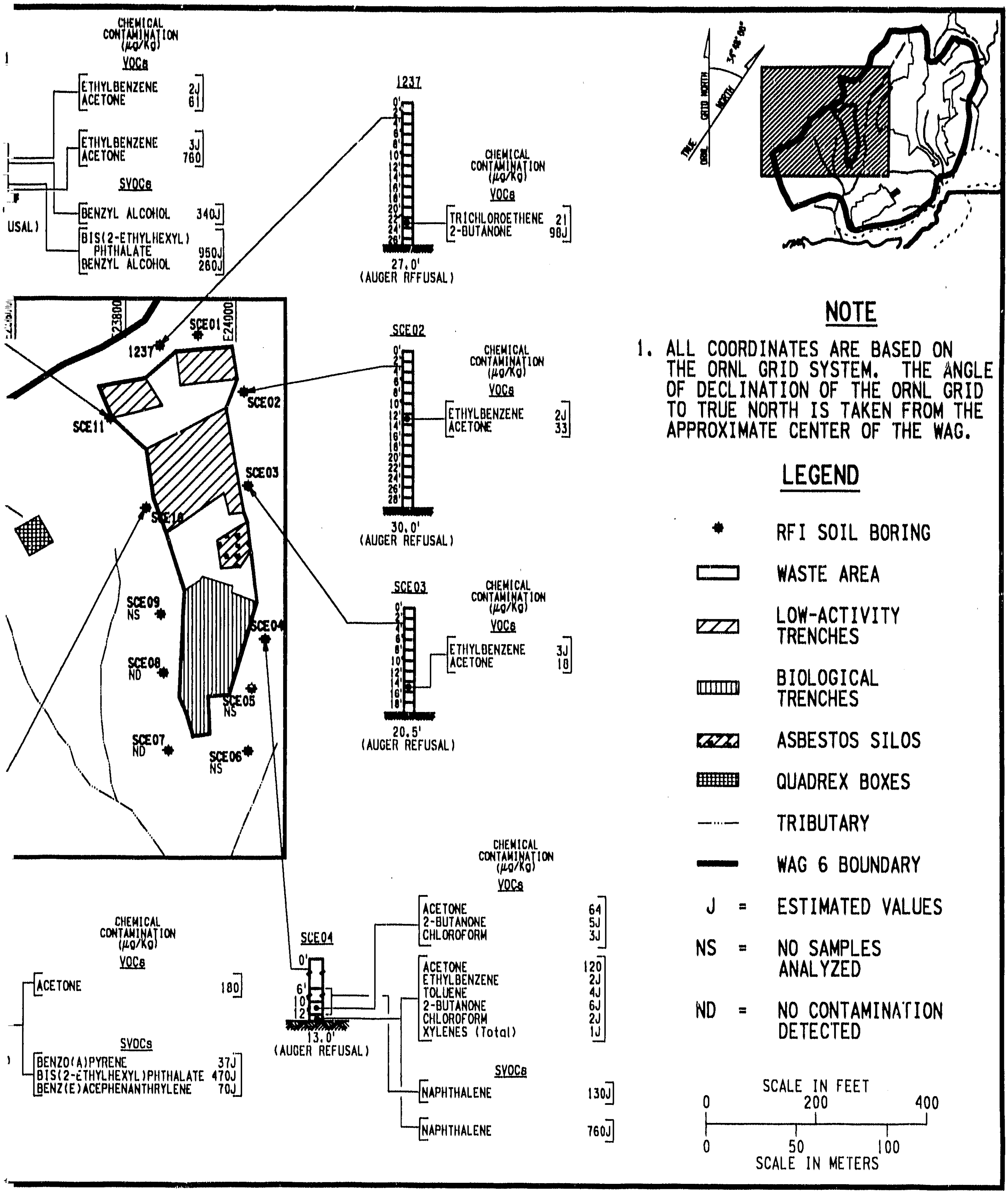
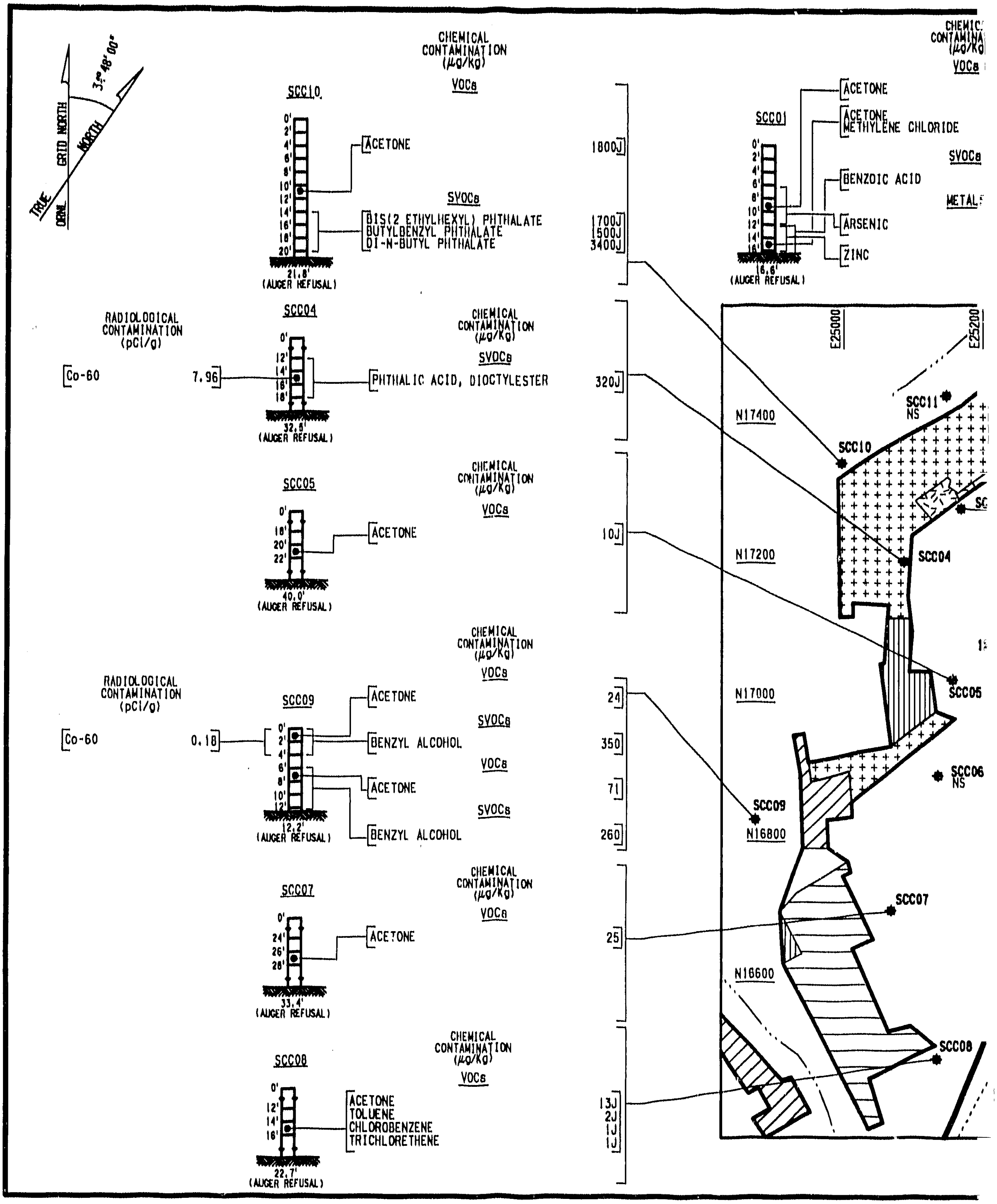


Fig. 4.48. Soil sample results from the western waste disposal area and the southwestern region of WAG 6.



WAG6 06F242.DGN

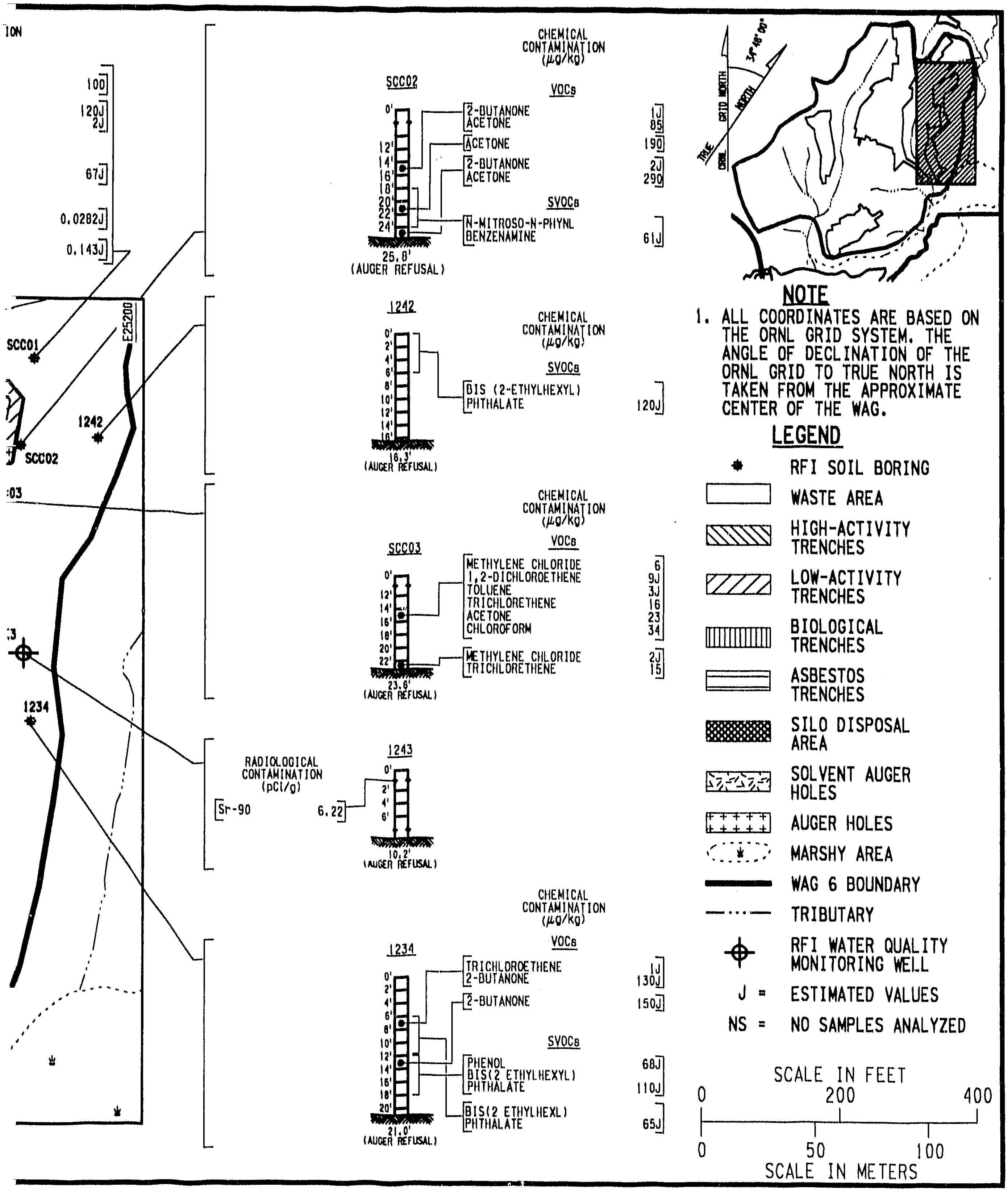
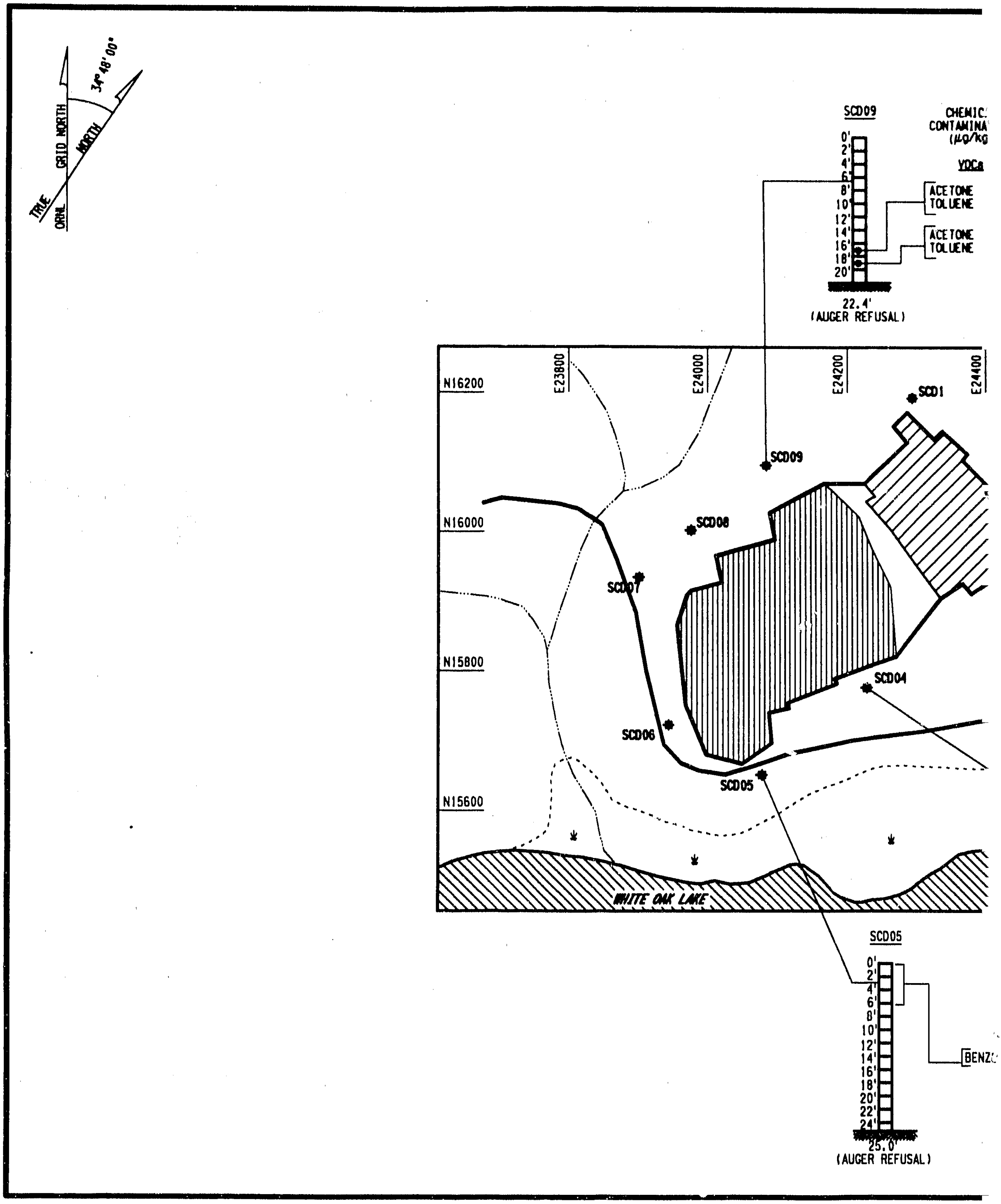


Fig. 4.49. Soil sample results from the eastern waste disposal area.



WAG6 06F238.DGN
9-1

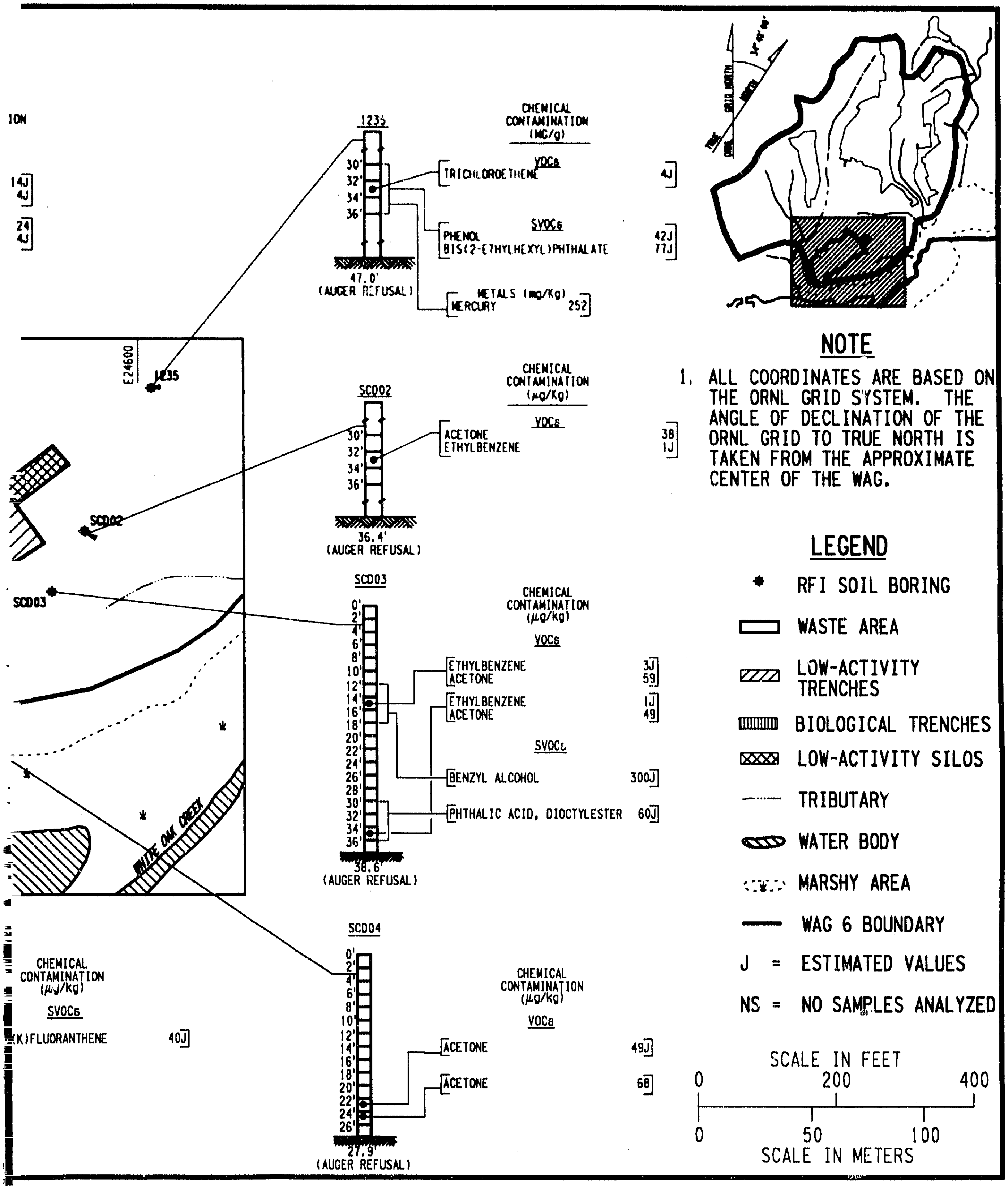
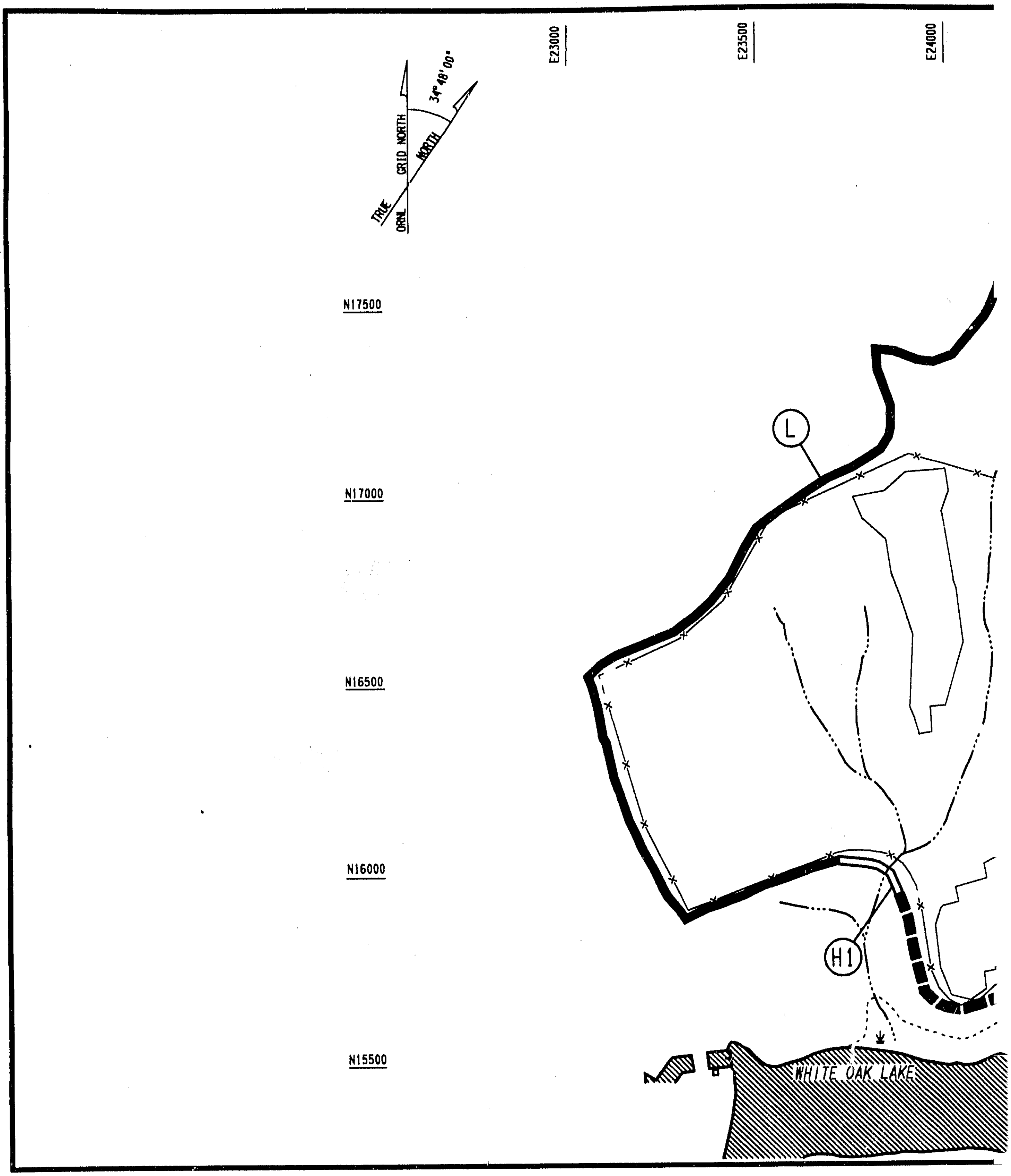


Fig. 4.50. Soil sample results from the southern waste disposal area, WAG 6.



WAG6 06F279.DGN
9-9

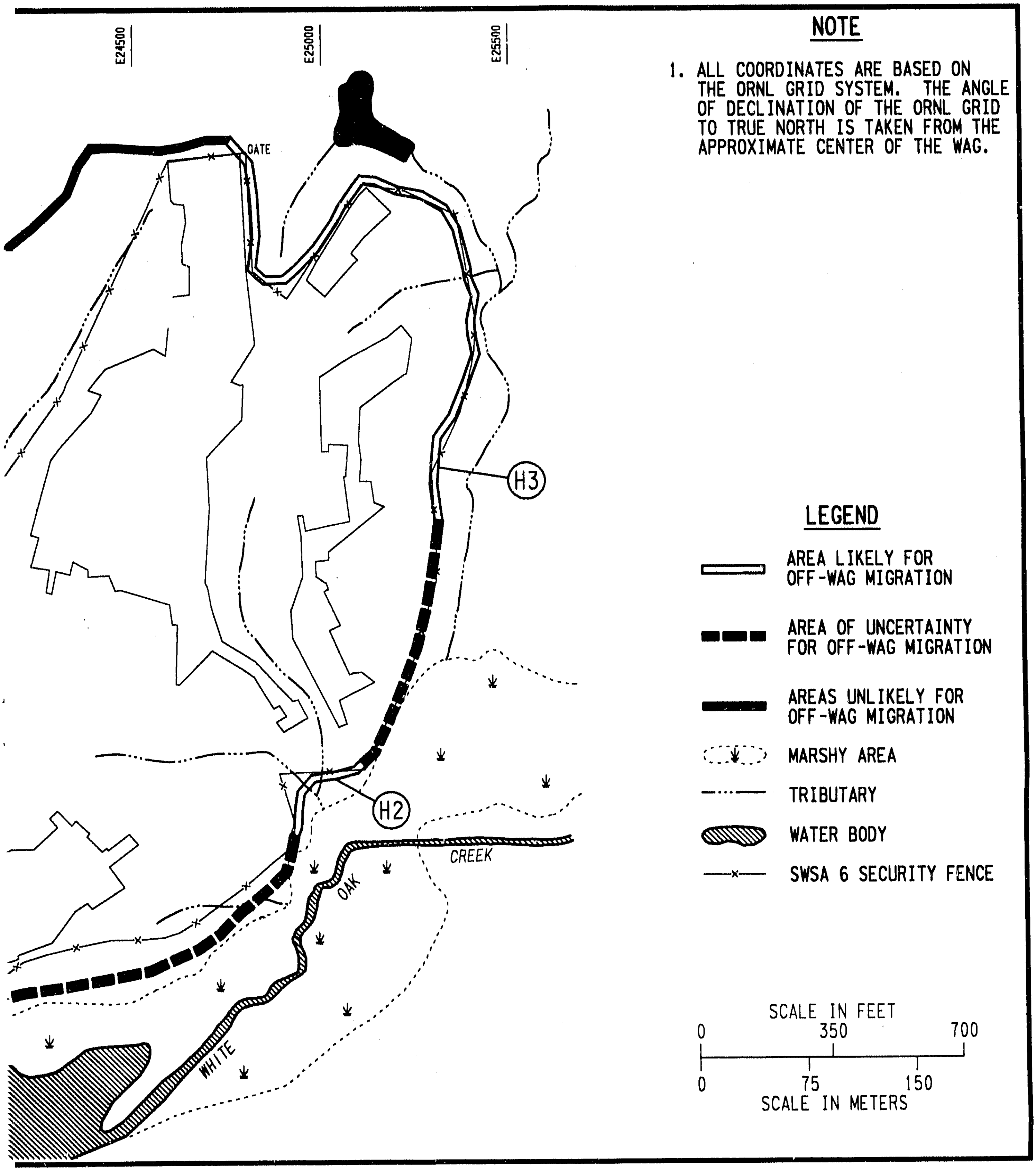
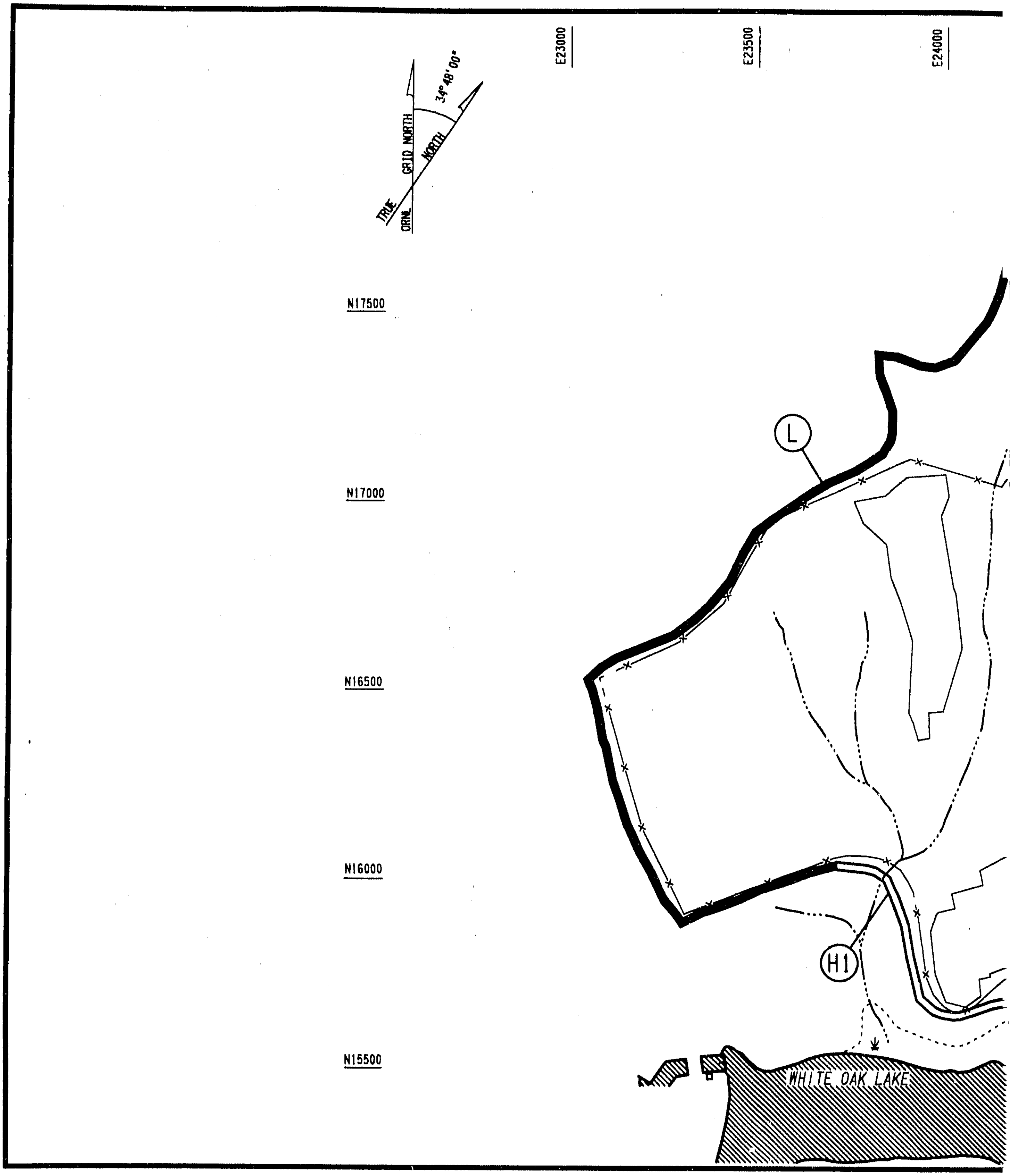
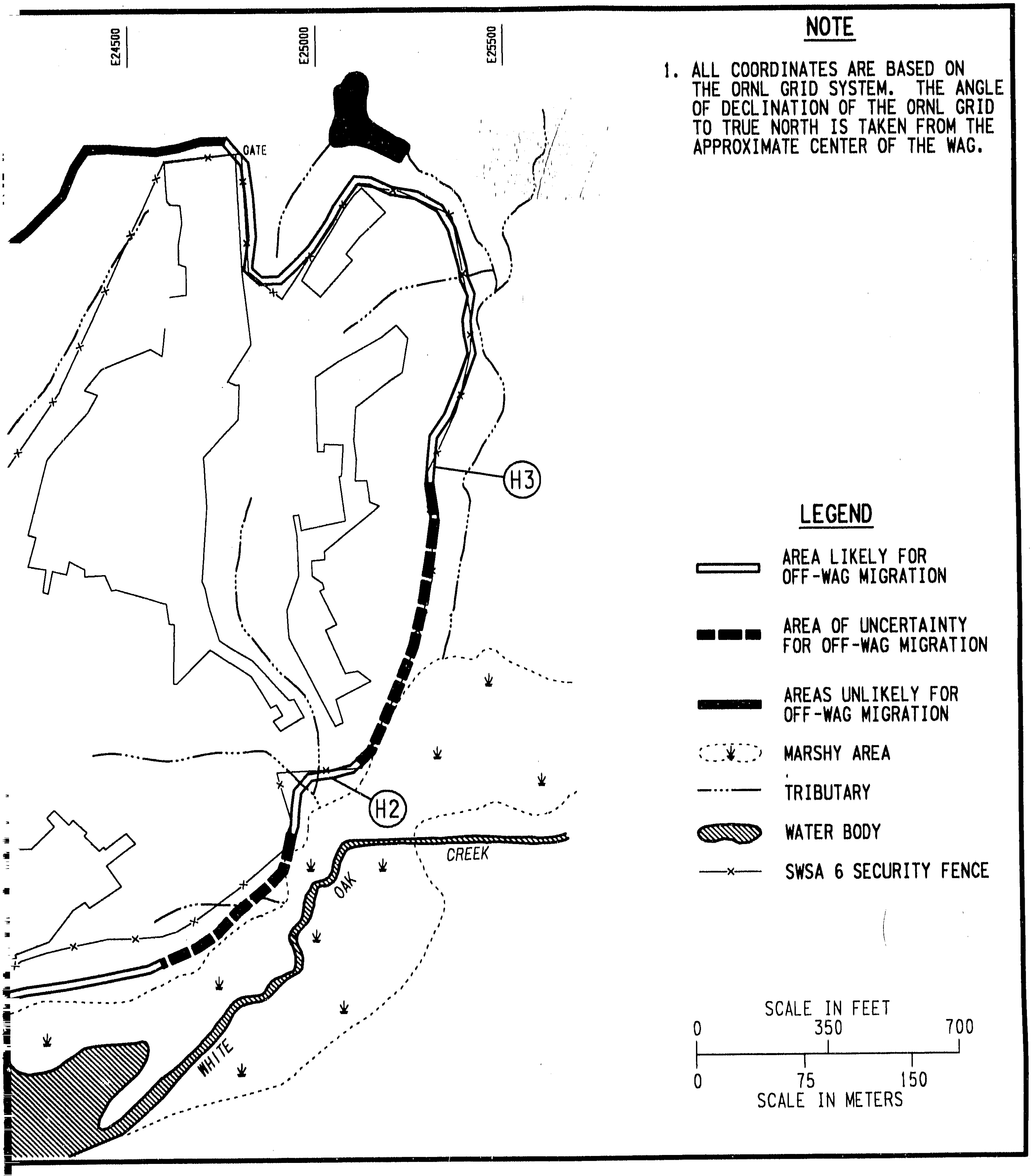


Fig. 4.51. Areas of WAG 6 boundary at which off-WAG migration of radionuclides in likely, uncertain, or unlikely.



WAG6 06F362.DGN
9-9



NOTE

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.

Fig. 4.52. Areas of WAG 6 boundary at which off-WAG migration of chemicals is likely, uncertain, or unlikely.

5. CONTAMINANT FATE AND TRANSPORT

5.1 INTRODUCTION

5.1.1 Scope of the Analysis

Contaminant fate and transport modeling was conducted to support the base line human health evaluation and to aid in definition of remedial alternatives. The base line human health evaluation (Sect. 6) considers hypothetical present day and future on-WAG and off-WAG receptors. For on-WAG receptors, predictions of contaminant concentrations in groundwater, surface water, sediment, soil, and air are required. For the off-WAG receptor, contaminant flux via surface water to WOL is assumed; this flux is used to calculate a corresponding concentration downstream in the Clinch River, adjacent to the location of the hypothetical off-WAG receptor.

To fulfill these data requirements, the fate and transport analysis described in this section quantified:

- Release rates of the contaminants from source areas into the environmental media
- Remaining quantities of the contaminants in source areas
- Persistence of contaminants in environmental media into which they are being released
- Transformation (decay and degradation) of contaminants in media, and their transfer between media
- Transport and migration of contaminants off of WAG 6

Modeling to predict future concentrations and fluxes was performed for radionuclides. Due to a lack of inventory data, prediction of future concentrations and fluxes for chemicals was not possible. However, fluxes of chemicals for current conditions were computed.

5.1.2 Methodology and Content

The methodology developed for the fate and transport analysis included computer modeling of contaminants in water and air. Because soil and sediment concentrations are a function of the transport of contaminants in water and air, no specific modeling methodology was developed for these media.

Computer modeling was chosen as the appropriate analytical tool for the analysis because it provided an exploratory as well as predictive capability. The exploratory capability of the modeling stemmed from the quantitative relationships developed that linked the observed contaminant concentrations to site and source characteristics. The predictive capability allowed for the estimation of future contaminant concentrations and fluxes that could not otherwise be derived from monitored concentrations alone.

The site conceptual model for WAG 6, discussed in Sect. 5.2, identifies the potential environmental pathways through which humans could be exposed to the radionuclides and chemicals. The model provided the basis for the analysis of fate and transport; also, exposure pathways identified in the model were used as the basis for performing the base line risk assessment presented in Sect. 6, and for identifying potential remedial measures.

Sections 5.3 and 5.4 address surface/subsurface water contaminant fate and transport modeling and air modeling, respectively. The conclusions of the fate and transport analysis appear in Sect. 5.5. The following appendixes supplement Sect. 5:

- Appendix 5A, Surface Water Modeling in WAG 6
- Appendix 5B, Analysis and Results of April-May 1990 Storm Sampling

5.2 SITE CONCEPTUAL MODEL

The site conceptual model formed the framework within which environmental pathways of potential concern for WAG 6 were identified and illustrated. Figure 5.1 is a conceptual model of WAG 6 that illustrates the environmental pathways by which humans can be exposed to contaminants released from the source areas where contaminants have been placed. The term source areas refers to the shallow land burial waste disposal units described in Sect. 1. In Fig. 5.2, these units are illustrated schematically within the general hydrologic setting. The following paragraphs discuss the basic elements of the conceptual model.

A contaminant release mechanism is defined as any process that results in migration of the contaminant from a source area into the immediate environment. Contaminants may be released via the mechanisms identified in Fig. 5.1 into the primary transport media: groundwater, surface water/sediments, soil, and air. As discussed in Sect. 4, the occurrence of contamination in all media except air was determined by observation. Air was included in the conceptual model since it can be a potential pathway, and air modeling was performed to characterize it.

Contaminants in source areas and in the environmental media are subject to transformation processes that degrade the contaminant concentrations. For the radionuclides, the process is radioactive decay; for organic chemicals transformation involves photochemical, biological, and chemical degradation processes.

Once in the environment, contaminants can be transferred among media (Fig. 5.3) and transported out of WAG 6. Humans are exposed to contaminant concentrations via the exposure routes of inhalation of air, ingestion, and direct contact. The environmental pathways involving each primary transport medium shown in Fig. 5.1 are discussed in the following subsections.

5.2.1 Groundwater Pathway

As discussed in Sect. 4, migration of contaminants was identified in WAG 6, extending from specific source areas toward on-WAG streams, the WAG 6 boundary, and WOL. The groundwater release mechanisms were identified as direct contact and percolation. In trenches saturated or partially saturated due to the rising water table, contaminants whose containers were breached or that have otherwise been exposed were in direct contact with groundwater. In disposal units located above the normal water table, leachate is generated when infiltrating moisture mixes with the waste material and then percolates to the groundwater.

Intermedia transfer of contaminants occurred between groundwater, soil, and surface water. Some contaminants in groundwater moved at velocities much lower than the characteristic flow velocity of groundwater because they were retarded by interaction with the solid matrix of the aquifer (sorption). The sorbed contaminants diffuse through the solid matrix, which may act as a source/sink for groundwater transport. When the groundwater intersected the streams, groundwater contaminants were transferred to surface waters by groundwater discharge. Seepage from the stream beds also transferred contaminants in surface water to the groundwater.

Contaminants can be transported in groundwater by advection and dispersion. Advection is the movement of contaminants at flow velocity. Dispersion occurs because of local velocity variations induced by the complex geometry of the actual flow path. The amount of dispersion depends on the mean flow velocity and is greater in the direction of flow than transverse to the flow. Therefore, a contaminant plume spreads in the direction of advection, resulting in longitudinally elongated plumes. As discussed in Sect. 3, the hydrogeology of WAG 6 is complex, and contaminant transport occurs primarily through secondary porosity features such as fractures.

Extensive refinement of the groundwater pathway conceptual model was not appropriate due to the considerable uncertainty associated with contaminant release rates and hydrogeologic parameters. As described in Sect. 4.2, because source areas comprise heterogeneous units with heterogeneous waste forms, packaging, and containers, contaminant release rates may vary unpredictably as containments randomly disintegrate. Hydrogeology exhibited characteristics of both porous media and fractured rock and contaminant transport was possibly dominated by matrix diffusion; these characteristics could not be readily characterized by field measurements on a local scale across the WAG and therefore were not modeled numerically.

5.2.2 Surface Water Pathway

Elevated levels of tritium, strontium-90, and chlorinated hydrocarbons were observed along the streams on WAG 6 and were being transported off-WAG, as discussed in Sect. 4.

The potential mechanisms identified for the release of these contaminants from the source areas were trench overflow and subsurface storm flow. Trench overflow occurs when a trench is saturated and spills leachate. Trench overflow mixes with overland flow and is carried into streams. Contaminant release into surface water may also occur via subsurface storm flow (Sect. 3.3). Leachate in bathtubbing trenches or in trenches within the groundwater table could mix with these subsurface storm flows, move laterally with the flows, and be discharged into the streams.

From surface waters, intermedia transfers occurred to air via volatilization, to groundwater via seepage from stream beds, and to soil via seepage. Stream widths on WAG 6 are relatively small, and streams carry water intermittently; therefore volatilization of contaminants is minimal. In the upper reaches of the streams, seepage into soil and groundwater occurred. Contaminants in stream water were also transferred to sediments in suspension and/or on the streambed.

Contaminants were also transferred to surface waters from air via precipitation/deposition; from soil via runoff, including overland flow, saturated overland flow, and subsurface storm flow; and from groundwater via groundwater discharge (base flow).

Transport of contaminants out of WAG 6 appeared to be relatively rapid, because streams exiting the site are quite short. Contaminant flux out of WAG 6 via stream flow was the major component of the total contaminant flux exiting WAG 6, since most of the flux from groundwater was discharged into site streams.

Further refinement of the surface water pathway was not appropriate, because contaminant release rates into surface waters could not be measured for the different release mechanisms. No models exist that address subsurface storm flow. As discussed in Sect. 3, subsurface storm flow exists in only parts of WAG 6. The stream flows in WAG 6 comprise overland flows, saturated overland flows, subsurface storm flows, and base flows. These individual flow components and contaminant concentrations associated with them were not estimated, because it is the contaminant flux out of WAG 6 that is of primary interest for surface water, and that can be monitored at stream outlets.

5.2.3 Soil Pathway

Because most of the source areas in WAG 6 contained disposal units located in the vadose zone, contaminants were released into soil by direct contact, diffusion, percolating water, and subsurface storm flows. Vertical and lateral movement of an advancing soil moisture front, especially from the unlined trenches, spread the dissolved contaminants in the vadose zone soil. As discussed in Sect. 4, contamination in the vadose zone was fairly limited and did not appear to represent significant secondary source areas. Contamination of surface soils appeared minimal.

From soil, contaminants were transferred to air via volatilization and particulate suspension; to groundwater via percolation; and to surface water via runoff and groundwater discharge. Also, contaminants were transferred to soil via precipitation/deposition from air; via sorption from groundwater; and via seepage from surface water.

No specific transport mechanism was considered for soil, since soil acts as a storage medium in which transport occurs mostly via water.

5.2.4 Air Pathway

Air was determined to be a minor pathway. Volatilization and particulate emissions (due to upwind erosion of source area soils) are the two mechanisms for contaminant releases to air. Other than the soil gas survey conducted during RFI activities at the perimeter of selected trench areas (to assess the presence and migration of VOCs), no data base was developed to quantify these release rates. Volatilization rates are contaminant-specific and vary according to waste characteristics, the characteristics of the waste units, natural and engineered barriers, and climatic and meteorological conditions. Biological trenches were identified as the areas where gas generation would most likely occur due to biological, chemical, and physical decomposition of the wastes.

Particulate emissions due to source area soil erosion would likely be minimal because of high annual rainfall, low wind speeds, extensive grass cover, and the rapid regrowth of vegetation in temporarily disturbed areas.

Contaminants released into the air could be transferred to both surface waters and soil surface by deposition (dry settling) and precipitation, thereby creating diffuse secondary sources that lead to further contamination of other media. For example, from the soil surface, infiltrating rainwater could carry the contaminants into the vadose zone, and from there into the groundwater. Air-soil and air-water transfers are functions of solubility, adsorption, particle size, and precipitation. Transport of contaminants in air occur by advection and dispersion in the predominant wind directions of southwest and northeast.

5.3 SURFACE/SUBSURFACE WATER CONTAMINANT FATE AND TRANSPORT

The surface water/subsurface water fate and transport model was developed and used to perform fate and transport analysis for radionuclides. For modeling purposes, the waste areas in WAG 6 (i.e., trenches/auger holes) were divided into 26 source areas. These areas (A to Z) are shown in Fig. 5.4. Modeling was also performed for one aboveground unit, Tumulus I.

The radionuclides modeled were tritium, cobalt-60, strontium-90, cesium-137, europium-152, europium-154, uranium-233, uranium-238, and thorium-232. The model was calibrated when possible, for each source area and for each radionuclide, using the concentration data from surface water and groundwater sampling and the radionuclide inventory for the source area. For each area, the model output consisted of 160 years of remaining radionuclide source inventory, surface water flux, and groundwater flux and concentration at calibration wells and points downgradient of wells within the WAG 6 boundary.

Radionuclide flux in streams was computed by combining the flux of surface runoff of the tributary areas (the output of the transport model) with the flux of groundwater discharging into streams. The radionuclide concentration in streams was then computed by dividing the stream radionuclide flux by the stream flow rate. The radionuclide plumes from source areas were superimposed to compute groundwater radionuclide flux and concentrations at the WAG 6 boundary.

The transport model is described in Sect. 5.3.1; modeling input data and parameters, calibration of the model, modeling results, and analysis of these results are presented in Sect. 5.3.2. Stream flow radionuclide concentrations and boundary fluxes are addressed in Sect. 5.3.3. and boundary fluxes in groundwater in Sect. 5.3.4. Conclusions are stated in Sect. 5.3.5. The sensitivity and uncertainty analysis are presented in Sect. 5.3.6.

5.3.1 Modeling Approach, Theoretical Basis, and the Computer Model

An integrated surface/subsurface water contaminant fate and transport model was developed for WAG 6. The model's level of complexity is appropriate for generating future estimates of annual average contaminant concentrations in surface waters and groundwater for a base line risk assessment, especially in an area where detailed definitions of contaminant release rates and the physical characteristics of the environmental media are not possible on a local scale. To directly incorporate the uncertainty into the analysis, a Monte Carlo simulation approach was chosen. The model included empirical and analytical algorithms for determining contaminant releases from a source area (i.e., a group of trenches and auger holes) into environmental media and contaminant transport to downgradient receptor points. Using field-monitored stream and well concentration data, the model's built-in calibration feature determined site-specific and contaminant-specific values of the model parameters.

5.3.1.1 General modeling assumptions

Modeling assumptions were as follows:

- The source area consisted of a group of waste trenches or auger holes located in the unsaturated soil zone.
- Storm runoff volumes were available for a typical year. A steady-state approach was taken so that the same storms are assumed to occur for each year of the simulation period.
- Release rates were empirically determined.

- Steady-state groundwater flow conditions were assumed, with known recharge, discharge, flow velocity, and water table elevation.
- Since site streams drain small areas, less than 40 acres, no in-stream transport processes were simulated. Instead, an empirical approach was developed which linked source term to observed contaminants in streams during storms.
- A one-dimensional analytical model was used to simulate contaminant fate and transport in the vadose zone and in the saturated zone.
- Field-monitored storm concentrations and well concentrations were available.

5.3.1.2 Monte Carlo simulation

Fate and transport modeling was carried out within a Monte Carlo simulation framework; the uncertainty of the model parameters was directly incorporated into the modeling process. The Monte Carlo process is depicted in Fig. 5.5. In each model application, 100 fate and transport simulation trials were performed. For each of the simulation trials, the values of the model input parameters were randomly generated from the input probability distributions of these parameters. The output of each of the 100 trials constituted an equally likely time series of future concentrations or fluxes. A probability distribution of the output values was then generated for each year of the simulation. For the purpose of exposure assessment, the annual mean value and its 95th percentile upper limit were computed.

5.3.1.3 Fate and transport model

An empirical/analytical approach was formulated to perform fate and transport analysis. The model had three components: a source component, a surface runoff component, and a groundwater component. A schematic diagram illustrating key geometric parameters is shown in Fig. 5.6.

Source inventory and depletion. The contaminants were assumed to be located in the unsaturated soil above the water table in lined or unlined trenches and auger holes. The contaminant inventory and its date of placement were assumed to be known. Within the source volume, which is defined by the dimensions of the trenches, wastes of similar form and packaging were assumed to be placed in similar containers distributed uniformly.

Initially, only a fraction of the waste was assumed to be in contact with soil moisture within the source volume. The time before onset of accelerated failure of the containers to inhibit soil moisture contact was specified. Once the containers start disintegrating at an accelerated rate, the fraction of waste in contact with water was assumed to increase linearly to a specified maximum.

Assuming that the contaminants were distributed in equilibrium between the solid and liquid phases, the contaminant concentration in the soil water at any time could then be expressed as:

$$C_w = f_l \times M / [V_t \times (K_d \times \rho_b + \alpha \times \Theta)], \qquad (1)$$

where

C_w = contaminant concentration in the soil water (g/m^3 or Ci/m^3),
f_l = leaching fraction ($0 < f_l < 1$),
M = inventory of the contaminant in any year (g or Ci),
V_t = volume of the source (m^3),
K_d = distribution coefficient in the source (m^3/kg),
ρ_b = dry bulk density of source matrix (kg/m^3),
α = degree of saturation,
Θ = porosity of source matrix.

The water that infiltrated into the source volume was the major medium for the release of the contaminant leachate from the source area. The pathways of release included percolation into the groundwater and surface runoff. Contaminants in the source volume would be depleted by releases into surface runoff and groundwater, as well as by decay due to physical, chemical, and biological processes.

The yearly source inventory can therefore be expressed as:

$$M_{t+1} = M_t \times \exp(-\lambda) - R_{sw} - R_{gw}, \qquad (2)$$

where

M_{t+1} = inventory in year t + 1,
M_t = inventory in year t,
λ = composite decay rate (1/yr),
R_{sw} = amount released to surface runoff (g/yr or Ci/yr),
R_{gw} = amount released to groundwater (g/yr or Ci/yr).

Surface water pathway. The amount of contaminant transferred to the stream by surface runoff during a storm is assumed to be a fraction (f_2) of the total leachate in the source volume, expressed as:

$$S = f_2 \times V_t \times \alpha \times \Theta \times C_w, \qquad (3)$$

where

S = Surface runoff contaminant flux (g or Ci),
f_2 = surface runoff leaching fraction, (the ratio of fraction of surface flux per storm to the exposed trench inventory),

all other terms were as previously defined.

Substituting the value of C_w from equation (1) into equation (3) gives:

$$S = f_1 \times f_2 \times M/[1 + (K_d \times \rho_d)/(\alpha \times \Theta)] \tag{4}$$

The annual amount of contaminant loading in surface runoff would be the sum of the amounts generated by all the storms that occurred within the year. Therefore, annual average contaminant concentration in surface runoff can be computed as:

$$\bar{C}_{sf} = \sum_{1}^{n} S \;/\; \sum_{1}^{n} V_r, \tag{5}$$

where

C_s = annual average contaminant concentration in surface runoff (g/m^3 or Ci/m^3),
V_r = storm runoff volume (m^3),
n = number of storms per year.

The annual average contaminant concentration of stream flow in a stream that receives base flow (groundwater discharge) is expressed as:

$$\bar{C}_{sf} = \left(\sum_{1}^{n} S + Q_b \times C_g\right) / \left(\sum_{1}^{n} V_r + Q_b\right), \tag{6}$$

where

C_{sf} = annual average contaminant concentration in stream flow (g/m^3 or Ci/m^3),
Q_b = base flow (m^3/yr),
C_g = contaminant concentration in base flow (g/m^3 or Ci/m^3).

Equation (4) suggests that an empirical relationship between f_2 and contaminant flux in storm runoff can be developed, assuming that all other terms of the equation are known. Developing this relationship would involve monitoring and sampling a few storms and determining the storm runoff volume and the corresponding contaminant flux. Continuous sampling would provide the storm hydrograph and the contaminant pollutograph from which contaminant flux and storm volume could be computed by integration.

If storm sampling were done in a stream that also carried base flow, the flux due to base flow must be separated from the total storm flux before the f_2 and surface runoff flux relationship is established. In this case, concurrent sampling of groundwater contaminant concentrations was necessary for computation of the flux from the base flow. The volume of the base flow was also isolated from the storm hydrograph by a storm hydrograph separation technique.

Groundwater pathway. Leachate from the bottom of a source area traveled with the percolating water vertically through the vadose zone into groundwater and moved with groundwater horizontally toward a well or discharge point, subject to retardation. Transport was assumed to be one-dimensional in both zones.

The vertical water velocity and the retardation factor were computed as follows:

$$V_v = I/(\alpha \times \Theta)$$
$$R_v = 1 + (\rho_b \times K_{dv}) / (\alpha \times \Theta), \qquad (7)$$

where

V_v = vertical velocity (m/yr),
R_v = vertical retardation factor,
K_{dv} = K_d in vadose zone (m^3/kg).

The horizontal velocity was input to the model, and the horizontal retardation factor was computed as:

$$R_h = 1 + \rho_b * K_{dh}/(\Theta) \qquad (8)$$

where

K_{dn} = K_d in saturated zone (m^3/kg)

Furthermore, the second term of the right side of equations (7) and (8) was replaced by f_3 as follows:

$$R_v = 1 + f_3$$
$$R_h = 1 + f_3$$

(f_3 was a lumped calibration parameter that was assumed to be the same for both vertical and horizontal transport).

The vertical and horizontal travel times, t_v and t_h, were then computed as follows:

$$t_v = D_v \times R_v/V_v, \qquad (9)$$

where

D_v = vertical distance from the bottom of source to water table (m) and other terms were as previously defined;

$$t_h = D_h \times R_h/V_h, \tag{10}$$

where

D_h = horizontal distance to well (m) and other terms were as previously defined.

The sum of the horizontal and vertical travel times equaled the average breakthrough time for a contaminant from the source area to appear in an observation well.

The inflow of contaminants to the vadose zone at the bottom of the source was calculated as follows:

$$Q_o = I \times C_w \times A, \tag{11}$$

where

Q_o = the rate of contaminant recharge to the vadose zone (Ci/yr),
I = the water percolation rate in the vadose zone (m/yr),
C_w = concentration of the contaminant in the percolating water at the bottom of the source (Ci/m^3),
A = effective cross-sectional area of the source (m^2).

The transport of radionuclides through the vadose and saturated zones was evaluated using Hung's groundwater transport model (Hung 1980). The assumptions of the model were as follows:

- Transport was one-dimensional; flow was vertical in the vadose zone and downgradient in the saturated zone.
- Longitudinal dispersion was accounted for by use of a dispersion parameter Z.
- No transverse dispersion occurred in the vadose zone. The transverse dispersion in the saturated zone was described by an angle of dispersion.
- Vertical dispersion in the saturated zone was accounted for by the effective saturated zone thickness at the receptor. The thickness represented the depth of the plume at the exposure point.
- The contaminant concentration was assumed to be uniform across the cross-sectional area of the plume at the exposure point (described by the depth of the plume and the transverse width).

The basic equations of the model were as follows:

$$Q = Q_o \text{ x } Z \text{ x } \exp(-\lambda \text{ x } t) \tag{12}$$

and

$$Z = \exp[P/2-(P/2) \text{ x } \sqrt{1+4 \text{ x } R \text{ x } L \text{ x } \lambda/(P \text{ x } V)}]/\exp(-R \text{ x } L \text{ x } \lambda/V), \tag{13}$$

where

Z = a correction factor to compensate for the dispersion effect,
R = retardation factor R_v or R_h,
P = Peclet number, $V_V D_V/d$ or $V_h D_h/d$,
L = length, D_V or D_H (m),
V = water flow velocity, V_v or V_h (m/yr),
t = transit time, t_v or t_h,
d = dispersion coefficient (m^2/yr),
Q = rate of contaminant transport at distance L form the source (Ci/yr),
Q_0 = rate of contaminant transport at the source (Ci/yr), evaluated at time = (T-R x L/V),
T = time of simulation (yr),
λ = decay constant (yr^{-1}).

and other terms were as previously defined.

To calculate the radionuclide concentration at the receptor, the rate of groundwater flow in the plume of contamination at the receptor point was calculated as follows:

$$W_a = V_h \text{ x } P_a \text{ x } D_a [\sqrt{A} + 2 \text{ x } \tan(a/2) \text{ x } D_h], \tag{14}$$

where

W_a = the rate of contaminated water available for removal at the receptor point (m^3/yr),
V_h = groundwater velocity (m/yr),
P_a = porosity of the shallow saturated zone,
D_a = effective thickness of the flow domain in the shallow saturated zone (m),
a = angle of spread of the contaminant plume in the shallow saturated zone (radians),
A = projected cross-sectional area of trench (m^2),
D_h = distance to the receptor (m).

The contaminant concentration in the groundwater at the exposure point was calculated by:

$$C_g = Q/W_a, \qquad (15)$$

where C_g is expressed in kg/m^3 or Ci/m^3.

5.3.1.4 Model calibration

The model calibration consisted of an iterative procedure in which the initial values of certain key parameters were adjusted to match the time of travel and the observed contaminant concentration in groundwater. These parameters were the surface water and groundwater release fractions, f_2 and f_1, respectively, and f_3, which was calibrated first. The value of f_1 was determined before that of f_2 because the formulation of f_2 included the f_1 term. An iterative procedure was necessary because the surface water and groundwater transport equations were coupled.

The calibration of f_3 was based on comparison of the simulated breakthrough time for both vadose and saturated zones with the observed time from waste emplacement well sampling. The initial value of f_3 was calculated using assumed or known values of K_d, Θ, ρ, and α. If the simulated breakthrough time exceeded the time when the sample was taken, the f_3 values were decreased so that the breakthrough time matched the observed time. If the simulated breakthrough time was less than the observed time of the sample, no adjustments were made to the f_3 values, assuming that the plume already extended beyond the downgradient well. The next adjustment was initiated by comparing the observed and simulated well concentration. The f_1 fraction was simply scaled so that the simulated well concentration matched the observed concentration. An assumed coefficient of variation was introduced to derive the distribution of the calibrated f_1 and f_3.

Values for f_2 were recomputed for each change in the value of f_1 since the formulation of f_2 includes f_1 as an independent variable.

5.3.1.5 Description of the computer program

The model was written in FORTRAN 77 language and was debugged and compiled using Lahey Fortran compiler and editor. A Compaq 386/25e microcomputer was used in developing and running the model. The model is composed of several subroutines structured around a main program. The flowchart shown in Fig. 5.7 presents the general structure of the model.

Description of subroutines. The following paragraphs provide brief descriptions of the subroutines used in the model.

Main program. The data input and output and the statistical analysis of the output data are performed in the main program. Calls to subroutines RNGEN, PARACORR, and SURTRAN are made from the main program.

RNGEN. Values of the specified number of Monte Carlo simulation parameters are generated in RNGEN. For the input distribution parameters (i.e., mean and standard deviation) and the specified probability distribution, RNGEN generates a random variable. The probability distributions presently supported are normal, log normal, and uniform distributions. RNGEN calls for the random number generator RANDOM before the selected probability distribution subroutine is called.

SURTRAN. This is the main computational subroutine where surface water and groundwater fate and transport analysis is carried out. A flowchart of SURTRAN is shown in Fig. 5.8. Routines are provided for calibration and no calibration options. The subroutines CALF2 and HUNGVH are called form SURTRAN.

DISTRIBUTE. The distribution of the output values, the mean, and the 95th percentile upper limit of the mean are determined in this subroutine.

AVEVAR. This subroutine computes the mean and standard deviation of the output values.

CALF2. This subroutine determines the f_2 parameters for a number of storms in a year. Input to the subroutine includes storm volumes and average contaminant concentrations for the storms.

FIT. This subroutine fits a function to the calibration storm values and the corresponding average contaminant concentrations.

HUNGVH. The transport in the vadose and saturated zones is performed in this subroutine. Travel times and contaminant concentration at the receptor point in the groundwater are computed.

HUNG FUNCTION. The dispersion correction factor Z is computed by this function, which is called by the HUNGVH subroutine.

RANDOM. This subroutine generates a random number between 0 and 1. A seed number (i.e., 19118) is entered to initiate the random number generator.

UNIFORM. This subroutine generates a uniformly distributed random number for a given set of minimum and maximum values.

NORMAL. This subroutine generates a normally distributed random number using the input mean and standard deviation.

LOGNORMAL. This subroutine generates a log normally distributed variate.

5.3.1.6 Model input parameters and output

The model input parameters are listed in Table 5.1 in the order of their appearance in the computer input file. A list of the Monte Carlo parameters is given in Table 5.2. The model output consisted of an echo of the input parameters and the time series of contaminant inventory and surface runoff and groundwater fluxes and/or concentrations. The time series values are given in terms of annual mean and 95th percentile of the mean.

5.3.2 Model Application

5.3.2.1 Input data and parameters

Storm runoff. The model input data included the storm runoff volumes for 1984 (a representative year, as demonstrated in Appendix 5A) and the storm runoff volumes and associated average radionuclide concentrations for the storms sampled in WAG 6 during April–May 1990. The 1984 storm runoff data, which were generated using a water balance model, are given in Appendix 5A. The storm runoff and radionuclide concentration data from 1990 sampling are given in Appendix 5B.

Waste inventory and groundwater data. The inventory of radionuclides for each of the 26 modeling areas is shown in Table 5.3. Time-averaged contaminant concentrations in the wells were used as input to the transport model. Groundwater sampling at various wells was conducted between 1988 and 1990 by ORNL and RFI personnel. Initially, the 800-series wells were sampled quarterly; later, some wells were sampled only semi-annually. The 1200-series wells were sampled by RFI personnel. Sampling events and concentration data are presented in Table 5.4.

Model input parameters. Model input parameters are listed in Tables 5.5 and 5.6. The determination of the average values and distribution of the model input parameters are described below.

TAREA. TAREA was the cross-sectional area of a waste source. Since each of the subareas in WAG 6 (A through Z) was a cluster of trenches/auger holes representing one source, TAREA was the effective area encompassing any such cluster of trenches/auger holes.

H. The average trench depth (H) was assumed to be 15 ft. Trenches were backfilled to above the water table wherever necessary; therefore, the depth of these trenches was less than 15 ft. The nominal depth of the auger holes is 20 ft. However, after backfilling, the actual depth ranges from 15 to 20 ft (Boegly 1984).

FINT. FINT was the initial value of the fraction (f_1) of the buried waste that was exposed to percolating water in the trench. For areas that included sampling wells, FINT was a calibration parameter. These values of FINT were used for other areas where no calibration wells existed, based on the similarity of trench or auger hole characteristics. FINT was assumed to be constant for a period of 30 years from the date of emplacement of the waste; thereafter, the integrity of the container was assumed to be breached.

Uncertainty was introduced by predicting the future corrosion rates of the containers and using estimated values of FINT for areas with no sampling wells. The sensitivity of the model to the value of FINT was also high; therefore, FINT was an important model parameter.

FFIN. FFIN was the final value of f_1 (the leaching fraction). The value of f_1 was assumed to increase linearly to FFIN at year 160. The value of FFIN is 0.1. The uncertainty in FFIN and its variation in time were important because the leaching fraction was a sensitive parameter; however, the effect of variation in FFIN was reduced significantly when the inventory was rapidly depleted by a high rate of decay.

ATREN. ATREN was the total inventory for individual radionuclides or chemicals in each waste area. The accuracy of inventory records prior to 1986 is poor. In some cases, radioactive decay prior to actual waste disposal has not been accounted for. Many curies of unidentified inventory have been logged. Some of the records could be in error by as much as an order of magnitude.

Inventory records include no contaminant entry for several trenches and auger holes. In such cases, an average value was determined from the inventory of similar trenches (high-level or low-level) or auger holes in other areas and was assigned to areas that lacked disposal records. Since 1986, waste disposal records have been well maintained. The radionuclide inventories (as recorded), for the modeling areas are given in Table 5.3.

DECAY. Natural radioactive decay is a constant for each radioactive isotope, a function of its half-life. Half lives for the radionuclides modeled are listed in Table 5.7.

XKDV, XKDH. Parameters XKDV and XKDH were the effective distribution coefficients for the contaminants in the unsaturated and saturated zones, respectively, and their calibrated values were equal. These parameters represented cationic retardation of the radioactive isotopes in the aquifer and were referred to collectively as the calibration parameter f_3.

Several values for retardation coefficients for radionuclides are available in the literature. These values differ widely based on soil chemistry and soil type. The distribution coefficients (K_d) in the ETF studies documented by Davis et al. (1984) were determined from laboratory batch studies. Because these values are too large to explain the actual contaminant travel times in the field, estimation of this parameter in situ was necessary. The calibrated value of f_3, which is much lower, thus represents an effective K_d. The values are listed in Tables 5.8 through 5.16.

There are several reasons why distribution coefficients reported from laboratory studies (Davis et al. 1984) were higher than the actual site-specific effective K_d values determined here. First, organics and solvents present in the field can complex with metals such as cobalt, strontium, and uranium, resulting in their mobilization. Second, the laboratory batch experiments provide suitable conditions for equilibrium to occur (e.g., mixing), while equilibrium may not be achieved in the field. Third, the rocks used for the batch studies were pulverized, thus increasing the level of intermedia contact. Finally, the large number of fractures allows contaminants to escape adsorption/retardation. For areas with no sampling wells, a representative value was selected from the calibrated distribution coefficients.

XKDT. XKDT was the distribution coefficient for contaminants in the trenches. A nominal value was chosen from the ETF studies or other reports in the literature. This parameter affected the leaching fraction via the source, and was effectively incorporated in the parameter FINT. Since FINT was calibrated using actual data, the precise value of XKDT was not critical for modeling purposes.

RPERC. RPERC was the annual groundwater recharge through the trench bottom, for which no estimates were available. A recharge value of 1.22 in./yr = 0.031 m was previously estimated for ORNL basins (Moore 1989a). In the water balance study performed in Appendix 5A using the CREAMS (chemicals, runoff, and erosion from agricultural management system) model, a value of 0.19 m/year was determined for WAG 6.

DVERT. DVERT was the vertical distance from the bottom of the trench or auger hole to the water table. This constituted the unsaturated zone or the vadose zone. In areas where the trenches were submerged, the vertical distance was assumed to be zero. Due to variation of trench depths and seasonal fluctuations in the water table, there was some uncertainty in the value of DVERT. Table 5.6 lists the estimated values of DVERT for all areas.

DHORZ. DHORZ was the horizontal distance between the centroid of a waste area and a receptor point (usually a well). This distance was considered to be the average distance to the receptor point. The difference in horizontal distance between the nearest and farthest trenches is 20 to 40 m.

EPORA, EPORV. The parameters EPORA and EPORV were the effective porosities in the saturated aquifer and the vadose zone above the water table, respectively. EPORV had an estimated average value of 0.0042, with a range of 0.0025 to 0.006 (Moore 1989a). EPORA values in the aquifer (up to 30 m) ranged from 0.001 to 0.002. ETF studies (Davis et al. 1984) showed an effective aquifer porosity of 0.03. An average value of 0.02 and a range of 0.004 to 0.04 were used for modeling purposes for both EPORV and EPORA.

PORT. PORT was the total porosity of the trench soil. It was assumed to be 0.5, which is representative of a silty clay soil.

PORV, PORA. PORV was the porosity in the vadose zone below the trench bottom, while PORA was the porosity of the saturated aquifer. The porosity varied based on rock material. Moore (1989) estimated a porosity of 0.1 to 0.15 for regolith, 10^{-3} to 0.1 for saprolite, and 10^{-5} to 10^{-3} for bedrock. A mean value of 0.1 was chosen for both PORV and PORA.

SSAT. SSAT was the degree of saturation of the vadose zone. The degree of saturation was calculated using the following equation (Clapp and Hornberger 1978):

$$\frac{\theta}{\theta_s} = \frac{q^{1/(2b+3)}}{K_s}$$

where

θ = volumetric water content in the unsaturated zone,
θ_s = volumetric water content of soil under saturated conditions,
q = percolation rate, m/d,
K_s = saturated hydraulic conductivity, m/d,
b = soil-specific exponential parameter.

The value for b obtained from the EPA *Superfund Exposure Assessment Manual* (EPA 1989e), for silt clay was 10.4. Using a K_s of 0.0216 m/day (Rawls 1986) and a percolation rate of 0.19m/year, the degree of saturation was computed to be 0.855.

The value of SSAT would actually have been lower below the soil zone at the trench bottom where the contaminant transport was being modeled. The lower limit of SSAT was taken as the field capacity for silty clay soil, which is prevalent in WAG 6, was estimated to be 0.39 (Rawls 1986). An average representative value of 0.5 was chosen for SSAT.

BDENS. BDENS was the bulk density of the soil matrix and the aquifer material. The ETF studies (Davis et al. 1984) gave an average value of 1340 kg/m^3 in the 0- to 2-m depth. In Sect. 3 of this report, the bulk density of the saprolite is given as 1590 kg/m^3, and that of the bedrock is given as 2630 kg/m^3. Because most of the flow (and, therefore, most of the transport) occurs through the weathered zone or fractured bedrock layers, a value of 1300 kg/m^3 for BDENS was considered reasonable for the transport modeling.

AQTHK. AQTHK was the thickness of the shallow aquifer. In the Rome, Conasauga, and Chickamauga Groups, the shallow aquifer thickness is about 30 m (Moore 1989a). Groundwater flow at WAG 6 is local and occurs predominantly in the shallow aquifer to a depth of about 30 m. However, the water-bearing interval through which most of the flow occurs is about 15 ft. Also, because the extent of vertical dispersion of the contaminants is much less than the depth of the aquifer, a lower value for AQTHK (ranging from 4.6 to 6.9 m) was chosen for modeling purposes, as shown in Table 5.5.

For the purpose of modeling, appropriate velocities for subareas were obtained—depending on whether the effective flow was in the regolith or bedrock aquifer—from the velocities given in Sect. 3 of this report. Since the velocities in Sect. 3 were calculated for high water table conditions, they represent the upper end of the range of velocities predicted for WAG 6. A lower value, 50% of these velocities, was used in transport modeling.

ADISP. ADISP was the effective angle of dispersion, assumed to be 0.3.

IYRC. IYRC was the age of the inventory to year 1989, when the groundwater samples were taken. Most waste areas were active for 3 to 6 years. The midpoint between the beginning and end of waste deposition in an area was used as a representative year for all the waste deposition.

SCONC. This was a measured concentration at the well/reception point. Average values of the concentrations from sampling events were used for calibration of the model parameters.

BAREA, WAREA. BAREA was the portion of the basin that contained the waste area. WAREA was the area of the watershed that contains BAREA and where storm runoff was sampled.

TINT. The start year for container degradation was assumed to be 30 years from the date of emplacement of the waste.

TFIN. The final year for which the model is run was 160 years.

5.3.2.2 Model calibration, results, and analysis

As described in Sect. 5.3.1 the model parameters f_1, f_2, and f_3 were calibrated using sampling data for groundwater concentration, travel time, and surface runoff concentration. Since the transport equations governing the surface and groundwater contaminant fluxes were coupled, an iterative approach was undertaken to calibrate these three parameters.

The parameter f_1 (leaching fraction) was the fraction of the waste in the disposal unit that was exposed to soil moisture. This leaching fraction directly influenced the groundwater concentration. The f_1 value was assumed to be constant for the first 30 years, after which the integrity of the container would be lost. f_1 was calibrated by matching the concentration predicted by the model with the actual measured data. After the first 30-year period, the factor f_1 increased at a steady rate to 0.1 at the end of 160 years. By this time, most of the contaminant would have been depleted in the disposal units. This predicted temporal variation of leaching (degradation rate) was impossible to verify and therefore introduced a measure of uncertainty.

The parameter f_2 actually depended on the functional form of the relation

$$f_2 = f(V_r),$$

where

f_2 = the ratio of the fraction of surface flux per storm to the exposed trench inventory,
V_r = the runoff in inches.

First, the functional form of the data for runoff concentration versus V_r was empirically determined. The model allowed three options for the type of curve used to empirically fit the concentration vs V_r data. The first option assumed a constant value of f_2 for all runoff values. The second option assumed a linear relationship, with concentration increasing linearly with runoff. The third option assumed an exponential relationship between f_2 and V_r.

In most cases, either an exponential or a uniform functional relationship was found reasonable. The former was based upon the rationale that the greater the runoff, the greater the amount of immobile particulate contaminants flushed out. However, in many cases the data were quite random and an average uniform concentration, regardless of runoff value, was appropriate.

The value of f_3 represented an effective K_d (distribution coefficient). It was, therefore, a measure of contaminant retardation in groundwater transport. The factor f_3 was related to the distribution coefficient. However, it differed greatly in magnitude from the values of K_d measured in the laboratory. The calibrated value of f_3 was used in the model, and resulted in conservative estimates of retardation.

The calibration of f_3 was based on the contaminant travel time. It was assumed that a contaminant first reached the sampling well when it was observed. However, the contaminant breakthrough at the well most likely occurred before the first sampling activity was implemented. Thus, considerable uncertainty may be involved and the effective K_d may be much lower.

5.3.2.3 Results and analysis

The calibrated model was used to predict concentrations of radionuclides in the groundwater and surface runoff for each modeled area. The model output for each of the

26 areas included: (1) predicted time-series of concentrations in the calibration well or at a downgradient receptor point; (2) predicted time-series of radionuclide inventory remaining in the source area; and (3) predicted time-series of radionuclide flux in surface runoff from the area. To illustrate typical results, the time-series plots for cobalt-60, cesium-137, europium-152, tritium, and strontium-90 for Areas A, B, C, D, and Y are shown in Figs. 5.9 through 5.33. All model output values presented correspond to the 95th percentile upper limit of the mean. As shown in Fig. 5.9, initially the concentrations in well water and surface water decreases with time due to decay and source inventory depletion. The subsequent increase in concentrations is caused by a total breakdown of all containers and full exposure of waste to trench waters, assumed to occur at the end of 30 years. The effect of total breakdown of containers was simulated by increasing the calibrated value of the leaching fraction to its maximum theoretical value at the end of 160 years.

The predicted concentrations for the first year in which the radionuclide is seen in the well (breakthrough year), the predicted peak value, and the year of occurrence are summarized in Tables 5.8 and 5.10 through 5.16 for all modeled areas. In the case of tritium (Table 5.9), the concentrations in 1989 are shown instead of breakthrough concentrations, because most of the breakthroughs occurred before 1989. The tables also show the initial inventory of the radionuclides in each area. Table 5.17 is a summary of the predicted inventories in the year 2020 and 2120 and the time to complete depletion.

The following paragraphs summarize the analysis of modeling results for groundwater contaminant concentrations for each of the modeled areas. All concentrations are reported as the upper 95 percentile limit of the mean.

Area A. The wastes were stored in a mixture of lined and unlined auger holes between 1974 to 1981. Table 5.3 gives the inventory of important radionuclides. There are large quantities of europium-152 (4176 Ci), europium-154 (1558 Ci), cobalt-60 (3357 Ci), and cesium-137 (1646 Ci).

The downgradient sampling well for Area A was Well 1242. The contaminant plume extended southeastward, not quite reaching Well 1242. Concentrations in this well were, therefore, much lower than those seen in Well 842, which monitors a similar area (Area B). The well was screened in the depth range of 12 to 25 ft and straddles the water table. Groundwater concentrations are given in Tables 5.8 to 5.13. The value was highest for tritium (316,000 pCi/L), and europium was not projected to show up until the year 1998. Peak concentrations of europium-152 (32.4 pCi/L) and europium-154 (2.2 pCi/L) were predicted for around the year 2030 because the aluminum cladding would prevent leaching of europium for the first 10 to 20 years. Europium has relatively reduced mobility because it does not complex with the organic solvents.

Area B. This area, adjacent to Area A, contains high-level auger holes and solvent wastes. The waste inventory includes low levels of tritium (32.8 Ci) and high levels of cobalt-60 (1305 Ci), strontium-90 (284 Ci), and cesium-137 (771 Ci) and europium-152 (16,270 Ci) and europium-154 (22,305 Ci). Area B was active from 1973 to 1983.

Well 842 monitors the shallow aquifer downgradient from Area B. The concentration in groundwater of tritium (656,000 pCi/L) at the well was high. There were also high concentrations of cobalt-60 (1030 pCi/L), moderate levels of cesium-137 (12.4 pCi/L), and low levels of strontium-90 (4.3 pCi/L). Peak concentrations of europium-152 and europium-154 were predicted to occur in the year 2030.

The high cobalt-60 concentrations may indicate that the distribution coefficient for cobalt-60 was lowered by complexation with organic solvents. The relatively low value for cesium-137 was due to its high retardation. The concentration of strontium-90 was predicted to reach a peak value of 418 pCi/L in 2052.

Area C. This area is located at the northern end of WAG 6. The wastes stored in Area C include large amounts of tritium (1071 Ci) and strontium-90 (105 Ci) and moderate quantities of cobalt-60 (15.2 Ci) and cesium-137 (34.1 Ci). There is also a small amount of uranium-233 (4.8 Ci). Waste was deposited in Area C from 1976 to 1978.

The general direction of the plume was southwest. The sampling Well 1225 was near (35 m from) the center of the area and monitored the plume. Significant quantities of all radionuclides were observed at the well. High levels of tritium (2.8 μCi/L) in the well were due to the large inventory of tritium. Extremely high levels of strontium-90 (5640 pCi/L) were also seen.

Area D. This area is adjacent to Area C. Waste was deposited in trenches in this area from 1978 to 1985. The initial inventory of cobalt-60 was high (3521 Ci). Moderate quantities of strontium-90 (68 Ci) and cesium-137 (53.3 Ci) were placed. The inventory of tritium was low (13.3 Ci). In addition, a small quantity of uranium-233 (2.8 Ci) was placed in Area D trenches. This area was open for waste deposition between 1978 and 1985.

The plume traveled southwestward toward the FB stream. There were no sampling wells downgradient of this area. The groundwater receptor point 50 m from the area showed low levels of cobalt-60, strontium-90, cesium-137 and uranium-233 because of low groundwater velocities. Cobalt-60, strontium-90, and cesium-137 did not reach the receptor point until the year 2024, 2055, 2126, respectively. However, large concentrations of tritium (1.04 μCi/L) were found.

Area E. Area E is located close to the northern boundary of WAG 6. The water table is about 4.6 m below the trench bottom. The area has relatively low quantities of radionuclides; its inventory includes 16.2 Ci of cobalt-60, 50.4 Ci of tritium, 3.2 Ci of strontium-90, 12.5 Ci of cesium-137, 4.6 Ci of uranium-233, and 7.7 Ci of uranium-238. Area E was active between 1972 and 1973.

The groundwater contaminant plume moved eastwardly toward the EWB. Well 867, located within 9 m of this area, contained low levels of cobalt-60 (9.6 pCi/L), strontium-90 (4.3 pCi/L) and cesium-137 (5.2 pCi/L). The strontium-90 concentration was predicted to peak at 17.2 pCi/L in the year 2049. Uranium-233 (7.3 pCi/L) and uranium-238 (19 pCi/L) were predicted to reach the well in the year 2038. Peak concentrations of 149 pCi/L for uranium-233 and 400 pCi/L for uranium-238 were predicted for the year 2197.

Area F. Area F, located at the northern end of WAG 6, close to the EWB, exhibited relatively small amounts of cobalt-60 and uranium-233 and moderate amounts of tritium, strontium-90, and cesium-137. The inventory of the major radionuclides consisted of 14.5 Ci of cobalt-60, 45.5 Ci of tritium, 93.3 Ci of strontium-90, 47.1 Ci of cesium-137, and 4.1 Ci of uranium-233. The area was active from 1972 to 1976.

Well 1241 is 65 m from the area. The contaminant plume migrated north toward the EWB. Groundwater concentrations at Well 1241 included low levels of cobalt-60 (7.4 pCi/L), strontium-90 (8.02 pCi/L), and cesium-137 (20 pCi/L). The concentration of tritium was also relatively low (4700 pCi/L). The peak concentration of tritium (35,000 pCi/L) was predicted to occur in the year 2009, with strontium-90 peaking at 268 pCi/L in the year 2032. Uranium-233 was projected to reach the well in the year 2217 (2.5 pCi/L). The uranium-233 concentration was predicted to increase to 60.6 pCi/L in the year 2376, the final year of simulation.

Area G. Area G, adjacent to Area F, contained a few high-level trenches. The waste inventory consisted of moderate quantities of tritium (13 Ci), cobalt-60 (4.13 Ci), strontium-90 (26.7 Ci), cesium-137 (13.5 Ci), and uranium-233 (1.2 Ci). Area G was active during 1975.

Sampling Well 843 is about 55 m east of the area. The direction of the hydraulic gradient of the underlying aquifer is toward the east. Groundwater well concentrations at Well 843 included high levels of cobalt-60 (105 pCi/L) and tritium (1.39 μCi/L). The trenches in Area G are relatively old, with breached containers. Also, retardation of cobalt-60 was high, possibly because of organic complexation. Only small amounts of strontium-90 and cesium-137 were present. The peak concentration of strontium-90 (496 pCi/L) was predicted to reach the well in the year 2053; uranium-233 was not expected to reach the well for a long time.

Area H. Area H, situated in the north-central part of WAG 6, contains auger holes and high-level silos. It had high inventories of cobalt-60 (82.2 Ci), strontium-90 (45 Ci), and cesium-137 (106 Ci). The quantity of tritium stored was relatively low (10 Ci). The waste was deposited between 1986 and 1990.

The plume has traveled west toward Stream FB. There is no sampling well to monitor the groundwater system. The results of modeling showed that most of the radionuclides had not reached the groundwater 50 m from the source. Only tritium was transported to that point and was present at a concentration of 843,000 pCi/L. Cobalt-60 and strontium-90 were predicted to reach a receptor point 50 m from the source in the year 1998. Cesium-137 was expected to reach the receptor point even later, in the year 2080, and was present at a very low concentration. Because this waste area is relatively new, cobalt-60 and most other radionuclides have not yet reached the well.

Area I. Area I is situated in the upper-central region of WAG 6. The inventory includes very small quantities of cobalt-60 and tritium, as well as strontium-90 (20.4 Ci) and cesium-137 (10.4 Ci). The area was active from 1986 to 1990.

There was no sampling well hydraulically downgradient of this area. The direction of the plume was southwest toward Stream FB. The concentration of tritium in the groundwater was predicted to be high (542,000 pCi/L) at a receptor point 50 m from the area in the year 1990, although very little tritium was deposited in the source area. This was because the trenches were relatively new, and the tritium source had undergone little depletion. Low levels of cobalt-60 were expected to reach the receptor point in the year 1999. Strontium-90 was predicted to reach the receptor point in the year 2013 at a concentration of 561 pCi/L. Low levels of cesium-137 were predicted to reach the receptor point in the year 2359.

Area J. Area J, situated in the north-central region of WAG 6, contains a large number of fissile waste auger holes and a few solvent waste auger holes. This area was active from 1986 to 1988 and had large inventories of cobalt-60 (250 Ci) and cesium-137 (702 Ci), a moderate amount of strontium-90 (43.7 Ci), and a small amount of tritium (5 Ci).

There was no sampling well to collect groundwater concentration data. The contaminant plume moved southeast toward Stream DB. Groundwater concentrations were modeled at a receptor point 50 m from the area. Cobalt-60 and strontium-90 were both predicted to reach this point in the year 1990 with concentrations of 96.6 pCi/L and 53.6 pCi/L, respectively. Strontium-90 was expected to reach a peak in the year 2049 at 130 pCi/L. Cesium-137, with low concentrations because of retardation, was predicted to reach this point in the year 2016. Tritium also showed low concentrations in the groundwater.

Area K. Area K, situated in the north-central region of WAG 6, had only a few auger holes. The area was active in the year 1981 and received a large deposition of europium-152 (26,801 Ci) and europium-154 (48,701 Ci). The radionuclide inventory also included cobalt-60 (20.9 Ci), strontium-90 (6.8 Ci), and cesium-137 (18.8 Ci). The quantity of tritium was very low.

There was no downgradient sampling well to monitor groundwater concentrations. The contaminant plume moved south toward Stream DB. The modeling results for a receptor point 50 m from that source showed the presence of cobalt-60 at a concentration of 7.56 pCi/L, strontium-90 at a concentration of 13.7 pCi/L, and very low levels of tritium and cesium-137. Europium-152 and europium-154 were predicted to reach the receptor point in the years 2010 and 1996 at concentrations of 0.17 pCi/L and 48.8 pCi/L, respectively. Europium has a higher retardation because it does not complex with organic solvents.

Area L. Area L, situated in the central region of WAG 6, had a radionuclide inventory that included cobalt-60 (7.2 Ci), tritium (2488 Ci), strontium-90 (38 Ci), cesium-137 (159 Ci), uranium-233 (101 Ci), and uranium-238 (1.72 Ci). This area was active from 1979 to 1984. Sampling Well 848 is 44 m downgradient from this area.

The contaminant plume moved southwest to the French Drain. A portion of the plume moved southeast to Stream DB. The concentration of strontium in the well was 59 pCi/L. Concentrations of cobalt-60 and cesium-137 were low because cobalt-60 had a low inventory and cesium-137 has high retardation. High levels of tritium were present (7 μCi/L). Uranium-233 and uranium-238 were predicted to reach the well in the year 2025. The concentration of uranium-233 was predicted to be 19.7 pCi/L; very little uranium-238 would be present.

Area M. Area M, located in the central region of WAG 6, was active between 1973 and 1976. The inventory of major radionuclides comprised 198 Ci of cobalt-60, 617 Ci of tritium, 39 Ci of strontium-90, 152 Ci of cesium-137, and 56 Ci of uranium-233. A small amount of thorium-232 (1.13 Ci) was also present.

Sampling Well 849, located about 58 m from the centroid of the area, monitored the contaminant plume from Area M. The contaminant plume migrated predominantly southwest toward Stream FB. A portion of the plume was intercepted by the French Drain. The observed concentration of cesium-137 was 11.5 pCi/L. The tritium level was quite high 3.4 μCi/L. Cobalt-60 and strontium-90 were present at low levels, and the concentration of thorium-232 was very low. Uranium-233 was predicted to reach the well in the year 2150 with a concentration of 27 pCi/L. The level of uranium-233 was predicted to increase over the years as the leaching rate increases (and decay is negligible).

Area N. Area N, situated in the western region of WAG 6, included low-level and asbestos trenches. The waste inventory consisted of large amounts of tritium (452 Ci) and cobalt-60 (645 Ci) and lesser amounts of strontium-90 (25 Ci), cesium-137 (35.4 Ci), uranium-233 (25 Ci), uranium-238 (16.3 Ci), and thorium-232 (16.2 Ci). This area was active between 1976 and 1979.

Sampling Well 853, about 56 m south of the centroid of Area N, missed the thickest portion of the contaminant plume that migrated southeast to drainage FB. Therefore, concentrations of all radionuclides present at the well were low. Uranium-233 and uranium-238 were predicted to reach the well in the year 2057. The concentration of uranium was expected to increase due to the increasing leaching rate and the lack of significant decay.

Area O. Area O, situated in the western region of WAG 6, was open between 1977 and 1978 and contains low-level trenches. The inventory contained significant amounts of tritium (284 Ci), cobalt-60 (391 Ci), strontium-90 (19.1 Ci), cesium-137 (115 Ci), and small quantities of thorium-232 (1 Ci) and uranium-238 (1.12 Ci). Sampling Well 854, about 54 m from the center of Area O, had low concentrations of radionuclides except for tritium (292,000 pCi/L). The well missed most of the plume because of its location near the groundwater divide at the southern tip of Area O.

Area P. Area P, in the eastern region of WAG 6, was active from 1986 to 1990 and included high-level auger holes. The inventory contained large amounts of cobalt-60 (21,064 Ci), tritium (2700 Ci), strontium-90 (129 Ci), cesium-137 (5822 Ci), europium-152 (250,865 Ci), europium-154 (187,368 Ci), and thorium-232 (6003 Ci). The area was monitored by two sampling wells, Well 845 (southwest) and Well 1243 (east). Since the predominant direction of groundwater flow was to the east, Well 1243 was more representative. High levels of cobalt-60 (8210 pCi/L), tritium (1.8 μCi/L), and cesium-137 (37 pCi/L) were present in groundwater samples from Well 1243. These concentrations reflected the high activity in the auger holes. Organic solvents were present in the solvent auger holes, and formation of complexes with organics increased the mobility of cobalt-60. Because most of the high-activity auger holes were lined, leaching was initially reduced, preventing the groundwater concentrations from reaching even higher levels.

Europium was predicted to reach the well in the year 2005, with the europium-152 concentration at 163 pCi/L and the europium-154 concentration at 321 pCi/L. Europium-152 was predicted to peak at 850 pCi/L in 2040, and europium-154 to peak at 584 pCi/L in 2036. Thorium-232 was not expected to reach the well for many years because of its relatively high distribution coefficient and consequent high retardation.

Area Q. Area Q, situated in the eastern region of WAG 6 just south of Area P, was capped. It comprises trenches and a few auger holes. The inventory consisted of small amounts of tritium (6.2 Ci) and large quantities of cobalt-60 (377 Ci), strontium-90 (50 Ci), and cesium-137 (265 Ci). Area Q was active between 1973 and 1974. The contaminant plume migrated east and was intercepted by sampling Well 840 to the east of the area.

The well concentration for tritium was 23,000 pCi/L. Other radionuclides present included cobalt-60 (34.6 pCi/L), strontium-90 (9.3 pCi/L), and cesium-137 (2.3 pCi/L). The tritium concentration was predicted to peak at 1.6 μCi/L in the year 2024.

Area R. Area R, located in the eastern region of WAG 6 near Area Q, includes a few trenches and auger holes. The waste inventory included large quantities of cobalt-60 (1001 Ci) and smaller quantities of tritium (19 Ci), strontium-90 (21 Ci), and cesium-137 (41 Ci). Area R was active from 1979 to 1986.

There was no sampling well monitoring the Area R contaminant plume, which flowed southeast. Radionuclides present at a receptor point 50 m downgradient from the area included cobalt-60 (82 pCi/L), tritium (49,500 pCi/L), and strontium-90 (22.8 pCi/L). The concentration breakthrough curve for cesium-137 showed it reaching the receptor point in the year 2025.

Area S. Area S, located in the western region of WAG 6, contains biological trenches. The inventory showed very low levels of radionuclides. Only tritium was deposited in a significant amount (30 Ci). Area S was open to waste deposition from 1989 to 1990. Sampling Well 851, located about 36 m southeast of Area S, showed relatively low levels of tritium (58,000 pCi/L) and insignificant concentrations of other radionuclides due to the small quantities of their sources in the trenches. The contaminant plume migrated southeast, and the groundwater velocity was low.

Area T. Area T, located in the western region of WAG 6 adjacent to Area S, includes biological trenches. It had an inventory of 34.5 Ci of tritium and very low quantities of other radionuclides. Area T was open to waste deposition between 1977 and 1979. Sampling Well 852, located 50 m south of this area, was in the contaminant flow path. The contaminant plume migrated south-southwest, and the groundwater velocity was low. The concentration of tritium in the well was 97,000 pCi/L. Other radionuclides were not present at significant levels.

Area U. Area U, located in the central region of WAG 6, includes mostly biological trenches. The inventory consisted of small amounts of radionuclides, with only tritium (61.6 Ci) present in significant amounts. This area was open between 1979 and 1981. Sampling Well 850, located about 48 m south of the area, monitored the contaminant plume extending from southeast (toward Stream DB) to south (toward Stream DA).

The concentration of tritium in the well was 63,000 pCi/L. The concentrations of other radionuclides were close to background levels and may have represented contributions of the Area L plume, which was directly upgradient.

Area V. Area V, located near the southeastern boundary of WAG 6, is a small area that included low-level trenches. The inventory for tritium was 50.4 Ci. There were small quantities of cobalt-60 (16.2 Ci), strontium-90 (3.2 Ci), cesium-137 (12.5 Ci), uranium-233 (4.6 Ci), and uranium-238 (7.7 Ci). This area was an active storage site from 1978 to 1979. The contaminant plume traveled southeast to Stream DB.

Well 838, which is 55 m from Area V, was a sampling well for this area. Well 838 was on a groundwater divide and did not intercept the main portion of the plume. The concentration of tritium in the well was 17,900 pCi/L. Low levels of other radionuclides were also present. Both uranium isotopes were predicted to reach the well in the year 2064.

Area W. Area W, situated near the southeastern boundary of WAG 6, contains asbestos trenches. The area was active between 1979 and 1985. The inventory of radionuclides was very small. There was no sampling well monitoring the plume, which extended from southeast (traveling toward the WAG boundary) to south (traveling toward Stream FB). Modeling results for a receptor point 50 m from the source area showed a low concentration of tritium (169,000 pCi/L).

Area X. Area X, at the southern end of WAG 6, includes animal trenches; it was active from 1972 to 1988. There was a significant inventory of cobalt-60 (25.7 Ci) and tritium (177 Ci). The other radionuclides were present in small quantities. The contaminant plume traveled toward WOL, and was monitored by Well 835.

The concentration of most radionuclides was low. Strontium-90 was predicted to exhibit a peak concentration of 41 pCi/L in the year 2049, and tritium was shown to be present at a concentration of 45,000 pCi/L.

Area Y. Area Y, near the southern boundary of WAG 6, comprises low-level trenches. It was active from 1984 to 1986. The trench inventory included large quantities of tritium (805 Ci) and cobalt-60 (6436 Ci). Moderate quantities of strontium-90 (34.4 Ci), cesium-137 (40.6 Ci), uranium-233 (124.7 Ci), and uranium-238 (209 Ci) are also present. The contaminant plume traveled toward WOL. Sampling Well 837, which is 58 m south of the area, monitored the contaminant plume.

The concentrations of all radionuclides in the well were relatively low. The concentration of cobalt-60 was predicted to peak at 60 pCi/L in the year 2024. The trenches in Area Y were, on the average, 3.0 m above the water table. The groundwater aquifer may possibly have been influenced by intrusion of water from WOL, thereby diluting the contaminant plume and also increasing adsorption (retardation) of radionuclides.

Uranium isotopes were predicted to reach the well in the year 2091. The concentration of uranium-233 was predicted to be 28.9 pCi/L, and that of uranium-238 to be 84 pCi/L. The uranium concentrations were expected to continue increasing with time due to their very long half-lives.

Area Z. Area Z, located in the southern region of WAG 6, just east of Area Y, contains low-level trenches and was active between 1986 and 1990. The inventory of radionuclide waste consisted of tritium (9.5 Ci), strontium-90 (10.6 Ci), cesium-137 (20.2 Ci), and small amounts of other radionuclides. There were no sampling wells downgradient from this area.

At a receptor point 50 m away, the concentration of tritium was 477,000 pCi/L. The concentration of strontium-90 was predicted to be 62 pCi/L in the year 2049. The other radionuclides were present at low levels.

Tumulus I. Tumulus I was the only aboveground unit simulated. This disposal unit, located in the south-central region of WAG 6, received waste stored in casks from 1986 to 1990. Small quantities of radionuclides were stored in the tumulus. Only cobalt-60

(10.4 Ci), tritium (2.7 Ci), strontium-90 (12.0 Ci), and cesium-137 (14.3 Ci) were present in significant quantities. The groundwater moved in a predominantly southwesterly direction toward Stream FB. There were six monitoring wells around Tumulus I.

Model calculations were made to determine the concentration at a receptor well 50 m downstream from the disposal unit. Since the unit's concrete lining was assumed to hold for 100 years, no significant concentrations were predicted to be observed at the receptor point for some time. Only small amounts of tritium were expected to reach the groundwater via diffusion through the concrete. Strontium-90, which has a longer half-life and higher retardation factor than tritium, was predicted to peak at 13.1 pCi/L in the year 2125 at the receptor point. Most of the other radionuclides were not expected to be present at significant concentrations at any time.

5.3.3 Stream Flow Radionuclide Concentrations and Boundary Flux

Radionuclide concentrations in the streams and the flux of each of the isotopes across the WAG 6 boundary were required for the off-WAG 6 risk assessment. The contaminants can leave the site in either surface water or groundwater. The flux in surface water could leave the site either in the streams draining the site or in overland flow. The groundwater flux could be intercepted by either the stream on the east side of WAG 6 or by WOL.

In the model, the flux in the surface water received contributions from two different pathways. The pathway was surface runoff, which consisted of overland flow, saturated overland flow, and subsurface storm flow. (The fluxes in the surface runoff are shown in the model output.) The second pathway was base flow, which results from the groundwater discharge.

The surface runoff fluxes were routed down to the stream. The flux due to base flow was determined by finding a point at which groundwater would intercept the stream. Not all of the groundwater will enter the stream as base flow; therefore, the groundwater flux was proportioned by the ratio of base flow to total percolation to derive the flux in the base flow. The rest of the flux will be transported in groundwater.

The radionuclide concentrations along the streams are listed in Tables 5.18 through 5.26. The predicted peak annual fluxes and the years of their occurrence are presented in Table 5.27. Plots of the time-series of radionuclide fluxes in surface water across the WAG boundary are shown in Figs. 5.34 through 5.42. Nine radionuclides were included: cobalt-60, cesium-137, tritium, strontium-90, europium-152, europium-154, thorium-232, uranium-233, and uranium-238. The time-series of the fluxes in surface water discharged into the EWB are shown in Figs. 5.43 through 5.48.

5.3.4 Boundary Fluxes in Groundwater

To satisfy the data requirements of the base line risk assessment, the total contaminant flux exiting the WAG 6 boundary was determined, as well as the contaminant flux into the EWB. The contaminant flux in groundwater at the boundary was calculated by the model for each of the plumes. Only the portion of the flux not discharged into the streams was considered to contribute to the total groundwater flux from WAG 6. The predicted peak annual fluxes and the years of their occurrence are presented in Table 5.28. A time-series plot of the contaminant flux across the WAG 6 boundary for each of the modeled constituents in groundwater is shown in Figs. 5.49 through 5.57. The fluxes in groundwater into the EWB are shown in Figs. 5.58 through 5.63.

5.3.5 Discussion of Results

5.3.5.1 Radionuclides in groundwater

Present-day and predicted future concentrations of the modeled radionuclides in groundwater (Tables 5.8 through 5.16) are discussed below. Cobalt-60 and tritium concentrations in groundwater were predicted to deplete monotonically over time due to decreasing leachate flux and radioactive decay. Maximum concentrations of cobalt-60 have already occurred and no significant peaks were predicted in the future. Except in areas F, N, Q, and V, the observed tritium concentrations in the wells were the maximum expected concentrations.

The peak strontium-90 concentrations in most wells were predicted to occur after the year 2032. The predicted peaks were within an order of magnitude of concentrations observed during WAG 6 RFI field sampling. The present-day concentration of strontium-90 in groundwater was highest in Area C. Also, existing groundwater plumes from areas Z and L showed significant strontium-90 concentrations. Because strontium-90 complexes with organics less readily than cobalt-60, it has relatively high retardation. Strontium-90 behaves as does calcium in competing for cationic adsorption sites in clays and minerals and has a longer half-life than cobalt-60 or tritium. Thus, in several areas with high strontium inventory (such as areas A, B, F, G, H, and P), the groundwater concentration was predicted to show a peak in the future after containers are breached. With the plumes from areas C, D, and H, there was a significant strontium-90 flux in groundwater discharging to stream FB.

There were relatively high concentrations of cesium-137 in the contaminant plumes downgradient of areas A, B, C, L, M, N, O, and P. The cesium concentration should decrease with time in most places except for the plume from Area P; peak concentration was predicted to occur in the year 2051, with about three times the present-day concentration.

Europium-152 and europium-154 were not present in most of the groundwater plumes. Most europium was placed in lined auger holes with control plates composed of a europium oxide-aluminum cermet clad with aluminum. The lining in the auger holes and the aluminum cladding has delayed and reduced leaching. There were high inventories of europium-152 in areas B and P and of europium-154 in areas B, K, and P. Significant concentrations of europium-152 were predicted to be present in the receptor wells downgradient of areas B and P around the year 2000. Around the same period, contaminant plumes from areas B, K, and P may show elevated levels of europium-154. Concentration peaks for europium were predicted to occur between 2030 and 2040, after the auger aluminum claddings were assumed to be ineffective.

Present concentrations of uranium in wells appeared negligible. There was a large inventory of uranium-233 reported for areas L, M, N, and Y. Uranium-238 is present in large quantities in areas E, N, V, and Y. For the source Area L, Sampling Well 848 was predicted to have a uranium-233 concentration of approximately 20 pCi/L in the year 2025. In Area Y, sampling well 837 was predicted to have a uranium-233 concentration of approximately 30 pCi/L and a uranium-238 concentration of approximately 84 pCi/L in year 2091. Peak concentrations of uranium-233 and uranium-238 were predicted to occur beyond the year 2180.

Thorium-232 inventory was high in Areas P (6003 Ci) and N (16.2 Ci). The thorium-232 concentrations, like those of uranium, continue to increase for many years (as thorium-232 has a very long half-life). However, retardation of thorium is almost an order of magnitude higher than retardation of the other radionuclides analyzed. Therefore, it should reach the wells or receptor points very slowly; its detected concentrations were negligible.

5.3.5.2 Depletion of radionuclide inventories

The fate of the radionuclides in the source areas discussed below was based on the remaining source inventory data presented in Table 5.17.

Within 55 years, the cobalt-60 inventory was predicted to be depleted in all waste areas in WAG 6 except Area P, which had the largest inventory. The cobalt-60 inventory was predicted to be completely depleted within 72 years. The tritium inventory was predicted to be depleted in all areas except Area L within 65 years and to be completely depleted in 96 years. Strontium-90 has a longer half-life and depletes slowly. By the year 2120 (within 129 years), only areas A, B, C, D, F, and P were predicted to have significant strontium-90 activity. Cesium-137 was predicted to remain in significant quantities in areas A, B, J, P and Q by the year 2120.

Europium-152 and europium-154 were present in areas A, B, K, and P. Europium-152 has a half-life of 13.6 years and was predicted to be present at significant levels in all source areas in the year 2120, when europium-154 (half-life of 8.8 years) would be depleted in all areas except Area P.

Uranium and thorium were present at significant levels in a few areas (see Table 5.10). Relatively high levels of uranium-233 were present in Areas L, M, N, and Y, and high levels of uranium-238 were present in areas M, N, and Y. High thorium-232 activity (6003 Ci) existed only in Area P. The source inventory of uranium-233, uranium-238, and thorium-232 was predicted to undergo little depletion by the year 2100 because of slow leaching. Since these radionuclides have long half-lives, depletion of the existing source through decay will take many years.

5.3.6 Sensitivity and Uncertainty Analysis

5.3.6.1 Sensitivity analysis

The sensitivity of a model output to a selected input parameter is the degree to which the modeling result is affected by changes in that parameter. A sensitivity analysis is conducted to identify sensitive input parameters, which should be given attention and defined with accuracy. Sensitivity analysis can range in complexity from a simple high-low screening approach to a complete factorial design using all combinations of levels of input parameter values.

The approach implemented in this study included an initial qualitative screening followed by a quantitative analysis. In a qualitative approach, the influence of the input parameters is ascertained by inspecting the model's equations. This qualitative appraisal identifies potentially sensitive as well as nonsensitive parameters and minimizes the number of parameters to be considered for a quantitative analysis. The sensitivity of the model can be appraised quantitatively by plotting the output parameter as a function of a single input parameter, holding the others constant. However, sensitivity to an input parameter may vary markedly under different sets of conditions (e.g., transient states).

Qualitative analysis. Surface water samples and preliminary surface water modeling results showed low levels of radionuclides except for tritium. The future groundwater concentration and the residual inventory in the trenches were the most important model output parameters considered for sensitivity analysis. Figs. 5.64 through 5.82 illustrate the sensitivity of model output to changes in various parameters.

Source inventories are depleted by radioactive decay and leaching to groundwater. Since radioactive elements decay exponentially at a constant, unalterable rate, only the parameters that affect the leaching required investigation. Leaching to groundwater was expressed as:

$$AL = (RPERC \times ATREN \times f_1)/[H \times (PORT \times SSAT + XKDT \times BDENS)]$$

where

AL = leachate flux (Ci),
RPERC = percolation (m/yr),

ATREN = present day trench inventory (Ci),
F = leaching fraction
H = depth of trench (in.),
PORT = porosity of trench,
SSAT = degree of saturation in trench,
XKDT = partition coefficient in trench (m^3/kg),
BDENS = bulk density (kg/m^3).

The trench depth (H) was documented to be more or less constant (15 ft). The bulk density (BDENS) usually had low variability. XKDT manifested large variations that would likely influence model output. The term PORT × SSAT was usually much smaller than the term XKDT × BDENS, so its effect on the model output is less significant. The terms RPERC, ATREN, and f_1 directly affected the leaching flux AL. The parameter ATREN was directly related to the initial source inventory ATRENI.

The groundwater concentration at any cross-section of the plume was determined primarily by the leachate flux AL, the water flux WA at that point, and the travel time TRTH.

GWC = AL/WA x exp(-DECAY x TRTH)
WA = GWV x EPORA x AQTHK x AN
AN = AN (ADISP, TAREA, DHORZ)
TRTH = TRTH (GWV, DHORZ, PORA, BDENS, XKDH)

where

GWC = groundwater concentration (Ci/m^3),
WA = water flux (m^3/yr),
GWV = groundwater velocity (m/yr),
EPORA = effective porosity in saturated zone
AQTHK = aquifer thickness (m),
AN = cross-sectional area of plume (m^2),
ADISP = angle of dispersion, (radians),
TAREA = area of the waste source area (m^2),
DHORZ = horizontal distance to well (m),
TRTH = travel time (yr),
XKDH = distribution coefficient in the aquifer (kg/m^3).

As explained earlier, the variation in PORA and BDENS was not expected to cause much variation in the output parameter. Therefore, the likely sensitive parameters were GWV, EPORA, AQTHK, ADISP, RPERC, ATRENI, F, TAREA, DHORZ, and XKDH.

Quantitative analysis. Sensitivity analysis done for tritium and cobalt-60, were representative of mobile and slow-moving isotopes, respectively. The sensitivity of the model output to change in the value of an input parameter was determined by the ratio of percent change in the output parameter to percent change in the input parameter. Table 5.29 lists ratios calculated using nominal values of input parameters for cobalt-60. Table 5.30 lists the corresponding values for tritium.

Inventory. In the analysis for trench inventory, both isotopes showed similar trends in changes in inventory with change in any input variable. However, inventory for cobalt-60 was less sensitive than the tritium inventory because radioactive decay is the predominant mechanism for loss of cobalt-60 from the trenches. Cobalt-60 inventory is insensitive to all parameters except ATREN (initial trench inventory).

Tritium is only moderately sensitive to RPERC and FINT (see Table 5.30) and quite insensitive to other parameters. Figs. 5.64 and 5.65 show the variation of inventory after 30 years of emplacement with variation in FINT and RPERC respectively.

Groundwater concentrations. The groundwater concentrations of both cobalt-60 and tritium are very sensitive to several input parameters, including ATREN, GWV, EPORA, RPERC, FINT, and AQTHK (Tables 5.29 and 5.30). Cobalt-60 is also very sensitive to IYRC and XKDT and moderately sensitive to DHORZ, XKDH, and ADISP. Tritium is moderately sensitive to DHORZ, IYRC, TAREA, and ADISP and very slightly sensitive to FFIN.

5.3.6.2 Uncertainty analysis

Uncertainty in fate and transport modeling can be related to (a) uncertainty in the input variables and (b) the accuracy with which environmental processes are represented. Uncertainty can be described quantitatively, semiquantitatively, or qualitatively. A quantitative approach requires that the probability distribution of the input parameters be known. Model results are highly dependent on the accuracy of the input variables used. The distribution of output will depend on the distribution of input variables. The wider the spread of the distribution, the larger the uncertainty.

For a semiquantitative treatment, only the range of parameter values is needed. Sensitivity analysis plots are used to determine the critical parameter and develop the bounds of the output parameter. In the qualitative treatment, the influence of individual parameter uncertainty and its sensitivity is analyzed. In this analysis, a quantitative and a qualitative treatment were considered.

Quantitative analysis. Uncertainty in the parameter value can be propagated through analytic or numerical (e.g., Monte Carlo simulation) methods. Except for simple formulations with few parameters, the analytical method is difficult to implement. Because of the large number of input parameters and the complex formulations in the model, a Monte Carlo simulation approach was used. The model equations were solved for a combination of different realizations of input parameter set. The mean of 100 simulation results with 95% upper limits were given in the computer printouts.

Qualitative analysis. Often, the most practical approach to characterizing parameter uncertainty is to develop a qualitative description of each parameter, indicating its influence on output result.

The contaminant release rate was represented by a leaching fraction (f_1) that is empirically calibrated. The actual breach mechanism was not modeled. Similarly, contaminant transport was modeled with a parameter (f_3) that represents a measure of retardation resulting from several inadequately understood transport processes due to fractures, solvent mobilization, and heterogeneity of media. However, because the model was calibrated with measured data for the leaching fraction (f_1) and the effective K_d (f_3), the description of the actual mechanism is not critical.

The nominal value, the range of values and a qualitative classification of the uncertainty of the parameter set are given in Table 5.31. Many of the parameters identified as uncertain are also identified as sensitive (see Table 5.30).

5.4 AIR MODELING

Air modeling was performed to determine average annual airborne contaminant concentrations over the WAG 6 area for several periods corresponding to hypothetical receptor scenarios defined by the base line risk assessment (Sect. 6). The modeling consisted of a screening level analysis of emissions and of atmospheric dispersion of the contaminants of concern, including tritium and selected VOCs. Tritium was found in surface waters of WAG 6 and was assumed to be emitted to the atmosphere as water evaporates. VOCs were emitted from waste burial grounds in WAG 6.

Volatile organic air emissions were estimated by EPA's landfill emissions estimation model CHEMDAT7 (EPA 1989a). Dispersion modeling was performed using EPA's Industrial Source Complex Long Term (ISCLT) atmospheric dispersion model (EPA 1987b). The data used in modeling included sampling data for tritium and VOCs, and meteorological data for the 5-year period (1986–1990) at ORNL's Station TA, located near WAG 6.

The paragraphs below describe the data and the models used in the analysis, the application of the models to the WAG, and the modeling results.

5.4.1 Environmental Characterization

5.4.1.1 Meteorological data

The meteorological data required included wind speed and direction frequencies sorted by atmospheric stability classification for a representative period of record. A search of existing meteorological data sets available for the WAG 6 area identified two ORNL stations (TA and TB) near the WAG 6 site. These two data sets were evaluated for suitability, and the TA station was chosen as the most representative for WAG 6 because it was closer (Fig. 5.83).

The data for the TA station covered a 5-year period (1986 to 1990). The data were obtained in raw form from ORNL and converted to model input using several computer programs. While most of this conversion process involved reformatting and unit conversion, atmospheric stability classifications had to be extracted from the data using a typical classification scheme as shown in Table 5.32 (EPA 1986b). These data were then summarized to produce a "joint frequency" of wind speed class versus wind direction class divided into their respective stability classifications. The choice of both the number of classes and the class limits was dependent on the requirements of the model used in this analysis. The resultant joint frequency analysis is presented in Table 5.33.

Included in Table 5.33 were both the joint frequencies for each stability class and the total wind frequency for the 5 years of data. The wind speed and direction were taken from a 10-m tower. Additionally, the listed wind speed value represented the approximate mid-point value of the specific wind speed class. The exceptions were the lowest wind speed class (1.5 m/s), which began at 1-m/s (below this value the wind speed sensor measurements tended to be unreliable), and the highest class (> 11 m/s), which summed all frequencies above this value. The wind directions indicated in the table represented the midpoint of a sector from which the wind was blowing to the station. The table includes summary statistics for the data processed. Calms were counted as valid data values below the 1-m/s wind sensor threshold. Missed observations included bad data points not processed into the joint frequency analysis.

Table 5.33 shows that the winds experienced by TA station were about 43% calm during the 5-year period. The wind direction pattern was bimodal and indicative of a valley type flow along a west-southwest to east-northeast axis.

5.4.1.2 Terrain

Terrain at and near the WAG 6 site generally consisted of rolling hills, ranging from 750 ft to 860 ft above sea level. However, for the purpose of screening level modeling, the terrain was assumed to be flat. Emissions, at ambient temperature, would not be buoyant and pollutant plumes should remain near the surface, where terrain should not be a factor.

5.4.2 Source Characterization

5.4.2.1 Source areas

Figure 5.84 shows the features of WAG 6 relevant for air modeling, including the surface waters and the waste areas.

Tritium. Tritium emissions at the WAG 6 site were assumed to originate primarily from surface water. Major sources consisted of the outfall of the 49-Trench area French Drain, the EWB, and streams. The French Drain collected groundwater in the central waste disposal area. This water flowed underground and was discharged to the surface where it flooded the ground. Except for stream FB, all streams were assumed to contain water only during and immediately (approximately 2 days) following rain events. Therefore, tritium was not continually emitted from these intermittent sources. Assumed perennial sources, including the EWB, French Drain, and stream FB, were assumed to have a constant emission rate throughout the year.

For modeling purposes, all sources were estimated as area sources. Because area sources for the programs used are, by definition, square in shape, source geometry requires each source to be approximated by a series of area sources.

The rectangular shapes of the EWB and French Drain outlet were easily characterized by square area sources. The outlet of the French Drain was modeled as a 50-ft by 50-ft area source. The waste basin, with an approximate surface area of 22,500 ft^2, was modeled by three adjacent area sources. The widths of these areas were 100 ft, 100 ft, and 50 ft for a total area of 25,000 ft^2. Overestimation of the EWB area resulted in slightly conservative predicted tritium concentrations.

Streams were more difficult to model because they are several hundred feet long and only a few feet wide and were not easily represented by a single area source. Several adjoining 100-ft by 100-ft area sources represented each stream. Streams were segmented such that each area source contained one segment. Emissions were calculated based on the characteristics of the stream segment. This approximation maintained the spatial variability of stream emissions.

VOCs. VOC emissions at the WAG 6 site were assumed to originate primarily from the SLB waste areas. SLB waste areas were considered as 12 separate areas. One area of migratory soil contamination was also considered. The caps on the landfills were modeled as area sources of 100 ft by 100 ft. VOCs considered are listed in Sect. 5.4.2.2.

5.4.2.2 Emissions

5.4.2.2.1 Tritium

Emission rates were developed for four periods which were specified by the risk assessment. The first was based on surface water sampling data taken in 1989. The other three were based on time average emission rates for the periods 1990 to 2019, 1990 to 2059, and 2100 to 2129.

Air emissions for 1989 were estimated from sampled tritium concentrations in surface water. Samples were available at each of the twelve sampling points. Figure 5.85 shows the sampling point locations. Sampling events were identified as:

- High water table
- Base flow—low water table
- Storm flow I—high water table
- Storm flow II—low water table

Table 5.34 contains tritium sampling data. The highest sampled concentrations were used in the analysis for conservatism.

Air emissions were estimated assuming that the dissolved tritium evaporated at the same rate as water. Assuming that one gram of tritium emitted 9600 Ci of radiation, the mass of tritium emitted to the atmosphere was obtained by multiplying the water concentration (pCi/L), the evaporated water volume (m^3), and the conversion rate of 9600 Ci/g (Jacobs 1968).

The volume of evaporated water from each stream was equal to the product of the source surface area and the annual evaporation rate. Evaporation rates for perennial sources were calculated assuming conservatively that 39 in. of water evaporated per year. Intermittent source evaporation rates were based on the number of days that the stream contained water. Assuming that streams flowed for 2 days after a rainfall and assuming an average of 128 days of rain per year, the evaporation rate was assumed to be 28 in. per year.

Stream surface areas, tritium concentrations, and emission rates are summarized below. Table 5.35 gives the predicted emission rates in 1989 and the parameters used for calculating these emissions for 51 source areas. The source areas are shown in Fig. 5.86.

Tritium concentrations were similarly calculated for the other three time periods. However, surface water concentration data for these periods were obtained from the modeling presented in Sect. 5.3. Modeling results gave tritium concentrations at one or several locations in each stream. Table 5.36 lists the surface water modeling data, stream lengths, and area sources used in the air modeling. Table 5.37 lists the tritium emissions for each source during each time period.

5.4.2.2.2 VOCs

Air emissions for VOCs were estimated using the landfill emissions estimation model CHEMDAT7.

Description of CHEMDAT7 model. The CHEMDAT7 model (EPA 1989a) was developed by EPA's Office of Air Quality Planning and Standards as part of an effort to develop an air emissions model for hazardous waste treatment, storage, and disposal facilities

(TSDF). The model primarily considers emissions of VOCs which escape into the atmospheric environment from typical TSDF sources such as surface impoundments, land treatment facilities, landfills, wastepiles, and wastewater treatment plant effluents. VOC emissions can be simulated in the model for a variety of mechanisms including volatilization, biological decomposition, adsorption, photochemical reaction, and hydrolysis.

The model includes an extensive chemical data base of more than 200 compounds and their associated chemical properties. The CHEMDAT7 model is implemented as a Lotus (TM) 1-2-3 spreadsheet allowing great user flexibility including many default values and the ability to utilize actual field data. The principal strength of the model lies in its ability to estimate emission levels comparable to actual field measurements from a limited amount of information. Additionally, estimated emissions can be adjusted for the age of the source months or years into the future.

CHEMDAT7 model parameters. The CHEMDAT7 model as applied in this analysis was confined to the "closed landfill" model. This section of the model is designed to estimate emissions from dilute aqueous sources contained in a closed (no longer active) landfill generally including an earthen layer or cap covering the landfill. CHEMDAT7 provides the ability to estimate emissions from the cap covering the landfill.

Ideally, CHEMDAT7 is designed to use as much information as is known about the potential source. For this detailed screening analysis, little source-specific information was available. Known or assumed data values used in the model included the following:

- Surface area of each landfill.
- Thickness of the covering or cap, assumed to be 1 ft (30.48 cm) and to be uniform for all landfills analyzed in each case.
- Cap air and total porosity values (0.1 and 0.5, respectively).
- The air porosity fraction of fixed waste (0.5 for all cases).
- Air and landfill temperatures and pressures (using default values of 15° C for temperature and 1013 mb for pressure both within and outside the landfill).
- Weight fractions of oil, water, and VOCs (assumed to be 0.0, 0.999999999, and 0.000000001, respectively). The VOCs were assumed to be equal to 0.000000001; and per model instructions, the water plus VOC values should equal one.
- Waste depth. Uniformly, all waste depths were assumed to be 15 ft (457 cm), which represents a reasonable approximation of the average depth of waste known to be present at WAG 6.

As stated above, site-specific data detailing amounts and varieties of emissions for each landfill site were unavailable for this analysis. However, a list of chemical constituents of concern for the air pathways analysis was generated based on results of trench leachate sampling reported by Solomon et al. (1988) and Ashwood and Spalding (1990). This list includes the following VOCs:

- Acetone
- Acrylamide
- Benzene
- Benzoic acid
- Chloroethane
- Chloroform
- Cresol (-o)
- Cresol (-p)
- 1,1 Dichloroethane
- 1,2 Dichloroethane
- 2,4 Dimethylphenol
- Ethylbenzene
- Methyl ethyl ketone
- Methylene chloride
- Napthalene
- Phenol
- 1,1,2,2 Tetrachloroethane
- Tetrachloroethene
- Toluene
- Trichloroethene
- Xylene (total)

The chemicals present in a landfill area were assumed to be the same as those detected in trench leachate samples from that area. If no trench leachate results were available for a landfill area, the results of trench leachate samples from the nearest adjoining landfill area were used.

CHEMDAT7 emissions. Base line emissions were calculated by using the base line parameters for CHEMDAT7. The emission rates for the capped landfills were divided by the area of each landfill. Table 5.38 lists the 89 sources as input to ISCLT and their emissions. These sources as modeled with ISCLT are shown in Fig. 5.86 for the base line emission sources.

5.4.3 Dispersion Modeling

5.4.3.1 Description of ISCLT model

The ISCLT model developed by EPA is a Gaussian dispersion model designed to calculate average concentration or total deposition for contaminants produced by emissions from multiple stack, volume, and area sources. The concentration or total deposition values can be calculated on a seasonal or annual basis for an unlimited number of sources. The program is capable of producing the seasonal and/or annual results for each individual source input as well as the combined results from multiple groups of selected sources. Program calculations are performed for an input set of receptor coordinates defining a fixed receptor grid system and/or for discrete (arbitrarily placed) receptor points. All points are relative to a user-defined grid origin (usually X=0, Y=0) in either a Cartesian or a polar coordinate system.

5.4.3.2 ISCLT model parameters

Input data for the ISCLT model consist of four categories: meteorological data, source data, receptor data, and program control parameters.

Meteorological data required for ISCLT consist of an annual joint frequency table, mixing height data, and temperature data. A joint frequency table is a matrix containing the frequencies of occurrence of 6 wind speed classes, 16 wind direction sectors, and 6 atmospheric stability categories. Mixing height data are required for each wind speed, stability category, and season. Temperature data are required for each stability category and season.

The program accepts three types of sources: point, area, and volume. Source data inputs include the location (in relation to a user-defined origin), elevation, and emission rate. For area sources, the area side length and effective emission height above ground level are also required. The source areas are shown in Fig. 5.87.

ISCLT receptor data consist of grid and discrete receptors. Grid receptors, spaced along a 25-m, Cartesian coordinate grid, are used to determine on-site concentrations. Discrete receptors are placed along the site boundary to determine concentrations at the site fenceline.

Program control parameters and options allow the user to choose one or more of the model options. These options define the types of sources, receptors, and output to be used in each run; they also allow the user to change model default values.

5.4.3.3 Predicted annual concentrations

Annual concentrations of tritium and VOCs were calculated for both boundary and on-WAG receptors. The boundary was delineated by receptors beginning at the southwest site border (177 m, 9 m) and continuing clockwise, approximately 25 m apart, to the northern border (347 m, 920 m). Receptors were not placed along the southern and eastern boundaries because these areas border other WAGs. On-WAG receptors were placed at 25-m intervals on a 25-by-40 grid ranging from coordinate (0 m, 0 m) at the southwest corner to coordinate (600 m, 975 m) at the northeast corner.

Tritium results. Annual tritium air pathway concentrations are shown in Figs. 5.88 through 5.91 for the periods 1989 (present), 1990 to 2019, 1990 to 2059, and 2100 to 2159, respectively. In each case, the maximum concentrations occurred near the French Drain. All other sources, with the exception of the EWB, had minimal impact on ambient concentrations. The predominant winds in the area caused concentrations to follow a southwest-northeast direction. Table 5.39 lists the maximum concentrations for both boundary and on-WAG receptors for all four time periods. Emissions based on 1989 sampling data resulted in the highest concentrations with a maximum of 64,670 pg/m^3 occurring a few meters east of the French Drain. Due to the radioactive decay of tritium, concentrations decreased for the later time periods.

VOC results. Annual average total VOC concentrations are shown in Fig. 5.92. The figure shows that the predominant winds for the WAG 6 area caused concentrations to generally elongate along a southwest-northeast axis. Table 5.39 lists the maximum concentration values for both boundary and on-WAG receptors. The maximum concentration of 435.4 pg/m^3 occurred in the 49-trench area.

5.4.4 Summary and Conclusions

5.4.4.1 Emission rates

Tritium. Tritium emission rates were calculated using conservative assumptions and were derived from 1989 water sampling data for 12 surface water sampling points located throughout the site. Each point was sampled up to 4 times. The sampling events are listed in Sect. 5.4.2.2. Table 5.34 lists the concentrations at each sampling location and identifies the samples used in this analysis.

Air emissions for the intermittent streams were calculated using the highest storm event measurement. This value was chosen to represent these streams because normally they have flow only during wet weather periods. Perennial stream (FB) emission calculations considered each of the four samples. As a conservative estimate, the highest of the four measurements was used to characterize each stream.

Each stream was sampled in only one or two locations. For streams sampled in only one location the emission rates were assumed constant throughout the length of the stream. Streams sampled at two locations, such as FB and FAB, were characterized as homogeneous upstream and downstream of the sampling points and heterogeneous in between. Upstream emissions were derived from measurements taken at the upstream sampling point, and downstream emissions were derived from measurements taken at the downstream sampling point. The section of stream between the sampling points was characterized by a concentration gradient with emissions ranging between the two measurements. Actual stream emissions were unlikely to be either homogeneous or gradational. Instead, the streams probably exhibited a more patchy distribution of tritium, depending on the location of underground tritium sources. However, the surface water measurements were the only tritium data available; therefore, the above generalizations were necessary to characterize the streams.

Tritium emissions for the periods 1990 to 2019, 1990 to 2059, and 2100 to 2129 were based on surface water modeling results; therefore, further uncertainty was introduced into the emission calculation.

VOCs. Results of the ISCLT model run for VOCs were used to calculate a 95 percentile on the median upper bound air concentration for both the boundary and on-WAG receptors. Since the modeled concentrations of total VOCs were not normally distributed, nonparametric statistical procedures, similar to those used for on-WAG data (Sect. 4.1.2), were employed for these calculations. VOC emissions for all exposure periods were assumed to be the same.

5.4.4.2 Transport

Area sources were used in both tritium and VOC modeling. Area sources were easily modeled by ISCLT, but line sources were not. To permit simulation of line sources with ISCLT, they were approximated as area sources. Although the total emissions released to the atmosphere were identical for both actual and simulated cases, the areas from which they were emitted were different. By using an area source to represent the streams or landfills (narrow or irregularly shaped source), emissions were spread over a comparatively large area. These approximations should not ordinarily cause noticeable differences in downwind concentrations far from the source, but some differences are likely at short distances. However, in this analysis, emissions from stream sources were practically negligible when compared to those from the French Drain and EWB; therefore, the modeling results were not expected to be affected by using area sources in approximating the emissions from streams.

Uncertainty existed with respect to the predicted receptor concentrations. The manual for the ISCLT model indicates that concentrations predicted at receptor distances less than 1.5 area source side lengths from an area source should be used with caution. Many of the receptors used in the on-WAG grid analysis posed this source-receptor distance concern for some of the sources. Specifically, the ISCLT model will calculate a zero concentration for a source if the source is located within 1.5 side lengths of the receptor. However, concentrations for other sources at greater distances from the receptor will still be calculated. For receptors located near this source-receptor distance limit yet beyond the limit, the model may overpredict concentrations.

5.5 CONCLUSIONS

The conclusions of the fate and transport analysis, which included the modeling of water and air in WAG 6, are summarized below.

- Water was the major transport mechanism for off-WAG migration of contaminants, and most of the radionuclide flux out of WAG 6 was expected to continue to occur via surface water.

- Cobalt-60, tritium, strontium-90, and cesium-137 were in most of the wells in WAG 6. Thorium-232 was in Area M wells.

- The first occurrences of europium and uranium in groundwater were predicted in years 1998 and 2025, respectively.

- Peak future groundwater concentrations were predicted to be within two orders of magnitude of the present-day concentrations.

- Air modeling, which was performed conservatively, indicated that air pathway contributed negligible amounts of exposure-point concentrations of contaminants on-WAG and off-WAG.

- Radioactive decay was the major mechanism for the depletion of source inventories of the radionuclides with short half-lives.

- Source inventories of cobalt-60 and tritium were predicted to be depleted in about 100 years.

- Source inventories of strontium-90 in areas A, B, C, D, F, and P were predicted to remain significant in the year 2120.

- Source inventories of cesium-137 in areas A, B, J, P, and Q were predicted to still be significant in the year 2120.

- Europium was present in source areas A, B, K, and P. By the year 2120, while europium-152 inventory was predicted to remain in significant quantities, europium-154 was expected to be depleted in all source areas except Area P.

- Radionuclides with long half-lives, such as uranium and thorium-232, were predicted to remain in the source areas far beyond the year 2120.

Section 5 Tables

Table 5.1. Model input parameters

TITLE 1:	Job title
TITLE 2:	Job title
TITLE 3:	Job title
NIND:	Number of Monte Carlo variables
NPAR:	Number of distribution parameters
MITER:	Number of Monte Carlo simulations
ICLB:	Maximum number of iterations for calibration
IDO:	Number of transects where well data are generated
IXO:	Seed number for the random number generator
NYRS:	Years of simulation
IFIT:	calibration option, yes = 1, no = 0
NNCLD:	Constituent simulated
TAREA:	Trench area in m^2
H:	Trench depth in m
FINT:	Initial value of f_1
FFIN:	Final value of f_1
TINT:	Initial time of f_1
TFIN:	Final time of f_1
ARTEN:	Trench inventory in curies
IYRC:	Age of source inventory in years
SCONC:	Calibration value of sample well concentration, pCi/L
DECAY:	Decay rate (1/yr)
XKDV:	K_d in vadose zone, m^3/kg
XKDH:	K_d in saturated zone, m^3/kg
XKDT:	K_d in trench/auger hole, m^3/kg
RPERC:	Recharge rate, m/yr
DVERT:	Vadose zone depth, m
DHORZ:	Distance to receptor, m
DHOREX:	Horizontal distance between transects, m
EPORV:	Effective porosity, vadose zone
EPORA:	Effective porosity, saturated zone
PORT:	Total porosity, trench

Table 5.1. (continued)

PORV:	Total porosity, vadose zone
SSAT:	Degree of saturation, vadose zone
BDENS:	Bulk density, kg/m^3
DISP:	Dispersion length, m
AQTHK:	Saturated zone thickness, m
PORA:	Saturated zone porosity
GWV:	Groundwater velocity, m/year
ADISP:	Dispersion angle in radians
BAREA:	Basin area, m^2
WAREA:	Tributary watershed area, m^2
NSR:	Number of storms per year
VR:	Storm volumes in inches (from 1st to NSR storm)
NCSR:	Number of sampled storms
WTRI:	Watershed inventory, Ci (watershed at sampling site)
ICAL:	Runoff curve fitting parameter
VRC:	Calibration storm volumes in inches
CBAR:	Calibration storm concentrations
TTR:	Ratio of travel time to age of inventory
TXKDV:	Minimum value of vadose zone K_d
TXKDH:	Minimum value of saturated zone K_d
CRMN:	Lower limit, ratio of simulated and recorded well concentrations
CRMX:	Upper limit, ratio of simulated and recorded well concentrations
NEW:	Flag to end input for one contaminant (1=next contaminant simulation starts)
S1:	Standard deviation for calibrated vadose zone K_d
S2:	Standard deviation for calibrated saturated zone K_d
S3:	Standard deviation for calibrated f1
IPARM:	Parameter number (see the list of Monte Carlo parameters)
DISTRIB:	Name of the distribution: supported names: NORMAL, LOGNORMAL, UNIFORM, LOGUNIFORM
SINF:	1st and 2nd parameters of the selected distribution

Table 5.2. List of Monte Carlo parameters

Parameter number		Description of parameters
1	TAREA:	Trench area in m^2
2	H:	Trench depth in m
3	FINT:	Initial value of f_1
4	FFIN:	Final value of f_1
5	ATREN:	Initial trench inventory, Ci
6	DECAY:	Decay rate, 1/yr
7	XKDV:	K_d in vadose zone, m^3/kg
8	XKDH:	K_d in saturated zone, m^3/kg
9	XKDT:	K_d in trench/auger hole, m^3/kg
10	RPERC:	Recharge rate, m/yr
11	DVERT:	Vadose zone depth, m
12	DHORZ:	Distance to receptor, m
13	EPORV:	Effective porosity, vadose zone
14	EPORA:	Effective porosity, saturated zone
15	PORT:	Total porosity, trench
16	PORV:	Total porosity, vadose zone
17	SSAT:	Degree of saturation, vadose zone
18	BDENS:	Bulk density, kg/m^3
19	DISP:	Dispersion length, m
20	AQTHK:	Saturated zone thickness, m
21	PORA:	Saturated zone porosity
22	GWV:	Groundwater velocity, m/yr
23	ADISP:	Dispersion angle in radians
24	BAREA:	Basin area in m^2
25	QBASE:	Base flow in m^3/yr
26	IYRC:	Age of source inventory, yr
27	SCONC:	Calibration value of well concentration, pCi/L
28	WTRI:	Watershed inventory, Ci

Table 5.3. WAG 6 inventory of modeled radionuclides (Ci)

Waste area	H-3	Co-60	Sr-90	Cs-137	Eu-152	Eu-154	Th-233	U-232	U-238
A	91.6	3357	416	1646	4175	1558	0.001	0	0.015
B	32.8	1305	284	771	16,270	22,305	0	0	0.001
C	1071	15.2	105	34.1	0.154	0.003	0	4.8	0
D	13.3	3521	68	53.3	0.146	0.014	0	2.8	0
E	50.4	16.2	3.2	12.5	0.153	0.020	0.093	4.6	7.7
F	45.5	14.5	93.3	47.1	0.431	0.009	0	4.1	0
G	13.0	4.13	26.7	13.5	0.123	0.003	0	1.2	0
H	10.0	82.2	45.0	106	6.7	2.4	0	0.004	0.010
I	4.9	1.9	20.4	10.4	4.1	2.1	0.008	0.354	0.014
J	5.0	250	43.7	702	0	0	0.001	0	0.017
K	0.35	20.9	6.8	18.8	26,801	48,701	0	0	0
L	2488	7.2	38.0	159	2.0	0	0.849	1.1	1.72
M	618	198	39	152	1.9	0.250	1.14	56.1	0.04
N	452	645	25.0	35.4	2.9	0.756	16.2	25.0	16.3
O	284	391	19.1	115	0	0.030	1.0	0.022	1.12
P	2700	21,064	129	5822	250,865	187,368	6003	0	0.024
Q	6.2	377	50.0	265	26.4	10.9	0	0	0.003
R	19.0	1001	21.0	41.0	0	0	0.100	0.004	0.510
S	30	0.005	0.001	0.011	0	0	0	0	0
T	34.5	0.001	0.005	0.001	0.001	0	0	0	0.126
U	61.6	0.004	0.012	0.002	0	0	0	0	0
V	50.4	16.2	3.2	12.5	0.153	0.020	0.093	4.6	7.7
W	3.5	0.173	0.021	0.021	0	0	0	0	0.016
X	177	25.7	11.1	16.6	0.022	0.003	0	0	0.046
Y	805	6436	34.4	40.6	2.4	0.730	0.593	124.7	209
Z	9.5	0.855	10.6	20.2	4.2	4.0	0.101	0.003	0.0002

Table 5.4 WAG 6 calibration well data for radionuclides^{a,b}

Well	Sampling 1	2	3	4	5	6	7	8	9
Cobalt-60 (pCi/L)									
745	2.703 U	0.2703	10.812	2.4327					
831	2.703 U		2.703 U	1.0812					2.703 U
832	2.703 U	0.5406	1.6218	2.703					2.703 U
833	2.703 U	2.703 U	2.1624	1.8921					2.703 U
835	2.703 U	8.109	2.703	2.703 U					2.703 U
836	2.703 U	2.703 U	2.703	2.703 U					2.703 U
837			2.703 U	2.703 U					2.703 U
838	0.5406	2.703 U	2.703 U	0.2703					2.703 U
839	0.8109	2.1624	18.921	0.2703					2.703 U
840	2.1624	4.0545	2.703	2.703 U					2.703 U
841	4.8654	2.703 U	1.0812	3.2436					2.703 U/120
842	351.39	324.36	267.597	297.33					2.703 U
843	11.0823	11.3526	51.357	11.3526					17.2992 U
844	1.3515	1.3515	16.218	2.4327					2.703 U
845	2.703 U	7.0278	3.2436						
846	2.703 U	0.8109	2.703 U	0.5406					
847	1.6218	2.703 U	5.406	1.3515					2.703 U
848	0.2703	2.4327	2.703 U						
849	4.0545	0.2703	2.703 U						
850	2.703 U	4.3248	16.218						
851	2.703 U	0.2703	2.703						
852	0.2703	2.9733	2.703						
853	0.8109	2.703 U	2.703 U						
854	2.1624	1.0812	2.703 U						
855	2.703 U	2.703 U	2.703	2.703 U					4.5951 U
856	2.703 U	2.1624	2.703 U	1.8921					2.703 U
857	2.703 U	0.5406	13.515	1.8921					2.703 U
858	2.703 U	2.703 U	3.7842	2.1624					2.703 U
859			35.139	2.703 U					2.703 U
860	2.703 U	2.703 U	29.733	2.703 U					2.703 U
1225				24		16.4	87.4	19.7	
1227									
1228									
1229									
1231									
1233									
1234									
1234A									
1236									
1237									
1238									
1239									
1240									
1241									
1242									
1243									1880
1244									

Table 5.4. (continued)

	Sampling Event								
Well	1	2	3	4	5	6	7	8	9
Cesium-137 (pCi/L)									
745	1.0812	0.2703	2.703 U	1.3515					
831	2.703 U		2.703 U	2.703 U					2.703 U
832	2.703 U	2.703 U	0.2703	2.4327					2.703 U
833	2.703 U	1.72992	0.5406	0.8109					2.703 U
835	2.703 U	5.9466	0.8109	2.703 U					2.703 U
836	0.2703	2.703	21.0834	2.703 U					2.703 U
837			2.703	2.703 U					2.703 U
838	2.1624	2.703 U	10.812	1.0812					2.703 U
839	0.2703	2.703 U	2.703 U	0.97308					2.703 U
840	2.9733	1.3515	8.109	0.8109					2.703 U
841	1.3515	0.8109	2.4327	5.1357					2.703/20 U
842	0.2703	2.703 U	13.515	2.703 U					2.703/20 U
843	1.6218	2.703 U	2.703	0.2703					2.703 U
844	1.6218	2.703 U	2.703 U	2.703 U					2.703 U
845	2.703 U	2.703 U	5.1357	20 U			20 U		
846	18.3804	2.703 U	2.9733	1.8921					
847	2.1624	4.0545	13.515	0.5406					2.703
848	5.9466	0.8109	0.2703	20 U			20 U		
849	0.5406	0.2703	8.109	20 U			20 U		
850	2.703 U	4.5951	8.109	20 U			20 U		
851	5.406	1.6218	2.9733	20 U	20 U		20 U		
852	2.703 U	1.3515	2.1624	20 U	20 U		20 U		
853	2.703 U	2.703 U	0.8109	20 U	20 U		20 U		
854	3.7842	2.703 U	2.703	20 U	20 U		20 U		
855	2.703 U	2.703	1.3515	1.3515					2.703U/20
856	7.8387	2.703 U	5.406	0.2703					2.703 U
857	2.703 U	2.4327	2.703 U	0.5406					2.703 U
858	3.7842	2.703 U	2.703 U	0.2703					2.703 U
859			2.703	2.1624					2.703 U
860	11.8932	2.703 U	8.109	0.5406					2.703 U
1225				20 U	20 U	20 U	20 U	20 U	
1227				20 U	20 U	20 U			
1228				20 U	20 U	20 U	20 U	20 U	20 U
1229				20 U	20 U	20 U			20 U
1231				20 U	20 U		20 U	20 U	
1233				20 U	20 U	20 U	20 U	20 U	
1234									
1234A									20 U
1236									20 U
1237									20 U
1239									20 U
1240									20 U
1241									20 U
1242									20 U
1243									20 U
1244									20 U

Table 5.4. (continued)

Well	Sampling event 1	2	3	4	5	6	7	8	9
Tritium (pCi/L)									
745	1270.41 U	35139	1270.41 U	1270.41 U					
831	594.66		1270.41 U	1270.41 U					864.96
832	108.12	973.08	1270.41 U	351.39					1270.41
833	1270.41 U	18650.7	1270.41 U	513.57					1270.41
835	24867.6	32436	20272.5	26219.1					18650.7
836	3513.9	13244.7	837.93	2243.49					1648.83
837			1892.1	1486.65					2432.7
838	14325.9	1270.41 U	11352.6	7027.8					24867.6
839	29733	8109	23786.4	26489.4					17839.8
840	21624	21353.7	11352.6	3243.6					4054.5
841	270300	243270	172992	216240					143259/16300
842	648720	675750	567630	459510					183804/21900
843	810900	919020	1243380	756840					324360
844	72981	75684	59466	64872					54060
845	26489.4	17569.5	40545	32000			230000		
846	270.3	108.12	1270.41 U	1946.16					
847	20002.2	89199	83793	81090					72981
848	4324800	4324800	2000220				4300000		
849	2459730	2973300	1027140	1500000	1600000		2600000		
850	43248	37842	29733	30600			26000		
851	43248	3784.2	40545	43900	50000		49000		
852	59466	59466	51357	56900	44000		7000		
853	3243.6	1270.41 U	1270.41 U	2660	2400		2600		
854	183804	297330	151368	185000	58000		33000		
855	1162.29	54.06	1270.41 U	378.42					864.96/500 U
856	324.36	1270.41 U	81.09	432.48					1270.41 U
857	810.9	432.48	243.27	108.12					1270.41 U
858	270.3	135.15	1270.41 U	1270.41 U					1270.41 U
859			405.45	946.05					1243.38 U
860	1270.41	1270.41 U	29.73	1270.41 U					1270.41 U
1225				1900000	1600000	1590000	1600000	879000	
1227				2600	2100	2360			
1228				9261	8800	7300	10000	10900	
1229				14100	15000	15800	16000	16400	21200
1231				51000	83000		54000	65200	
1233				770000	440000	246000	760000	561000	
1234									500 U
1234A									500 U
1236									500 U
1237									500 U
1238									500 U
1239									500 U
1240									801
1241									2800
1242									472000
1243									794000
1244									51200
1245									500 U

Table 5.4. (continued)

	Sampling event								
Well	1	2	3	4	5	6	7	8	9
Strontium-90 (pCi/L)									
745	1.3515	0.8109	4.0545 U	4.0545 U					
831	4.5951		2.35161	2.9733					5.1357
832	3.2436	3.2436	1.18932	4.0545 U					4.0545 U
833	1.6218	6.2169	0.83793	0.2703					4.0545 U
835	0.2703	3.2436	4.0545 U	4.0545 U					4.0545 U
836	2.9733	0.8109	0.29733	3.2436					4.0545 U
837			4.0545 U	0.2703					4.0545 U
838	4.0545 U	0.2703	6.2169	4.0545 U					4.0545 U
839	4.0545 U	4.0545 U	0.02703	0.2703					4.0545 U
840	5.1357	4.5951	3.5139	4.0545 U					4.0545 U
841	4.8654	4.8654	0.02703	1.3515					4.0545/5 U
842	4.0545	4.0545 U	0.24327	0.72981					4.0545 U/5.98
843	1.0812	4.0545 U	1.64883	2.64894					4.0545 U
844	3.5139	6.7575	0.56763	4.0545 U					4.0545 U
845	4.0545 U	0.2703	1.40556	5 U			5U		
846	2.9733	15.1368	0.13515	4.0545 U					
847	4.0545 U	2.1624	1.94616	0.5406					
848	51.357	11.3526	5.6763	30.2			40		
849	2.703	2.4327	1.02714	5 U	5U		5U		
850	10.5417	3.2436	1.43259	5 U			5U		
851	4.3248	4.0545 U	4.0545 U	5 U	5U		5U		
852	2.703	4.0545 U	0.59466	5 U	5U		5U		
853	3.5139	0.5406	4.0545 U	5 U	5U		5U		
854	3.5139	0.5406	4.0545 U	5 U	5U		5U		
855	2.51379	1.8921	0.40545	4.0545 U					4.0545/5 U
856	0.43248	3.2436	1.8921	4.0545 U					4.0545 U
857	3.7842	3.5139 U	0.67575	0.18921					4.0545 U
858	0.45951	10.5417	1.70289	1.6218					4.0545 U
859			4.0545 U	8.109					4.0545 U
860	4.5951	1.3515	4.0545 U	4.0545 U					4.0545 U
1225				1700	1900	3031	5400	4410	
1227				5 U	5U	5U			
1228				5 U	5U	5U	5U	5U	5U
1229				5 U	5U	5U	5U	5U	5U
1231				5 U			5U	5U	5U
1233				5 U	5U	5U	5U	5U	
1234									
1234A									5U
1236									5U
1237									8.3
1238									5U
1239									5U
1240									5U
1241									5U
1242									5U
1243									5U
1244									5U

Table 5.4. (continued)

Well	Sampling event 1	2	3	4	5	6	7	8	9
Thorium-232 (pCi/L)									
845				1 U			1 U		
848				1 U			1 U		
849					1 U		1 U		
850				1 U			1 U		
851				1 U	1 U				
852				1 U	1 U		1 U		
853				1 U	1 U		1 U		
854				1 U	1 U		1 U		
1225					1 U	2.26	3.01	1 U	
1227				1.8	1 U	1 U			
1228				1 U	1.5	2.04	1 U	1 U	
1229				1 U	1.2	1.55	1 U	1 U	
1231				6	1 U		1 U	1 U	
1233				4.42	1 U	2.39	2.2	1 U	5.42
1234									1.5
1234A									1 U
1236									1.19
1237									1 U
1238									1 U
1239									6.47
1240									1.83
1241									9.13
1242									1 U
1243									1 U
1244									1 U
Uranium 238 (pCi/L)									
845				1			1 U		
848				1			1 U		
849					1		1 U		
850				1 U			1 U		
851				1 U	1 U				
852				1 U	1 U		1 U		
853				1 U	1 U		1 U		
854				1 U	1 U		1 U		
1225				1 U	1 U	1 U	1 U	1 U	
1227				1 U	1 U	1 U			
1228				1 U	1	1 U	1 U	1 U	
1229				1.16	1.2	1 U	1 U	1 U	1.91
1231				1.97	1 U		1 U	1 U	
1233				1.38	1.1	1.13	1 U	1 U	
1234A									1 U
1236									1 U
1237									1.24
1238									1 U
1239									5
1240									1 U
1241									7.83
1242									1.58
1243									1 U
1244									1 U

[a] Letter "u" means undetected.

[b] Where two concentration values appear for a particular sampling event at an individual well, the first value reflects analysis by ORNL and the second analysis under the RFI program.

Table 5.5. Parameters common to all waste areas in WAG 6

Parameter[a]	Nominal value
H	4.6 m
RPERC	0.19 m/yr
EPROV	0.02
EPORA	0.02
PORT	0.5
PORV	0.1
SSAT	0.5
BDENS	1300 kg/m^3
DISP	5 m
AQTHK	4.6 m
PORA	0.1
GWV	Varies
ADISP	0.3 radians

[a] Defined in Sect. 5.3.2.

Table 5.6. Physical parameters TAREA, DHORZ, DVERT, and GWV[a]

Areas	Well no.	TAREA m^2	DHORZ m	DVERT m	GWV m/yr
A	1242	1481	53	3.0	75
B	842	717	58.6	3.0	75
C	1225	1292	35	0	20
D		2084	50	3.0	10
E	847	1868	9	4.6	20
F	1241	2093	65	6.1	27
G	843	557	55	6.1	7
H		1430	50	0	29
I		757	50	1.5	29
J		2424	50	0	90
K		441	50	0	90
L	848	5377	44	0	57
M	849	5731	58	0	29
N	853	3047	56	0	62
O	854	2363	54	0	62
P	1243	1791	63	1.5	62
Q	840	1438	52	1.5	76
R		1397	50	0	62
S	851	1315	34	0	44
T	852	1468	50	0	44
U	850	2784	48	0	34
V	838	1357	55	0	29
W		3702	50	0	22
X	835	8417	57	0	18
Y	837	3580	58	3.0	56
Z		688	50	3.5	56

[a] Parameters defined in Sect. 5.3.2.

Table 5.7. Radionuclide decay rates

Radionuclide	Half-life (yr)
Co-60	5.3
Cs-137	30.2
Eu-152	13.6
Eu-154	8.8
H-3	12.3
Sr-90	28.6
Th-232	1.4E5
U-233	1.6E5
U-238	4.4E9

Table 5.8. Summary model results—cobalt-60

				Inventory		Breakthrough concentration		Peak concentration	
Area	Well no.	f_1[a]	f_3[b]	Ci	Year	pCi/L	Year[c]	pCi/L	Year[d]
A	1242	1.0E-5	1.0E-3	3357	1978	7.6		17.6	2024
B	842	4.0E-4	7.0E-4	1305	1978	1030			
C	1225	1.5E-2	7.0E-4	15.2	1977	37.3			
D		1.0E-3	1.0E-3	3521	1982	590			
E	867	1.0E-3	1.5E-3	16.2	1973	9.6			
F	1241	3.5E-3	3.0E-4	14.5	1974	7.4			
G	843	1.2E-2	1.0E-4	4.13	1975	105			
H		3.0E-3	4.0E-4	82.2	1988	57.6	1998		
I		3.0E-2	4.0E-4	1.9	1988	8.6	1999		
J		3.0E-3	4.0E-4	250	1987	96.6	2024		
K		3.0E-3	4.0E-4	20.9	1981	7.56			
L	848	5.0E-3	5.0E-4	7.2	1982	2.0			
M	849	2.6E-4	5.0E-4	198	1975	2.7			
N	853	1.0E-4	5.0E-4	645	1978	3.3	2024	3.3	2024
O	854	1.0E-4	5.0E-4	391	1978	3.2			
P	1243	1.5E-3	2.0E-4	21,064	1985	8210			
Q	840	3.6E-3	5.0E-4	377	1974	34.6			
R		1.0E-3	4.0E-4	1001	1983	82			
S	851			0.005	1979				
T	852			0.001	1978				
U	850			0.004					
V	838	1.0E-3	4.0E-4	16.2	1979	1.7			
W				0.17	1982				
X	835	1.7E-3	2.0E-4	25.7	1979	3.4			
Y	837	1.0E-5	1.5E-4	6436	1985	8.8		60	2024
Z				0.855	1988				

[a] Leaching fraction.
[b] Calibration factor accounting for retardation.
[c] Unless otherwise indicated, breakthrough year is 1989.
[d] Unless otherwise indicated, peak occurred in breakthrough year.

Table 5.9. Summary model results—tritium

			Inventory		Concentration in 1989	Peak concentration	
Area	Well no.	f_1[a]	Ci	Year	pCi/L	pCi/L	Year
A	1242	8.0E-3	92	1978	316,000	605,000	1979
B	842	4.0E-2	32	1978	656,000	1.37E+6	1979
C	1225	1.0E-3	1071	1977	2.8E+6	4.97E+6	1978
D		3.0E-2	13.2	1982	1.04E+6	1.36E+6	1987
E	867	1.5E-3	50.4	1973	114,000	3.79E+5	1973
F	1241	1.0E-3	45.5	1974	4700	35,000	2009
G	843	1.6E-2	13	1975	1.39E+6	2.317E+6	1983
H		3.0E-2	10	1988	843,000	8.43E+5	1989
I		3.0E-2	4.9	1988	542,000	5.42E+5	1990
J		3.0E-2	5	1987	116,000	1.32E+5	1987
K		3.0E-2	0.35	1981	10,000	1.67E+4	1981
L	848	5.0E-3	2488	1982	7.0E+6	1.11E+7	1982
M	849	1.0E-2	617	1975	3.4E+6	8.4E+6	1977
N	853	2.0E-5	456	1978	4000	7.03E+3	1979
O	854	2.0E-3	283	1978	292,000	4.4E+5	1978
P	1243	1.0E-3	2700	1985	1.8E+6	1.86E+5	1986
Q	840	1.0E-2	6.2	1974	20,000	5.50E+4	1974
R		3.0E-3	19	1983	49,500	7.09E+4	1983
S	851	3.0E-3	30	1979	58,200	1.83E+5	1979
T	852	3.0E-3	34.5	1978	97,000	1.78E+5	1979
U	850	1.0E-3	61.6	1980	53,100	8.84E+4	1981
V	838	3.0E-4	50.4	1979	17,900	3.11E+4	1980
W		3.0E-2	3.5	1982	157,000	2.26E+5	1984
X	835	2.0E-4	177	1979	36,500	5.71E+5	1982
Y	837	1.0E-4	804	1985	2700	2.55E+5	2018
Z		3.0E-2	9.5	1988	477,000	5.57E+5	1989

[a] Leaching fraction.

Table 5.10. Summary model results—strontium-90

				Inventory		Breakthrough concentration		Peak concentration	
Area	Well no.	f$_1$[a]	f$_3$[b]	Ci	Year	pCi/L	Year[c]	pCi/L	Year[d]
A	1242	5.0E-5	1.0E-03	416	1978	4.6		718	2052
B	842	5.0E-5	7.0E-6	284	1978	4.3		418	2052
C	1225	4.0E-2	7.0E-4	105	1977	5640			
D		2.0E-3	1.0E-3	68	1982	41	2055	117	2116
E	867	1.0E-3	1.5E-3	3.2	1973	4.3		17.2	2049
F	1241	1.5E-4	3.0E-4	93.3	1974	8.0		268	2032
G	843	4.0E-5	1.0E-4	26.7	1975	4.5		496	2053
H		3.0E-3	4.0E-4	45	1988	163	1998	419	2058
I		3.0E-2	4.-E-4	20.4	1988	561	2013		
J		3.0E-3	4.0E-4	43.7	1987	53.6	1990	130	2049
K		3.0E-3	4.0E-4	6.8	1981	13.7			
L	848	4.5E-3	5.0E-4	38	1982	59		75	2045
M	849	4.0E-4	5.0E-4	39	1975	7.3		100	2051
N	853	1.0E-3	5.0E-4	25	1978	8.5		46.2	2050
O	854	1.0E-3	5.0E-4	19.1	1978	8.6		39.4	2050
P	1243	3.0E-4	2.0E-4	129	1985	12.0		201	2048
Q	840	5.0E-4	5.0E-4	50	1974	9.3		79.2	2051
R		3.0E-3	4.0E-3	21	1983	22.8			
S	851			0.01	1979				
T	852			0.05	1978				
U	850			0.012	1980				
V	838	2.5E-3	4.0E-4	3.2	1979	7.5		15.8	2045
W				0.02	1982				
X	835	8.0E-4	2.0E-4	11	1979	6.2		41	2049
Y	837	4.0E-4	1.5E-4	34	1985	4.6		48.3	2049
Z		3.0E-2	4.0E-3	10.5	1988	62	2049		

[a] Leaching fraction.
[b] Calibration factor accounting for retardation.
[c] Unless otherwise indicated, breakthrough year is 1989.
[d] Unless otherwise indicated, peak occurred in breakthrough year.

Table 5.11. Summary model results—cesium-137

				Inventory		Breakthrough concentration		Peak concentration	
Area	Well no.	f_1[a]	f_3[b]	Ci	Year	pCi/L	Year[c]	pCi/L	Year[d]
A	1242	7.0E-4	1.0E-3	1646	1978	12.2			
B	842	5.0E-4	7.0E-4	771	1978	12.4		12.5	2045
C	1225	1.0E-1	7.0E-4	34.1	1977	23.7			
D		2.0E-3	4.0E-3	53	1982	0.62	2126	1.4	2187
E	867	6.6E-2	1.5E-3	12.5	1973	5.2			
F	1241	1.5E-1	3.0E-4	47.1	1974	20			
G	843	1.0E-2	1.0E-6	13.5	1975	2.9			
H		3.0E-3	4.0E-3	106	1988	0.7	2080	1.1	2140
I		3.0E-2	4.0E-3	10.4	1988	0.03	2359		
J		3.0E-3	4.0E-3	702	1987	3.7	2016	7.5	2077
K		3.0E-3	4.0E-3	18.8	1981	0.17	2010	0.36	2072
L	848	5.0E-2	5.0E-4	159	1982	11.5			
M	849	3.0E-2	5.0E-4	152	1975	11			
N	853	2.5E-1	5.0E-4	35	1978	12.7			
O	854	7.0E-2	5.0E-4	115	1978	13.3			
P	1243	3.5E-3	2.0E-4	5822	1985	37		92	2051
Q	840	1.0E-3	5.0E-4	265	1974	3.4			
R		3.0E-3	4.0E-3	41	1983	0.1	2025	0.24	2086
S	851			0.011	1979				
T	852			0.001	1978				
U	850			0.002	1980				
V	838	9.0E-2	4.0E-4	12.5	1979	4.4			
W				0.021	1982				
X	835	2.7E-1	2.0E-4	3.7	1979	3.6			
Y	837	4.4E-2	1.5E-4	40.5	1985	2.8			
Z		3.0E-2	4.0E-3	20.2	1988	0.12	2597		

[a] Leaching fraction.

[b] Calibration factor accounting for retardation.

[c] Unless otherwise indicated, breakthrough year is 1989.

[d] Unless otherwise indicated, peak occurred in breakthrough year.

Table 5.12. Summary model results—europium-152

				Inventory		Breakthrough concentration		Peak concentration	
Area	Well no.	f_1[a]	f_3[b]	Ci	Year	pCi/L	Year	pCi/L	Year
A	1242	3.0E-5	1.0E-3	4175	1978	7.7	1998	32.4	2033
B	842	3.0E-5	1.0E-3	16,270	1978	15.8	2000	85.2	2036
K		3.0E-3	4.0E-3	26,801	1981	0.17	2010	0.36	2072
P	1243	3.0E-5	1.0E-3	250,000	1985	163	2005	850	2040

[a] Leaching fraction.
[b] Calibration factor accounting for retardation.

Table 5.13. Summary model results—europium-154

Area	Well no.	f_1[a]	f_3[b]	Inventory		Breakthrough concentration		Peak concentration	
				Ci	Year	pCi/L	Year	pCi/L	Year
A	1242	3.0E-5	1.0E-3	1558	1978	0.85	1998	2.2	2028
B	842	3.0E-5	1.0E-3	22,300	1978	47.7	2000	85.4	2029
K		3.0E-5	1.0E-3	48,700	1981	48.8	1996	23.5	2021
P	1243	3.0E-5	1.0E-3	187,368	1985	321	2005	584	2036

[a] Leaching fraction.

[b] Calibration factor accounting for retardation.

Table 5.14. Summary model results—uranium-233

Area	Well no.	f_1[a]	f_3[b]	Inventory		Breakthrough concentration		Peak concentration	
				Ci	Year	pCi/L	Year	pCi/L	Year
C	1225	3.0E-3	4.0E-3	4.8	1977	7.6	2131	201	2290
D		3.0E-3	4.0E-3	2.8	1982	2.5	2219	61.4	2378
E	847	3.0E-3	4.0E-3	4.6	1973	7.3	2038	149	2197
F	1241	3.0E-3	4.0E-3	4.1	1974	2.5	2217	60.6	2376
G	843	3.0E-3	4.0E-3	1.2	1975	4.1	2671	100	2830
L	848	3.0E-3	4.0E-3	100	1982	19.7	2025	524	2184
M	849	3.0E-3	4.0E-3	56	1975	27	2150	780	2309
N	853	3.0E-3	4.0E-3	25	1978	8.3	2057	221	2216
V	838	3.0E-3	4.0E-3	4.6	1979	2.24	2064	59	2223
Y	837	3.0E-3	4.0E-3	124.7	1985	28.9	2091	695	2250

[a] Leaching fraction.

[b] Calibration factor accounting for retardation.

Table 5.15. Summary model results—uranium-238

Area	Well no.	f_1[a]	f_3[b]	Inventory		Breakthrough concentration		Peak concentration	
				Ci	Year	(pCi/L)	Year	pCi/L	Year
E	847	3.0E-3	4.0E-3	7.7	1973	19	2038	400	2197
L	848	3.0E-3	4.0E-3	1.72	1982	0.3	2025	8.7	2184
N	853	3.0E-3	4.0E-3	16.3	1978	4.9	2057	144	2216
O	854	3.0E-3	4.0E-3	1.12	1978	0.4	2055	8.2	2215
V	838	3.0E-3	4.0E-3	7.7	1979	3.5	2064	100	2223
Y	837	3.0E-3	4.0E-3	209	1985	84	2091	1240	2250

[a] Leaching fraction.

[b] Calibration factor accounting for retardation.

Table 5.16. Summary model results—thorium-232

Area	Well no.	f_1	f_3	Inventory		Breakthrough concentration		Peak concentration	
				Ci	Year	pCi/L	Year	pCi/L	Year
M	849	2.0E-4	4.0E-2	1.1	1975	0.4	1989	0.83	2148
N	853	3.0E-3	4.0E-3	16.2	1978	4.9	2057	144	2216
O	854	4.0E-2	1.0E-3	1.0	1978	4.2	1989	4.2	1989
P	1243	3.0E-3	4.0E-2	6000	1985	57.8	2948	1080	3107

[a] Leaching fraction.
[b] Calibration factor accounting for retardation.

Table 5.17. Predicted radionuclide inventories (Ci) in source areas

Area A

Year	Co-60	H-3	Sr-90	Cs-137	Eu-152	Eu-154
2020	16.6	6.8	168	768	581	65.6
2120			14.6	76.2	3.2	
Year of depletion	2042	2044	>2138	>2138	>2138	2074

Area B

Year	Co-60	H-3	Sr-90	Cs-137	Eu-152	Eu-154
2020	6.44	2.0	128	343	1960	894
2120			11.1	34.1	10.8	0.3
Year of depletion	2035	2030	>2138	>2138	>2138	2106

Area C

Year	Co-60	H-3	Sr-90	Cs-137	U-233
2020		35.9	46.1	14.4	5.56
2120			4.04	1.43	5.53
Year of depletion	1999	2042	2137	2136	>2137

Area D

Year	Co-60	H-3	Sr-90	Cs-137	U-233
2020	29.4	1.04	32.8	26.0	3.28
2120			2.8	2.6	3.27
Year of depletion	2046	2021	>2142	>2142	>2142

Area E

Year	Co-60	H-3	Sr-90	Cs-137	U-233	U-238
2020		1.0	1.24	4.76	5.54	9.59
2120				0.5	5.52	9.59
Year of depletion	1996	2021	2029	2088	>2133	>2133

Area F

Year	Co-60	H-3	Sr-90	Cs-137	U-233
2020		0.5	39	18.2	4.79
2120			3.4	1.8	4.77
Year of depletion	1996	2017	>2134	>2134	>2134

Table 5.17. (continued)

Area G

Year	Co-60	H-3	Sr-90	Cs-137	U-233
2020		0.65	11	5.4	1.37
2120			0.95	0.5	1.36
Year of depletion	1988	2015	2118	2093	>2135

Area H

Year	Co-60	H-3	Sr-90	Cs-137
2020	1.25	1.29	23.6	57.8
2120			2.0	5.7
Year of depletion	2022	2024	>2148	>2148

Area I

Year	Co-60	H-3	Sr-90	Cs-137
2020		0.6	11.5	5.66
2120			1.00	0.56
Year of depletion	1994	2013	2121	2096

Area J

Year	Co-60	H-3	Sr-90	Cs-137
2020	3.33	0.68	22.4	375
2120			1.97	37.2
Year of depletion	2030	2014	>2147	>2147

Area K

Year	Co-60	H-3	Sr-90	Cs-137	Eu-152	Eu-154
2020	0.14		3.0	8.7	4100	2860
2120			0.26	0.87	22.6	1.08
Year of depletion	2005		2066	2114	>2141	2122

Table 5.17. (continued)

Area L

Year	Co-60	H-3	Sr-90	Cs-137	U-238	U-238
2020		249	25.7	16.2	11.7	2.18
2120				1.36	11.6	2.18
Year of depletion	1998	2087	2056	2133	>2142	>2142

Area M

Year	Co-60	H-3	Sr-90	C-137	U-233	U-238
2020	0.63	25.7	16.2	61.4	71.1	103
2120			1.36	6.1	70.8	103
Year of depletion	2017	2056	2133	2135	>2135	>2135

Area N

Year	Co-60	H-3	Sr-90	Cs-137	U-233	U-238
2020	3.05	3.93	11.2	15.3	29.0	20.7
2120			0.95	1.5	28.9	20.7
Year of depletion	2029	2024	2118	2138	>2138	>2138

Area O

Year	Co-60	H-3	Sr-90	Cs-137	U-233	Th-232
2020	1.85	22.1	8.55	49.7	1.41	1.01
2120			0.71	4.93	1.41	1.01
Year of depletion	2025	2055	2107	2138	>2138	>2138

Area P

Year	Co-60	H-3	Sr-90	Cs-137	Eu-152	Eu-154
2020	261	283	67.3	2930	47700	15000
2120			5.88	290	263	5.66
Year of depletion	2063	2047	>2145	>2145	>2145	2143

Area Q

Year	Co-60	H-3	Sr-90	Cs-137
2020	1.1	0.26	20	104
2120			1.75	10.3
Year of depletion	2021	2004	>2137	>2137

Table 5.17. (continued)

Area R

Year	Co-60	H-3	Sr-90	Cs-137
2020	9.1	2.4	10.7	20.1
2120			0.94	2.0
Year of depletion	2037	2033	2118	>2143

Area S

Year	Co-60	H-3	Sr-90	Cs-137
2020		0.37		
2120				
Year of depletion		2017		

Area T

Year	Co-60	H-3	Sr-90	Cs-137
2020		2.1		
2120				
Year of depletion		2035		

Area U

Year	Co-60	H-3	Sr-90	Cs-137	U-233	U-238
2020		4.1			5.31	9.79
2120					5.29	9.79
Year of depletion		2032			>2139	>2139

Area V

Year	Co-60	H-3	Sr-90	Cs-137
2020		2.0	1.44	5.5
2120				0.5
Year of depletion	2002	2024	2035	2094

Table 5.17. (continued)

Area W

Year	Co-60	H-3	Sr-90	Cs-137
2020		0.3		
2120				
Year of depletion		2004		

Area X

Year	Co-60	H-3	Sr-90	Cs-137
2020	0.14	3.7	5.05	1.65
2120			0.4	
Year of depletion	2005	2025	2086	2046

Area Y

Year	Co-60	H-3	Sr-90	Cs-137	U-233	U-238
2020	80.5	61.3	18.3	20.4	14.6	2.55
2120			1.54	2.0	14.6	255
Year of depletion	2054	2035	2138	>2145	>2145	>2145

Area Z

Year	Co-60	H-3	Sr-90	Cs-137
2020		1.28	5.86	10.9
2120			0.5	1.08
Year of depletion		2024	2093	2173

Tumulus I

Year	Co-60	H-3	Sr-90	Cs-137
2020	157	0.44	5.63	7.01
2120			0.049	0.7
Year of depletion	2006	2006	2092	2105

Table 5.18. Cobalt-60 concentrations in surface water (pCi/L)

Distance upstream (m)[a]	Time period		
	1990-2019	1990-2059	2100-2129
Drainage DA			
209	4.21	2.37	
140	3.55	1.98	
110	2.29	1.28	
98	1.45	0.81	
17	1.25	0.70	
0	1.29	0.72	
French Drain outlet	0.66	0.42	
Drainage DB			
261	67.03	146.78	0.04
180	46.99	101.79	0.03
123	35.76	70.91	0.02
110	31.38	62.12	0.02
0	23.18	45.50	0.01
Drainage FA			
184	34.83	26.22	
87	12.93	9.74	
67	9.36	7.01	
17	4.69	3.51	
Drainage FB			
512	8.23	3.98	0.07
461	23.51	10.99	0.05
397	17.87	8.32	0.04
343	9.94	4.63	0.02
196	11.93	7.03	0.01
95	10.67	6.28	0.01
17	11.83	10.97	0.01
EWB			
	0.23	0.11	

[a] Measured from the WAG 6 boundary.

Table 5.19. Cesium-137 concentrations in surface water (pCi/L)

Distance upstream (m)[a]	Time period 1990-2019	Time period 1990-2059	Time period 2100-2129
Drainage DA			
209	42.96	32.05	6.00
140	35.99	26.83	5.00
110	24.27	18.10	3.38
98	15.93	11.88	2.22
17	13.81	10.29	1.92
0	12.73	9.44	1.72
French Drain outlet	9.55	7.26	1.50
Drainage DB			
261	16.44	51.85	44.63
180	25.46	46.21	32.71
123	17.26	31.37	22.22
110	15.21	27.53	19.46
0	12.27	20.91	14.31
Drainage FA			
184	222.29	155.49	20.66
87	82.55	57.74	7.67
67	60.05	41.79	5.49
17	30.11	20.95	2.75
Drainage FB			
512	32.97	25.54	5.44
461	27.33	24.51	8.52
397	21.38	18.96	6.44
343	11.90	10.55	3.58
196	23.58	17.78	3.62
95	21.08	15.89	3.24
17	19.45	14.62	2.93
EWB			
	3.40	2.32	0.26

[a] Measured from the WAG 6 boundary.

Table 5.20. Europium-152 concentrations in surface water (pCi/L)

Distance upstream (m)[a]	Time period		
	1990-2019	1990-2059	2100-2129
Drainage DB[b]			
261	31.39	42.37	3.09
180	21.70	29.29	2.14
123	14.63	19.74	1.44
110	12.81	17.29	1.26
0	9.35	12.62	0.92

[a] Measured from the WAG 6 boundary

[b] The model predicted no occurrences of europium-152 in drainages DA, FA, and FB and the EWB in each of the periods evaluated.

Table 5.21. Europium-154 concentrations in surface water (pCi/L)

Distance upstream (m)[a]	Time period		
	1990-2019	1990-2059	2100-2129
Drainage DB[b]			
261	17.39	24.22	0.29
180	12.02	16.75	0.20
123	8.10	11.29	0.13
110	7.09	9.88	0.12
0	5.18	7.22	0.09

[a] Measured from the WAG 6 boundary.

[b] The model predicted no occurrences of europium-152 in drainages DA, FA, and FB and the EWB in each of the periods evaluated.

Table 5.22. Tritium concentrations in surface water (pCi/L)

Distance upstream (m)[a]	Time period		
	1990-2019	1990-2059	2100-2129
Drainage DA			
209	456,080	259,748	228
140	421,554	240,522	206
110	328,190	187,252	160
98	241,952	138,048	118
17	216,492	123,522	106
0	194,531	110,923	93
French Drain outlet	2,669,856	1,531,075	1378
Drainage DB			
261	731,451	699,760	2
180	809,609	659,545	170
123	548,035	446,332	119
110	489,081	396,451	104
0	363,693	293,111	77
Drainage FA			
184	192,805	119,013	57
87	71,603	44,199	21
67	56,440	34,740	19
17	28,295	17,416	10
Drainage FB			
512	796,833	405,032	3
461	583,725	296,755	5
397	416,667	211,806	4
343	231,832	117,848	2
196	254,589	125,395	5
95	227,605	112,104	4
17	215,221	109,530	4
EWB			
	15,158	7020	0

[a] Measured from the WAG 6 boundary.

Table 5.23. Strontium-90 concentrations in surface water (pCi/L)

Distance upstream (m)[a]	Time period		
	1990-2019	1990-2059	2100-2129
Drainage DA			
209	107.22	198.12	109.28
140	89.85	165.37	91.06
110	60.60	111.54	61.42
98	39.77	73.20	40.30
17	34.46	63.44	34.93
0	30.61	56.54	31.19
French Drain outlet			
	29.49	57.09	33.79
Drainage DB			
261	20.60	114.57	98.26
180	47.17	130.34	95.56
123	34.66	93.42	67.97
110	30.62	81.97	59.53
0	23.88	62.88	45.20
Drainage FA			
184	214.90	534.60	314.25
87	79.81	198.54	116.70
67	60.33	143.78	83.48
17	30.25	72.08	41.85
Drainage FB			
512	264.00	479.14	270.13
461	234.11	432.23	258.40
397	181.32	333.65	201.60
343	100.88	185.64	112.17
196	92.44	189.51	113.44
95	82.64	169.42	101.42
17	73.95	156.43	95.33
EWB			
	43.12	106.28	60.48

[a] Measured from the WAG 6 boundary.

Table 5.24. Thorium-232 concentrations in surface water (pCi/L)

Distance upstream (m)[a]	Time period		
	1990-2019	1990-2059	2100-2129
Drainage DA			
209	0.19	0.23	0.37
140	0.16	0.19	0.30
110	0.10	0.13	0.21
98	0.07	0.08	0.13
17	0.06	0.07	0.12
0	0.05	0.06	0.10
French Drain outlet	0.09	0.10	0.16
Drainage DB			
261	11.08	104.60	523.61
180	7.66	72.32	362.04
123	5.16	48.74	244.00
110	4.52	42.68	213.65
0	3.30	31.16	155.96
Drainage FA			
184	2.57	2.29	1.22
87	0.95	0.85	0.45
67	0.68	0.61	0.32
17	0.34	0.30	0.16
Drainage FB			
512			
461			
397			
343			
196	0.31	0.28	0.15
95	0.28	0.25	0.14
17	0.24	0.22	0.12
EWB			

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[a] Measured from the WAG 6 boundary.

Table 5.25. Uranium-233 concentrations in surface water (pCi/L)

Distance upstream (m)[a]	Time period 1990-2019	 1990-2059	 2100-2129
Drainage DA			
209	5.96	27.33	118.04
140	4.97	22.77	98.33
110	3.35	15.36	66.32
98	2.20	10.08	43.52
17	1.91	8.73	37.72
0	1.70	7.74	33.31
French Drain outlet	3.53	16.09	153.62
Drainage DB			
261			
180	2.11	10.24	44.10
123	1.42	6.90	29.72
110	1.24	6.05	26.02
0	1.00	4.78	20.45
Drainage FB[b]			
512	0.78	2.88	10.69
461	0.57	2.09	7.77
397	0.40	1.49	5.52
343	0.23	0.83	3.07
196	0.89	3.26	11.98
95	0.79	2.91	10.71
17	0.84	3.23	12.47
EWB			
	0.19	0.61	2.09

[a] Measured from the WAG 6 boundary.

[b] The model predicted no occurrences of uranium-233 in Drainage FA in each of the periods evaluated.

Table 5.26. Uranium-238 concentrations in surface water (pCi/L)

Distance upstream (m)[a]	Time period		
	1990-2019	1990-2059	2100-2129
Drainage DA			
209	10.28	45.58	190.75
140	8.56	41.30	158.91
110	5.78	27.86	107.18
98	3.79	18.28	70.33
17	3.29	15.84	60.96
0	2.79	13.48	51.85
French Drain outlet	3.87	26.68	115.08
Drainage DB			
261			
180	0.08	0.38	1.60
123	0.06	0.25	1.08
110	0.05	0.22	0.94
0	0.37	1.49	5.89
Drainage FA			
184	2.86	10.53	39.46
87	1.06	3.91	14.66
67	0.76	2.80	10.48
17	0.38	1.40	5.25
Drainage FB			
512	0.27	0.86	2.91
461	0.20	0.62	2.11
397	0.14	0.44	1.50
343	0.08	0.25	0.83
196	3.95	18.66	71.59
95	3.53	16.68	64.00
17	3.26	15.63	60.85
EWB			
	0.07	0.23	0.84

[a] Measured from the WAG 6 boundary.

Table 5.27. Predicted peak annual flux in surface water in WAG 6

Radionuclide	Peak flux exiting WAG 6		Peak flux discharge to EWB	
	μCi	Year	μCi	Year
H-3	60,000,000	2018	820,000	2028
Co-60	7700	2026	12	1990
Sr-90	43,000	2042	3400	2038
Cs-137	3200	1990	120	1990
Eu-152	930	2020		
Eu-154	430	2024		
Th-232[a]	5000	2130		
U-233[a]	2200	2130	46	2130
U-23[a]	6000	2130	18	2130

[a] Peak occurs sometime after 2130.

Table 5.28. Predicted peak annual flux in groundwater discharge in WAG 6

Radionuclide	Peak flux exiting WAG 6 μCi	Peak flux exiting WAG 6 Year	Peak flux discharge to EWB μCi	Peak flux discharge to EWB Year
H-3	2,750,000	1992	4480	2007
Co-60	1700	1990	0.2	1995
Sr-90	450	2052	13	2043
Cs-137	62	2050	0.3	1995
Eu-152	200	2050		
Eu-154	52	2036		
Th-232	0.4	2064		
U-233[a]	92	2205	2	2205
U-238[a]	160	2210	3	2205

[a] Peak occurs after 2205.

Table 5.29. Sensitivity analysis for cobalt-60

Output parameter (yr)	Input parameter	Nominal value	Ratio of % change in output to % change in input	Sensitivity[a]
GW Conc (1)[b]	DHORZ	53 m	- 0.37	Medium
GW Conc (1)	ATREN	3357 Ci	1	High
GW Conc (1)	IYRC	11 yr	- 0.96	High
GW Conc (1)	GWV	21.5 m/yr	- 1.22	High
GW Conc (1)	GWV	46 m/yr	- 1.27	High
GW Conc (1)	GWV	60.4 m/yr	- 0.776	Medium
GW Conc (1)	XKDH	0.00055 m^3/kg	- 0.24	Medium
GW Conc (1)	EPORA	0.014	- 0.81	High
GW Conc (1)	EPORA	0.03	- 0.96	High
GW Conc (1)	FFIN	0.1	0	
GW Conc (31)	FFIN	0.1	0.985	High
GW Conc (1)	RPERC	0.19 m/yr	1	High
GW Conc (1)	RPERC	0.399 m/yr	1.07	High
GW Conc (1)	FINT	0.000026	1	High
GW Conc (1)	AQTHK	4.6 m	- 0.85	High
GW Conc (1)	ADISP	0.3	- 0.28	Medium
GW Conc (1)	TAREA	1481 m^2	- 0.36	Medium
GW Conc (1)	XKDT	0.782 m^3/kg	1.001	High
Inventory (31)	RPERC	0.19	0	Low
Inventory (31)	FINT	0.000026	0	Low

[a] Criteria for low, medium, and high sensitivity based on ratio of percent change of parameters: low (0-0.1), medium (0.1-0.8) and high (> 0.8).

[b] GW Conc (1) = groundwater concentration, year 1.

Table 5.30. Sensitivity analysis for tritium

Output parameter (yr)	Input parameter	Nominal value	Ratio of % change in output to % change in input	Sensitivity[a]
GW Conc (1)[b]	DHORZ	53 m	- 0.35	Medium
GW Conc (1)	ATREN	91.6 Ci	1.00	High
GW Conc (1)	IYRC	11 yr	- 0.5	Medium
GW Conc (1)	GWV	25.7 m/yr	- 1.07	High
GW Conc (1)	GWV	46 m/yr	- 1.025	High
GW Conc (1)	EPORA	0.0129	- 1.59	High
GW Conc (1)	EPORA	0.03	- 1.4	High
GW Conc (1)	EPORA	0.0497	- 0.917	High
GW Conc (31)	FFIN	0.1	0.088	Low
GW Conc (1)	RPERC	0.19 m/yr	0.955	High
GW Conc (1)	FINT	0.021	1.0	High
GW Conc (1)	AQTHK	4.6	- 0.97	High
GW Conc (1)	ADISP	0.3	- 0.28	Medium
GW Conc (1)	TAREA	1481	- 0.356	Medium
Inventory (31)	RPERC	0.19	- 0.1	Low
Inventory (31)	FINT	0.021	- 0.105	Low

[a] Criteria for low, medium, and high sensitivity based on ratio of percent change of parameters: low (0-0.1), medium (0.1-0.8) and high (> 0.8).

[b] GW Conc (1) = groundwater concentration, year 1 medium (0.1-0.8) and high (> 0.8).

Table 5.31. Parameter range and uncertainty

Parameter	Nominal value	Distribution	Range	Uncertainty
TAREA	Varies	Normal	Cov[a] = 0.1	Low
H	4.6 m	Normal	Cov = 0.2	Low
FINT	Varies	Normal	Cov = 1.0	High
FFIN	0.1	Normal	Cov = 0.5	High
ATREN	Varies	Normal	Cov = 0.5	High
DECAY	Varies			Low
XKDV	Varies	Normal	Cov = 1.0	High
XKDH	Varies	Normal	Cov = 1.0	High
XKDT	Varies	Normal	Cov = 0.5	High
RPERC	0.19 m/yr	Normal	Cov = 1.0	High
DVERT	Varies	Normal	Cov = 0.2	Medium
DHORZ	Varies	Normal	Cov = 0.1-1.0	Low
EPORV	0.02	Uniform	0.004-0.04	High
EPORA	0.02	Uniform	0.004-0.04	High
PORT	0.5	Normal	Cov = 0.5	Low
PORV	0.1	Normal	Cov = 0.5	Low
SSAT	0.5	Normal	Cov = 0.5	Medium
BDENS	1300 kg/m^3	Normal	Cov = 0.25	Low
DISP	5 m	Normal	Cov = 0.2	Medium
AQTHK	4.6 m	Uniform	4.6-6.4	High
PORA	0.1	Normal	Cov = 0.5	Medium
GWV	Varies	Normal	Cov = 0.5	Medium
ADISP	0.3 radians	Normal	Cov = 0.2	Medium
IYRC	Varies	Normal	Cov = 0.2-0.5	Low
WTRI	Varies	Normal	Cov = 0.5	Medium

[a] Cov: coefficient of variation.

Table 5.32. Classification of atmospheric stability

Stability classification	Pasquill categories	Degrees
Extremely Unstable	A (1)	$\sigma\Theta \geq 22.5$
Moderately Unstable	B (2)	$22.5 > \sigma\Theta > 17.5$
Slightly Unstable	C (3)	$17.5 > \sigma\Theta > 12.5$
Neutral	D (4)	$12.5 > \sigma\Theta > 7.5$
Slightly Stable	E (5)	$7.5 > \sigma\Theta > 3.8$
Moderately Stable	F (6)	$3.8 > \sigma\Theta > 0.0$

Nighttime Modified Sigma Theta Scheme

If Stability is	and Wind Speed is	Then Stability Class is
A	$WS < 2.9$	F(6)
	$2.9 \leq WS < 3.6$	E (5)
	$3.6 \leq WS$	DN (7)[a]
B	$WS < 2.4$	F (6)
	$2.4 \leq WS < 3.0$	E (5)
	$3.0 \leq WS$	DN (7)[a]
C	$WS < 2.4$	E (6)
	$2.4 \leq WS$	DN (7)[a]
D	N/A	D
E	N/A	E
F	N/A	F

[a] DN = D stability for nighttime regime. This is counted as stability class D in the joint frequency analysis.

Table 5.33. Frequency of wind speed and direction for atmospheric stability categories at ORNL Tower TA, 1986-1990[a]

Frequency (%) of wind speed and direction for stability class A

speed (m/s)[b]

Direction	1.5	2.5	4.5	7.0	9.5	>11	Total
N	0.0541	0.0313	0.0000	0.0000	0.0000	0.0000	0.8555
NNE	0.0855	0.0313	0.0000	0.0000	0.0000	0.0000	0.1168
NE	0.1282	0.0798	0.0057	0.0000	0.0000	0.0000	0.2137
ENE	0.1567	0.1083	0.0000	0.0000	0.0000	0.0028	0.2679
E	0.2223	0.0513	0.0000	0.0000	0.0000	0.0000	0.2736
ESE	0.1282	0.0342	0.0000	0.0000	0.0000	0.0000	0.1624
SE	0.0855	0.0256	0.0000	0.0000	0.0000	0.0000	0.1111
SSE	0.1054	0.0199	0.0000	0.0000	0.0000	0.0000	0.1254
S	0.1852	0.0427	0.0028	0.0000	0.0000	0.0000	0.2308
SSW	0.2736	0.2166	0.0313	0.0000	0.0000	0.0000	0.5215
SW	0.4217	0.2650	0.0171	0.0000	0.0000	0.0000	0.7038
WSW	0.3790	0.1653	0.0028	0.0000	0.0000	0.0000	0.5471
W	0.1995	0.2052	0.0057	0.0000	0.0000	0.0000	0.4103
WNW	0.1140	0.1624	0.0085	0.0000	0.0000	0.0000	0.2850
NW	0.0541	0.0855	0.0057	0.0000	0.0000	0.0000	0.1453
NNW	0.0484	0.0513	0.0000	0.0000	0.0000	0.0000	0.0997
Total	2.6416	1.5758	0.0798	0.0000	0.0000	0.0028	

Calms = 320 Total class observations = 4.30%

Frequency (%) of wind speed and direction for stability class B

speed (m/s)

Direction	1.5	2.5	4.5	7.0	9.5	>11	Total
N	0.0912	0.0541	0.0000	0.0000	0.0000	0.0000	0.1453
NNE	0.1368	0.0484	0.0000	0.0000	0.0000	0.0000	0.1852
NE	0.2679	0.1368	0.0000	0.0000	0.0000	0.0000	0.4046
ENE	0.5614	0.2365	0.0028	0.0000	0.0000	0.0000	0.8007
E	0.4901	0.2166	0.0000	0.0000	0.0000	0.0000	0.7067
ESE	0.3419	0.0769	0.0000	0.0000	0.0000	0.0000	0.4189
SE	0.2080	0.0513	0.0000	0.0000	0.0000	0.0000	0.2593
SSE	0.2451	0.0313	0.0000	0.0000	0.0000	0.0000	0.2764
S	0.2622	0.0513	0.0000	0.0000	0.0000	0.0000	0.3135
SSW	0.6611	0.3277	0.0484	0.0114	0.0000	0.0000	1.0486
SW	1.3535	0.5044	0.0256	0.0028	0.0000	0.0000	1.8864
WSW	0.9261	0.3704	0.0114	0.0028	0.0000	0.0000	1.3108
W	0.5984	0.4445	0.0399	0.0000	0.0000	0.0000	1.0828
WNW	0.2793	0.2622	0.0285	0.0000	0.0000	0.0000	0.5699

Table 5.33 (continued)

NW	0.1881	0.0969	0.0028	0.0000	0.0000	0.0000	0.2878
NNW	0.1539	0.0769	0.0057	0.0000	0.0000	0.0000	0.2365
Total	6.7649	2.9864	0.1653	0.0171	0.0000	0.0000	

Calms = 852　　Total class observations = 9.93%

Frequency (%) of wind speed and direction for stability class C

speed (m/s)

Direction	1.5	2.5	4.5	7.0	9.5	>11	Total
N	0.1311	0.0228	0.0000	0.0000	0.0000	0.0000	0.1539
NNE	0.1852	0.0655	0.0000	0.0000	0.0000	0.0000	0.2508
NE	0.5414	0.3306	0.0142	0.0000	0.0000	0.0000	0.8862
ENE	0.9774	0.5813	0.0057	0.0000	0.0000	0.0000	1.5644
E	0.6782	0.2451	0.0028	0.0000	0.0000	0.0000	0.9261
ESE	0.2964	0.0627	0.0000	0.0000	0.0000	0.0000	0.3590
SE	0.1738	0.0256	0.0028	0.0000	0.0000	0.0000	0.2023
SSE	0.1311	0.0171	0.0000	0.0000	0.0000	0.0000	0.1482
S	0.1824	0.0228	0.0000	0.0000	0.0000	0.0000	0.2052
SSW	0.7152	0.3704	0.0456	0.0057	0.0000	0.0000	1.1370
SW	1.6243	0.7494	0.0883	0.0028	0.0000	0.0000	2.4649
WSW	1.1199	0.4075	0.0199	0.0000	0.0000	0.0000	1.5473
W	0.5984	0.4331	0.0456	0.0000	0.0000	0.0000	1.0771
WNW	0.1795	0.1197	0.0114	0.0000	0.0000	0.0000	0.3106
NW	0.0912	0.0256	0.0000	0.0000	0.0000	0.0000	0.1168
NNW	0.0769	0.0114	0.0028	0.0000	0.0000	0.0000	0.0912
Total	7.7024	3.4907	0.2394	0.0085	0.0000	0.0000	

Calms = 1429　　Total class observations = 11.44%

Frequency (%) of wind speed and direction for stability class D

speed (m/s)

Direction	1.5	2.5	4.5	7.0	9.5	>11	Total
N	0.0826	0.0114	0.0000	0.0000	0.0000	0.0000	0.0940
NNE	0.2622	0.0684	0.0028	0.0000	0.0000	0.0000	0.3334
NE	1.2681	0.9432	0.0199	0.0000	0.0000	0.0000	2.2312
ENE	1.9406	0.7751	0.0114	0.0000	0.0000	0.0000	2.7270
E	0.6583	0.1140	0.0028	0.0000	0.0000	0.0000	0.7751
ESE	0.1653	0.0142	0.0000	0.0000	0.0000	0.0000	0.1795
SE	0.1225	0.0085	0.0000	0.0000	0.0000	0.0000	0.1311
SSE	0.1225	0.0057	0.0000	0.0000	0.0000	0.0000	0.1282
S	0.1596	0.0014	0.0028	0.0000	0.0000	0.0000	0.1783

Table 5.33 (continued)

SSW	0.6981	0.4445	0.0456	0.0000	0.0000	0.0000	1.1883
SW	1.5302	0.4901	0.0342	0.0000	0.0000	0.0000	2.0545
WSW	2.1885	0.4417	0.0114	0.0000	0.0000	0.0000	2.6416
W	0.7466	0.4217	0.0114	0.0000	0.0000	0.0000	1.1797
WNW	0.1539	0.0228	0.0000	0.0000	0.0000	0.0000	0.1795
NW	0.0598	0.0028	0.0000	0.0000	0.0000	0.0000	0.0627
NNW	0.0513	0.0000	0.0000	0.0000	0.0000	0.0000	0.0513
Total	10.2100	3.7757	0.1453	0.0000	0.0000	0.0000	

Calms = 4682 Total class observations = 14.13%

Frequency (%) of wind speed and direction for stability class E

Direction	speed (m/s) 1.5	2.5	4.5	7.0	9.5	>11	Total
N	0.1282	0.0114	0.0000	0.0000	0.0000	0.0000	0.1396
NNE	0.2308	0.0484	0.0000	0.0000	0.0000	0.0000	0.2793
NE	1.0230	0.1881	0.0028	0.0000	0.0000	0.0000	1.2139
ENE	1.7325	0.2878	0.0000	0.0000	0.0000	0.0000	2.0203
E	0.4759	0.0256	0.0000	0.0000	0.0000	0.0000	0.5015
ESE	0.0712	0.0114	0.0000	0.0000	0.0000	0.0000	0.0826
SE	0.0513	0.0085	0.0000	0.0000	0.0000	0.0000	0.0598
SSE	0.0826	0.0057	0.0000	0.0000	0.0000	0.0000	0.0883
S	0.0769	0.0171	0.0000	0.0000	0.0000	0.0000	0.0940
SSW	0.1767	0.1140	0.0000	0.0000	0.0000	0.0000	0.2907
SW	1.0771	0.2422	0.0000	0.0000	0.0000	0.0000	1.3194
WSW	2.0175	0.2337	0.0000	0.0000	0.0000	0.0000	2.2512
W	0.8520	0.4189	0.0028	0.0000	0.0000	0.0000	1.2738
WNW	0.1852	0.0741	0.0000	0.0000	0.0000	0.0000	0.2593
NW	0.0598	0.0199	0.0000	0.0000	0.0000	0.0000	0.0798
NNW	0.1054	0.0171	0.0000	0.0000	0.0000	0.0000	0.1225
Total	8.3464	1.7240	0.0057	0.0000	0.0000	0.0000	

Calms = 5181 Total class observations = 10.08%

Frequency (%) of wind speed and direction for stability class F

Direction	speed (m/s) 1.5	2.5	4.5	7.0	9.5	>11	Total
N	0.1026	0.0171	0.0000	0.0000	0.0000	0.0000	0.1197
NNE	0.0798	0.0399	0.0000	0.0000	0.0000	0.0000	0.1197
NE	0.7494	0.0741	0.0000	0.0000	0.0000	0.0000	0.8235
ENE	1.7553	0.0798	0.0000	0.0000	0.0000	0.0028	1.8380

Table 5.33 (continued)

E	0.2992	0.0028	0.0057	0.0000	0.0000	0.0028	0.3106
ESE	0.0142	0.0000	0.0000	0.0000	0.0000	0.0000	0.0142
SE	0.0342	0.0057	0.0000	0.0000	0.0000	0.0000	0.0399
SSE	0.0228	0.0028	0.0000	0.0000	0.0000	0.0000	0.0256
S	0.0541	0.0171	0.0057	0.0000	0.0000	0.0000	0.0769
SSW	0.0598	0.0798	0.0427	0.0000	0.0000	0.0000	0.1824
SW	0.3106	0.1681	0.0370	0.0000	0.0000	0.0028	0.5186
WSW	1.1541	0.1282	0.0085	0.0000	0.0000	0.0000	1.2909
W	0.4645	0.2992	0.0285	0.0000	0.0000	0.0000	0.7922
WNW	0.1311	0.1539	0.0142	0.0000	0.0000	0.0000	0.2992
NW	0.1254	0.0541	0.0000	0.0000	0.0000	0.0000	0.1795
NNW	0.0855	0.0228	0.0000	0.0000	0.0000	0.0000	0.1083
Total	5.4427	1.1455	0.1425	0.0000	0.0000	0.0085	

Calms = 2759 Total class observations = 6.74%

Frequency (%) of wind speed and direction for all stability classes

speed (m/s)

Direction	1.5	2.5	4.5	7.0	9.5	>11	Total
N	0.5899	0.1482	0.0000	0.0000	0.0000	0.0000	0.7380
NNE	0.9803	0.3021	0.0028	0.0000	0.0000	0.0000	1.2852
NE	3.9780	1.7525	0.0427	0.0000	0.0000	0.0000	5.7732
ENE	7.1239	2.0688	0.0199	0.0000	0.0000	0.0057	9.2184
E	2.8239	0.6554	0.0114	0.0000	0.0000	0.0028	3.4936
ESE	1.0173	0.1995	0.0000	0.0000	0.0000	0.0000	1.2168
SE	0.6753	0.1254	0.0028	0.0000	0.0000	0.0000	0.8036
SSE	0.7095	0.0826	0.0000	0.0000	0.0000	0.0000	0.7922
S	0.9204	0.1624	0.0114	0.0000	0.0000	0.0000	1.0942
SSW	2.5846	1.5530	0.2137	0.0171	0.0000	0.0000	4.3684
SW	6.3175	2.4193	0.2023	0.0057	0.0000	0.0028	8.9477
WSW	7.7850	1.7468	0.0541	0.0028	0.0000	0.0000	9.5888
W	3.4594	2.2227	0.1339	0.0000	0.0000	0.0000	5.8160
WNW	1.0429	0.7950	0.0655	0.0000	0.0000	0.0000	1.9035
NW	0.5785	0.2850	0.0085	0.0000	0.0000	0.0000	0.8720
NNW	0.5215	0.1795	0.0085	0.0000	0.0000	0.0000	0.7095
Total	41.1079	14.6981	0.7779	0.0256	0.0000	0.0114	

Calms = 15,223 h

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[a] Total observations (35,093 hours) includes calms; missed observations totalled 8083 hours. The total days counted are for the entire data set and numbered 1799.

[b] NODC wind speed ranges (m/s): 1.5 (0-1.8); 2.5 (1.8-3.3); 4.5 (3.3-5.4); 7.0 (5.4-8.5); 9.5 (8.5-11)

Date of run: 3/28/91

Table 5.34. WAG 6 surface water sampling results—tritium[a]

Source	Sampling point	Sampling Period High water table	Storm event I	Base flow	Storm event II
Stream FB	WFBB1	3,100,000	2,200,000	3,600,000	3,910,000[c]
	WFBB3	NS[b]	190,000[c]	NS	161,000
Stream FA	WFBA1	86,000	96,000	NS	103,000[c]
Stream FAA	WBAA1	NS	1500[c]	NS	NS
Stream FAB	WBAB1	NS	130,000[c]	NS	122,000
	WBAB2	NS	390,000	NS	393,000[c]
French Drain	W49TS	12,000,000	8,500,000	14,000,000[c]	4,270,000
Stream DA	WDA1	2,050,000	160,000	2,900,000	2,320,000[c]
Stream DB	WDB1	NS	73,000[c]	NS	NS
Stream B	SWB1	30,000	2700[c]	NS	NS
Stream A2	SWA1	NS	2700[c]	NS	2250
EWB	WEWB	NS	NS	190,000[c]	169,000

[a] Concentrations are in pCi/L.
[b] Not sampled.
[c] Used to calculate present-day emissions.

Table 5.35. Tritium source data—1989

Source	Source area	Length (ft)	Width (ft)	Area (ft^2)	Evaporation (m^3/yr)	Tritium concentration (pCi/L)	Area source side length (ft)	Emission rate ($g/m^2/s$)
FA		285	2	570	44			
	FA1	143	2	285	22	103,000	100	6.95E-18
	FA2	143	2	285	22	103,000	100	6.95E-18
FAA		530	1	530	41			
	FAA1	133	1	133	10	1500	100	4.71E-20
	FAA2	133	1	133	10	1500	100	4.71E-20
	FAA3	133	1	133	10	1500	100	4.71E-20
	FAA4	133	1	133	10	1500	100	4.71E-20
FAB[a]		320	1	320	25			
	FAB1	80	1	80	6	393,000	100	7.44E-18
	FAB2	80	1	80	6	305,333	100	5.78E-18
	FAB3	80	1	80	6	217,667	100	4.12E-18
	FAB4	80	1	80	6	130,000	100	2.46E-18
FB[b]		2356	0.75-2.25	3764	416			
	FB1	70	1	70	8	62,000	100	4.49E-18
	FB2[c]	70	1	50	6	62,000	100	3.21E-18
	FB3	140	0.75	105	12	62,000	100	6.73E-18
	FB4	159	1.5	238	26	62,000	100	1.53E-17
	FB5	159	1.5	238	26	62,000	100	1.53E-17
	FB6	159	1.5	238	26	62,000	100	1.53E-17
	FB7	159	1.5	238	26	411,818	100	4.25E-17
	FB8	159	1.5	238	26	761,636	100	6.97E-17
	FB9	159	1.5	238	26	1,111,455	100	9.69E-17
	FB10	159	1.5	238	26	1,461,273	100	1.24E-16
	FB11	159	1.5	238	26	1,811,091	100	1.51E-16
	FB12[d]	165	1.5	274	30	2,160,909	100	2.06E-16
	FB13	83	2	165	18	2,510,727	100	1.42E-16
	FB14	83	2	165	18	2,860,545	100	1.61E-16
	FB15	83	2	165	18	3,210,364	100	1.80E-16
	FB16	83	2	165	18	3,560,182	100	1.99E-16
	FB17	103	2.25	232	26	3,910,000	100	3.07E-16
	FB18	103	2.25	232	26	3,910,000	100	3.07E-16
	FB19	103	2.25	232	26	3,910,000	100	3.07E-16
French Drain		50	50	2500	276			
	FD1	50	50	2500	276	14,000,000	50	4.73E-14

Table 5.35. (continued)

Source	Source area	Length (ft)	Width (ft)	Area (ft^2)	Evaporation (m^3/yr)	Tritium concentration (pCi/L)	Area source side length (ft)	Emission rate (g/m^2/s)
DA		300	1.5	450	35			
	DA1	100	1.5	150	12	2,320,000	100	8.24E-17
	DA2	100	1.5	150	12	2,320,000	100	8.24E-17
	DA3	100	1.5	150	12	2,320,000	100	8.24E-17
DB		860	2	1720	133			
	DB1	123	2	246	19	73,000	100	4.25E-18
	DB2	123	2	246	19	73,000	100	4.25E-18
	DB3	123	2	246	19	73,000	100	4.25E-18
	DB4	123	2	246	19	73,000	100	4.25E-18
	DB5	123	2	246	19	73,000	100	4.25E-18
	DB6	123	2	246	19	73,000	100	4.25E-18
	DB7	123	2	246	19	73,000	100	4.25E-18
B		480	1	480	37			
	B1	120	1	120	9	2700	100	7.67E-20
	B2	120	1	120	9	2700	100	7.67E-20
	B3	120	1	120	9	2700	100	7.67E-20
	B4	120	1	120	9	2700	100	7.67E-20
A2		450	2	900	70			
	A21	113	2	225	17	2700	100	1.44E-19
	A22	113	2	225	17	2700	100	1.44E-19
	A23	113	2	225	17	2700	100	1.44E-19
	A24	113	2	225	17	2700	100	1.44E-19
EWB[e]		150	150	22500	2487			
	WB1	50	50	2500	276	190,000	50	6.41E-16
	WB2	100	100	10000	1106	190,000	100	6.41E-16
	WB3	100	100	10000	1106	190,000	100	6.41E-16

[a] FAB1 is based on WBAB2 concentration; FAB2 and FAB3 are based on the concentration gradient between WBAB1 and WBAB2; FAB4 is based on WBAB1 concentration.

[b] FB1 through FB6 are based on WFBB3 concentration; FB7 through FB16 are based on the concentration gradient between WFBB2 and WFBB1; FB17 through FB19 are based on WFBB1 concentration.

[c] Length includes a 20-ft section of stream flowing underground; area does not account for this section.

[d] Includes a 6-ft by 6-ft pool in stream.

[e] Surface area is 22,467 ft^2.

Table 5.36. Surface water modeling results—tritium

Stream	Stream length (ft)	Downstream distance (ft)	Surface water modeling concentrations 1990-2019 (pCi/L)	1990-2059 (pCi/L)	2100-2129 (pCi/L)	Area sources[a] (Used in air modeling)
DA[b]	300	0	194,531	110,923	93	
		24	216,492	123,522	106	DA3
		141	241,952	138,048	118	
		158	328,190	187,252	160	DA2
		201	421,554	240,522	206	DA1
		300	456,080	259,748	228	
DB	860	0	363,693	293,111	77	DB7
		363	489,081	396,451	104	DB5, DB6
		406	548,035	446,332	119	DB4
		594	809,609	659,545	170	DB2, DB3
		860	731,451	699,760	2	DB1
FA/FAB	605	56	28,295	17,416	10	FA2
		220	56,440	34,740	19	FA1
		285	71,603	44,199	21	FAB3, FAB4
		605	192,805	119,013	57	FAB1, FAB2
FB	2356	79	215,221	109,530	4	FB17 - FB19
		438	227,605	112,104	4	FB13 - FB16
		902	254,589	125,395	5	FB9 - FB12
		1578	231,832	117,848	2	FB6 - FB8
		1826	416,667	211,806	4	FB4, FB5
		2120	583,725	296,755	5	FB3
		2356	796,833	405,032	3	FB1, FB2
French Drain	none	none	2,669,856	1,531,075	1378	FD1
EWB	none	none	15,158	7020	0	WB1 - WB3

[a] Area sources correspond to the nearest modeled downstream concentration (using air modeling distance).

[b] Surface water modeling assumed stream DA was wet for entire length. Observations indicate that uppermost section of the stream is a dry channel.

Table 5.37. Predicted surface water tritium emissions ($g/m^2/s$)

Source	1989	Period 1990 - 2019	Period 1990 - 2059	Period 2100 - 2129
FA1	6.95E-18	3.81E-18	2.34E-18	1.28E-21
FA2	6.95E-18	1.91E-18	1.18E-18	6.75E-22
FAA1	4.71E-20	0.00E+00	0.00E+00	0.00E+00
FAA2	4.71E-20	0.00E+00	0.00E+00	0.00E+00
FAA3	4.71E-20	0.00E+00	0.00E+00	0.00E+00
FAA4	4.71E-20	0.00E+00	0.00E+00	0.00E+00
FAB1	7.44E-18	3.65E-18	2.25E-18	1.08E-21
FAB2	5.78E-18	3.65E-18	2.25E-18	1.08E-21
FAB3	4.12E-18	1.36E-18	8.37E-19	3.98E-22
FAB4	2.46E-18	1.36E-18	8.37E-19	3.98E-22
FB1	4.49E-18	1.88E-17	9.57E-18	7.09E-23
FB2	3.21E-18	1.34E-17	6.84E-18	5.06E-23
FB3	6.73E-18	2.07E-17	1.05E-17	1.77E-22
FB4	1.53E-17	3.35E-17	1.70E-17	3.22E-22
FB5	1.53E-17	3.35E-17	1.70E-17	3.22E-22
FB6	1.53E-17	1.87E-17	9.48E-18	1.61E-22
FB7	4.25E-17	1.87E-17	9.48E-18	1.61E-22
FB8	6.97E-17	1.87E-17	9.48E-18	1.61E-22
FB9	9.69E-17	2.05E-17	1.01E-17	4.02E-22
FB10	1.24E-16	2.05E-17	1.01E-17	4.02E-22
FB11	1.51E-16	2.05E-17	1.01E-17	4.02E-22
FB12	2.06E-16	2.36E-17	1.16E-17	4.63E-22
FB13	1.42E-16	1.27E-17	6.24E-18	2.23E-22
FB14	1.61E-16	1.27E-17	6.24E-18	2.23E-22
FB15	1.80E-16	1.27E-17	6.24E-18	2.23E-22
FB16	1.99E-16	1.27E-17	6.24E-18	2.23E-22
FB17	3.07E-16	1.69E-17	8.60E-18	3.14E-22
FB18	3.07E-16	1.69E-17	8.60E-18	3.14E-22
FB19	3.07E-16	1.69E-17	8.60E-18	3.14E-22
FD1	4.73E-14	9.01E-15	5.17E-15	4.65E-18
DA1	8.24E-17	1.50E-17	8.54E-18	7.32E-21
DA2	8.24E-17	1.17E-17	6.65E-18	5.68E-21
DA3	8.24E-17	7.69E-18	4.39E-18	3.76E-21
DB1	4.25E-18	4.26E-17	4.07E-17	1.16E-22
DB2	4.25E-18	4.71E-17	3.84E-17	9.89E-21
DB3	4.25E-18	4.71E-17	3.84E-17	9.89E-21
DB4	4.25E-18	3.19E-17	2.60E-17	6.92E-21
DB5	4.25E-18	2.85E-17	2.31E-17	6.05E-21
DB6	4.25E-18	2.85E-17	2.31E-17	6.05E-21
DB7	4.25E-18	2.12E-17	1.71E-17	4.48E-21
B1	7.67E-20	0.00E+00	0.00E+00	0.00E+00
B2	7.67E-20	0.00E+00	0.00E+00	0.00E+00
B3	7.67E-20	0.00E+00	0.00E+00	0.00E+00
B4	7.67E-20	0.00E+00	0.00E+00	0.00E+00
A21	1.44E-19	0.00E+00	0.00E+00	0.00E+00
A22	1.44E-19	0.00E+00	0.00E+00	0.00E+00
A23	1.44E-19	0.00E+00	0.00E+00	0.00E+00
A24	1.44E-19	0.00E+00	0.00E+00	0.00E+00
WB1	6.41E-16	5.12E-17	2.37E-17	0.00E+00
WB2	6.41E-16	5.12E-17	2.37E-17	0.00E+00
WB3	6.41E-16	5.12E-17	2.37E-17	0.00E+00

Table 5.38. Baseline CHEMDAT7 sources and emissions

Area[a]	Total area (cm^2)	No. of sources	Receptor no.	X (m)	Y (m)	Emissions rate (g/m^2/s)
49-Trench	69,675,000					3.646E-10
		10	1	244	366	3.646E-10
			2	274	366	3.646E-10
			3	244	396	3.646E-10
			4	274	396	3.646E-10
			5	305	396	3.646E-10
			6	244	427	3.646E-10
			7	274	427	3.646E-10
			8	305	427	3.646E-10
			9	244	46	3.646E-10
			10	274	46	3.646E-10
West Disposal Area	92,900,000	14				4.778E-08
			11	244	244	4.778E-08
			12	213	27	4.778E-08
			13	244	274	4.778E-08
			14	183	305	4.778E-08
			15	213	305	4.778E-08
			16	244	305	4.778E-08
			17	122	335	4.778E-08
			18	152	335	4.778E-08
			19	183	335	4.778E-08
			20	213	366	4.778E-08
			21	122	366	4.778E-08
			22	152	366	4.778E-08
			23	183	366	4.778E-08
			24	152	396	4.778E-08
Central Disposal Area	11,612,500	3				5.692E-08
			25	427	396	5.692E-08
			26	457	396	5.692E-08
			27	488	396	5.692E-08
North-East Disposal Area	18,580,000	4				2.763E-10
			28	427	549	2.763E-10
			29	457	549	2.763E-10
			30	396	579	2.763E-10
			31	427	579	2.763E-10
17-Trench	20,902,500	3				1.595E-09
			32	305	701	1.595E-09

Table 5.38. (continued)

Area[a]	Total area (cm^2)	No. of sources	Receptor no.	X (m)	Y (m)	Emissions rate (g/m^2/s)
			33	305	732	1.595E-09
			34	305	762	1.595E-09
HATA	20,902,500	3				7.715E-10
			35	183	671	7.715E-10
			36	183	701	7.715E-10
			37	213	701	7.715E-10
Groundwater	27,870,000	4				2.763E-10
			38	427	671	2.763E-10
			39	427	701	2.763E-10
			40	46	701	2.763E-10
			41	427	732	2.763E-10
South Disposal Area	137,027,500	18				2.066E-08
			42	335	122	2.066E-08
			43	335	152	2.066E-08
			44	366	61	2.066E-08
			45	366	91	2.066E-08
			46	366	122	2.066E-08
			47	366	152	2.066E-08
			48	366	183	2.066E-08
			49	366	213	2.066E-08
			50	396	61	2.066E-08
			51	396	61	2.066E-08
			52	396	122	2.066E-08
			53	396	152	2.066E-08
			54	396	183	2.066E-08
			55	396	213	2.066E-08
			56	427	122	2.066E-08
			57	427	152	2.066E-08
			58	427	183	2.066E-08
			59	427	213	2.066E-08
South-Central Disposal Area	41,805,000	5				5.690E-08
			60	366	396	5.690E-08
			61	366	427	5.690E-08
			62	396	366	5.690E-08
			63	396	396	5.690E-08
			64	396	427	5.690E-08
Middle Central Disposal Area	74,320,000	9				4.922E-10
			65	274	488	4.922E-10

Table 5.38. (continued)

Area[a]	Total area (cm^2)	No. of sources	Receptor no.	X (m)	Y (m)	Emissions rate ($g/m^2/s$)
			66	274	518	4.922E-10
			67	305	457	4.922E-10
			68	305	488	4.922E-10
			69	305	518	4.922E-10
			70	305	549	4.922E-10
			71	335	427	4.922E-10
			72	335	457	4.922E-10
			73	335	488	4.922E-10
North-Central Disposal Area	23,225,000	2				7.715E-10
			74	213	640	7.715E-10
			75	244	640	7.715E-10
South-East Disposal Area	71,997,500	8				7.715E-10
			76	427	488	6.229E-09
			77	427	518	6.229E-09
			78	457	457	6.229E-09
			79	457	488	6.229E-09
			80	488	427	6.229E-09
			81	488	457	6.229E-09
			82	518	396	6.229E-09
			83	518	427	6.229E-09
North-West Auger Hole Area	55,740,000	6				2.762E-10
			84	366	610	2.762E-10
			85	366	640	2.762E-10
			86	366	671	2.762E-10
			87	396	610	2.762E-10
			88	396	640	2.762E-10
			89	396	671	2.762E-10

[a] Locations are shown in Fig. 5.87.

Table 5.39. Predicted concentrations of tritium and VOCs

Tritium

Period	Maximum boundary receptor Concentration (pg/m^3)	Maximum boundary receptor Concentration (pCi/m^3)	Maximum on-WAG receptor Concentration (pg/m^3)	Maximum on-WAG receptor Concentration (pCi/m^3)
1989 (present)	620	5,952,000	64,670	620,832,000
1990-2019	109	1,046,400	12,326	118,329,600
1990-2059	64	614,400	7,077	67,939,200
2100-2129	0.05	480	6	57,600

VOCs

Case	Maximum boundary receptor Concentration (pg/m^3)	95 percentile of mean, boundary Concentration (pg/m^3)	Maximum on-WAG receptor Concentration (pg/m^3)	95 percentile of mean, on-WAG Concentration (pg/m^3)
Baseline	168,450	26,450	435,000	28,800

Section 5 Figures

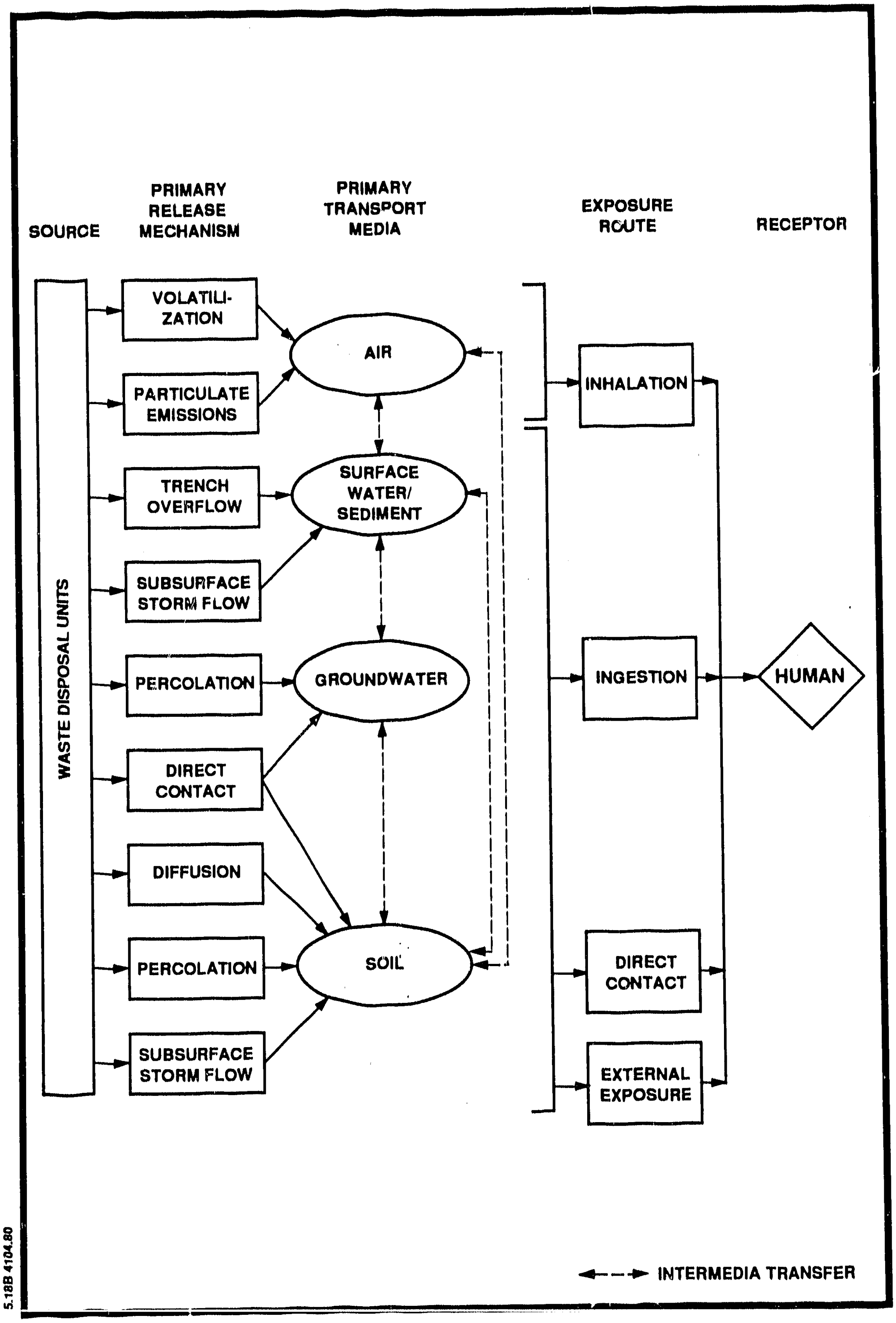


Fig. 5.1. Site conceptual model.

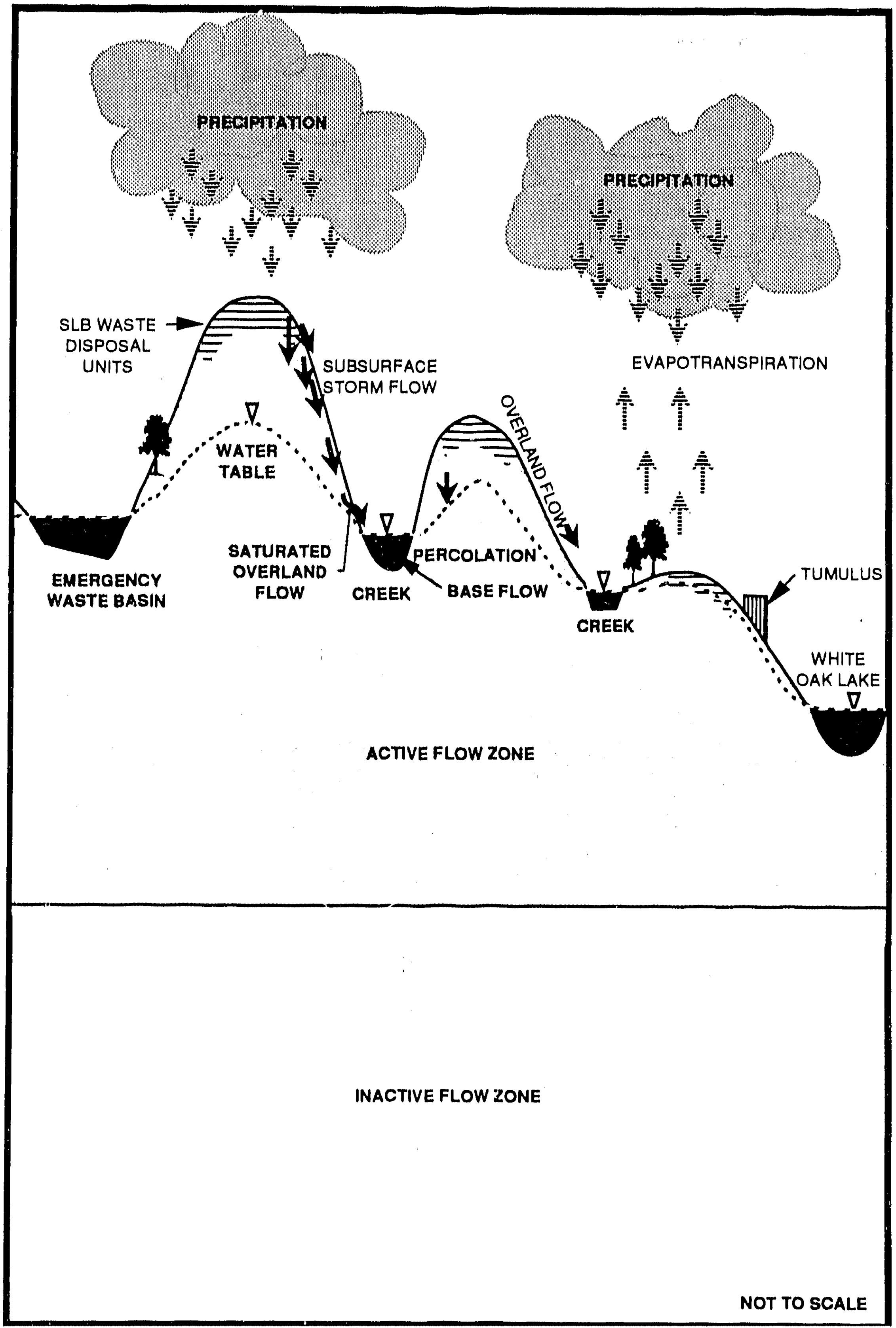


Fig. 5.2. Hydrologic features of WAG 6.

5 18 4104.16

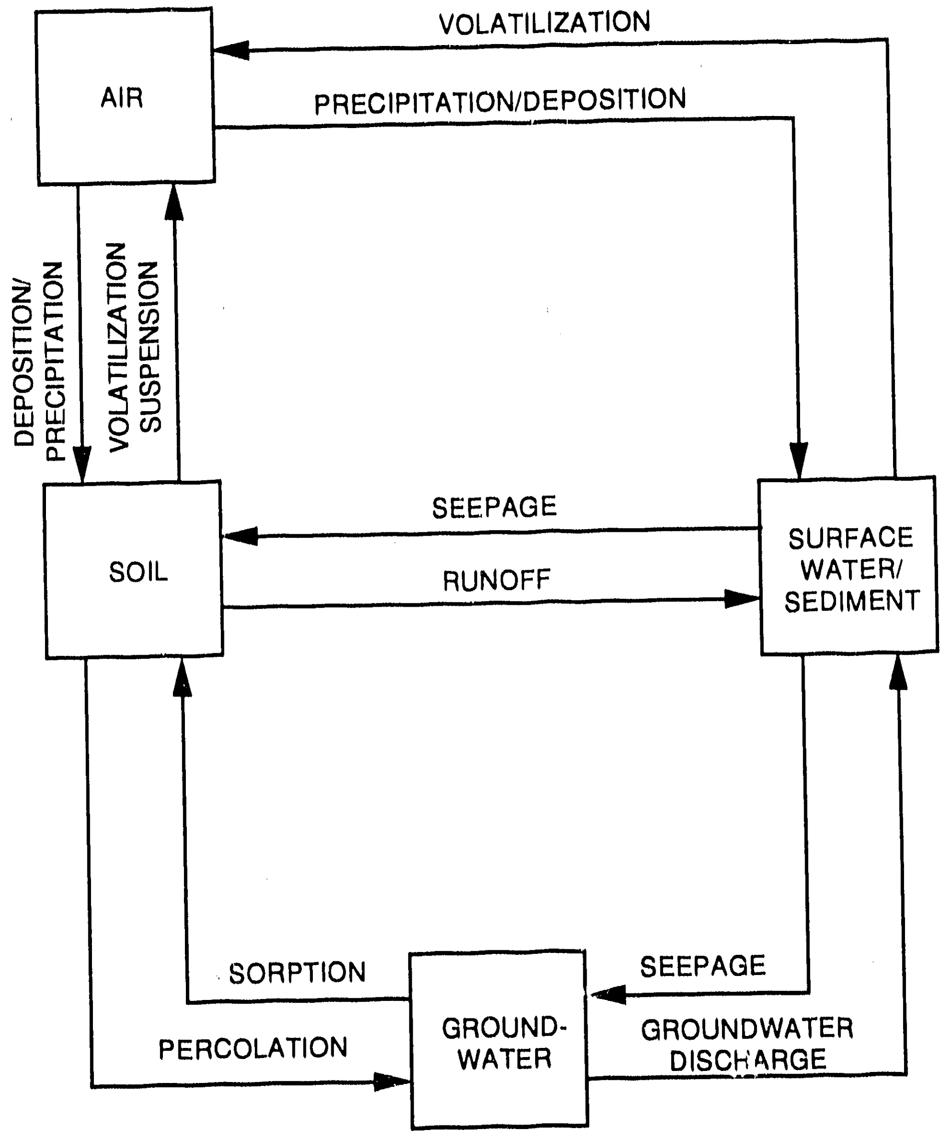
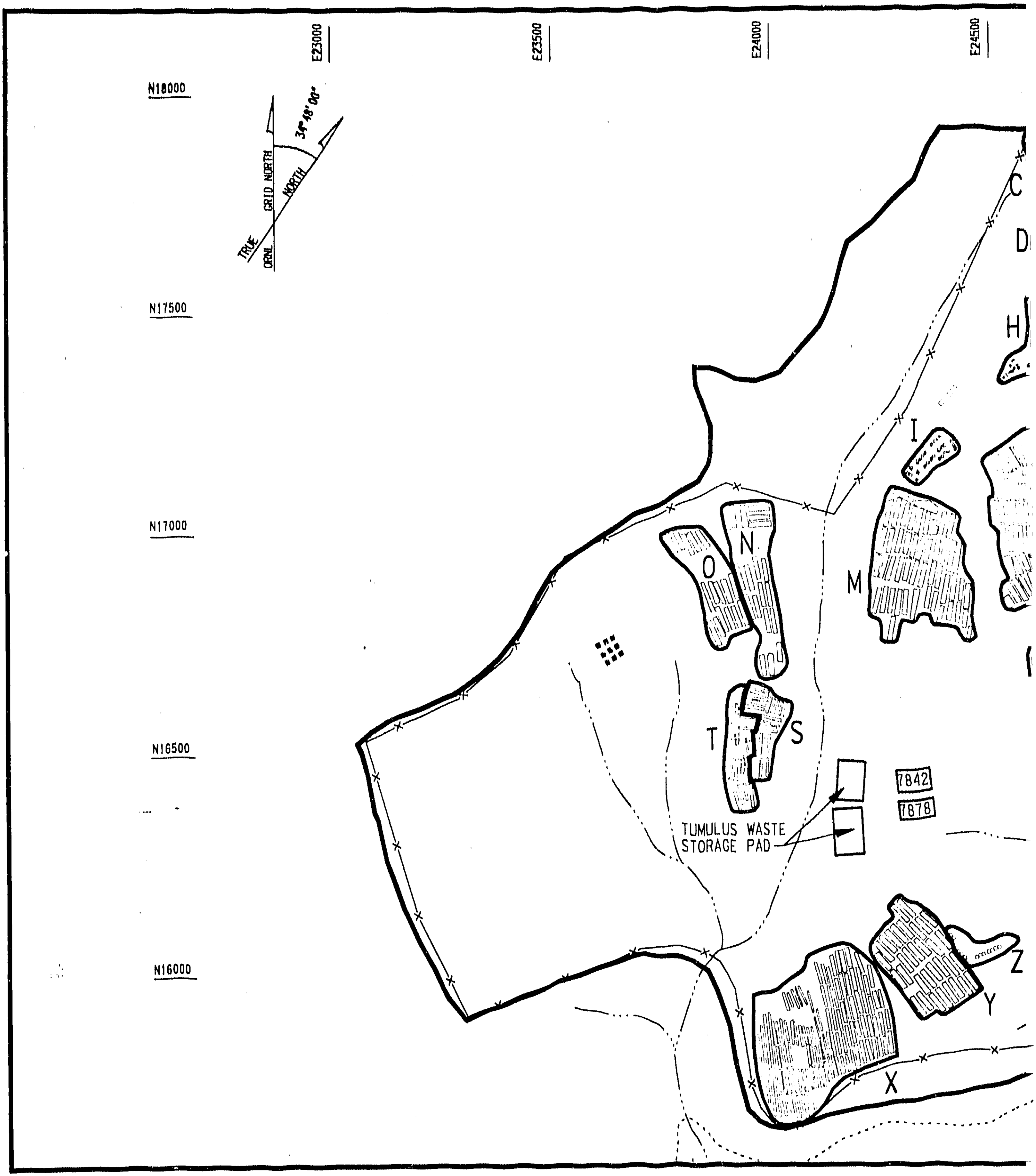
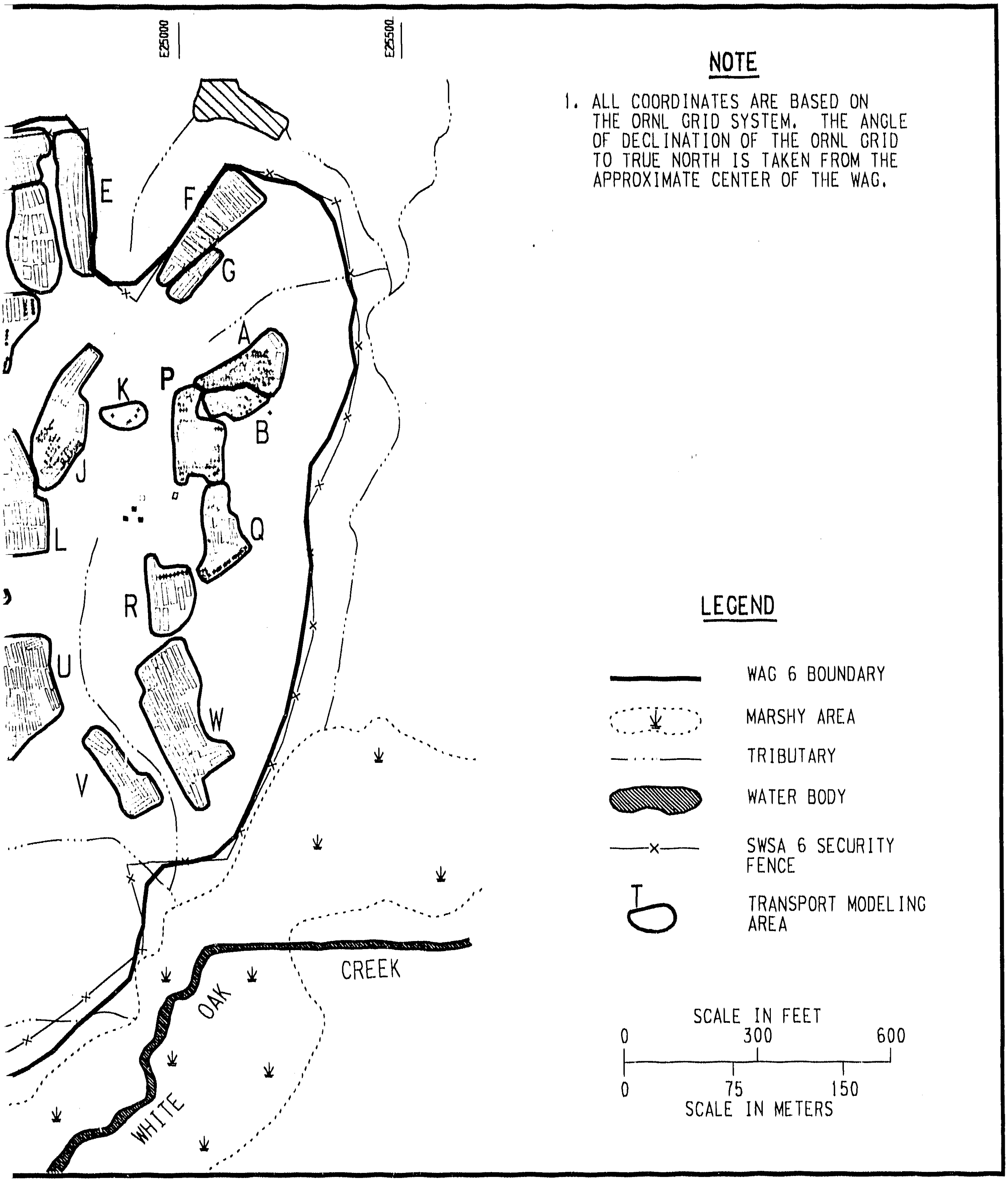


Fig. 5.3. Intermedia transfer processes.

5.18 4104.17



WAG6 06F367.DGN



NOTE

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.

Fig. 5.4. WAG 6 transport modeling areas.

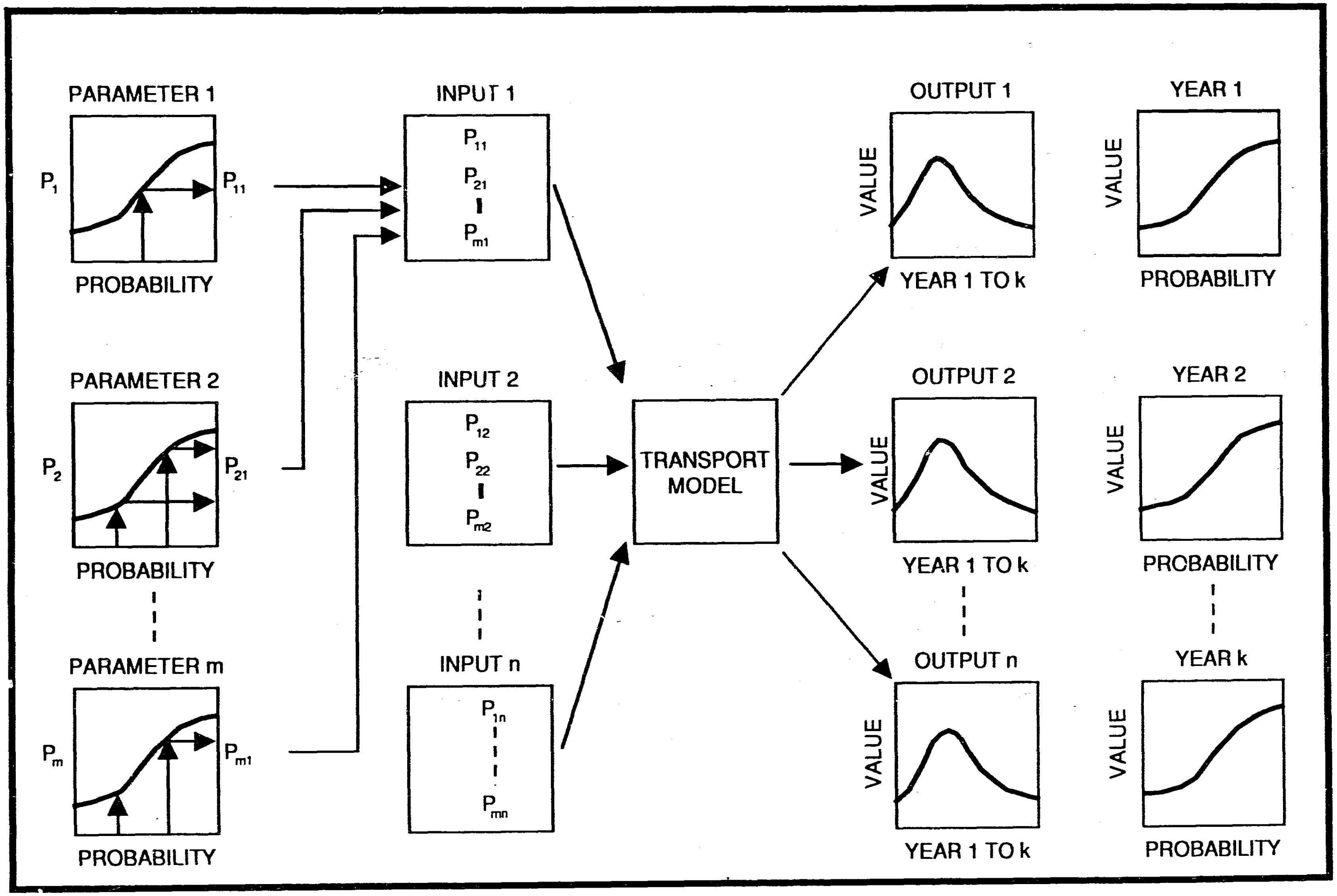


Fig. 5.5. Monte Carlo simulation.

5.18A 4104.36

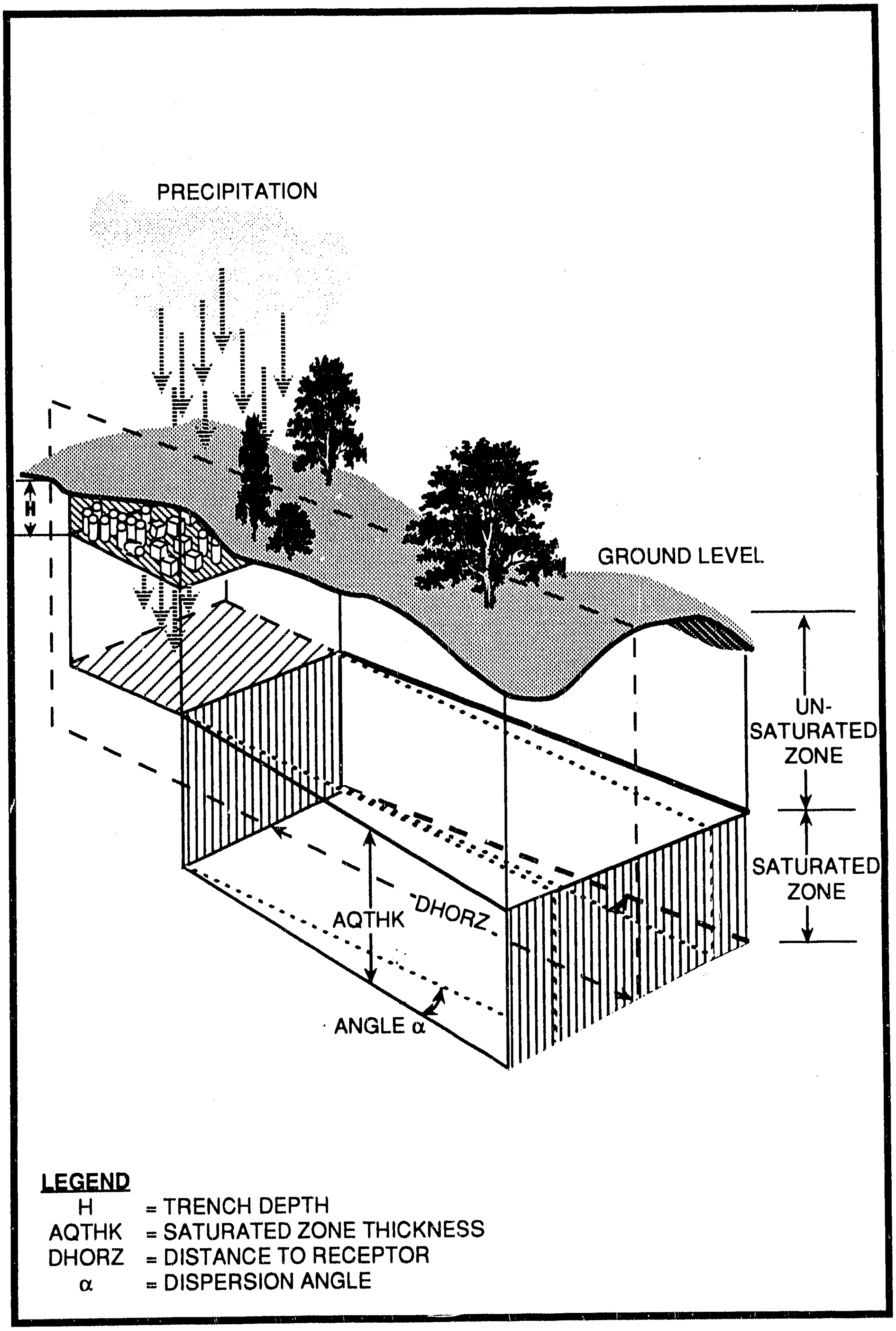


Fig. 5.6. Schematic diagram of the transport model.

5.18A 4104.33

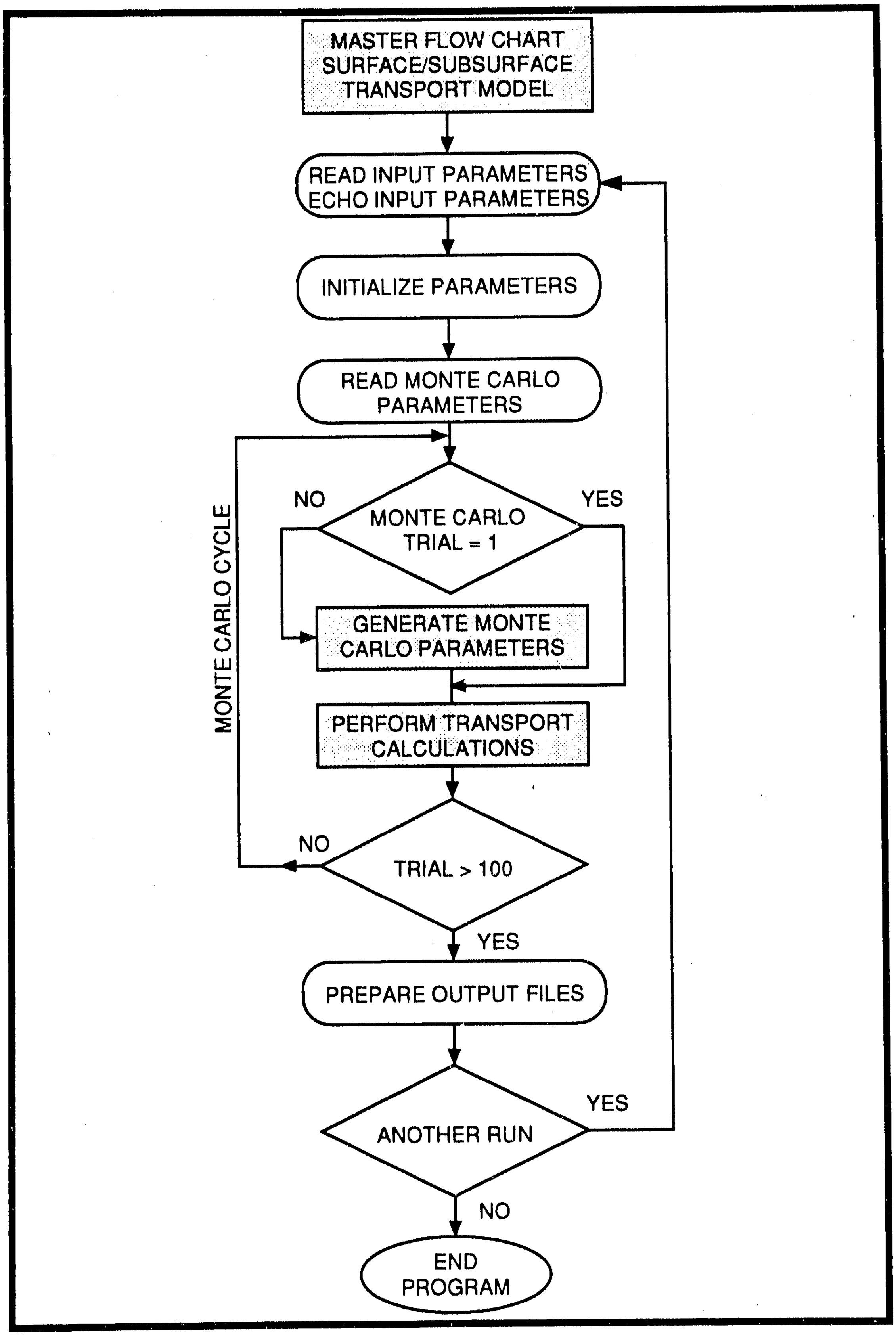


Fig. 5.7. General structure of the model.

5.18A 4104.34

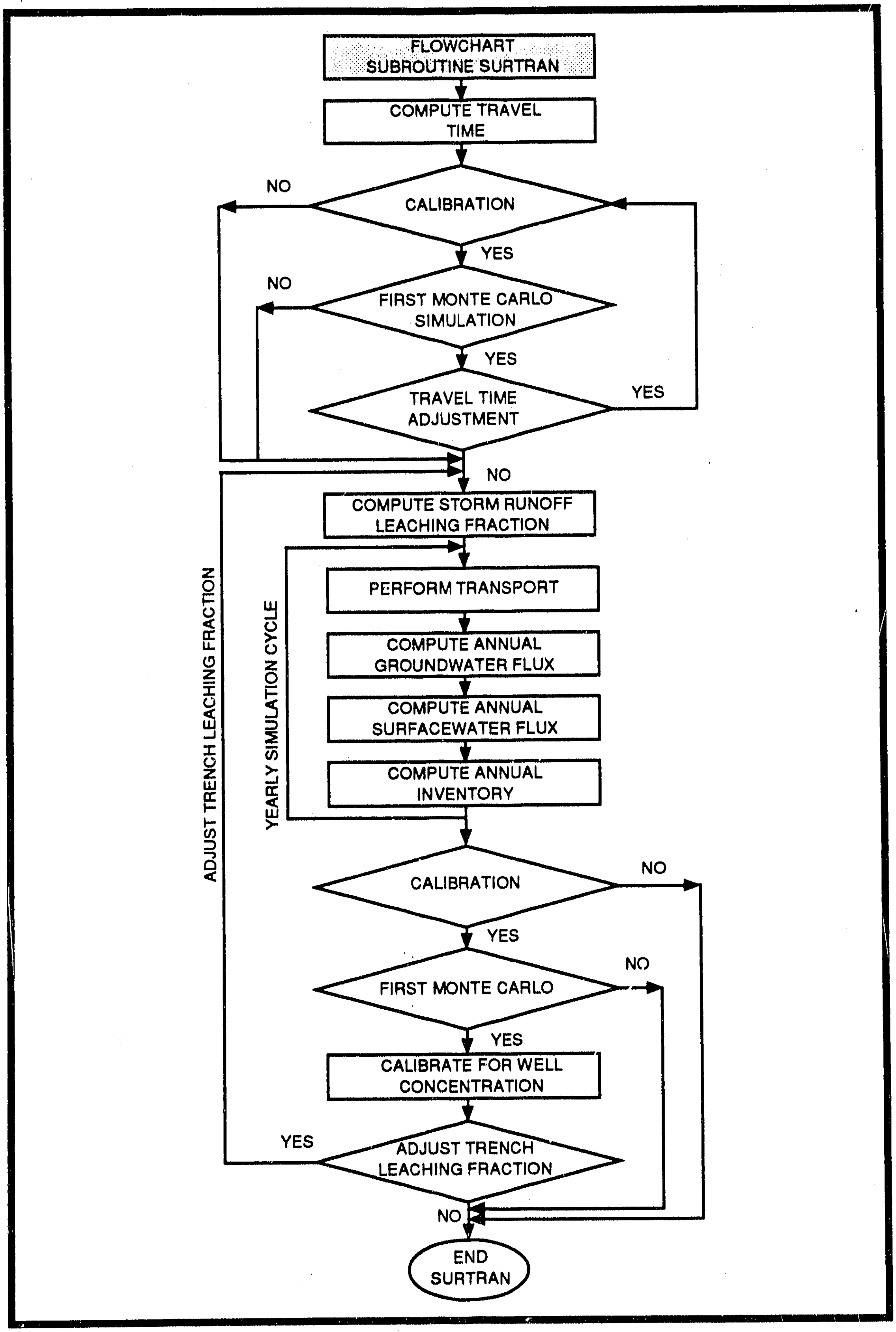


Fig. 5.8. Flow chart of the SURTRAN subroutine.

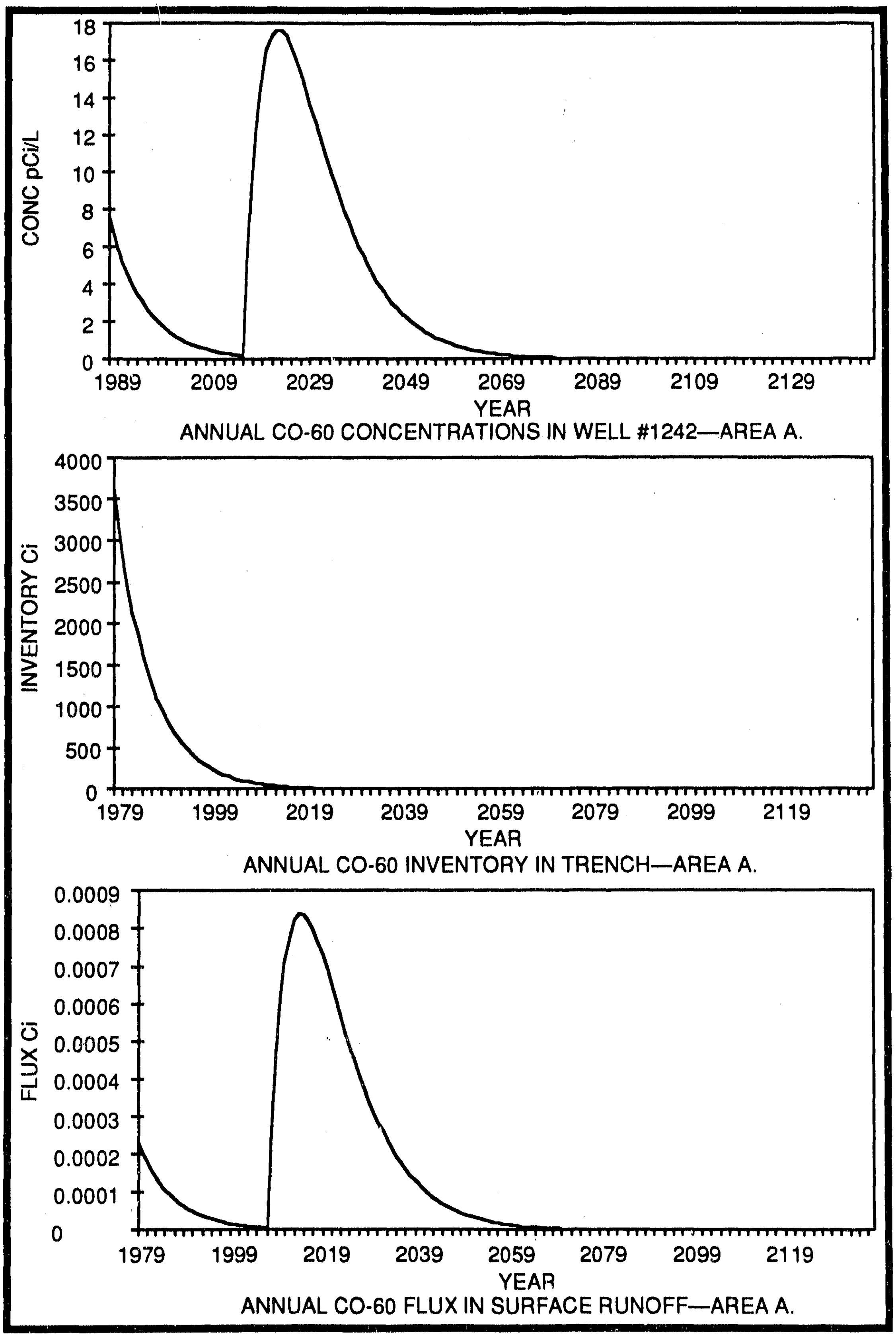
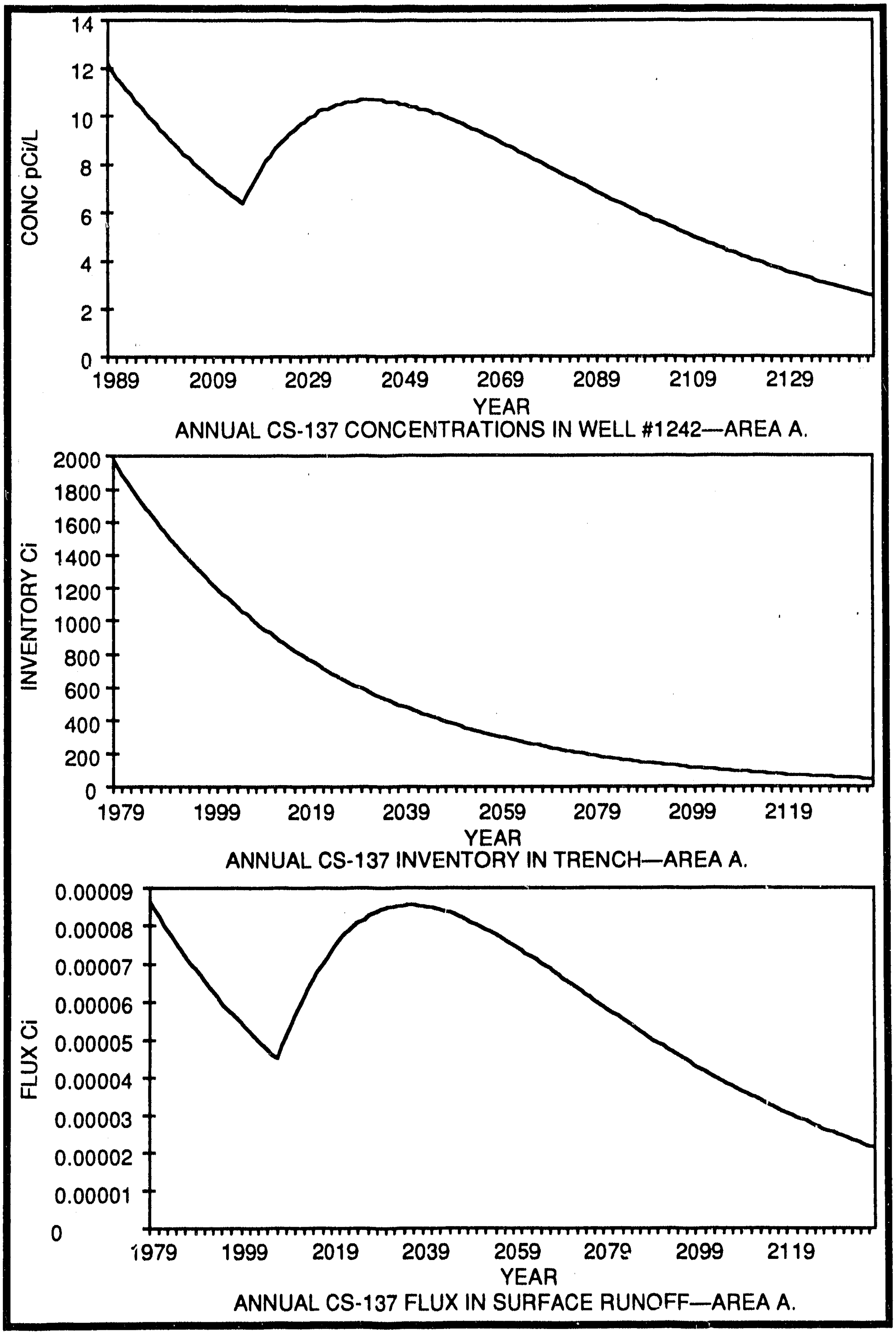


Fig. 5.9. Modeling output for cobalt-60, Area A.

5.18A 4104.37

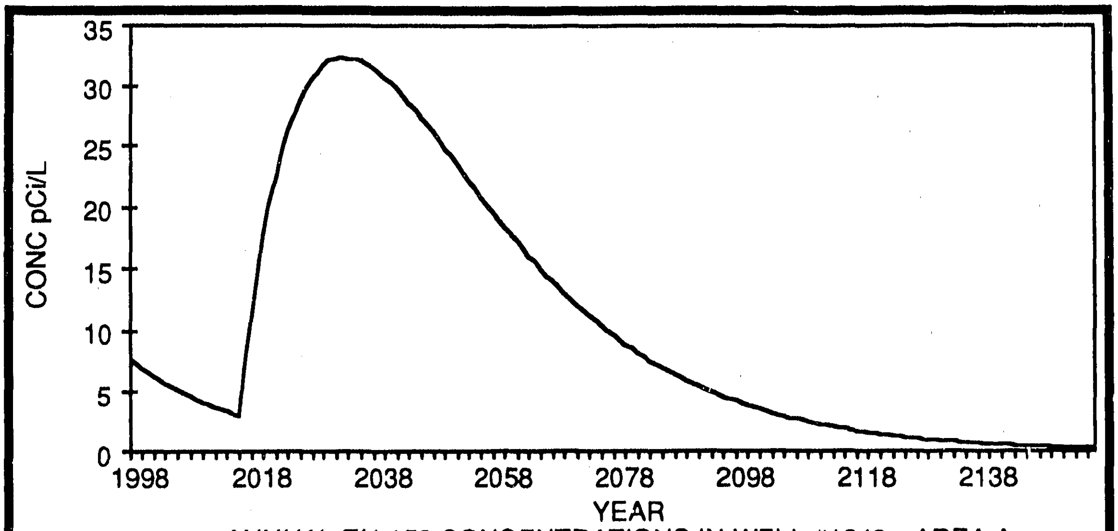


ANNUAL CS-137 CONCENTRATIONS IN WELL #1242—AREA A.

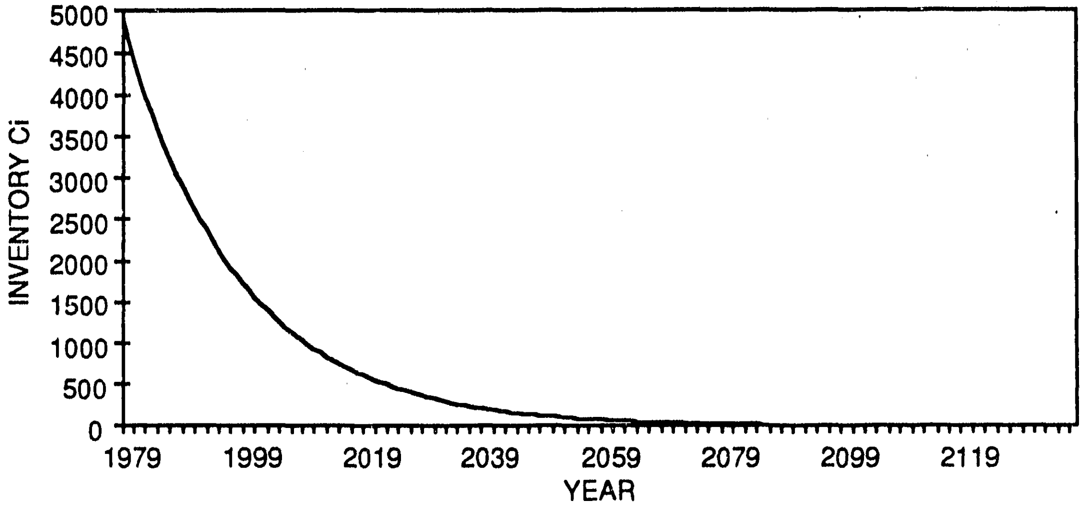
ANNUAL CS-137 INVENTORY IN TRENCH—AREA A.

ANNUAL CS-137 FLUX IN SURFACE RUNOFF—AREA A.

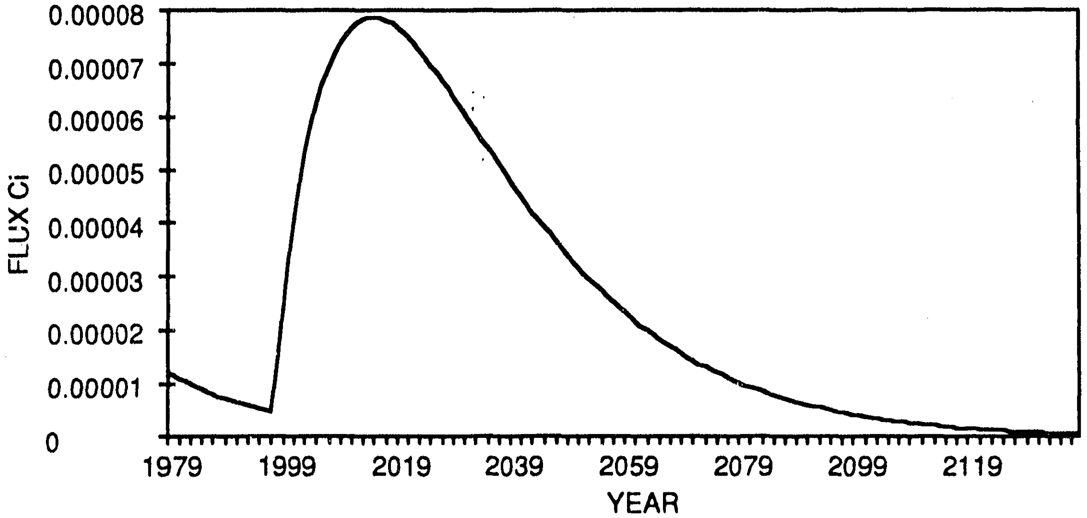
Fig. 5.10. Modeling output for cesium-137, Area A.



ANNUAL EU-152 CONCENTRATIONS IN WELL #1242—AREA A.



ANNUAL EU-152 INVENTORY IN TRENCH —AREA A.



ANNUAL EU-152 FLUX IN SURFACE RUNOFF—AREA A.

Fig. 5.11. Modeling output for europium-152, Area A.

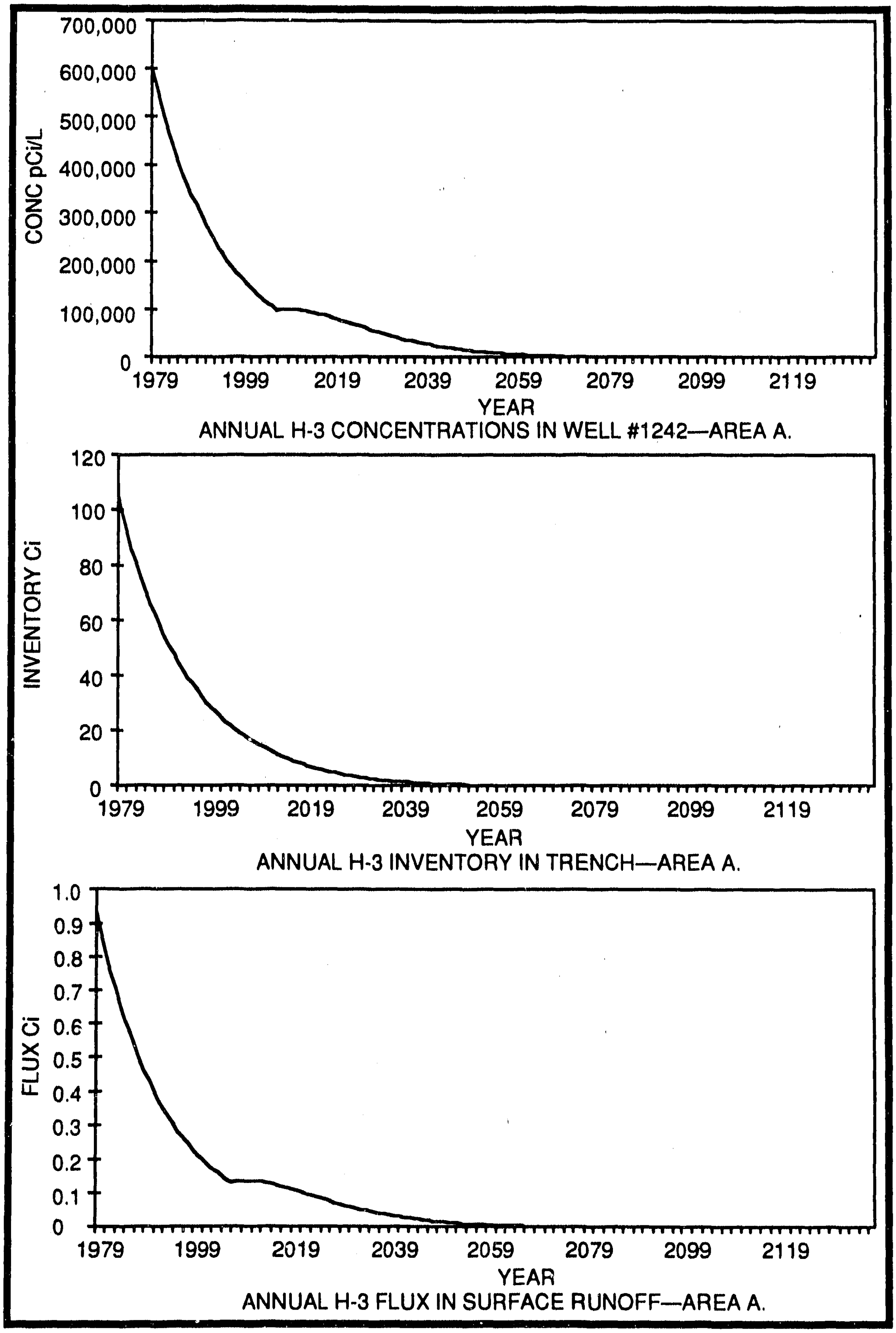
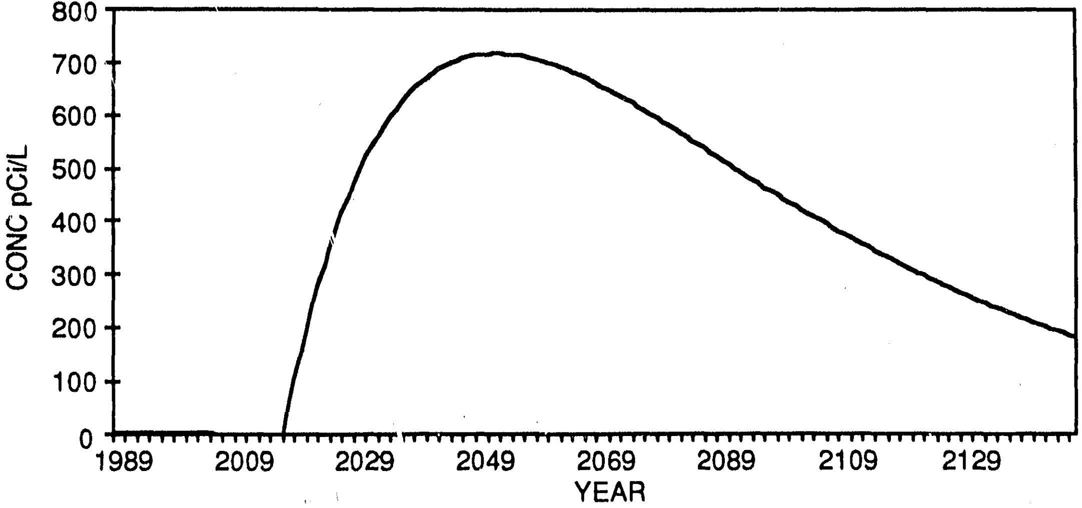
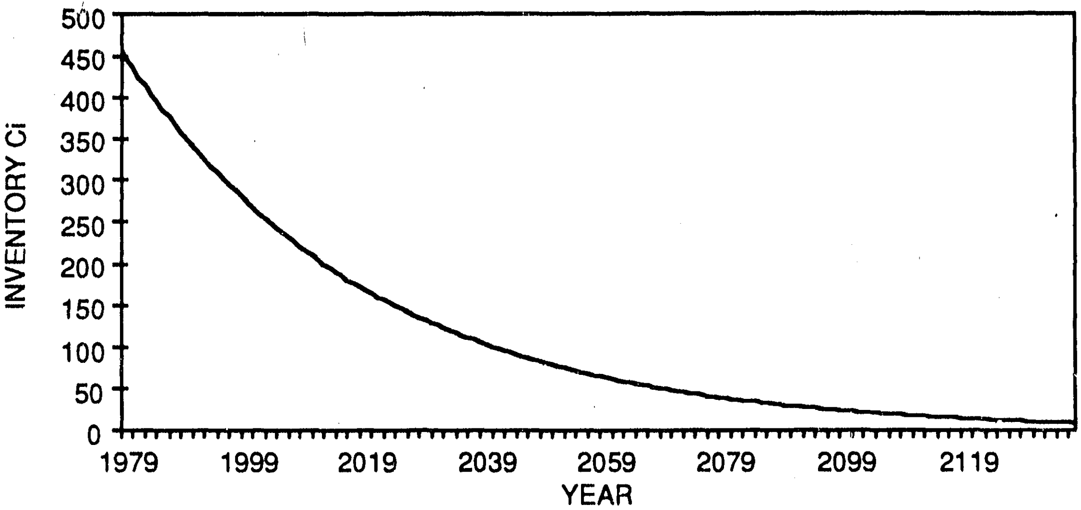


Fig. 5.12. Modeling output for tritium, Area A.

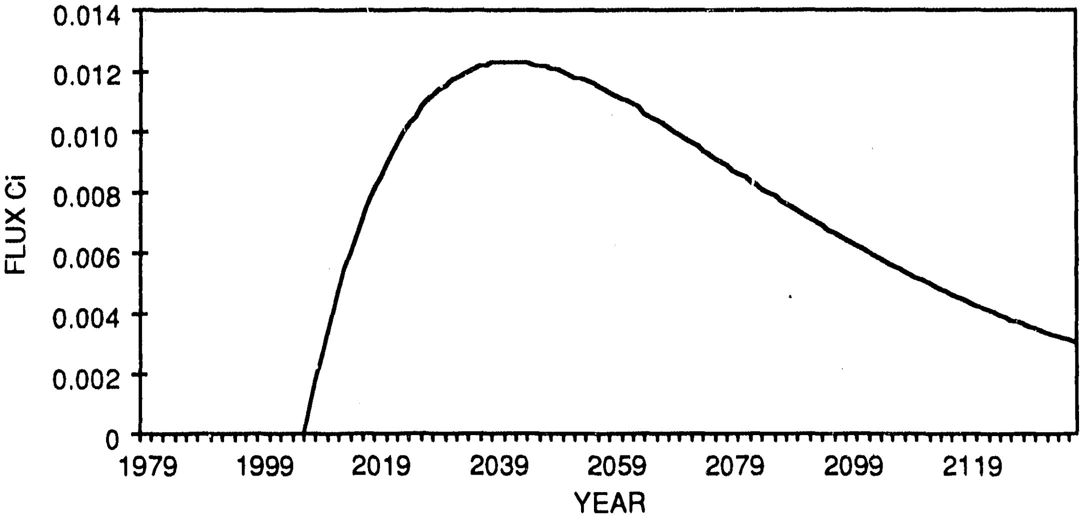
5.18A 4104.40



ANNUAL SR-90 CONCENTRATIONS IN WELL #1242—AREA A.



ANNUAL SR-90 INVENTORY IN TRENCH—AREA A.



ANNUAL SR-90 FLUX IN SURFACE RUNOFF—AREA A.

Fig. 5.13 Modeling output for strontium-90, Area A.

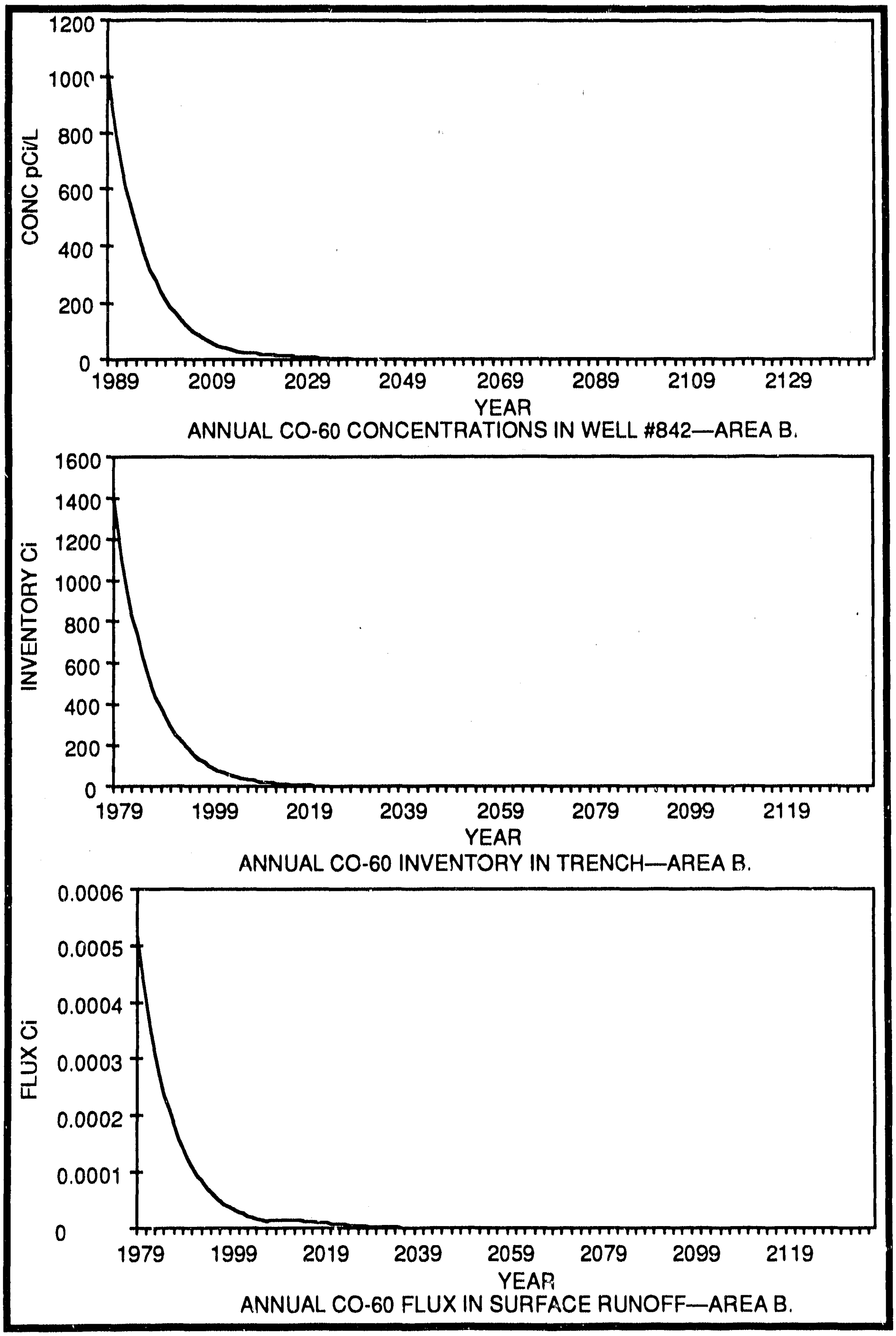
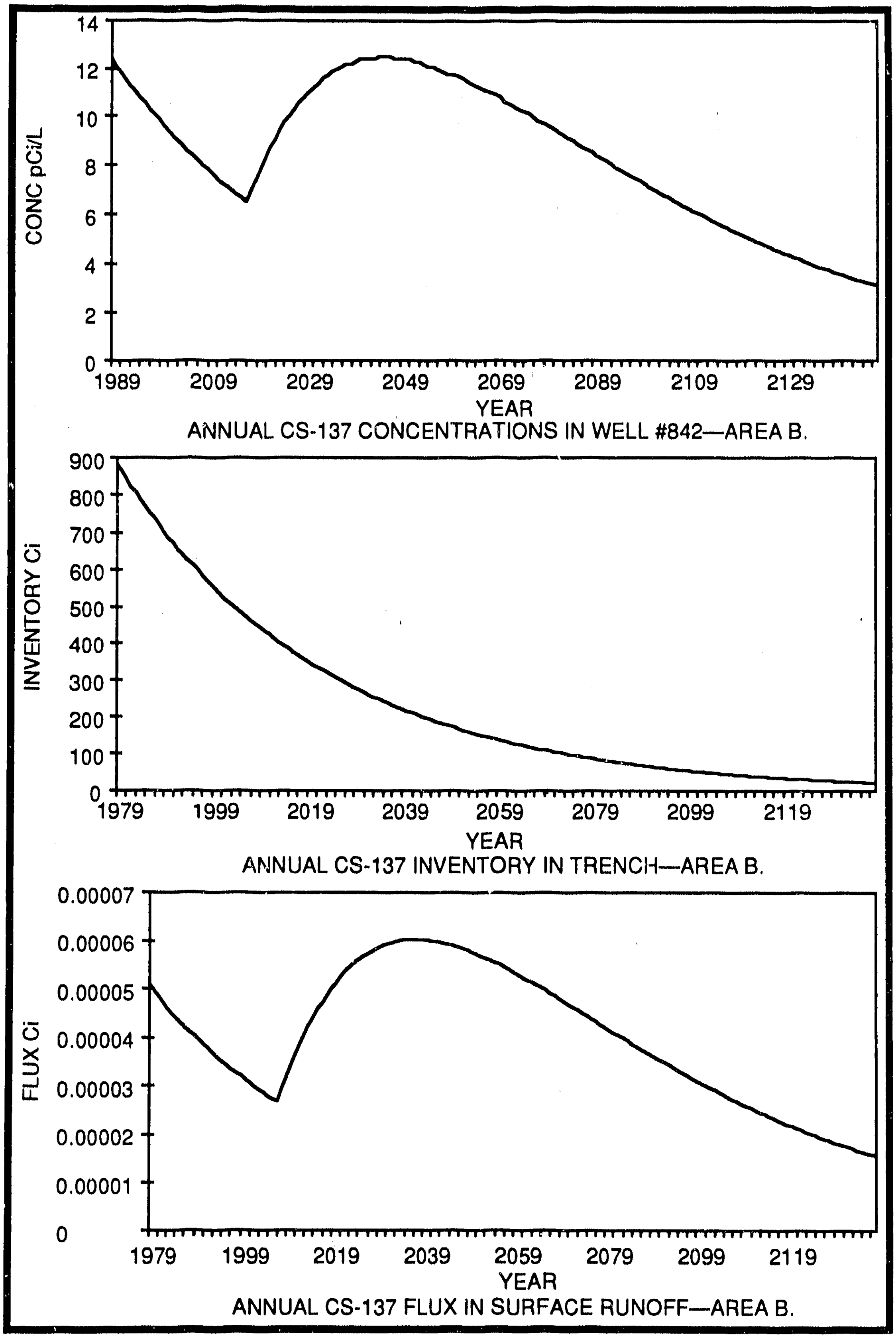


Fig. 5.14. Modeling output for cobalt-60, Area B.

5.18A 4104.42



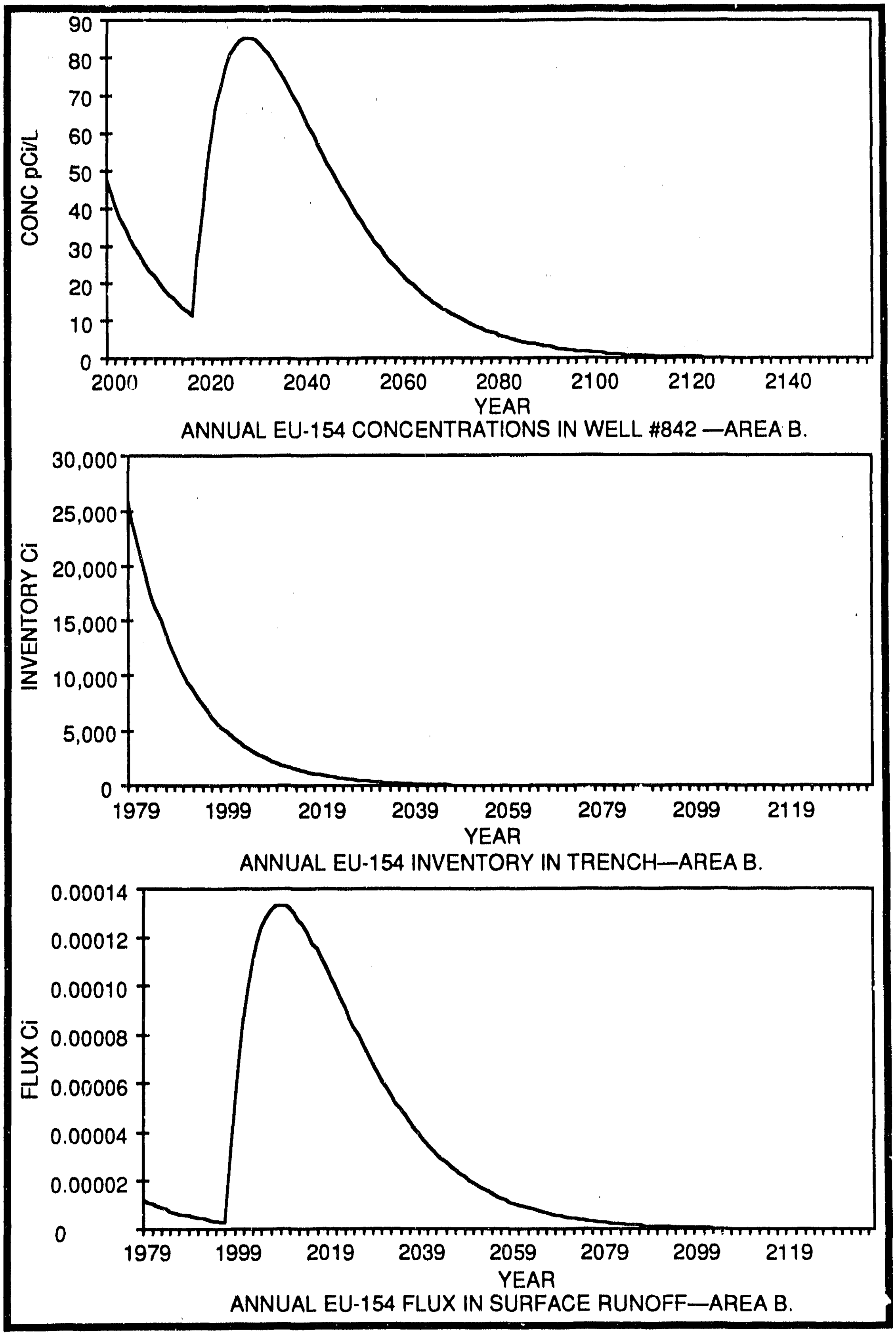
ANNUAL CS-137 CONCENTRATIONS IN WELL #842—AREA B.

ANNUAL CS-137 INVENTORY IN TRENCH—AREA B.

ANNUAL CS-137 FLUX IN SURFACE RUNOFF—AREA B.

Fig. 5.15. Modeling output for cesium-137, Area B.

5.18A 4104.43

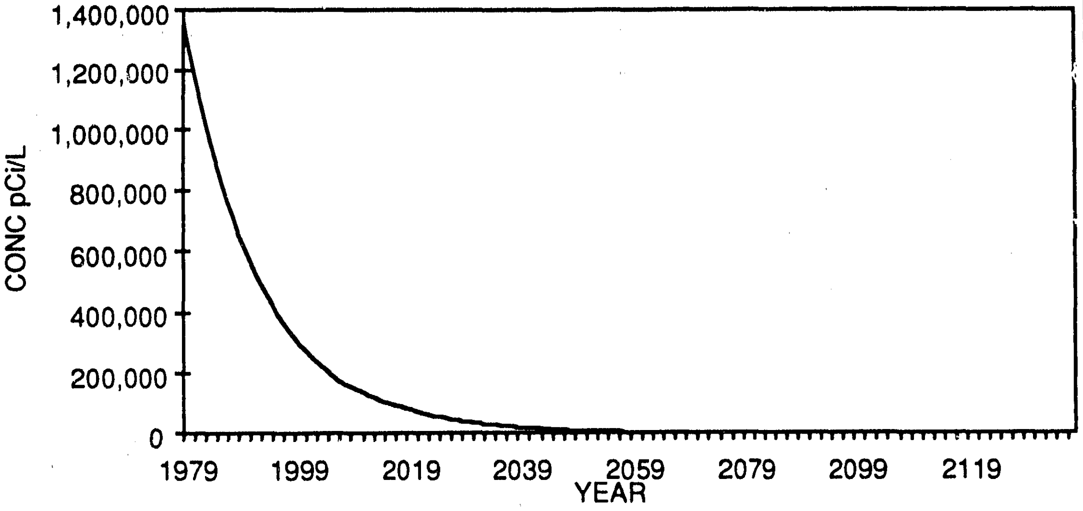


ANNUAL EU-154 CONCENTRATIONS IN WELL #842 —AREA B.

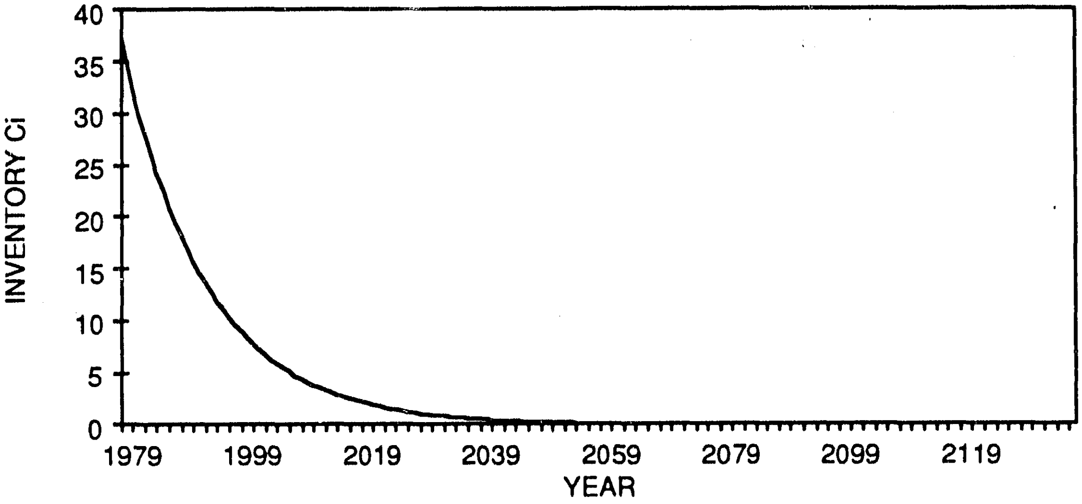
ANNUAL EU-154 INVENTORY IN TRENCH—AREA B.

ANNUAL EU-154 FLUX IN SURFACE RUNOFF—AREA B.

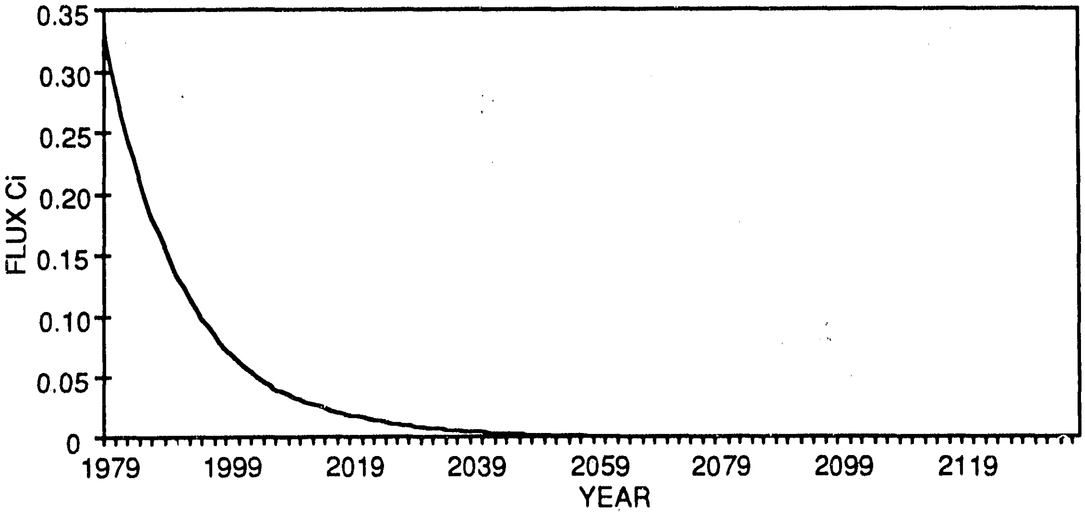
Fig. 5.16. Modeling output for europium-154, Area B.



ANNUAL H-3 CONCENTRATIONS IN WELL #842—AREA B.

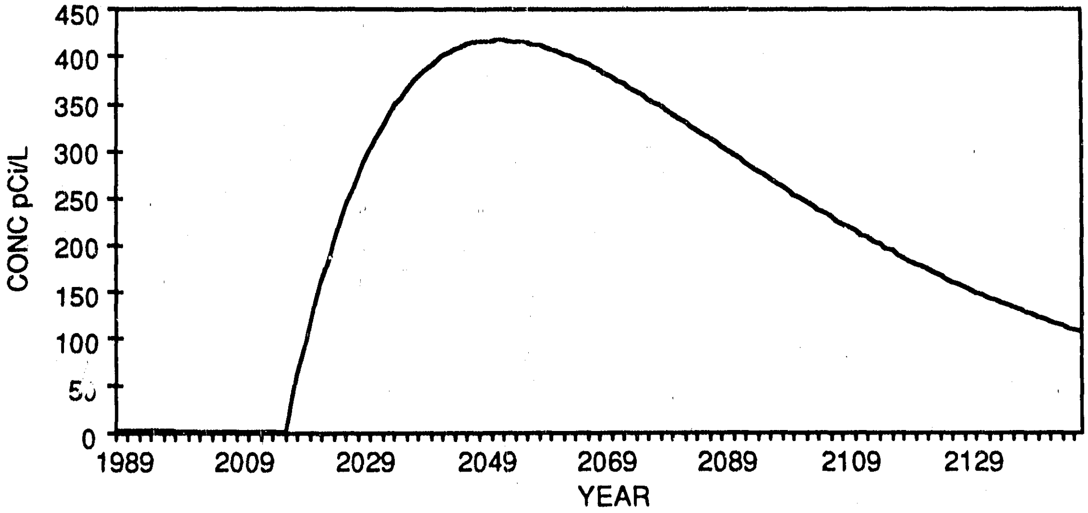


ANNUAL H-3 INVENTORY IN TRENCH—AREA B.

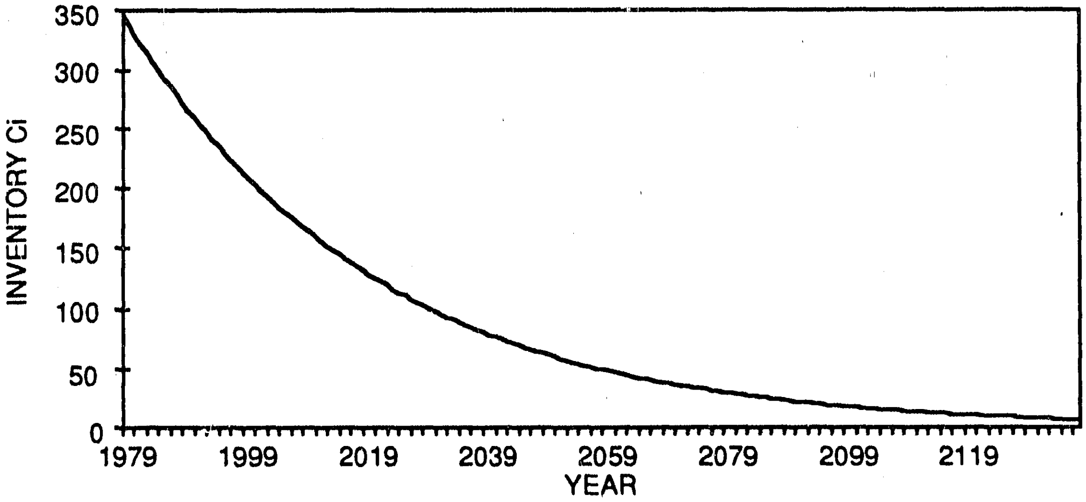


ANNUAL H-3 FLUX IN SURFACE RUNOFF—AREA B.

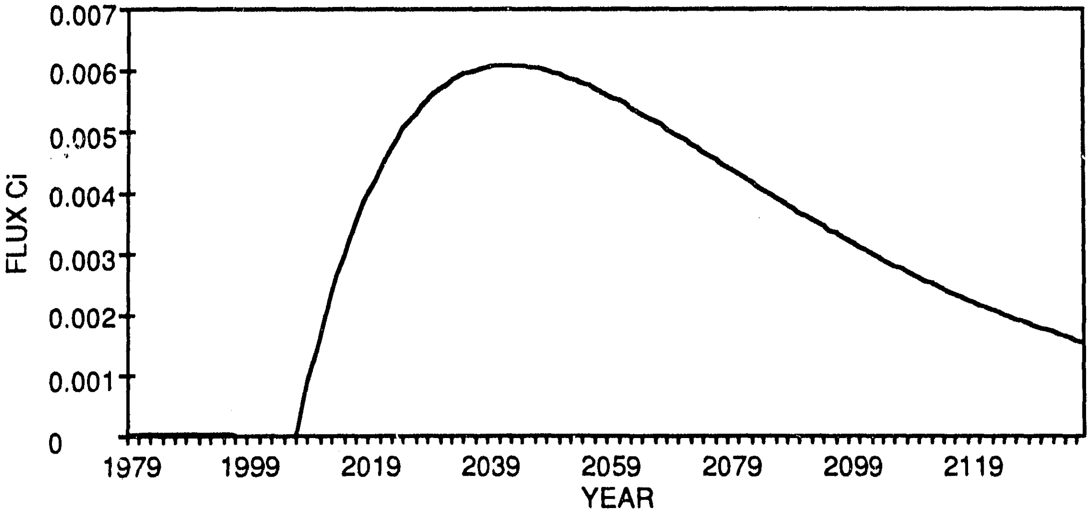
Fig. 5.17. Modeling output for tritium, Area B.



ANNUAL SR-90 CONCENTRATIONS IN WELL #842—AREA B.



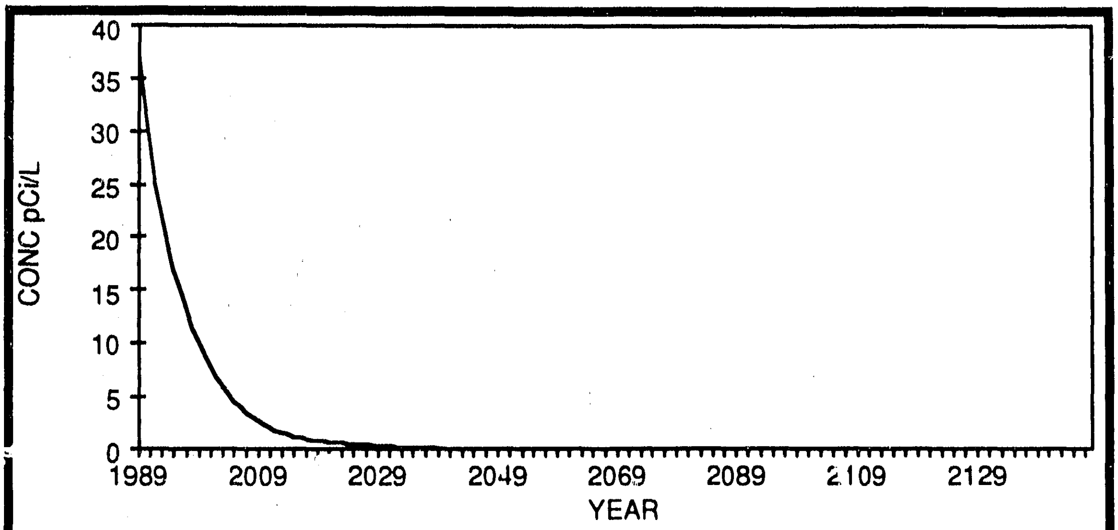
ANNUAL SR-90 INVENTORY IN TRENCH—AREA B.



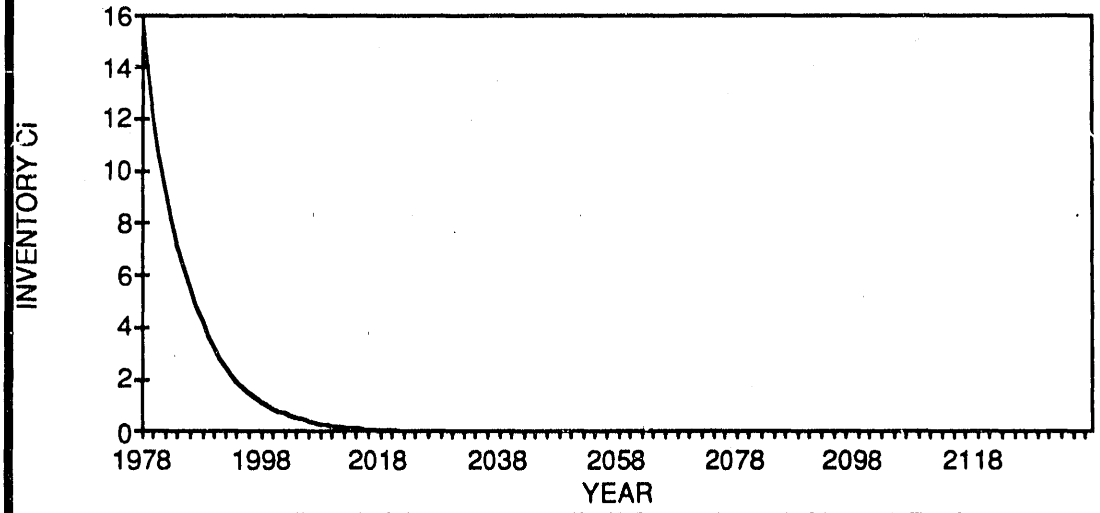
ANNUAL SR-90 FLUX IN SURFACE RUNOFF—AREA B.

Fig. 5.18. Modeling output for strontium-90, Area B.

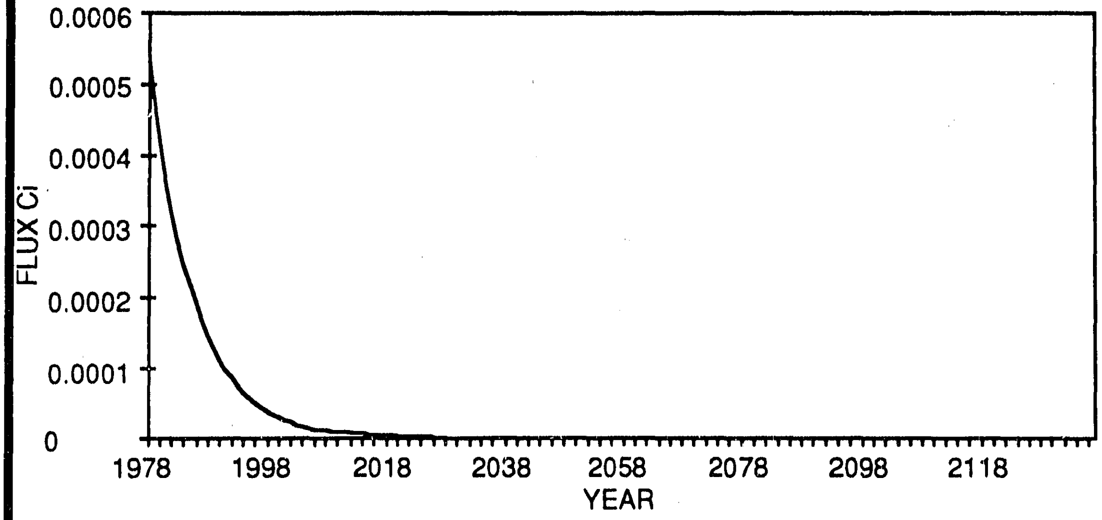
5.18A 4104.46



ANNUAL COBALT-60 CONCENTRATIONS IN WELL #1225—AREA C.



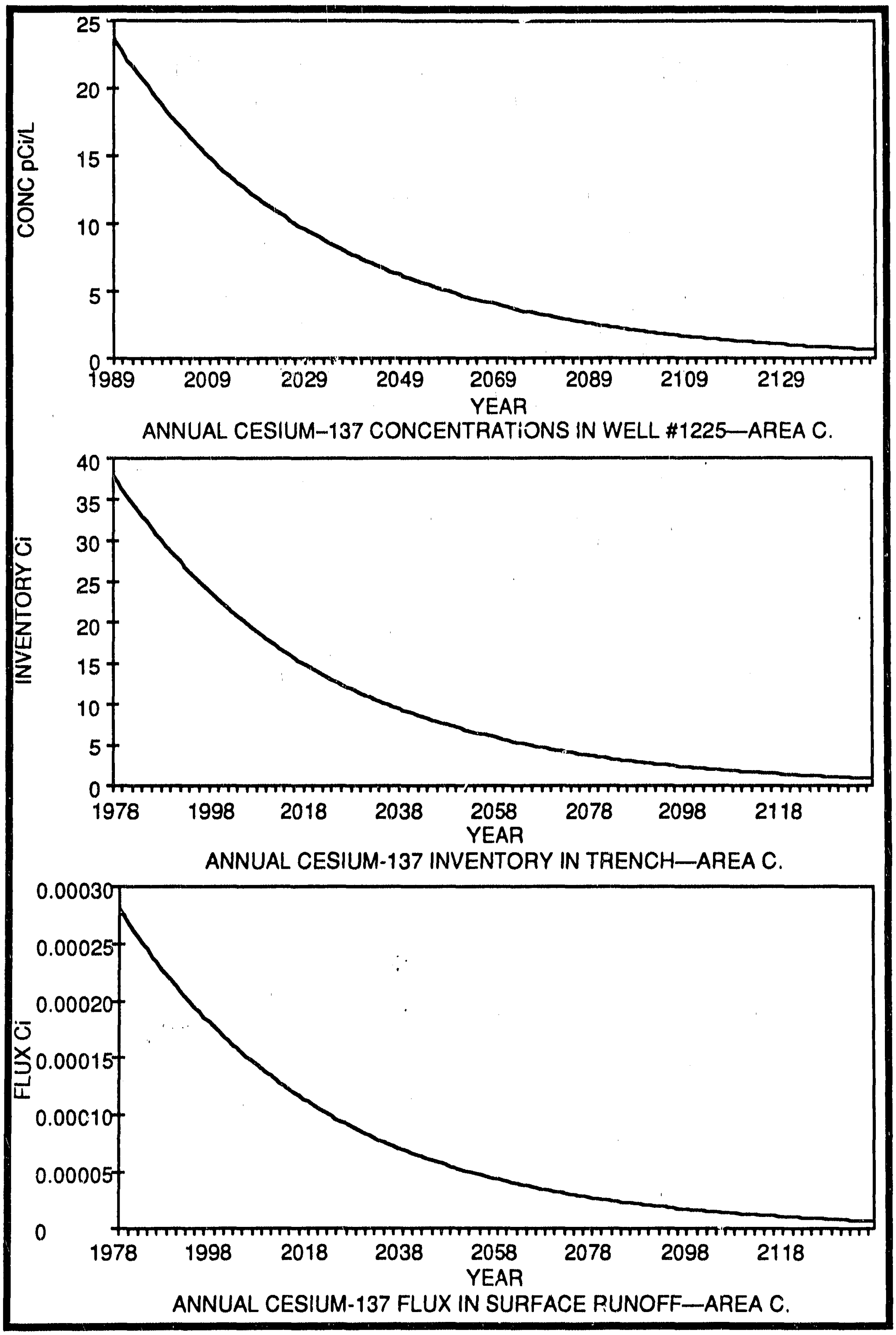
ANNUAL COBALT-60 INVENTORY IN TRENCH—AREA C.



ANNUAL COBALT-60 FLUX IN SURFACE RUNOFF—AREA C.

Fig. 5.19. Modeling output for cobalt-60, Area C.

5.18A 4104.47

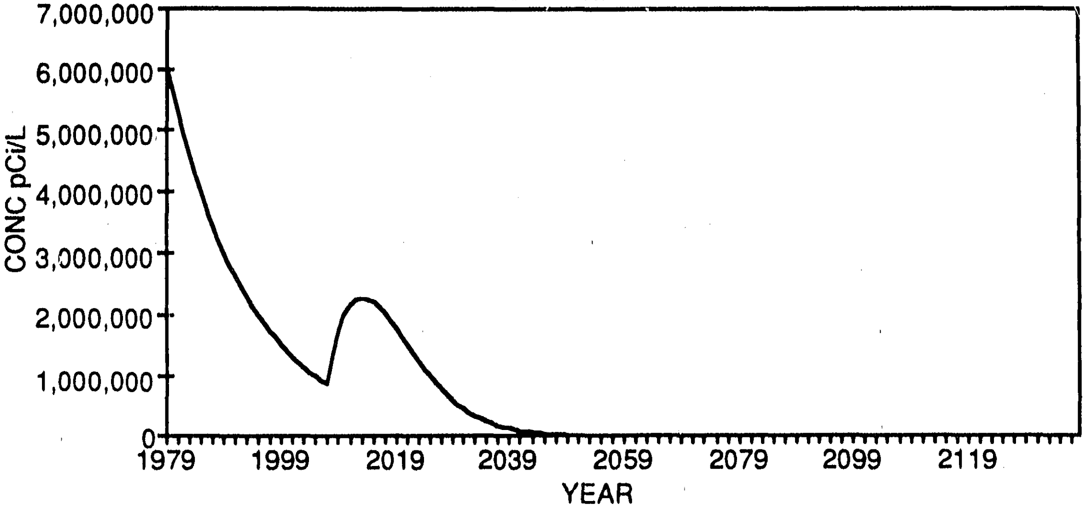

ANNUAL CESIUM-137 CONCENTRATIONS IN WELL #1225—AREA C.

ANNUAL CESIUM-137 INVENTORY IN TRENCH—AREA C.

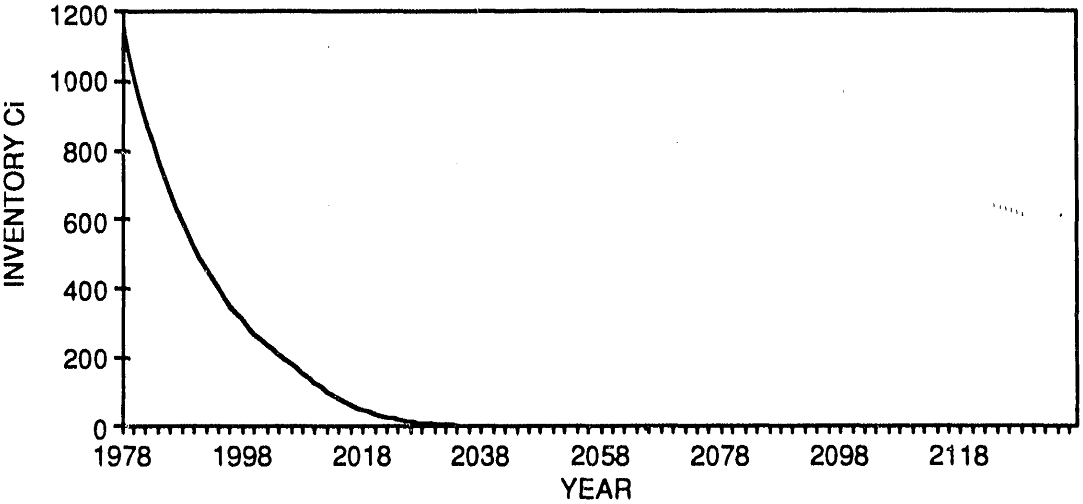
ANNUAL CESIUM-137 FLUX IN SURFACE RUNOFF—AREA C.

Fig. 5.20. Modeling output for cesium-137, Area C.

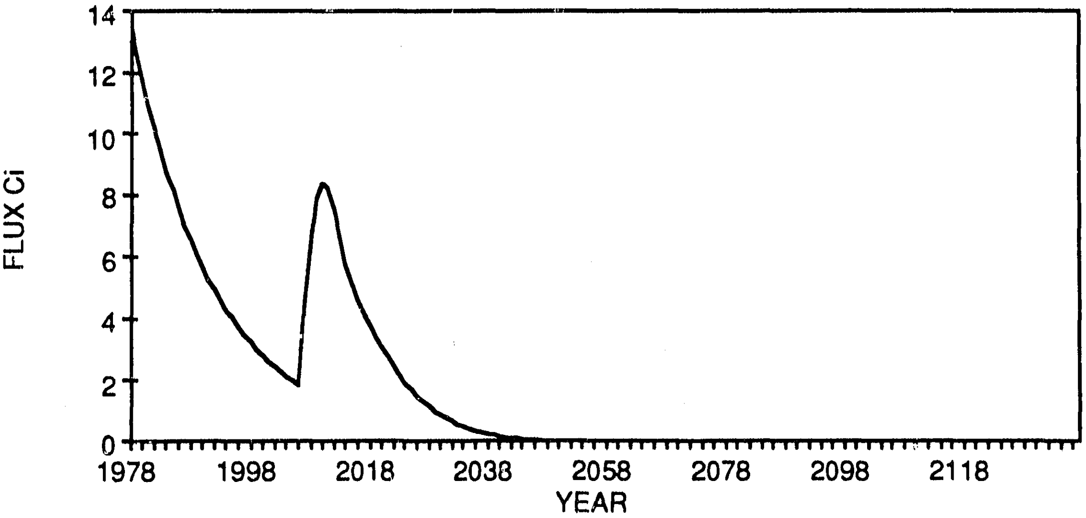
5.18A 4104.48



ANNUAL H-3 CONCENTRATIONS IN WELL #1225—AREA C.



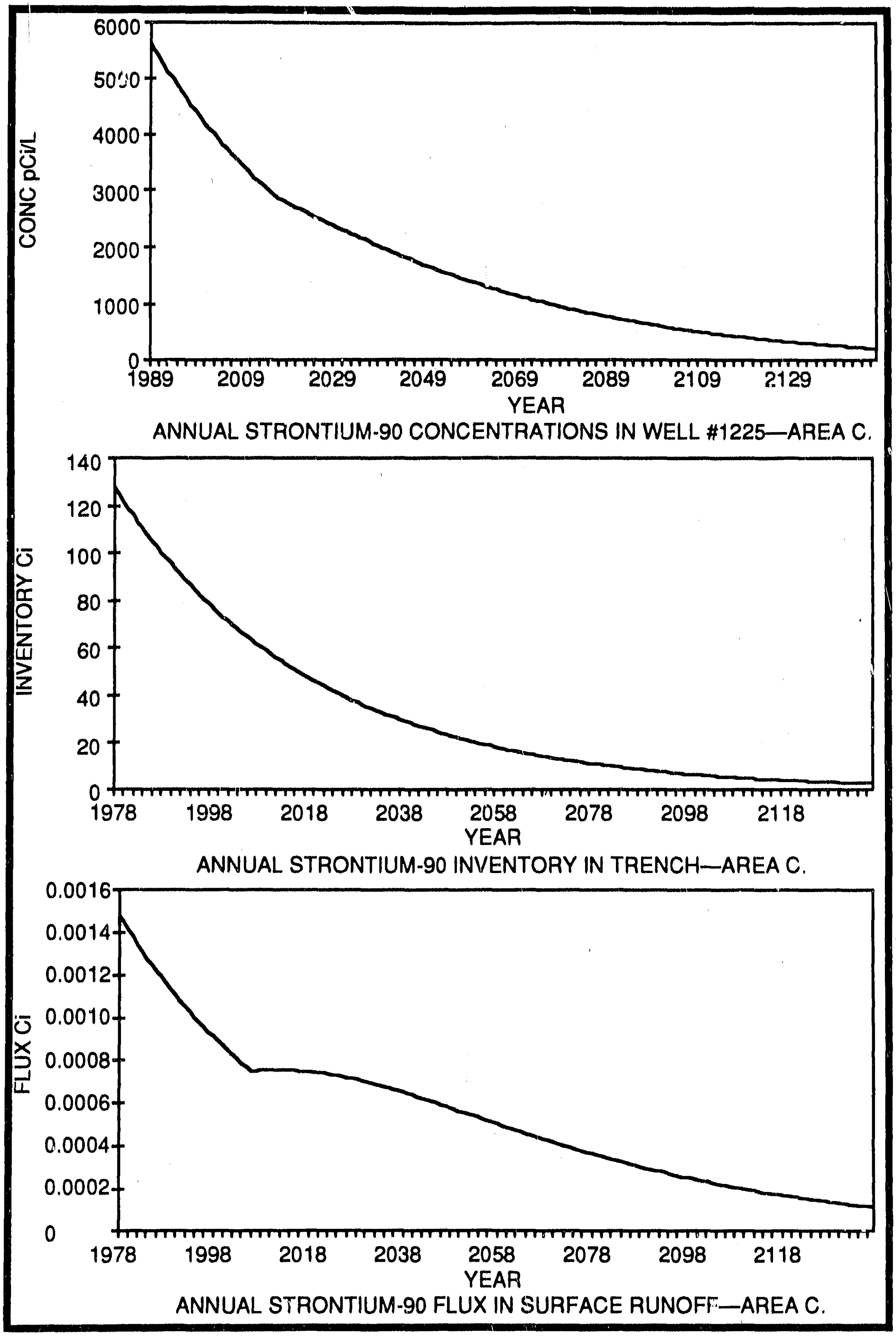
ANNUAL H-3 INVENTORY IN TRENCH—AREA C.



ANNUAL H-3 FLUX IN SURFACE RUNOFF—AREA C.

Fig. 5.21. Modeling output for tritium, Area C.

5.18A 4104.49



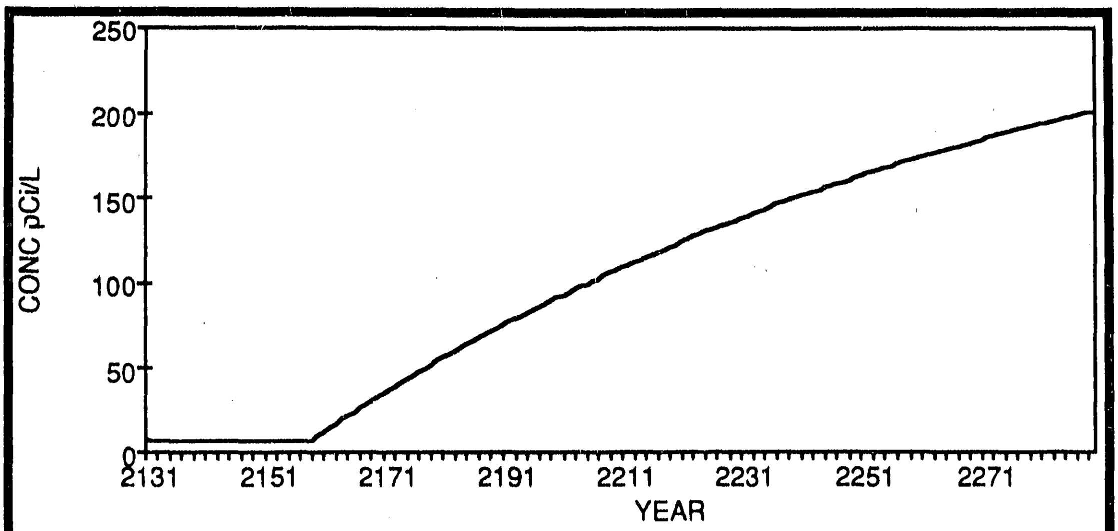
ANNUAL STRONTIUM-90 CONCENTRATIONS IN WELL #1225—AREA C.

ANNUAL STRONTIUM-90 INVENTORY IN TRENCH—AREA C.

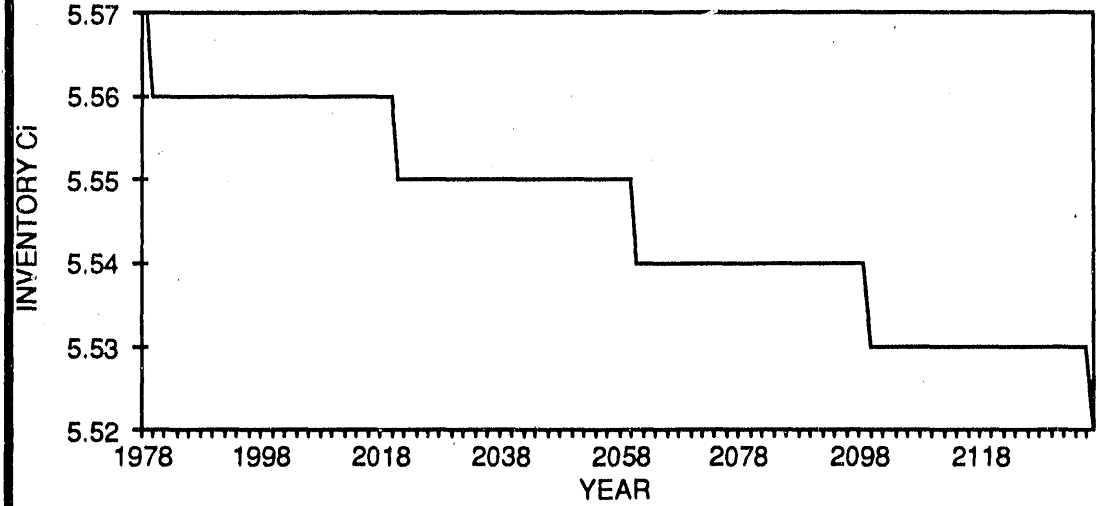
ANNUAL STRONTIUM-90 FLUX IN SURFACE RUNOFF—AREA C.

Fig. 5.22. Modeling output for strontium-90, Area C.

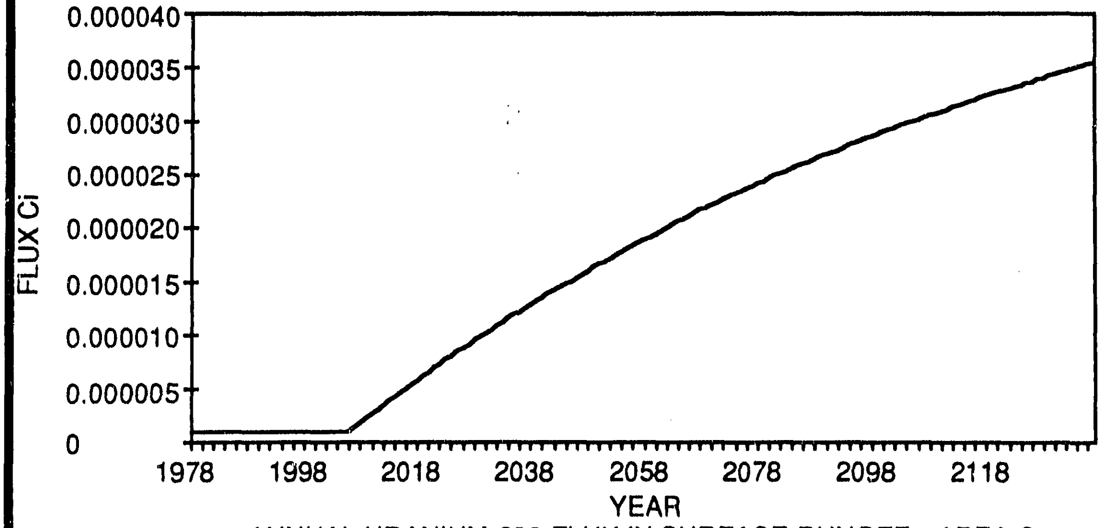
5.18A 4104.50



ANNUAL URANIUM-233 CONCENTRATION IN WELL #1225—AREA C.



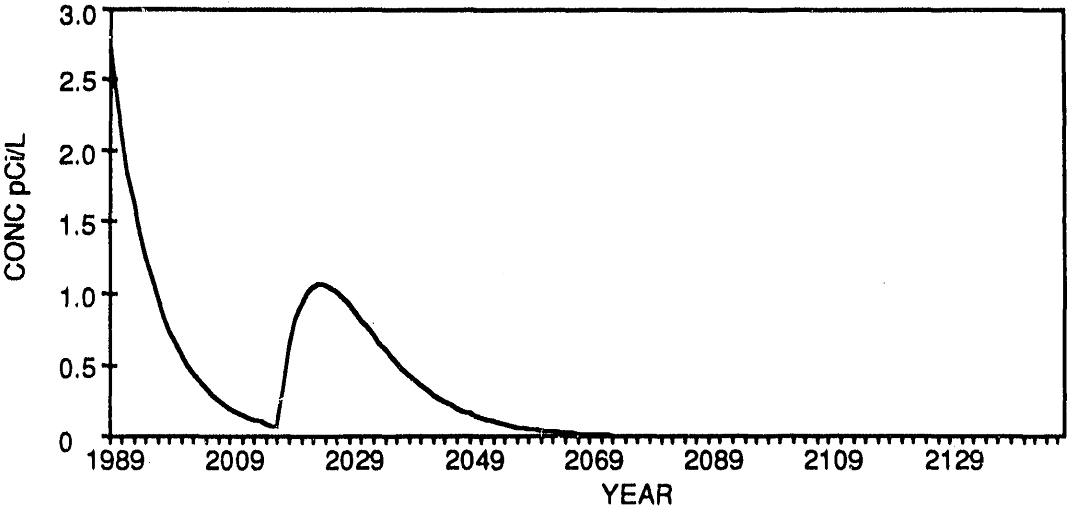
ANNUAL URANIUM-233 CONCENTRATION INVENTORY IN TRENCH—AREA C.



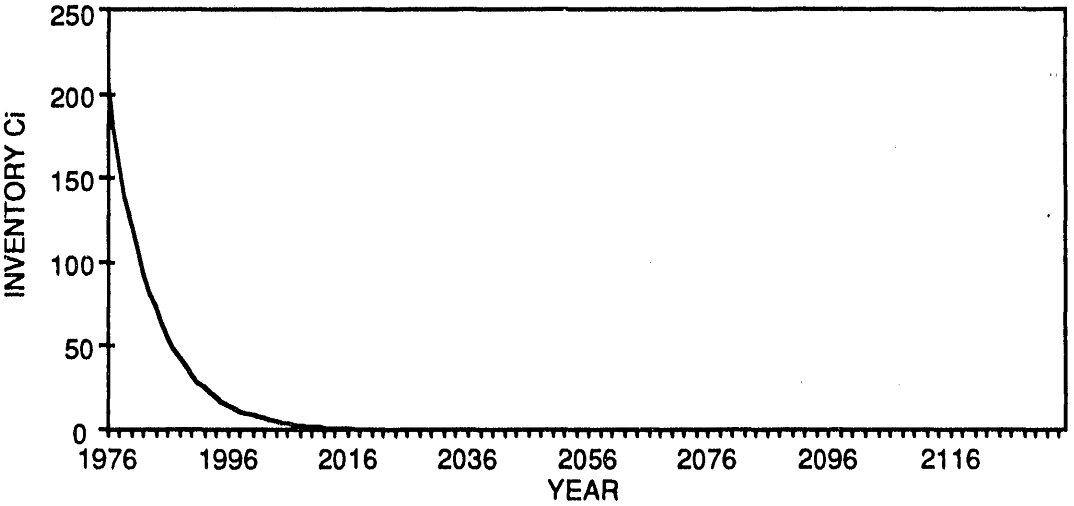
ANNUAL URANIUM-233 FLUX IN SURFACE RUNOFF—AREA C.

Fig. 5.23. Modeling output for uranium-233, Area C.

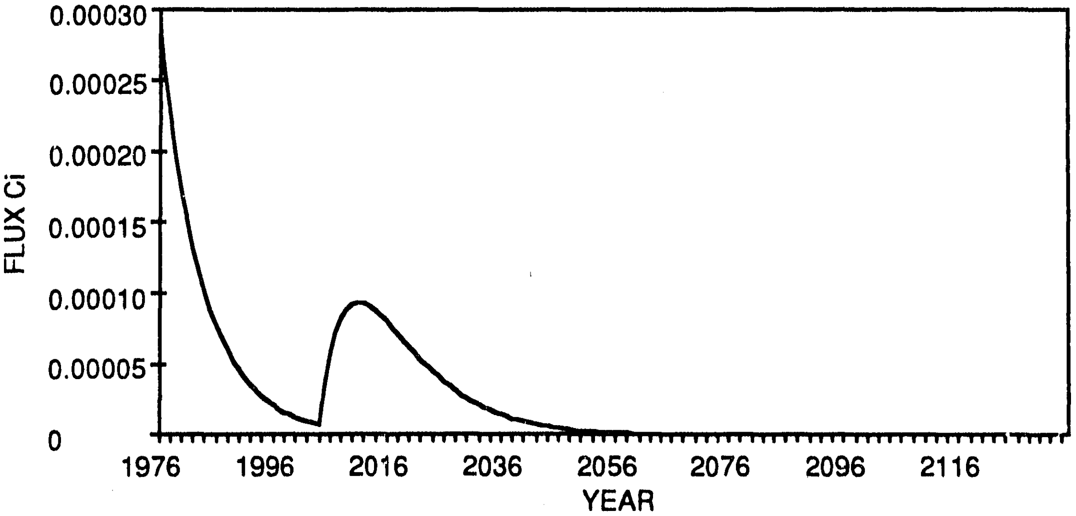
5.18A 4104.51



ANNUAL COBALT-60 CONCENTRATION IN WELL #849—AREA M.



ANNUAL COBALT-60 CONCENTRATION INVENTORY IN TRENCH—AREA M.



ANNUAL COBALT-60 FLUX IN SURFACE RUNOFF—AREA M.

Fig. 5.24. Modeling output for cobalt-60, Area M.

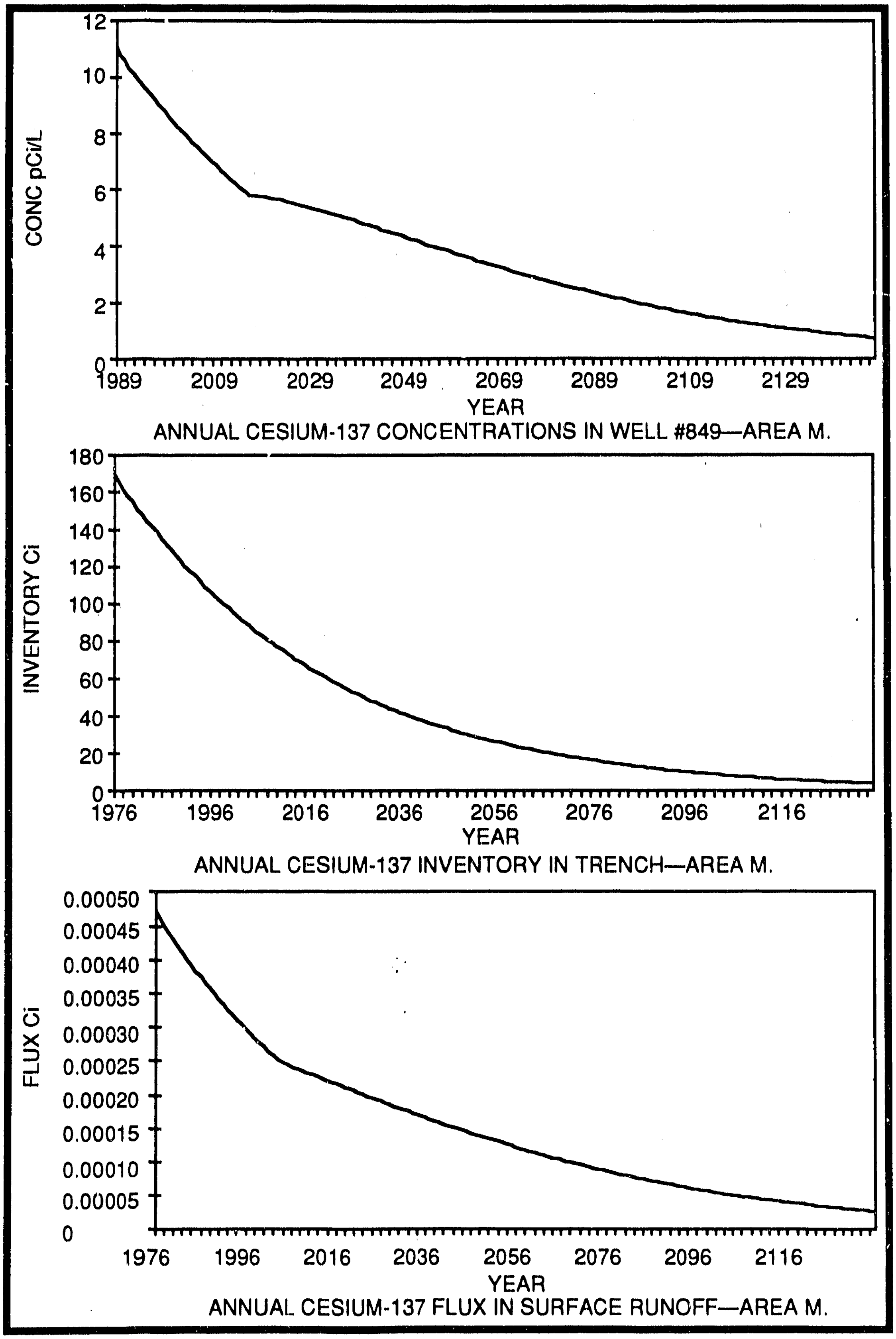
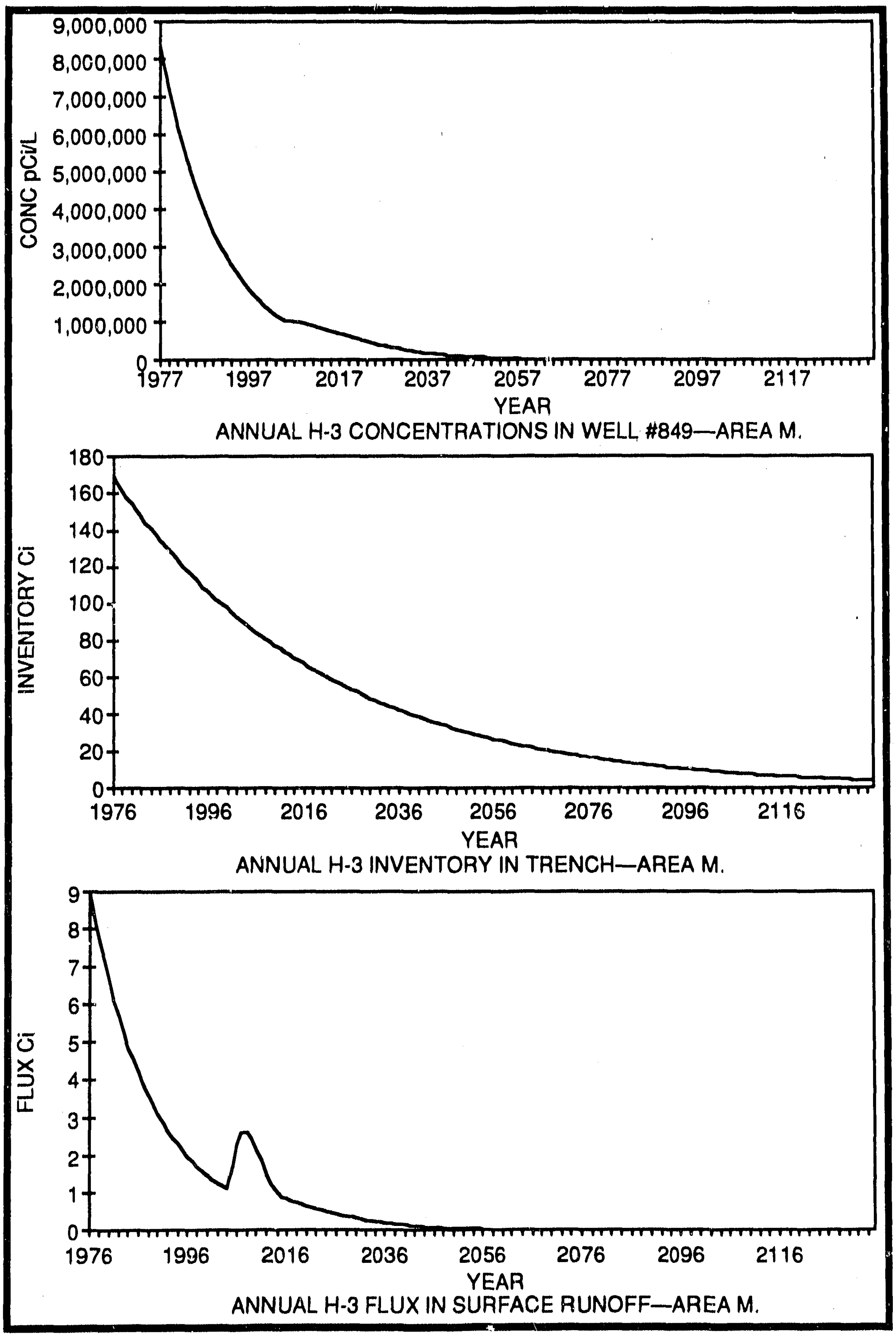


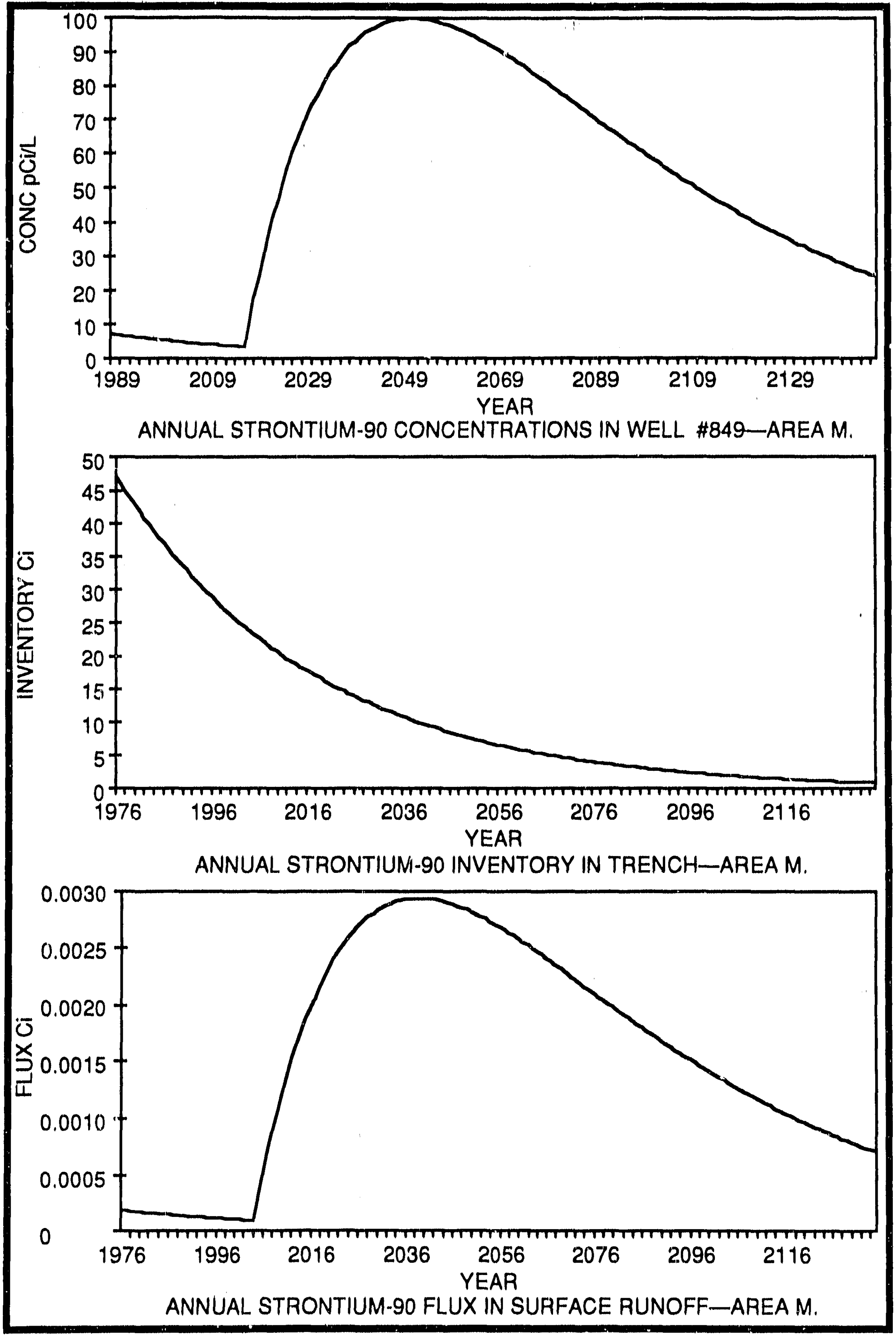
Fig. 5.25. Modeling output for cesium-137, Area M.


ANNUAL H-3 CONCENTRATIONS IN WELL #849—AREA M.

ANNUAL H-3 INVENTORY IN TRENCH—AREA M.

ANNUAL H-3 FLUX IN SURFACE RUNOFF—AREA M.

Fig. 5.26. Modeling output for tritium, Area M.



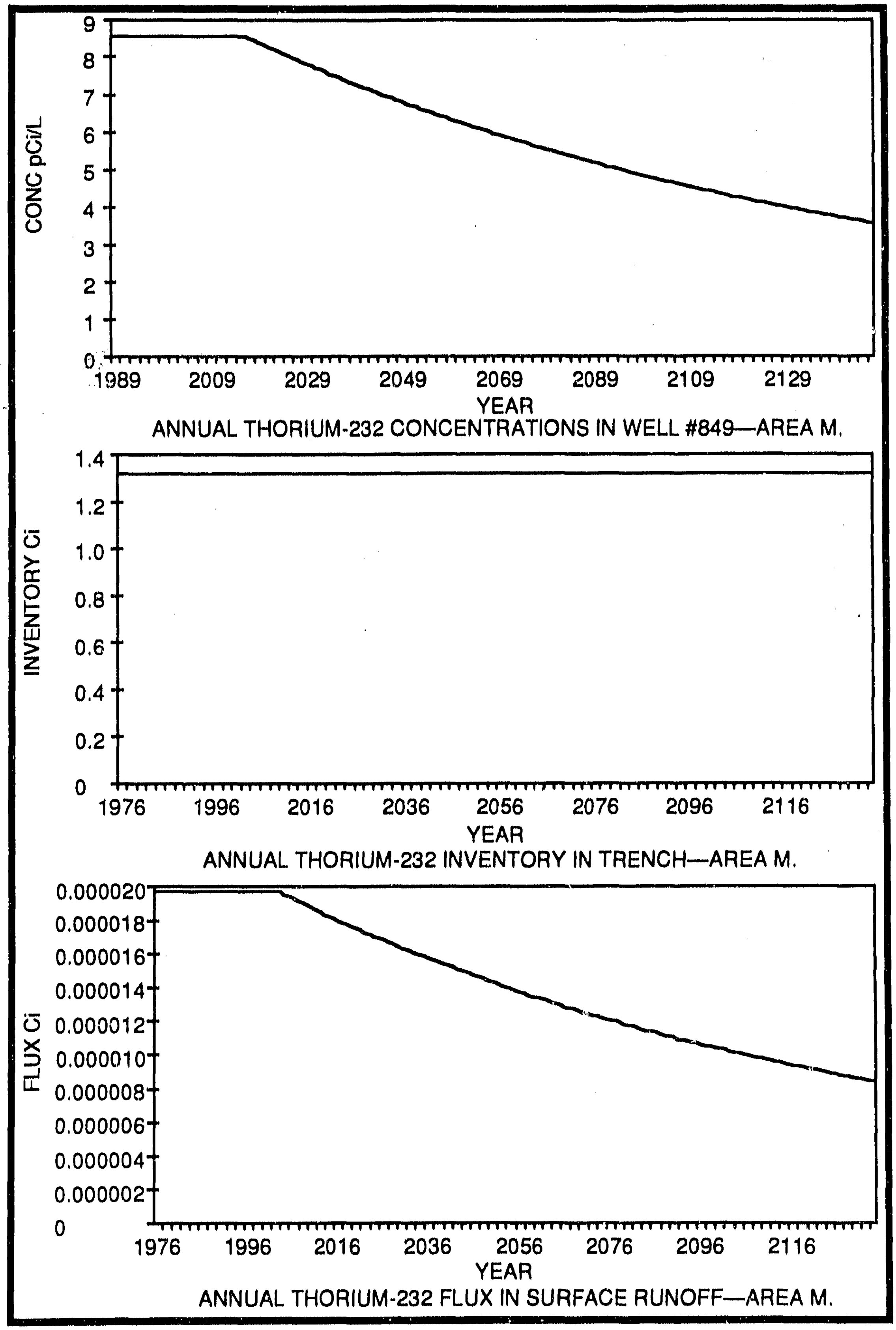
ANNUAL STRONTIUM-90 CONCENTRATIONS IN WELL #849—AREA M.

ANNUAL STRONTIUM-90 INVENTORY IN TRENCH—AREA M.

ANNUAL STRONTIUM-90 FLUX IN SURFACE RUNOFF—AREA M.

Fig. 5.27. Modeling output for strontium-90, Area M.

5.18A 4104.55

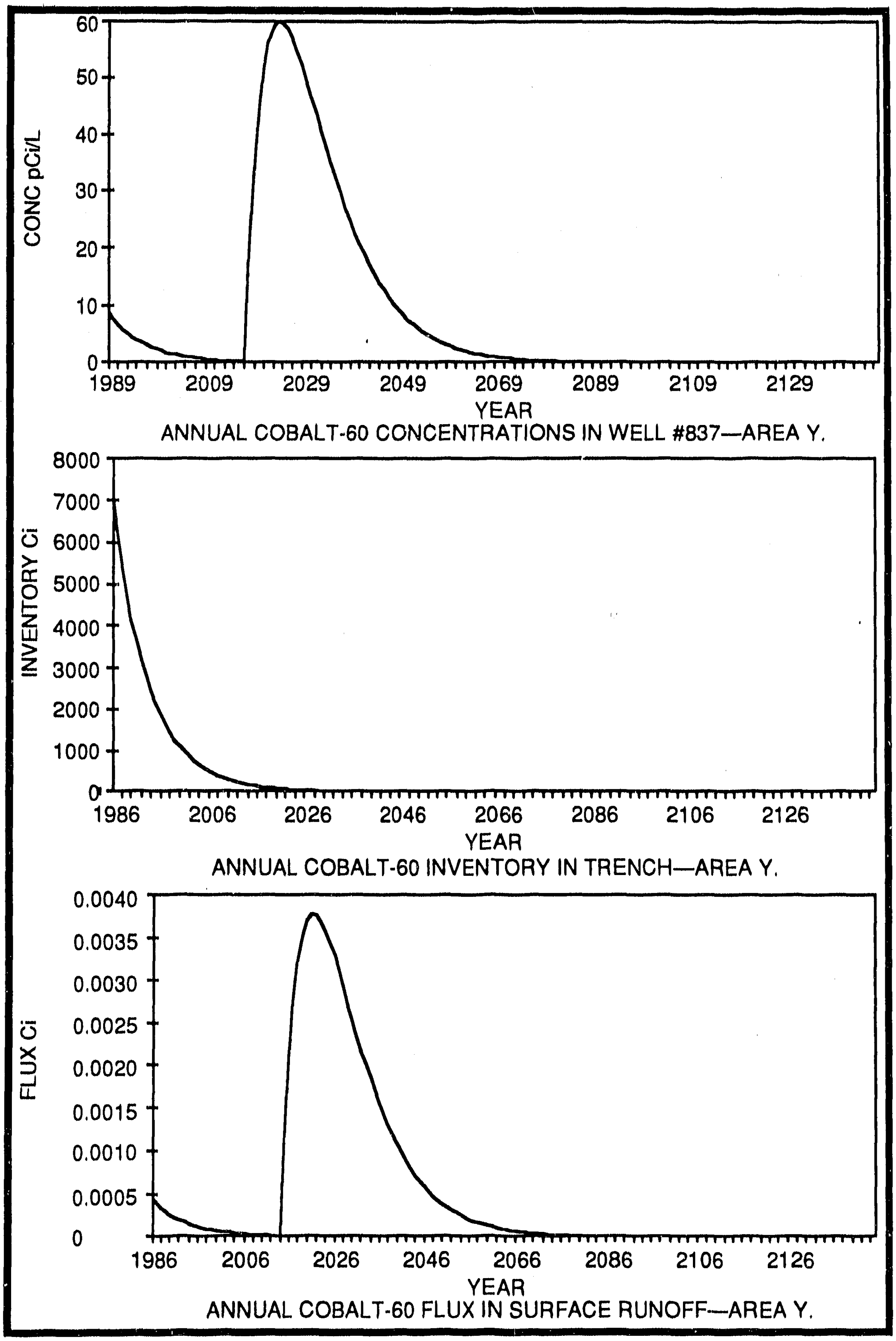


ANNUAL THORIUM-232 CONCENTRATIONS IN WELL #849—AREA M.

ANNUAL THORIUM-232 INVENTORY IN TRENCH—AREA M.

ANNUAL THORIUM-232 FLUX IN SURFACE RUNOFF—AREA M.

Fig. 5.28. Modeling output for thorium-232, Area M.



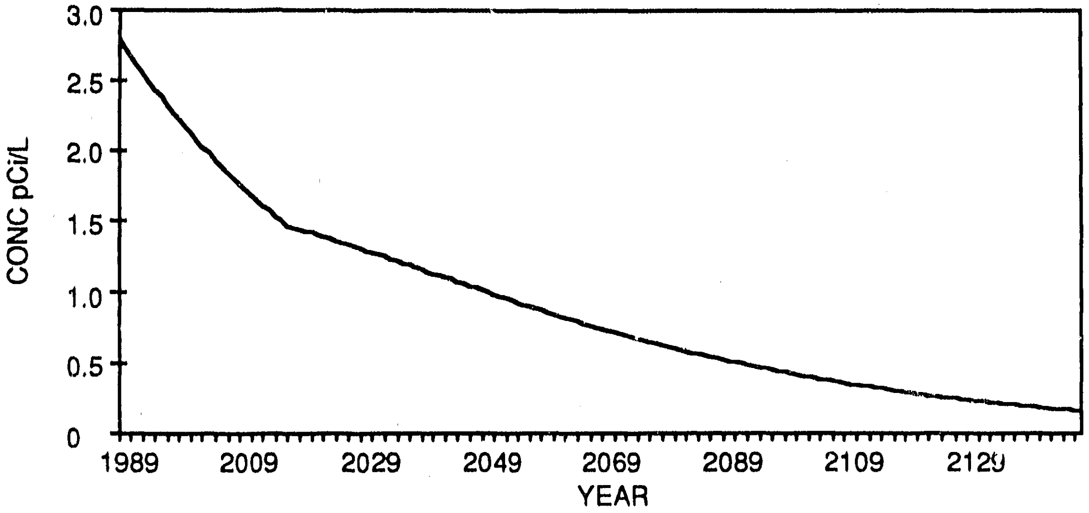
ANNUAL COBALT-60 CONCENTRATIONS IN WELL #837—AREA Y.

ANNUAL COBALT-60 INVENTORY IN TRENCH—AREA Y.

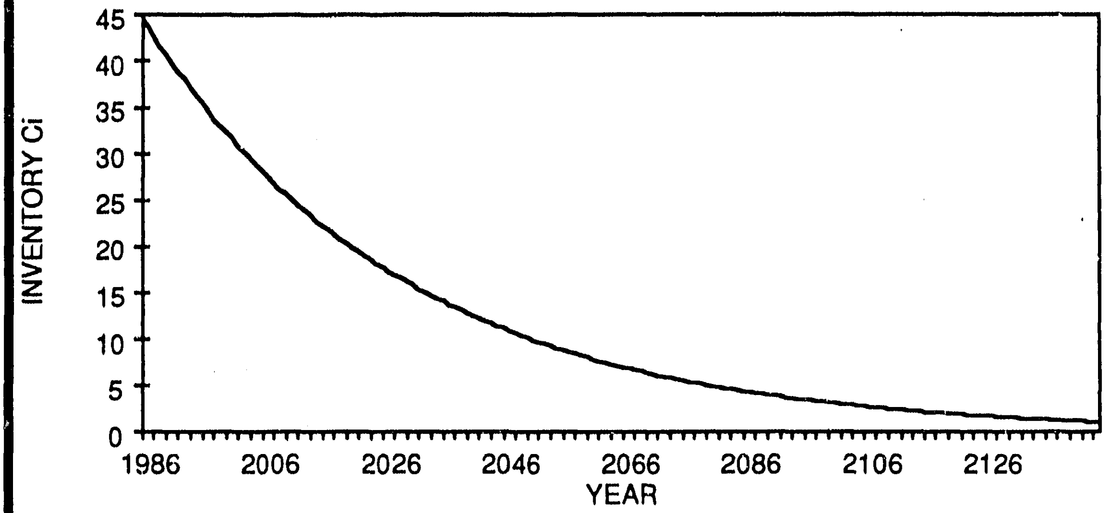
ANNUAL COBALT-60 FLUX IN SURFACE RUNOFF—AREA Y.

Fig. 5.29. Modeling output for cobalt-60, Area Y.

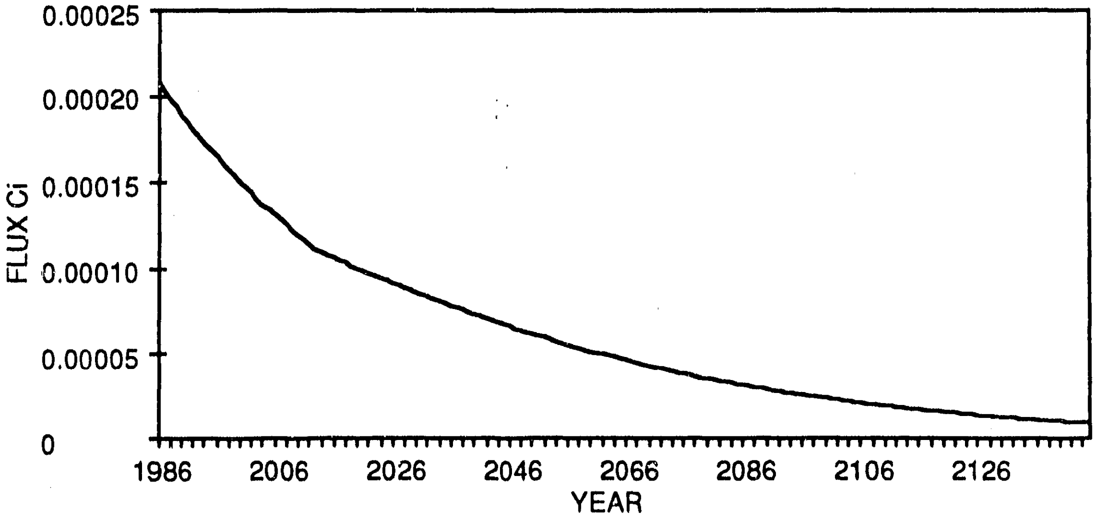
5.18A 4104.57



ANNUAL CS-137 CONCENTRATIONS IN WELL #837—AREA Y.



ANNUAL CS-137 INVENTORY IN TRENCH—AREA Y.



ANNUAL CS-137 FLUX IN SURFACE RUNOFF—AREA Y.

Fig. 5.30. Modeling output for cesium-137, Area Y.

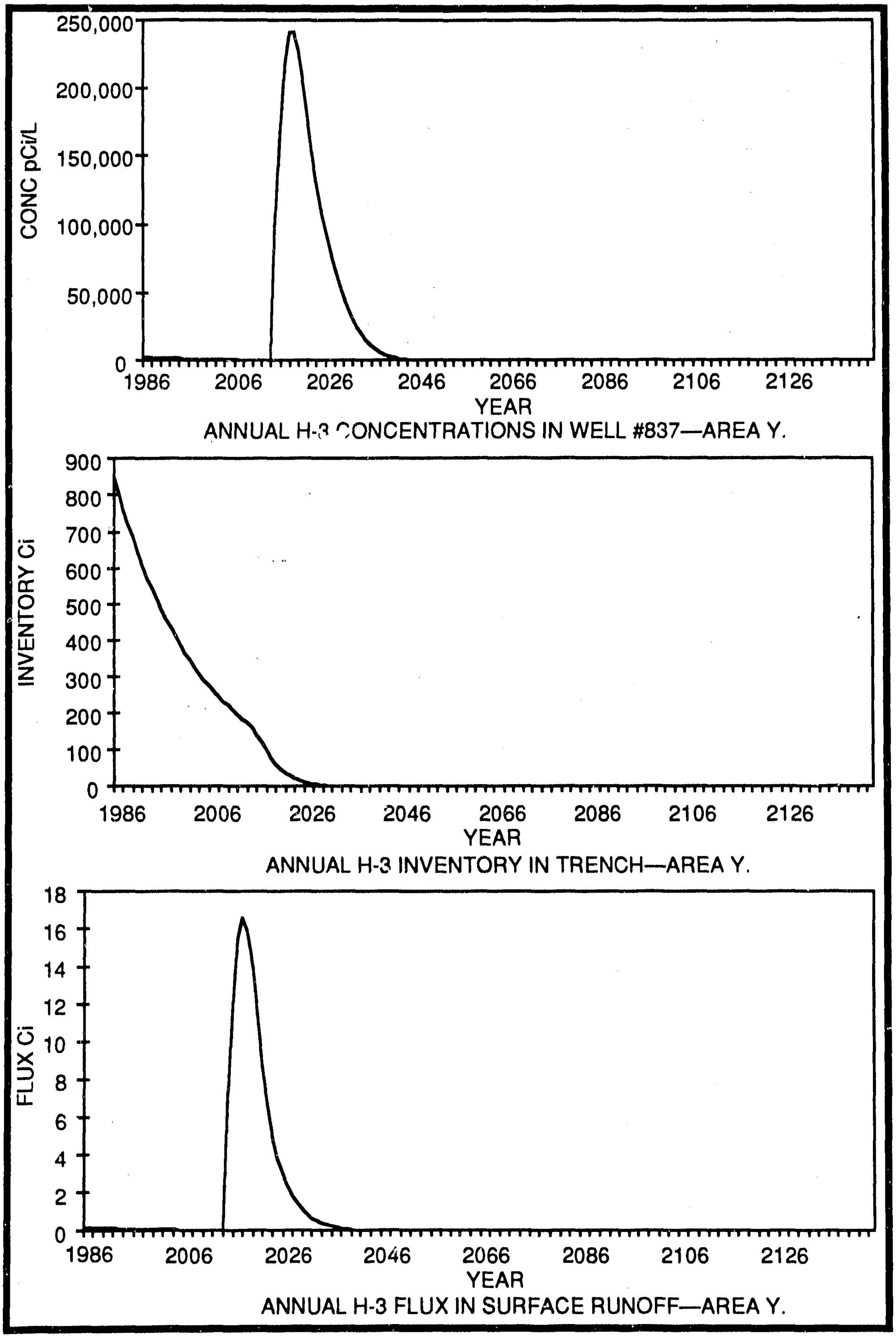
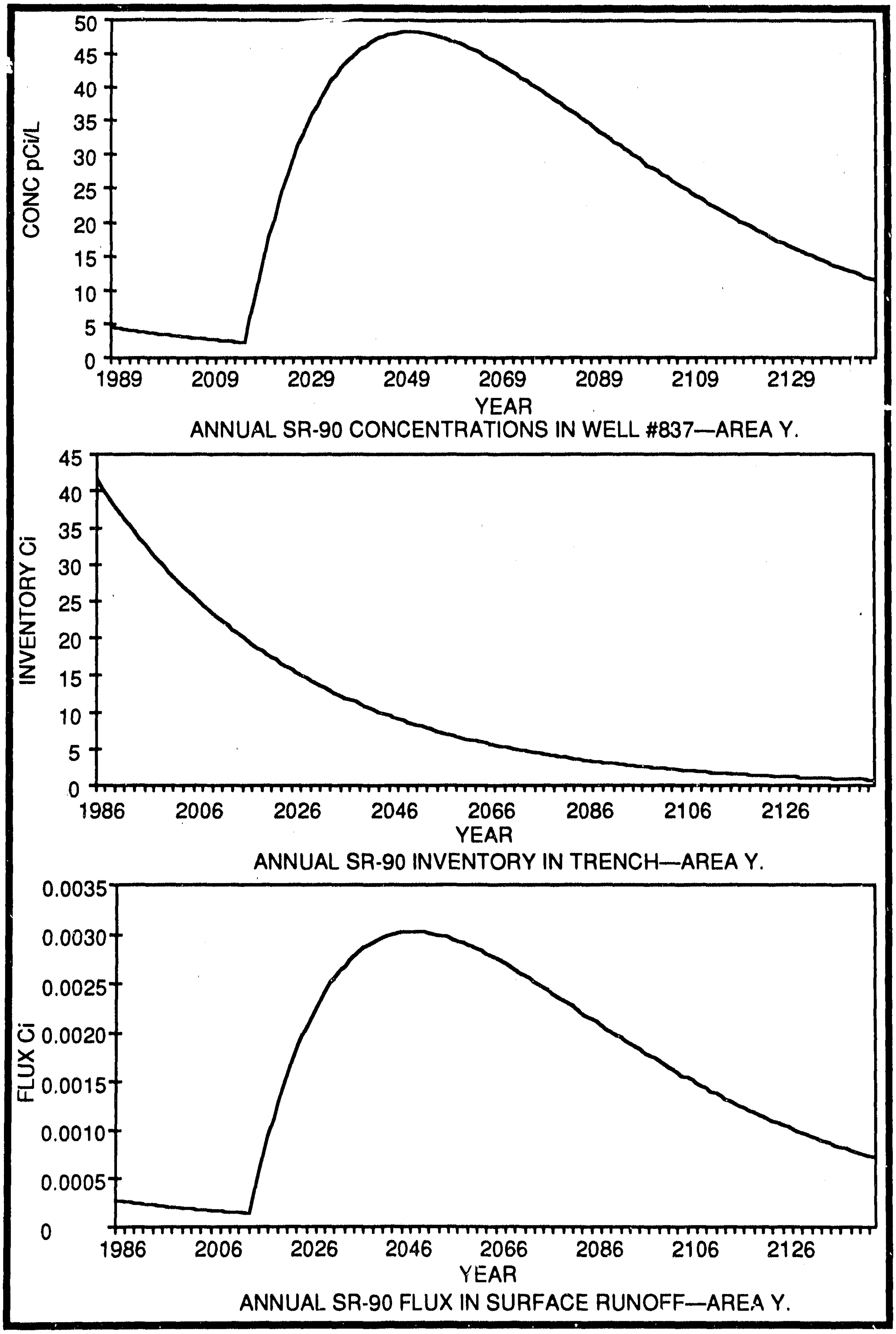


Fig. 5.31. Modeling output for tritium, Area Y.

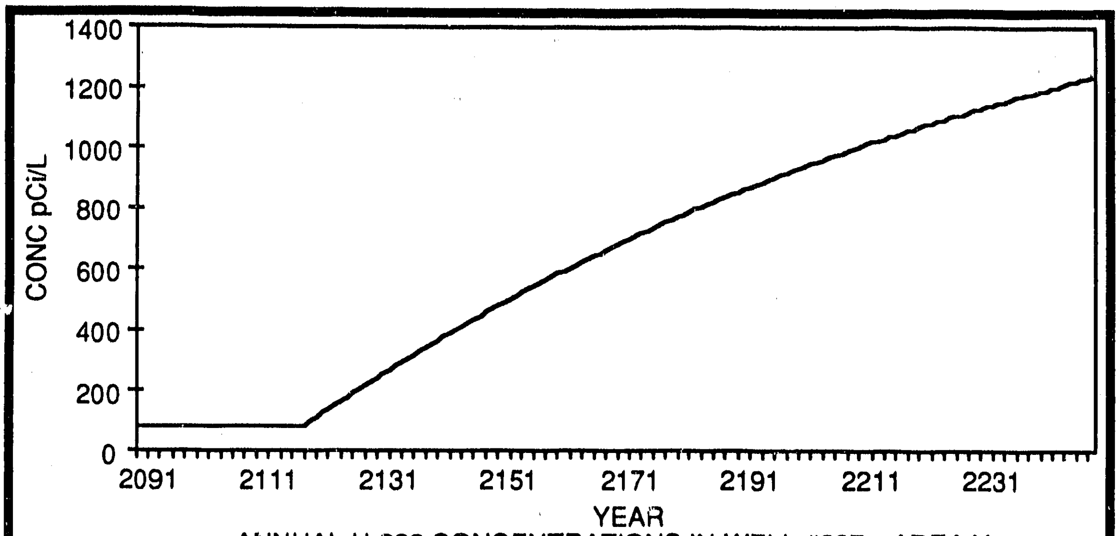


ANNUAL SR-90 CONCENTRATIONS IN WELL #837—AREA Y.

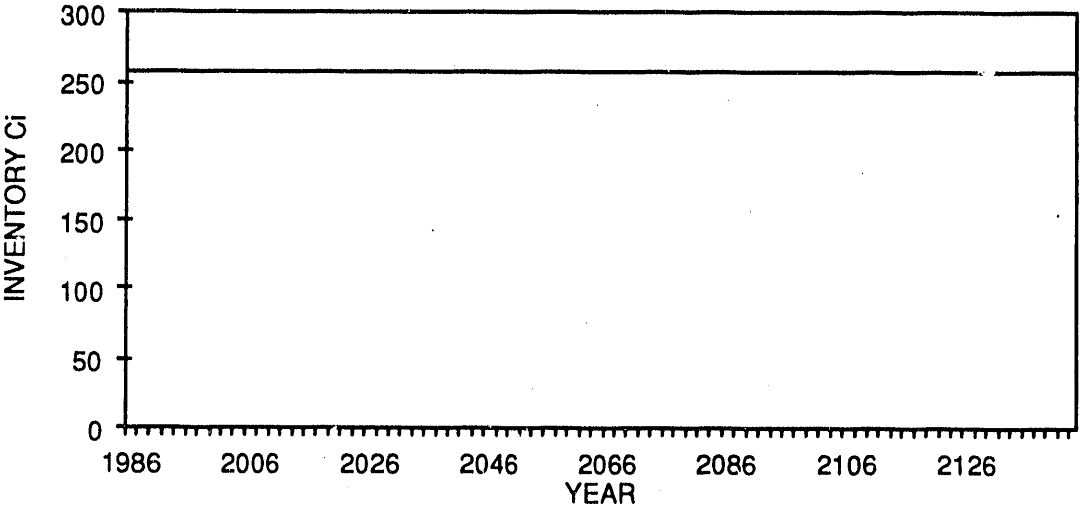
ANNUAL SR-90 INVENTORY IN TRENCH—AREA Y.

ANNUAL SR-90 FLUX IN SURFACE RUNOFF—AREA Y.

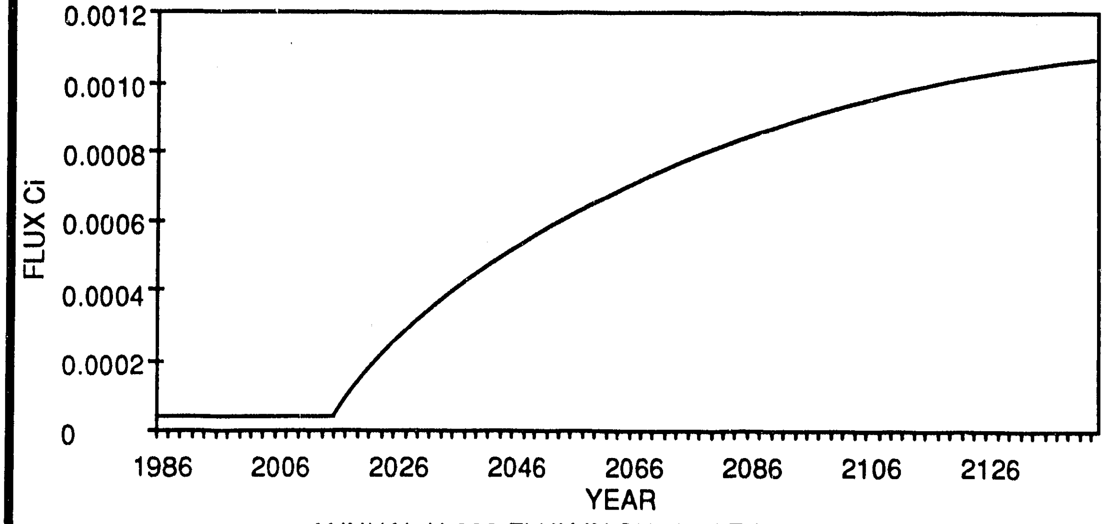
Fig. 5.32. Modeling output for strontium-90, Area Y.



ANNUAL U-238 CONCENTRATIONS IN WELL #837—AREA Y.



ANNUAL U-238 INVENTORY IN TRENCH—AREA Y.



ANNUAL U-238 FLUX IN SURFACE RUNOFF—AREA Y

Fig. 5.33. Modeling output for uranium-238, Area Y.

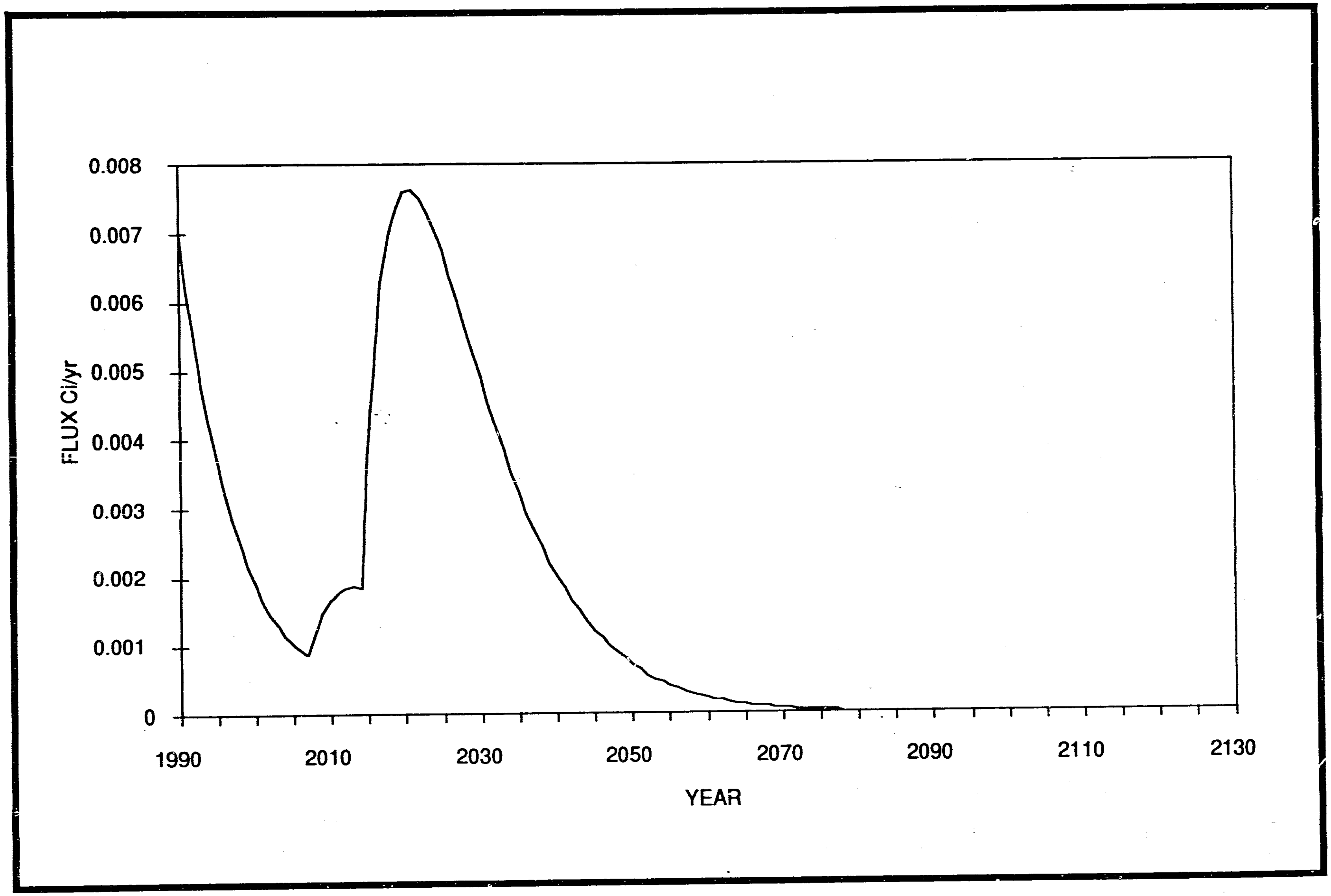


Fig. 5.34. Total cobalt-60 flux in surface water across the WAG 6 boundary.

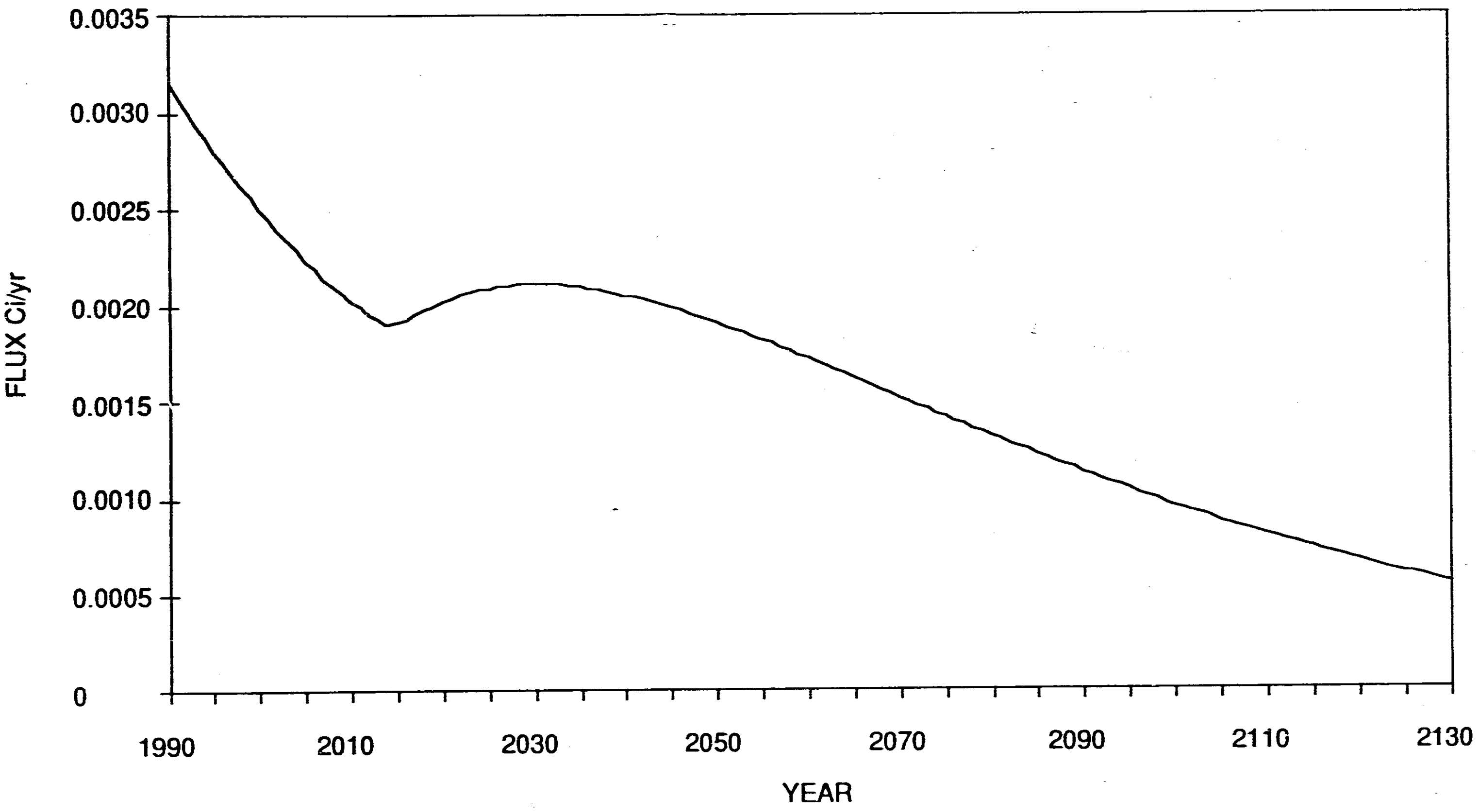


Fig. 5.35. Total cesium-137 flux in surface water across the WAG 6 boundary.

5.18D 4104.107

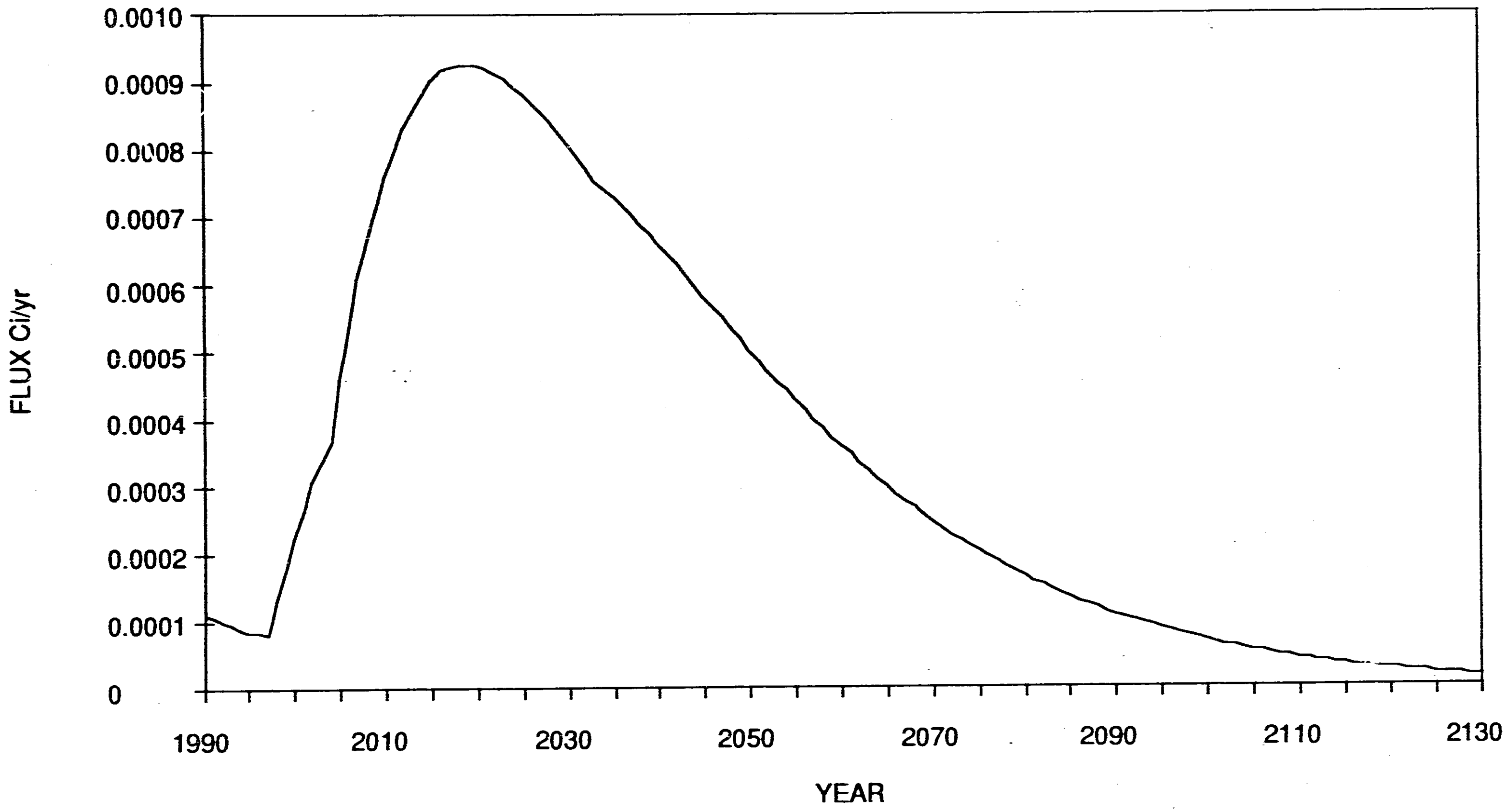


Fig. 5.36. Total europium-152 flux in surface water across the WAG 6 boundary.

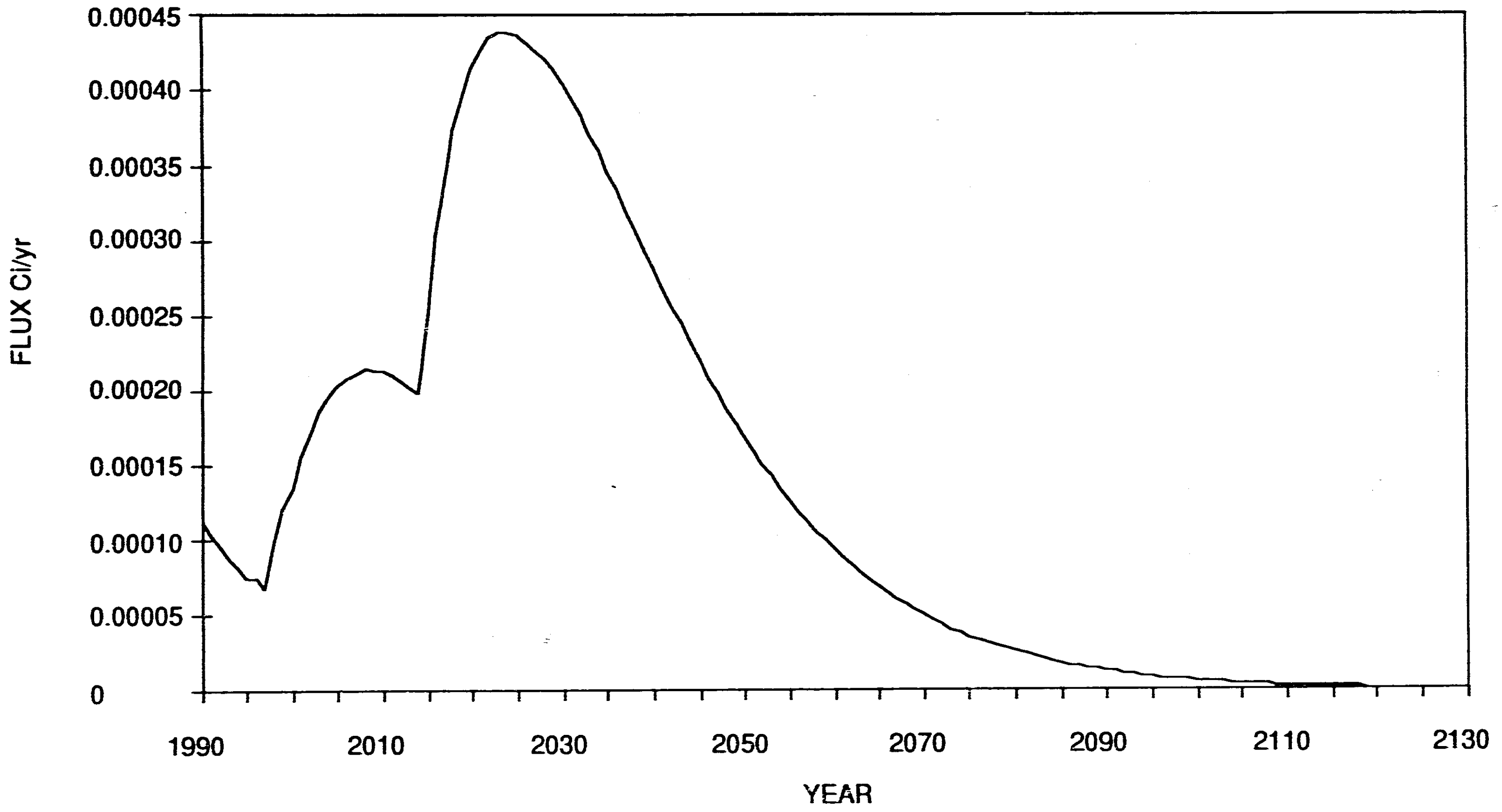


Fig. 5.37. Total europium-154 flux in surface water across the WAG 6 boundary.

5.18D 4104.109

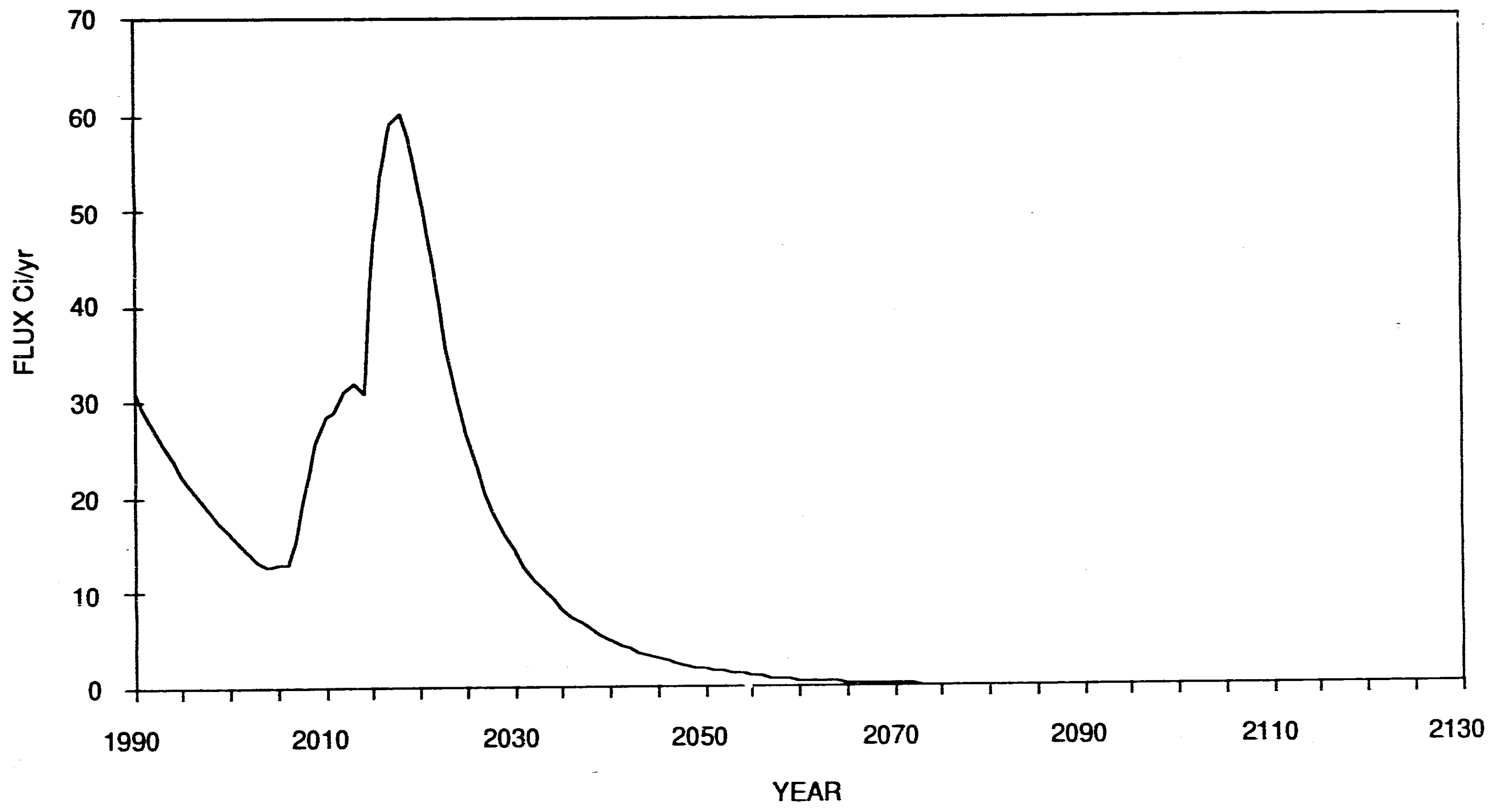


Fig. 5.38. Total tritium flux in surface water across the WAG 6 boundary.

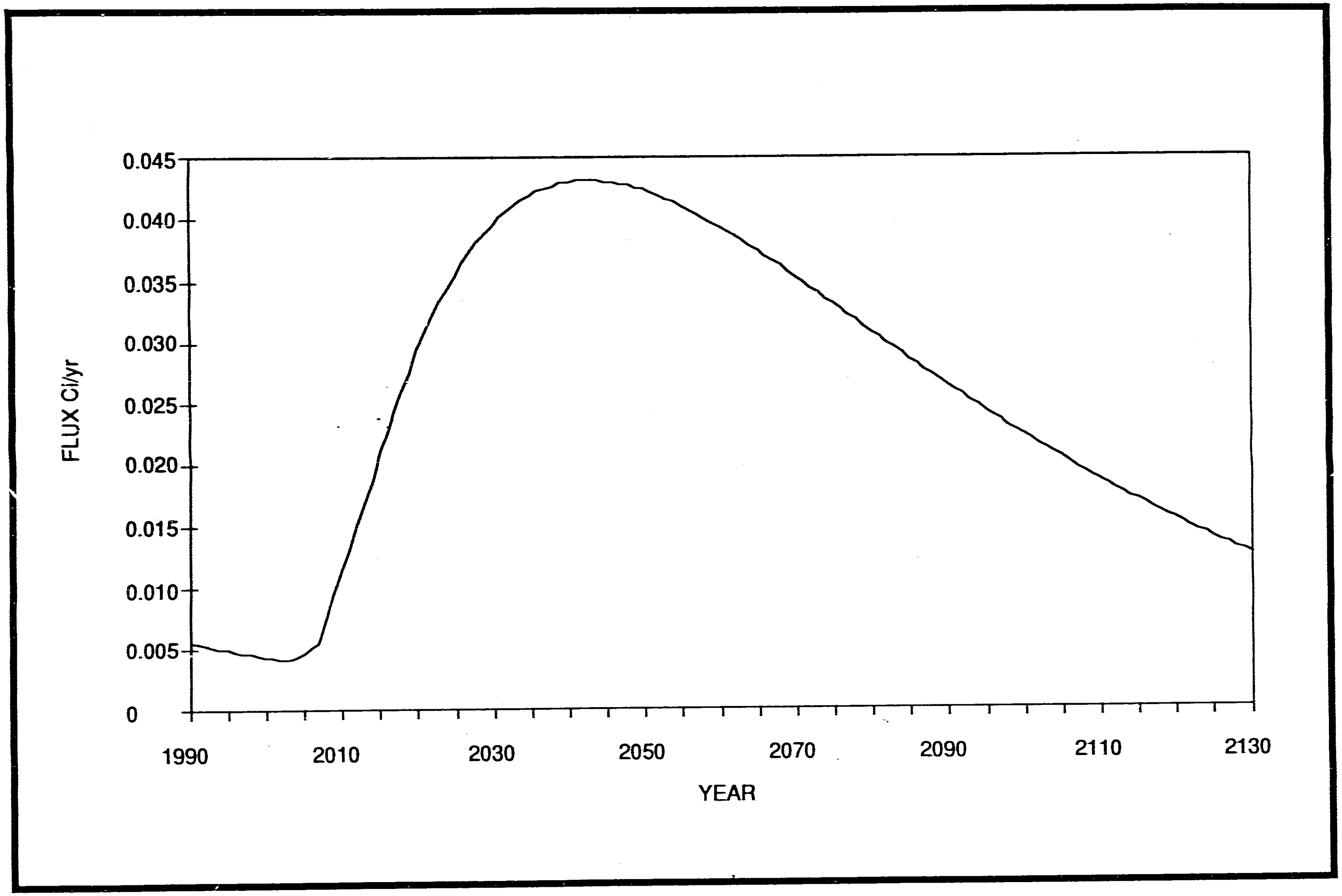


Fig. 5.39. Total strontium-90 flux in surface water across the WAG 6 boundary.

5.18D 4104.111

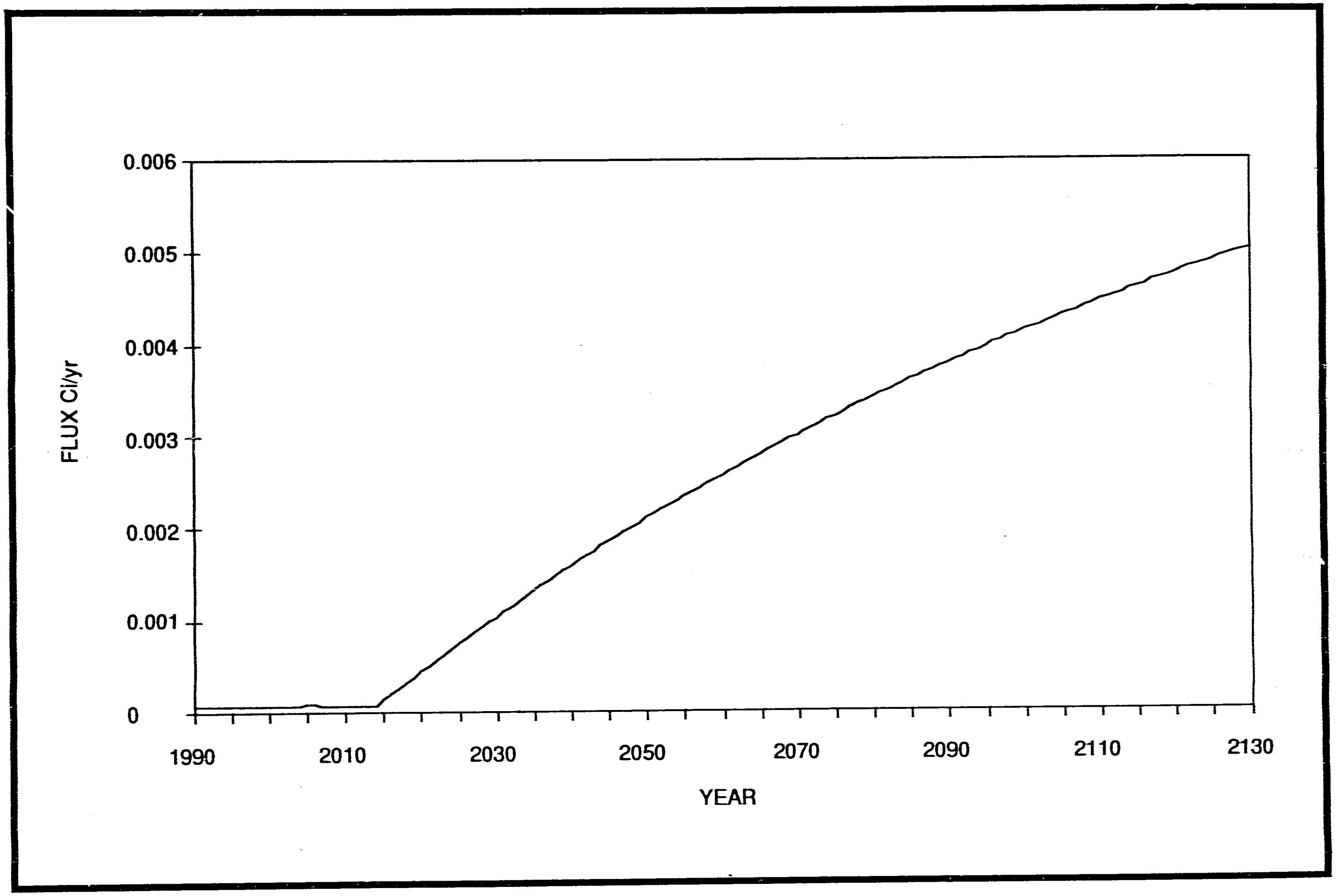


Fig. 5.40. Total thorium-232 flux in surface water across the WAG 6 boundary.

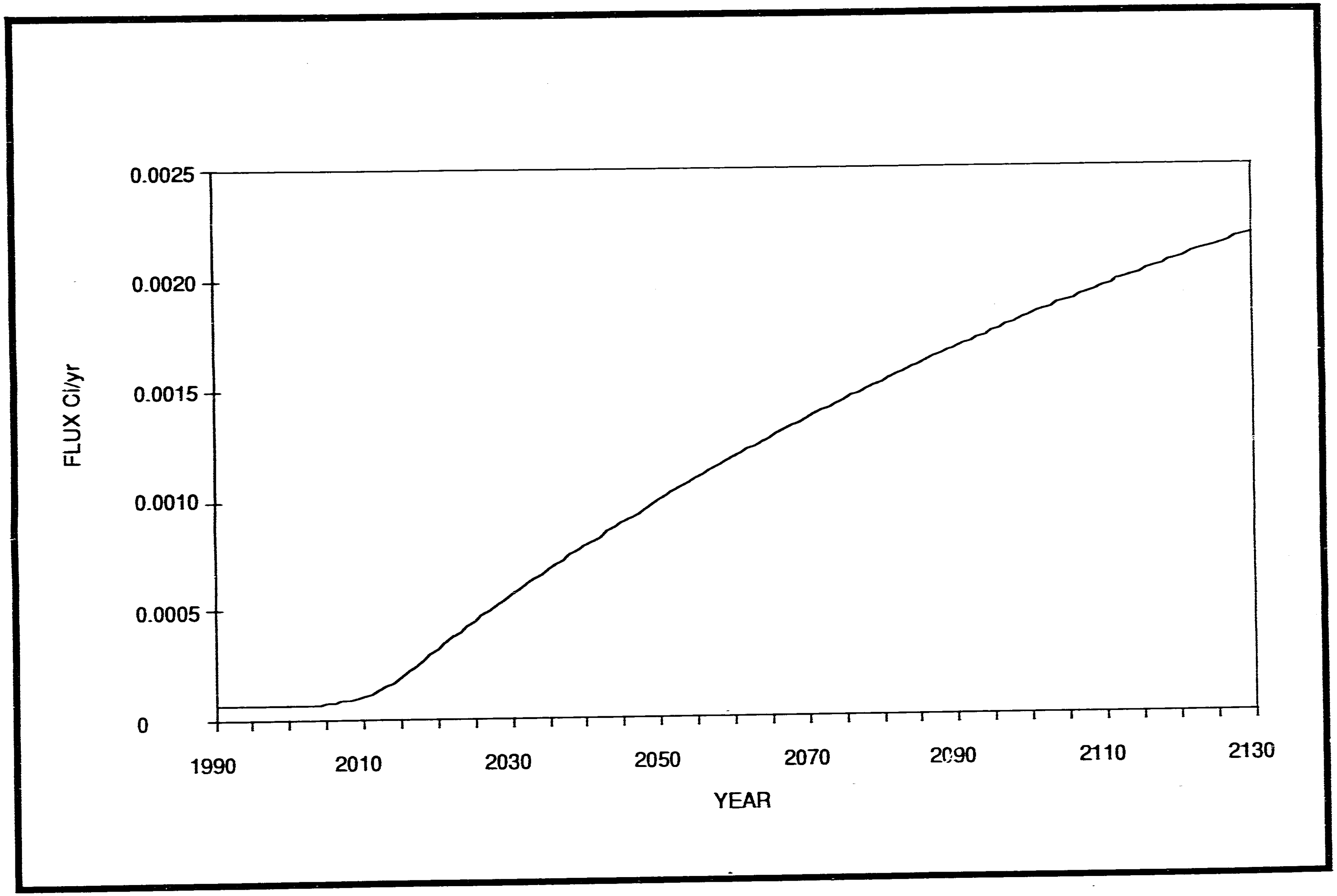


Fig. 5.41. Total uranium-233 flux in surface water across the WAG 6 boundary.

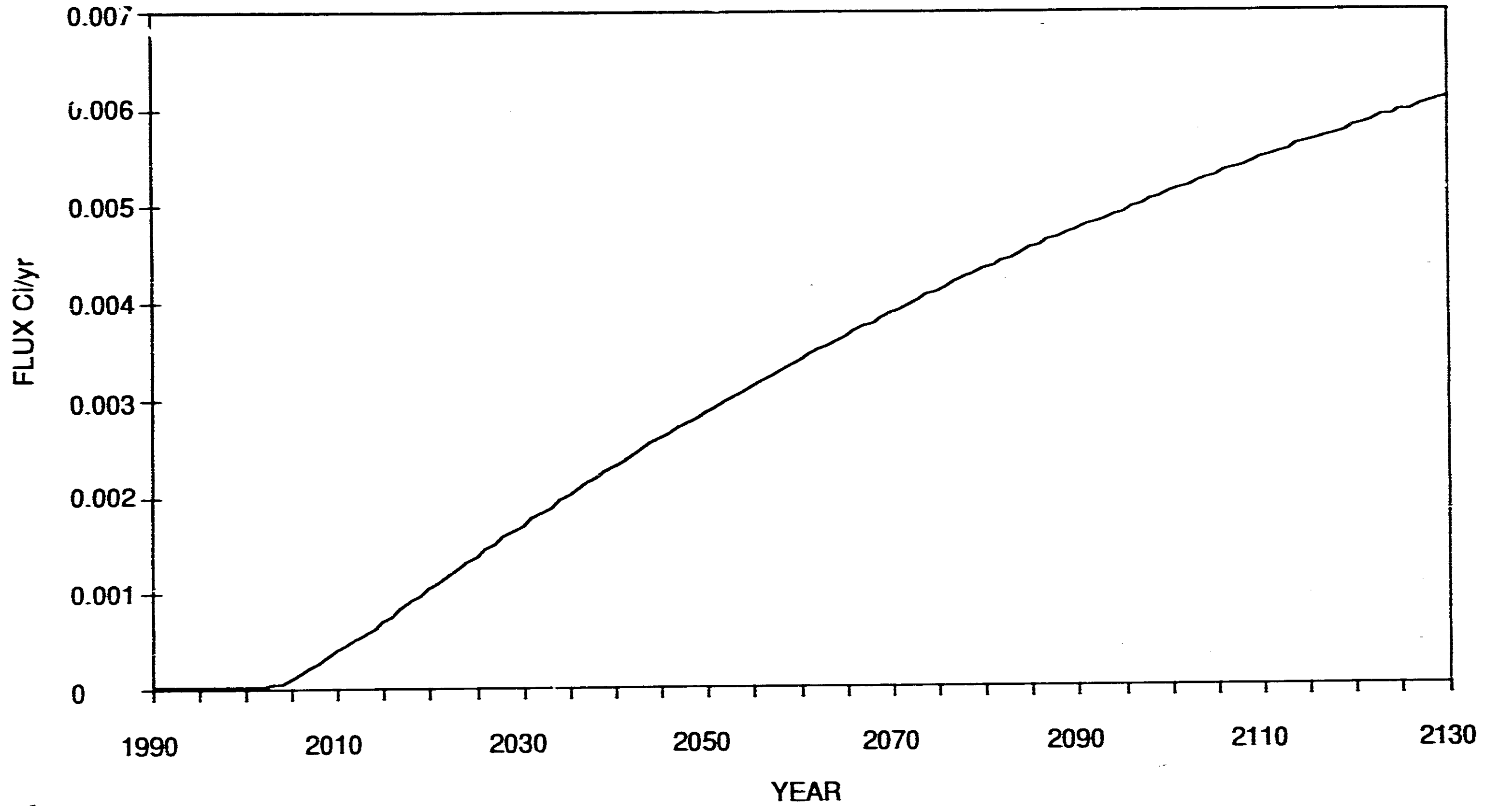


Fig. 5.42. Total uranium-238 flux in surface water across the WAG 6 boundary.

5.18D 4104.114

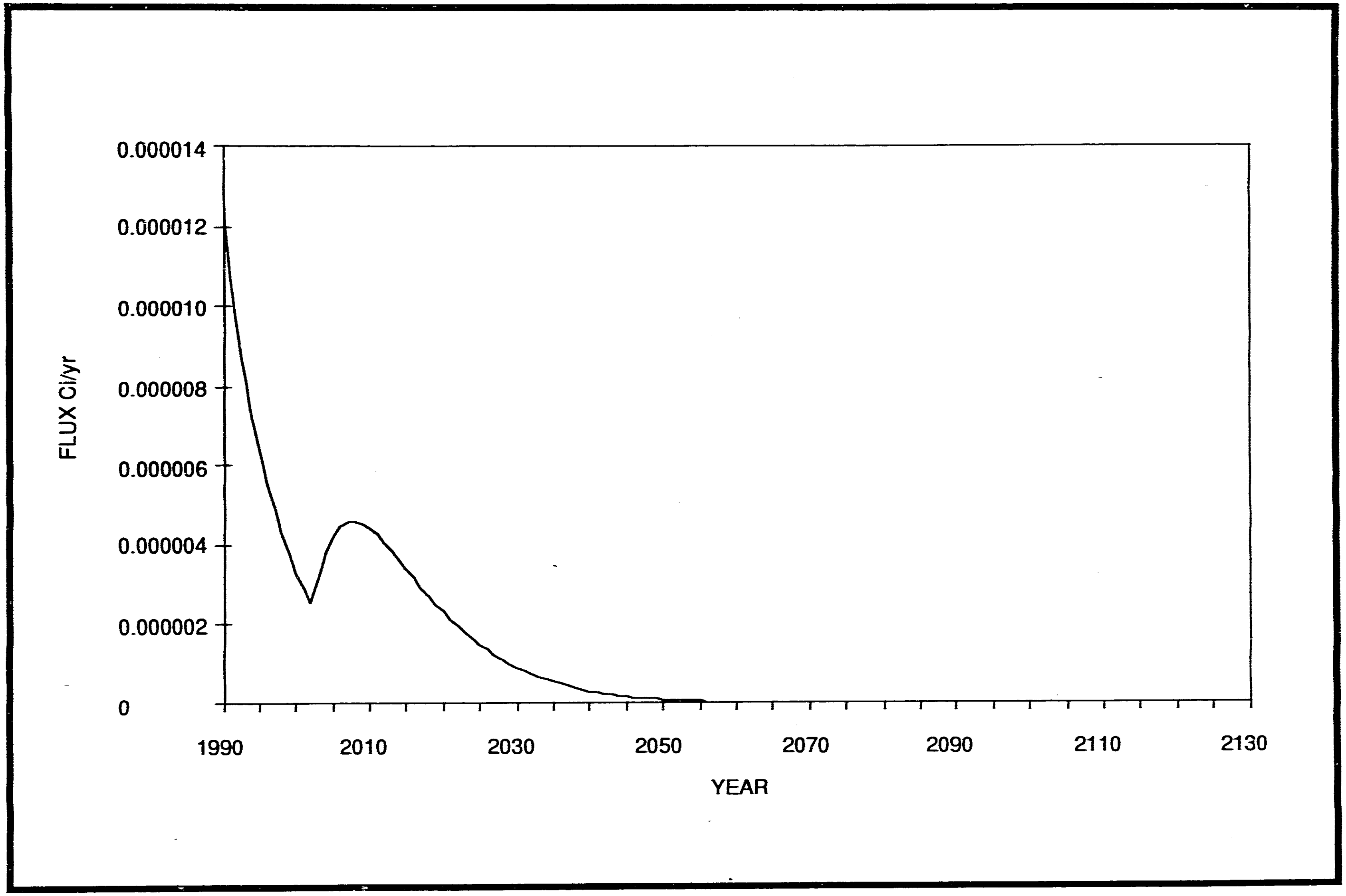


Fig. 5.43. Total cobalt-60 flux in surface water into the EWB.

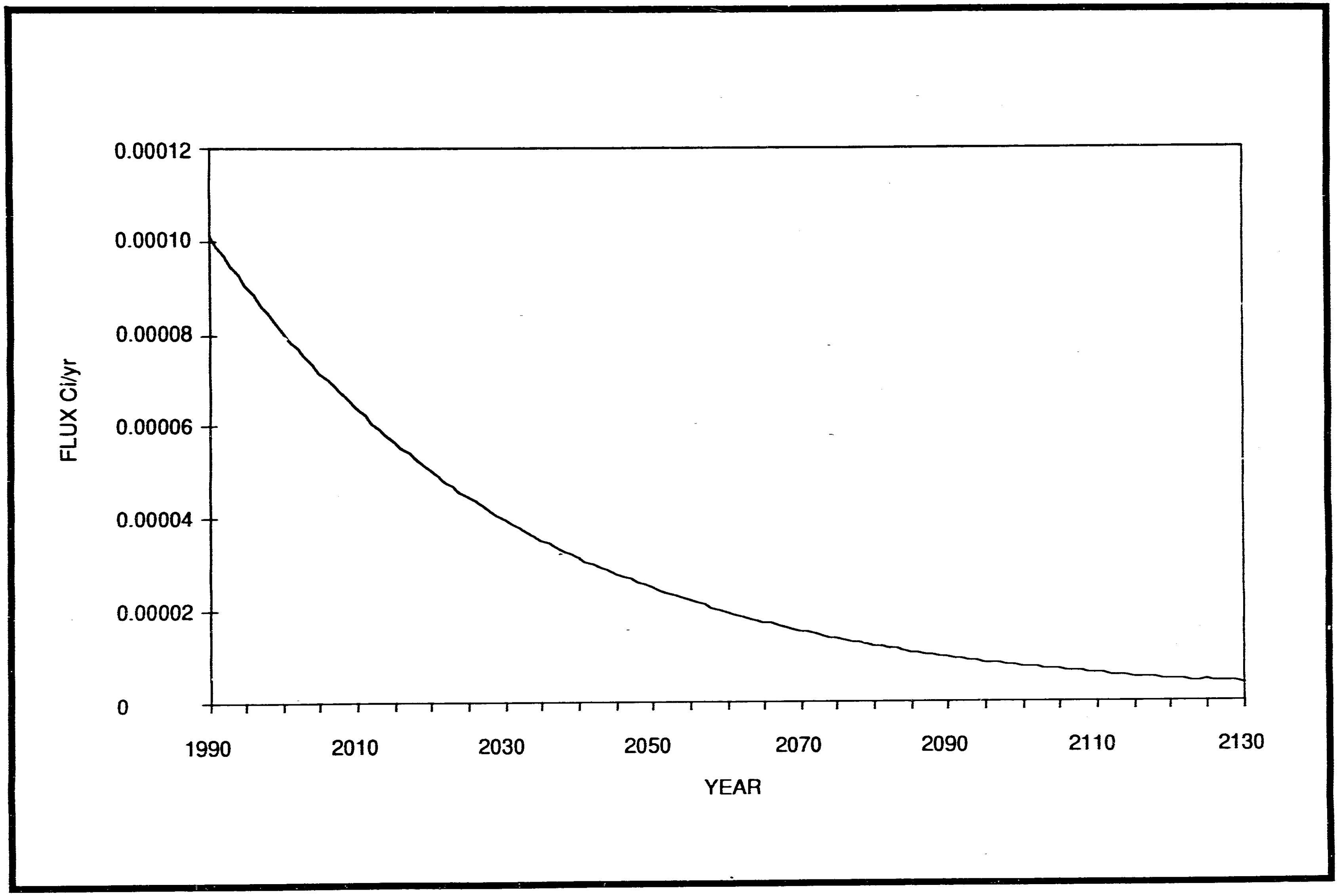


Fig. 5.44. Total cesium-137 flux in surface water into the EWB.

5.18D 4104.116

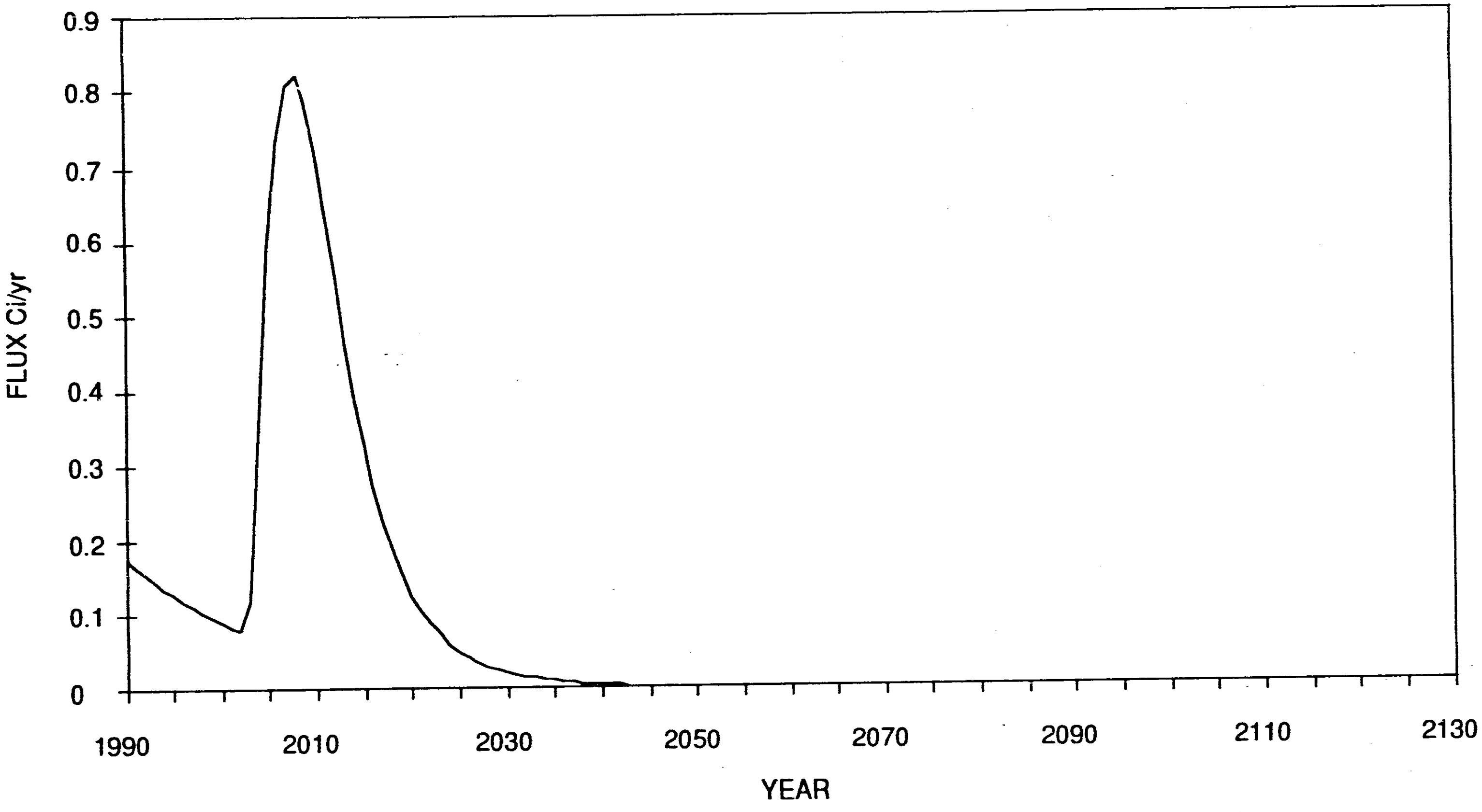


Fig. 5.45. Total tritium flux in surface water into the EWB.

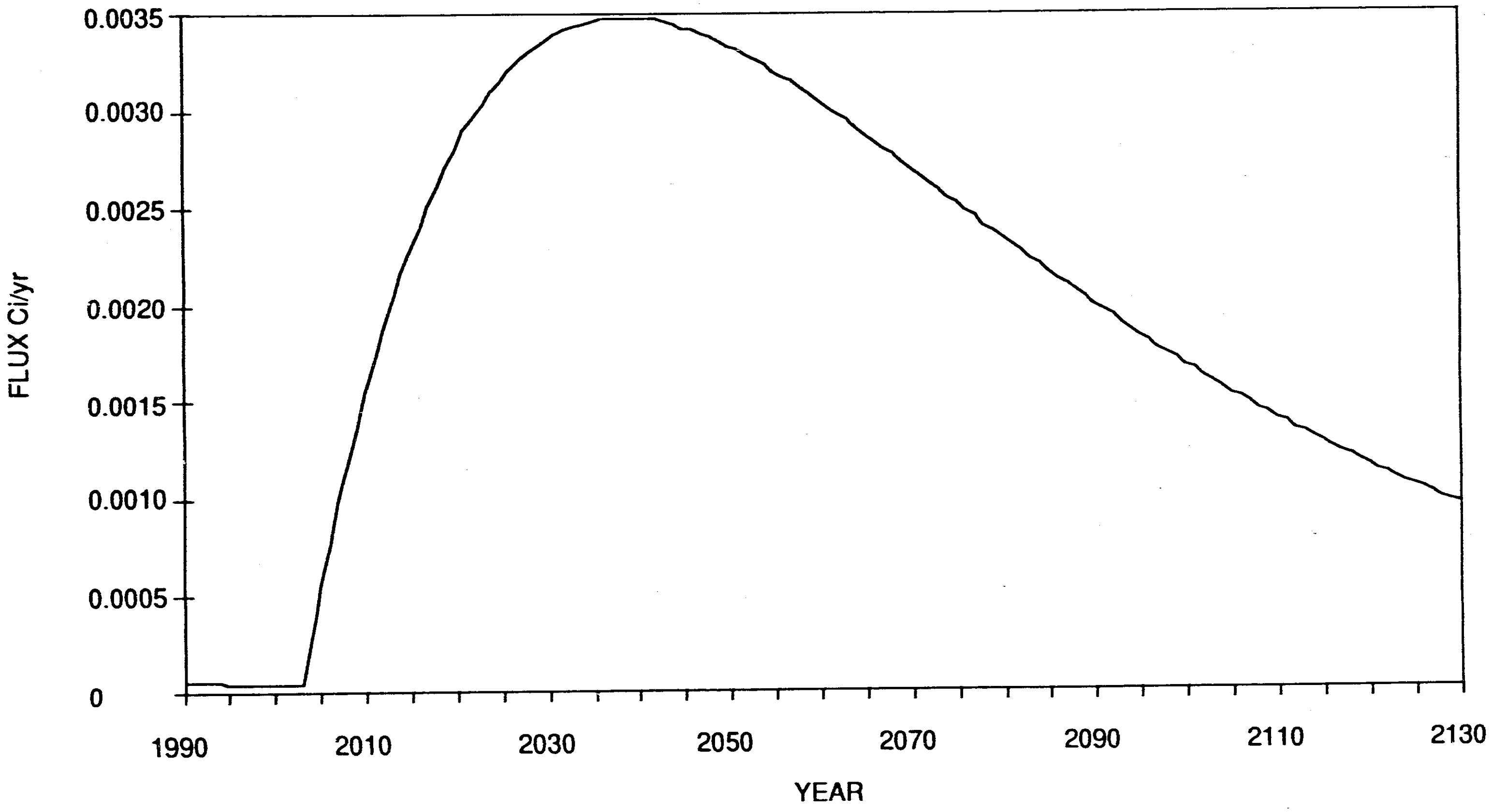


Fig. 5.46. Total strontium-90 flux in surface water into the EWB.

5.18D 4104.118

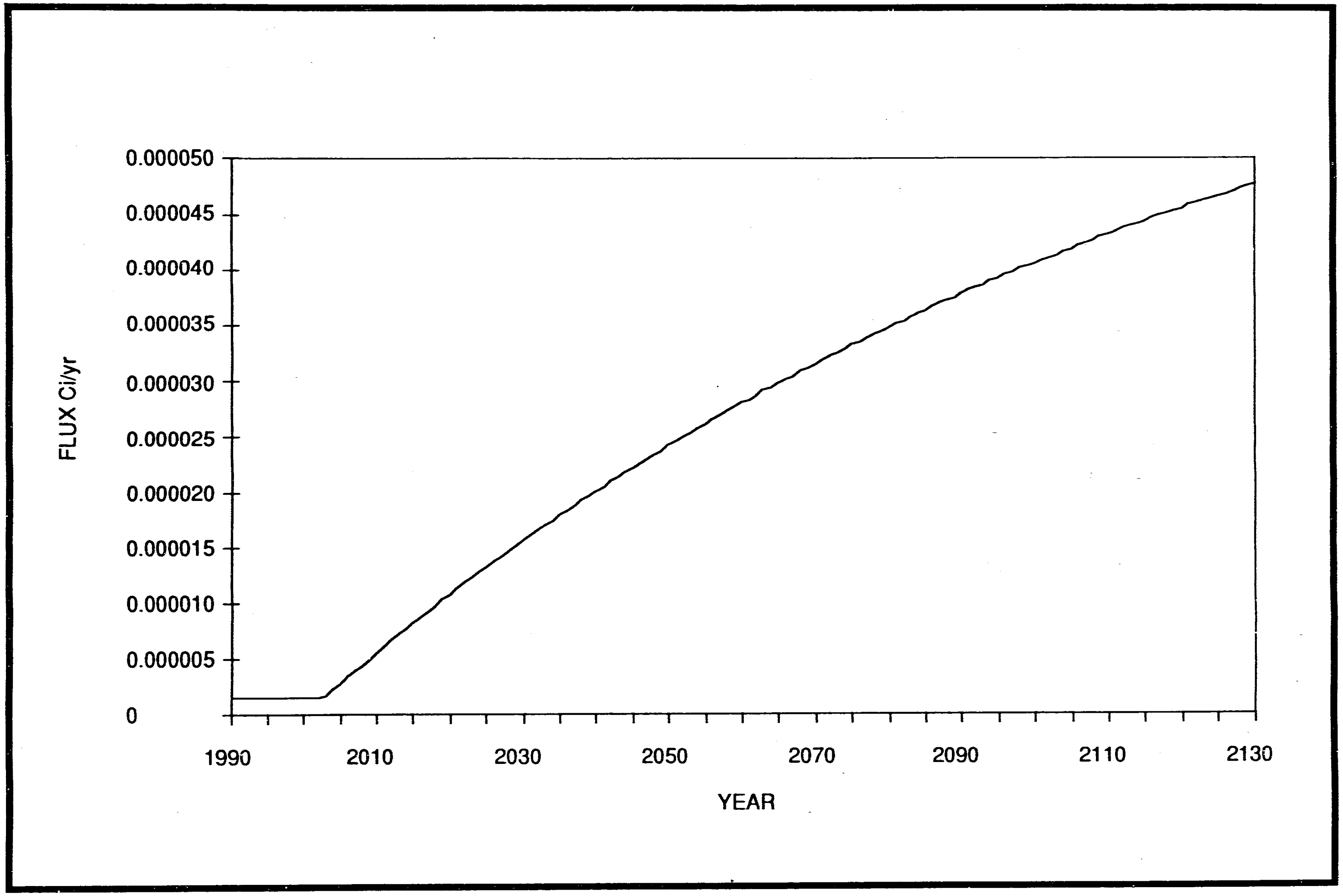


Fig. 5.47. Total uranium-233 flux in surface water into the EWB.

5.18D 4104.119

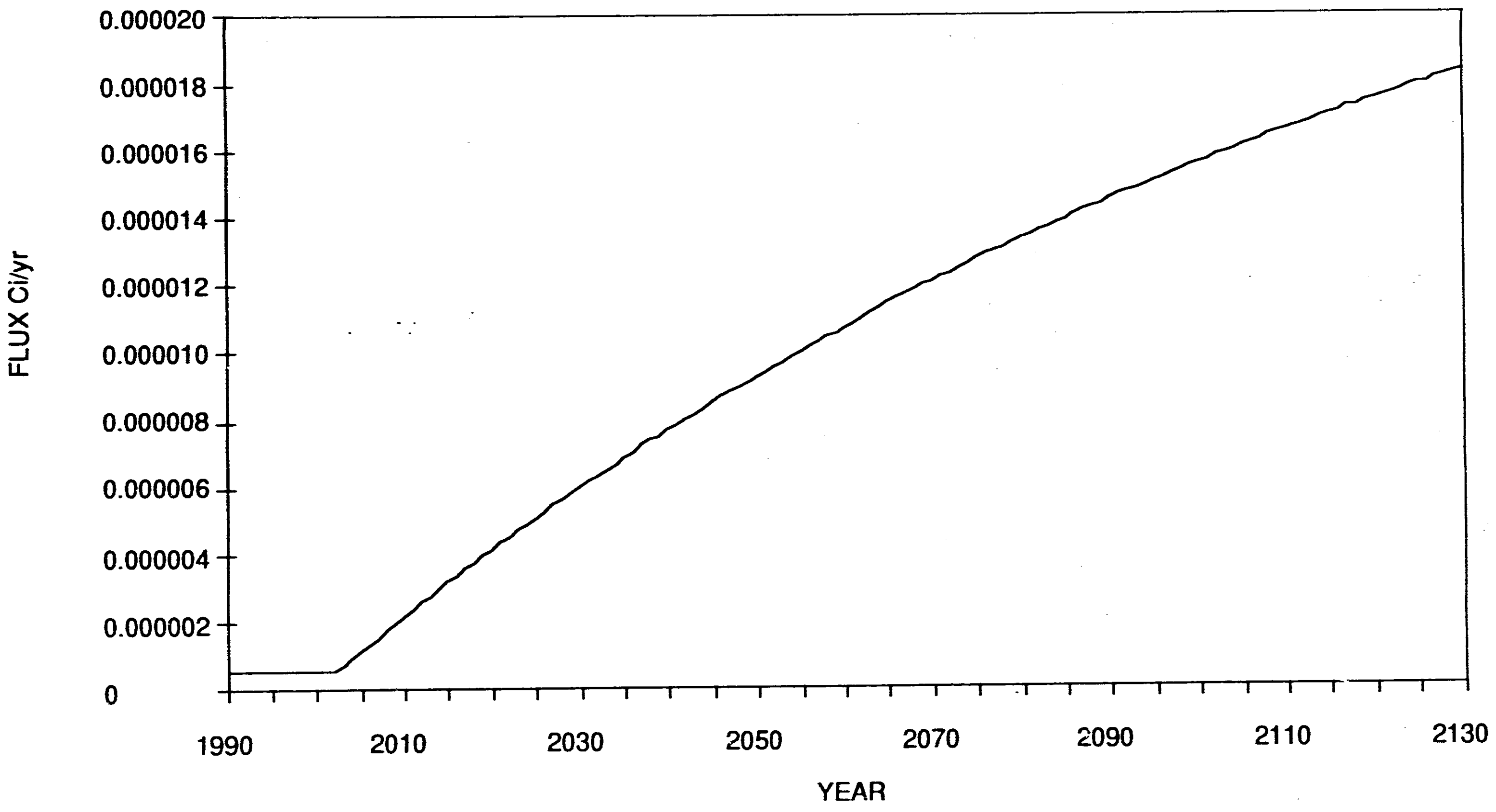


Fig. 5.48. Total uranium-238 flux in surface water into the EWB.

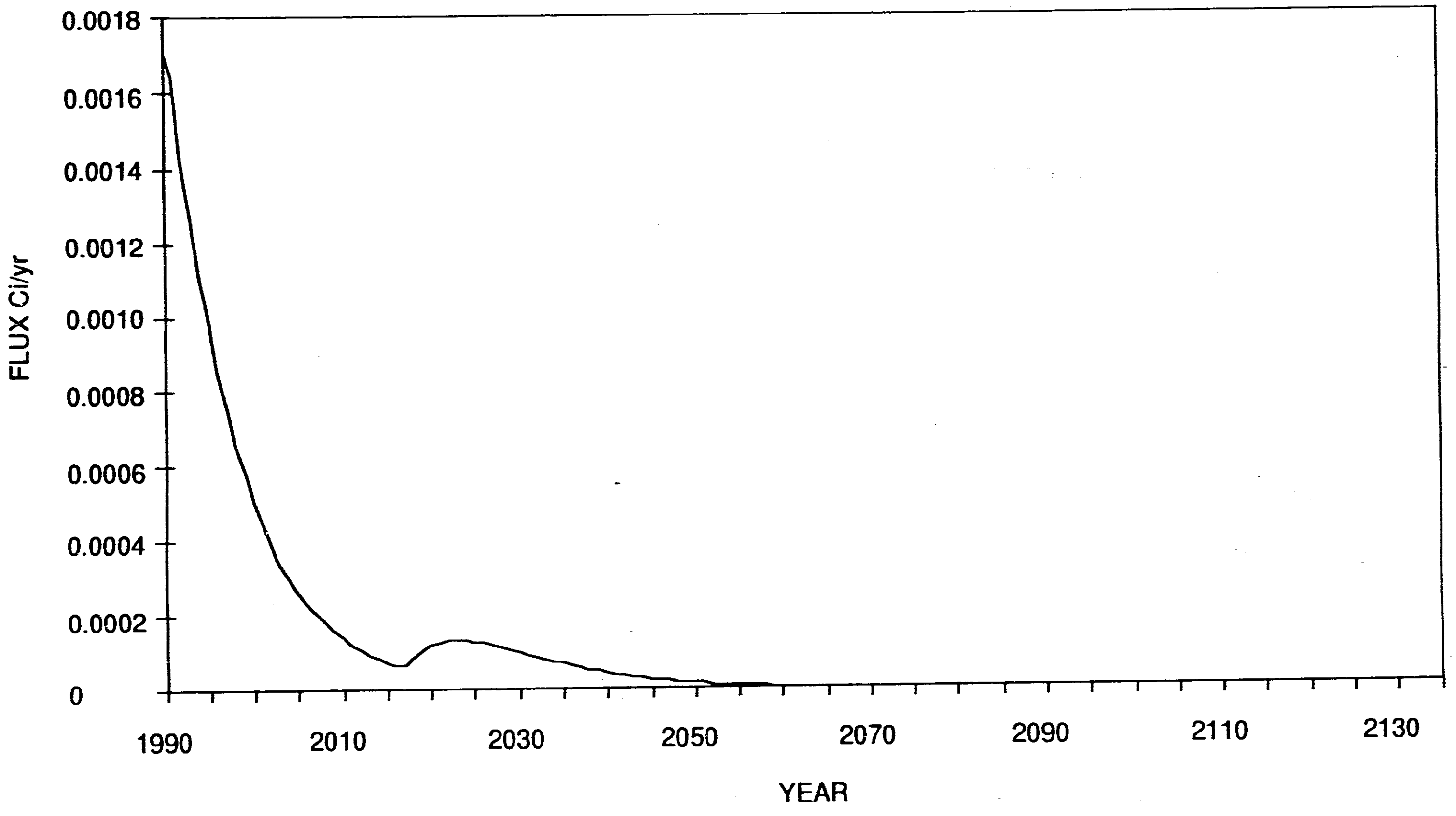


Fig. 5.49. Cobalt-60 flux in groundwater across the WAG 6 boundary.

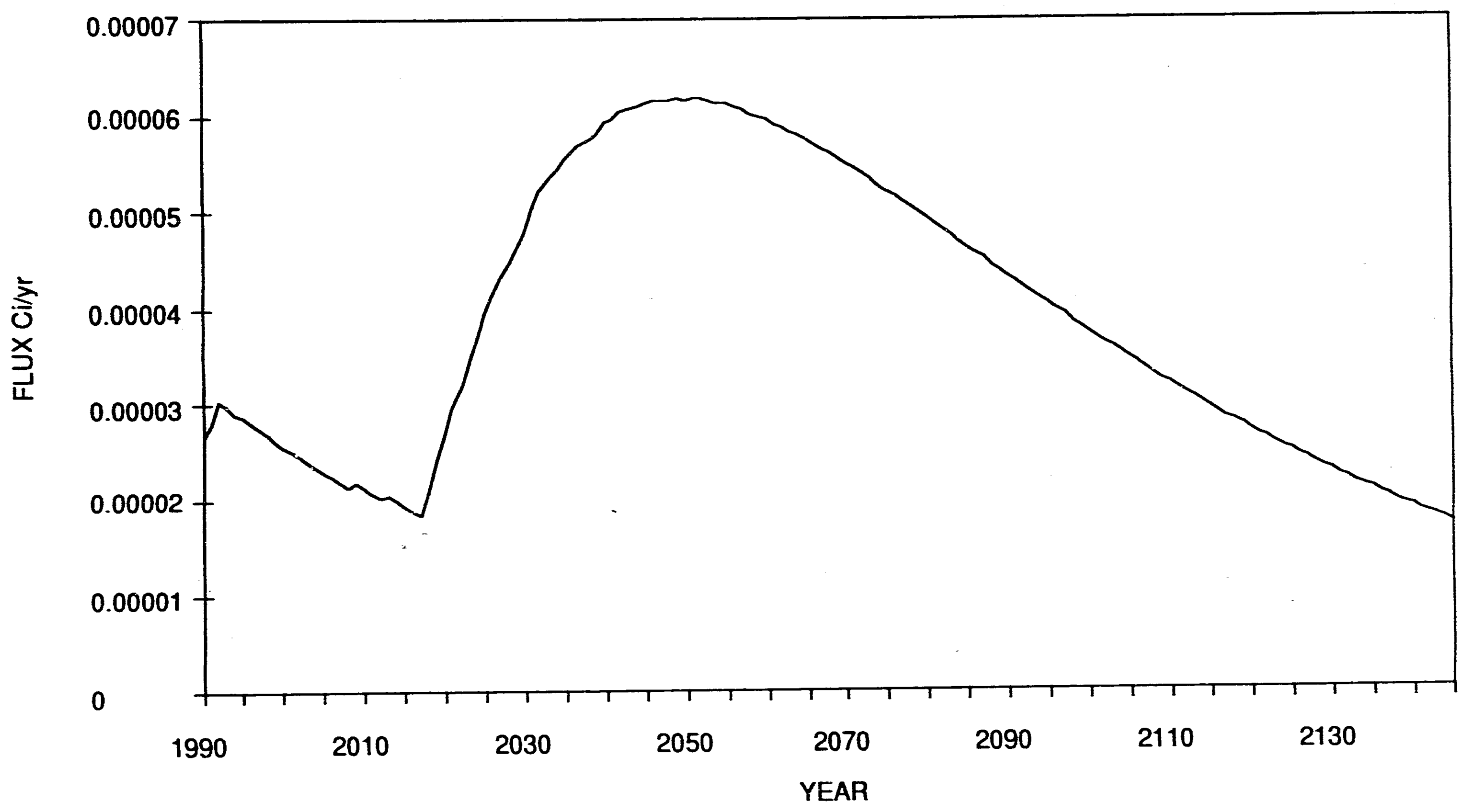


Fig. 5.50. Cesium-137 flux in groundwater across the WAG 6 boundary.

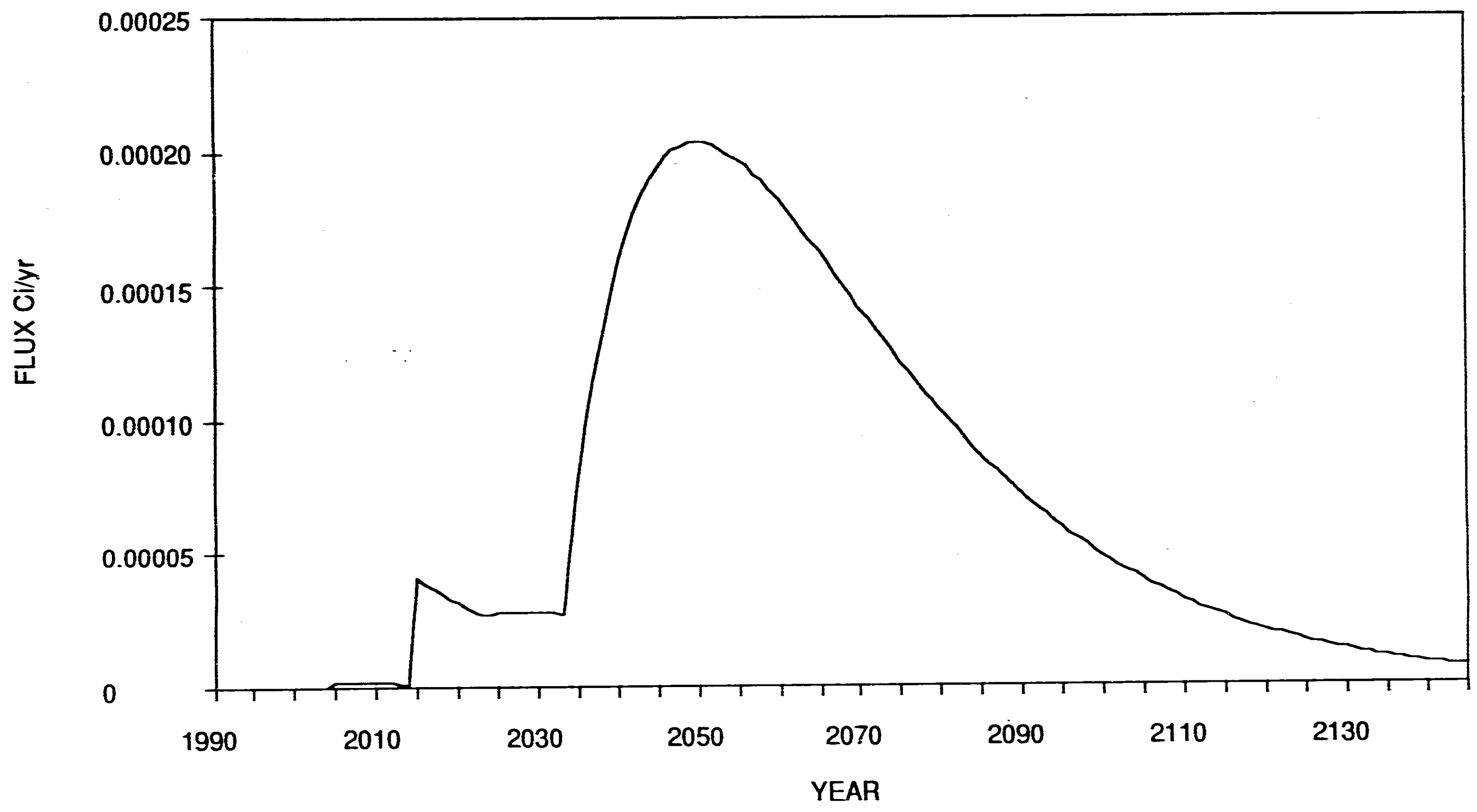


Fig. 5.51. Europium-152 flux in groundwater across the WAG 6 boundary.

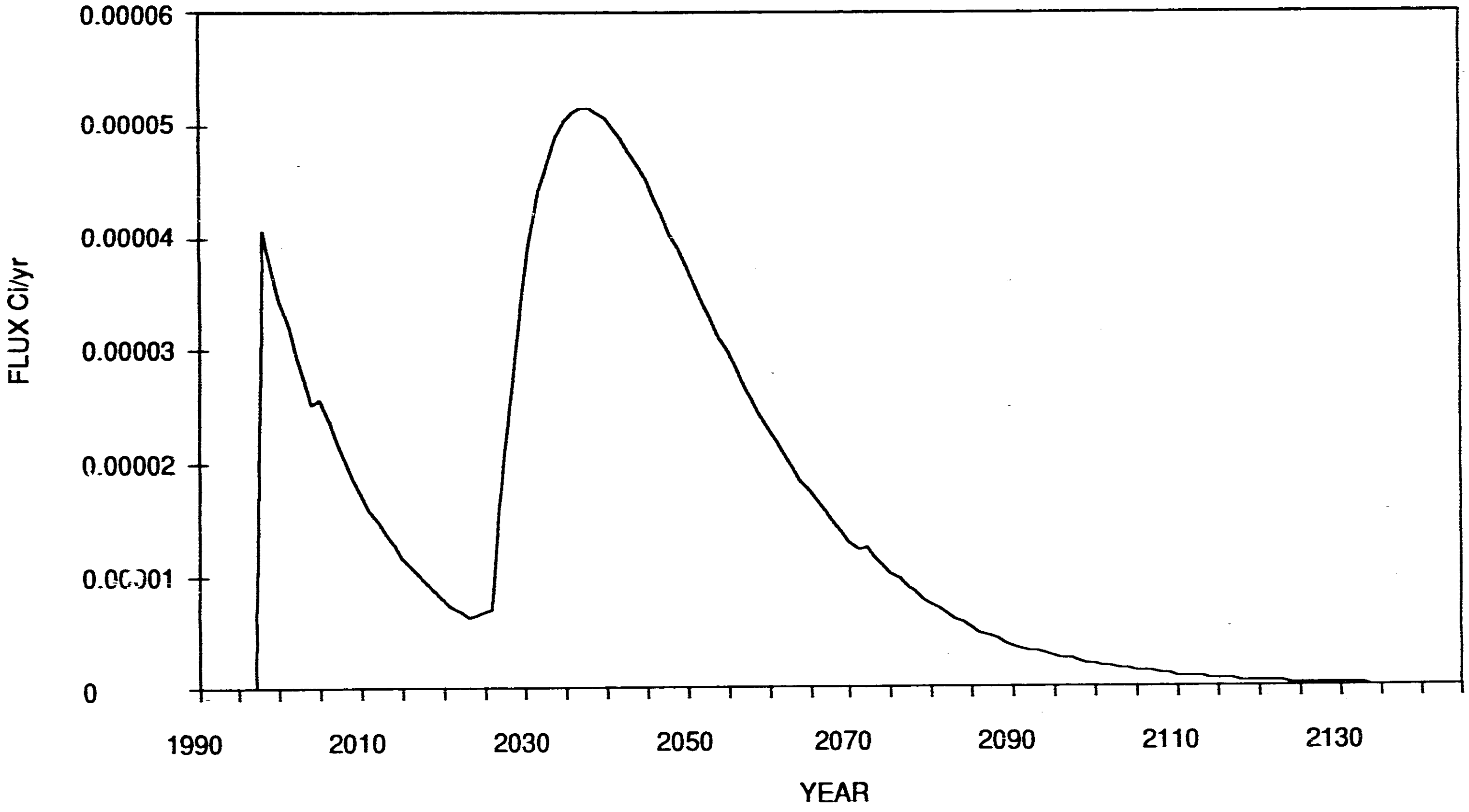


Fig. 5.52. Europium-154 flux in groundwater across the WAG 6 boundary.

5.18D 4104.124

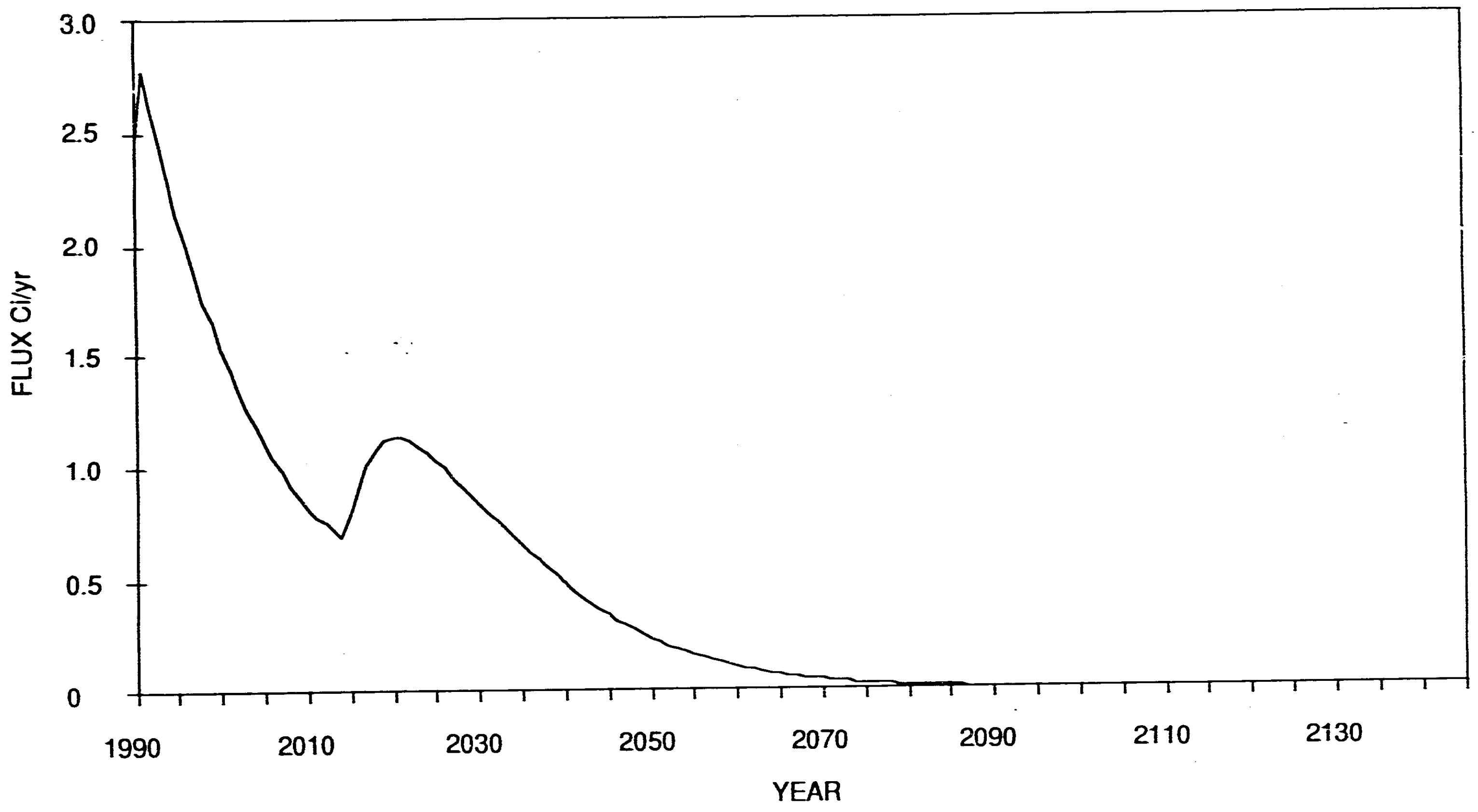


Fig. 5.53. Tritium flux in groundwater across the WAG 6 boundary.

5.18D 4104.125

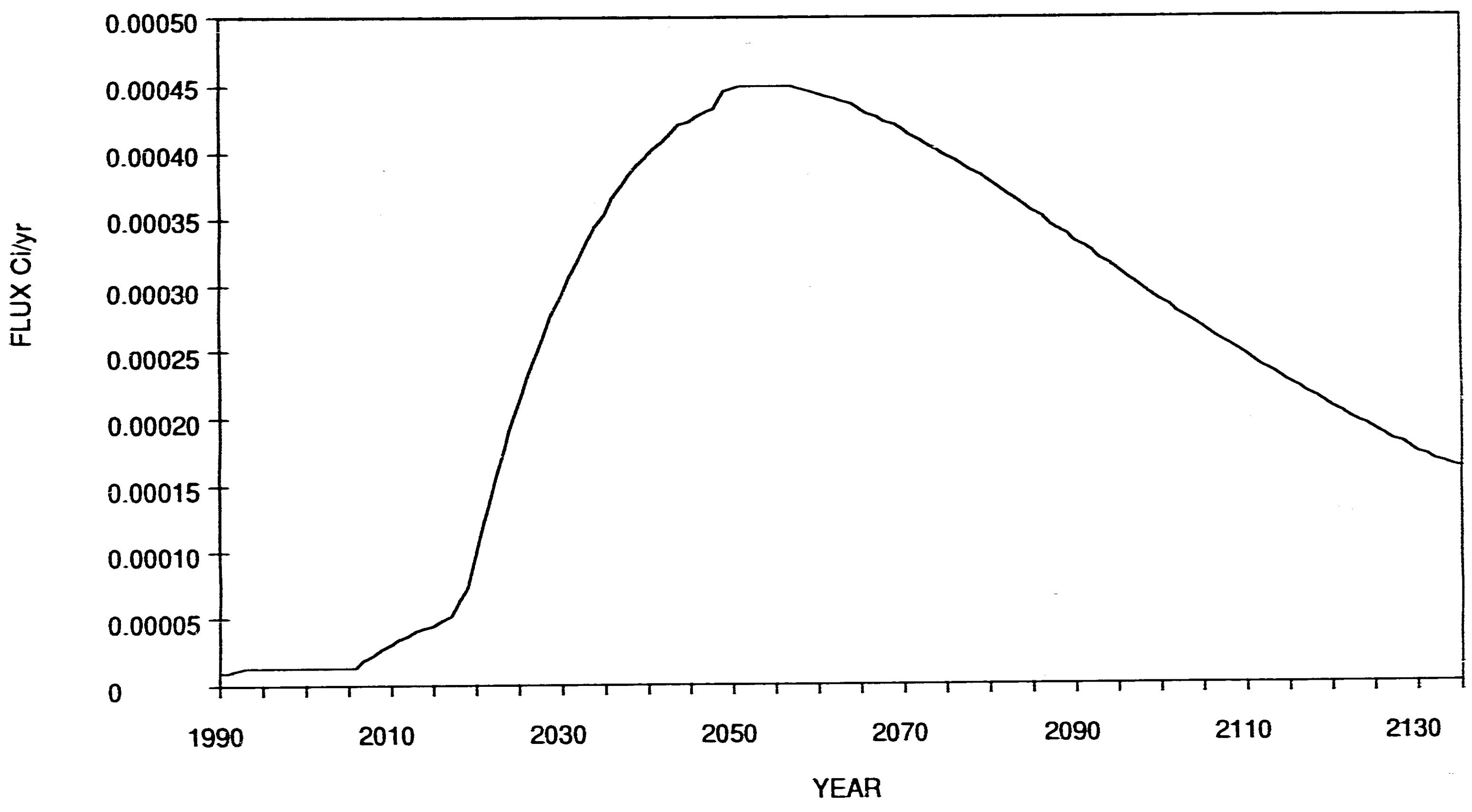


Fig. 5.54. Strontium-90 flux in groundwater across the WAG 6 boundary.

5.18D 4104.126

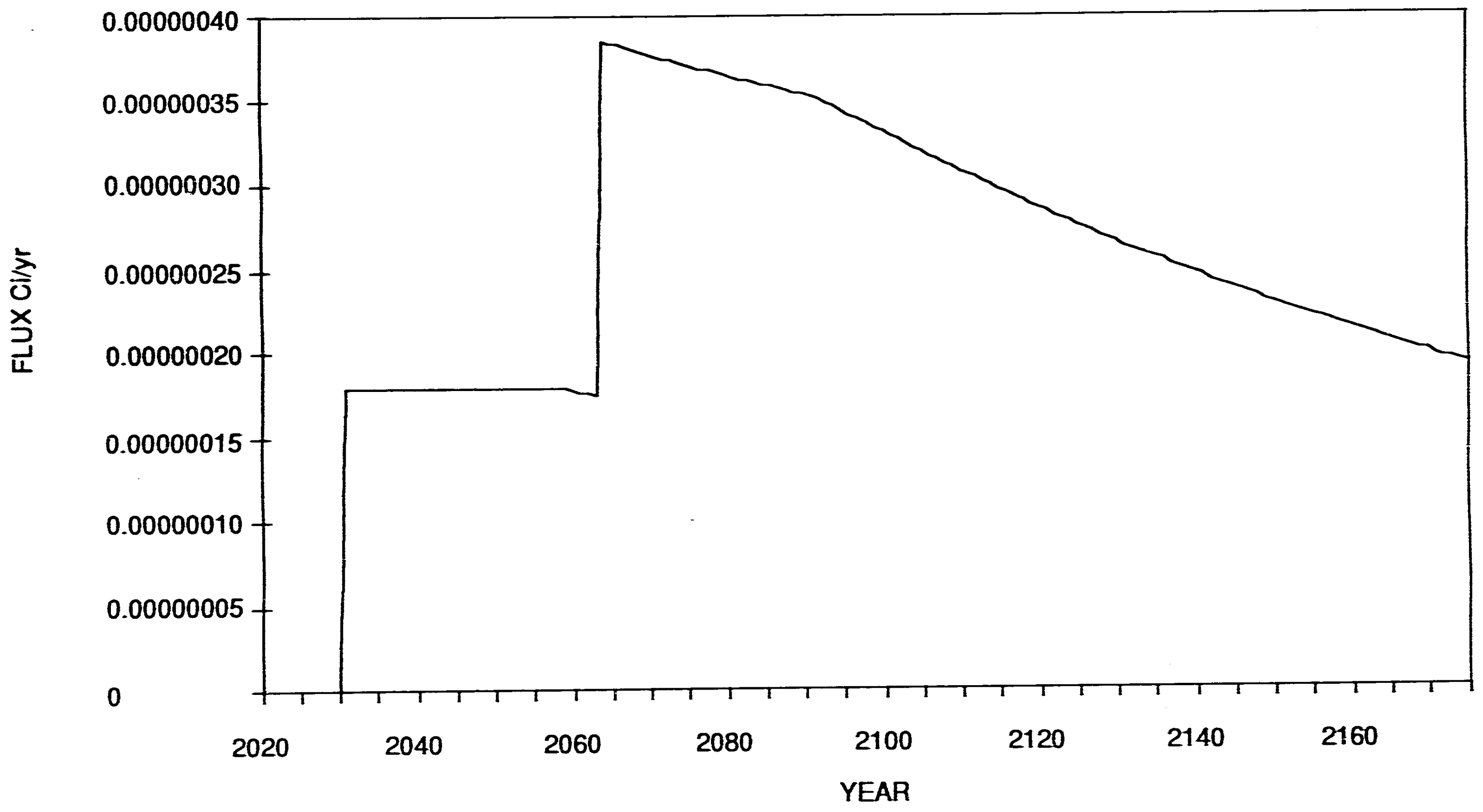


Fig. 5.55. Thorium-232 flux in groundwater across the WAG 6 boundary.

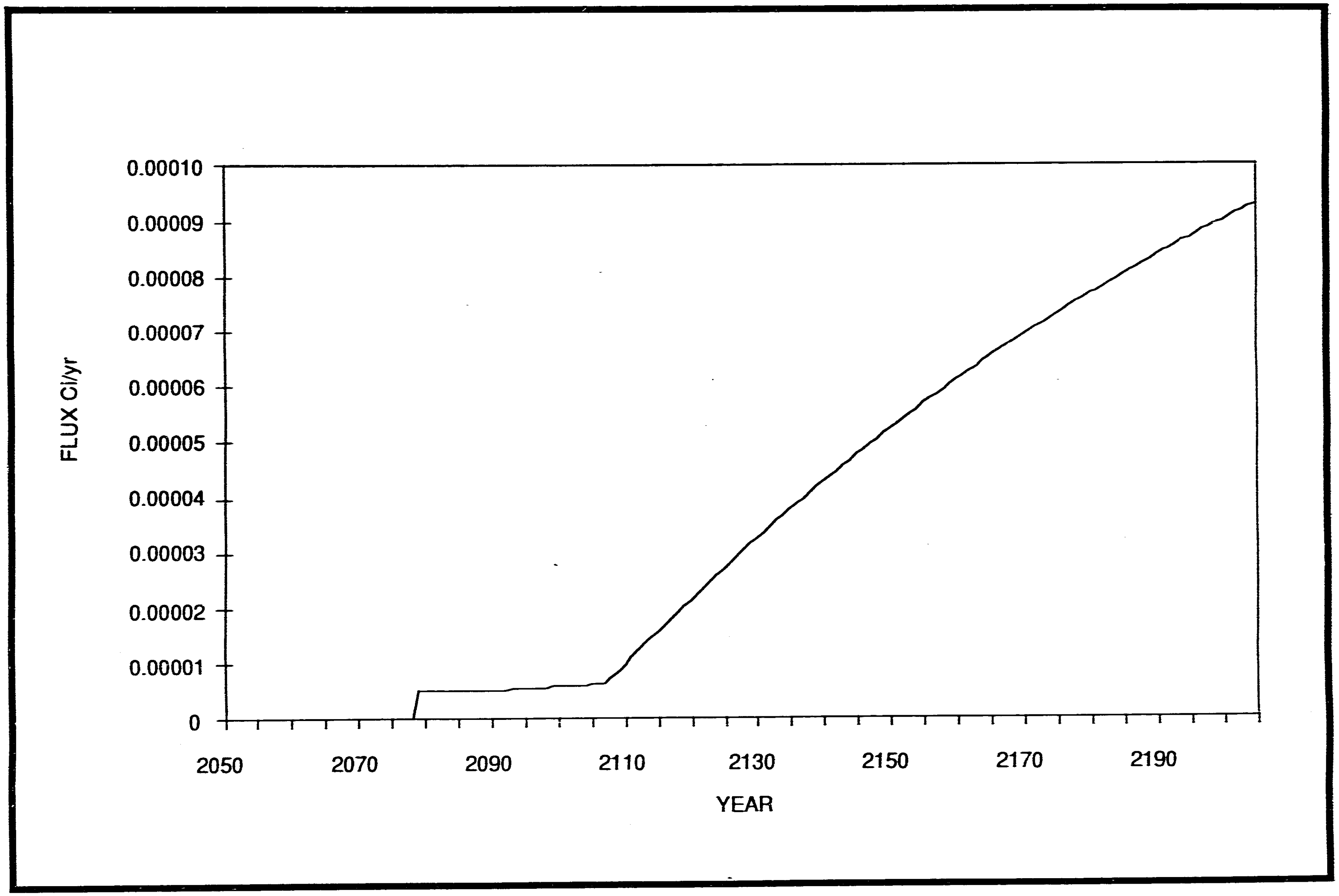


Fig. 5.56. Uranium-233 flux in groundwater across the WAG 6 boundary.

5.18D 4104.128

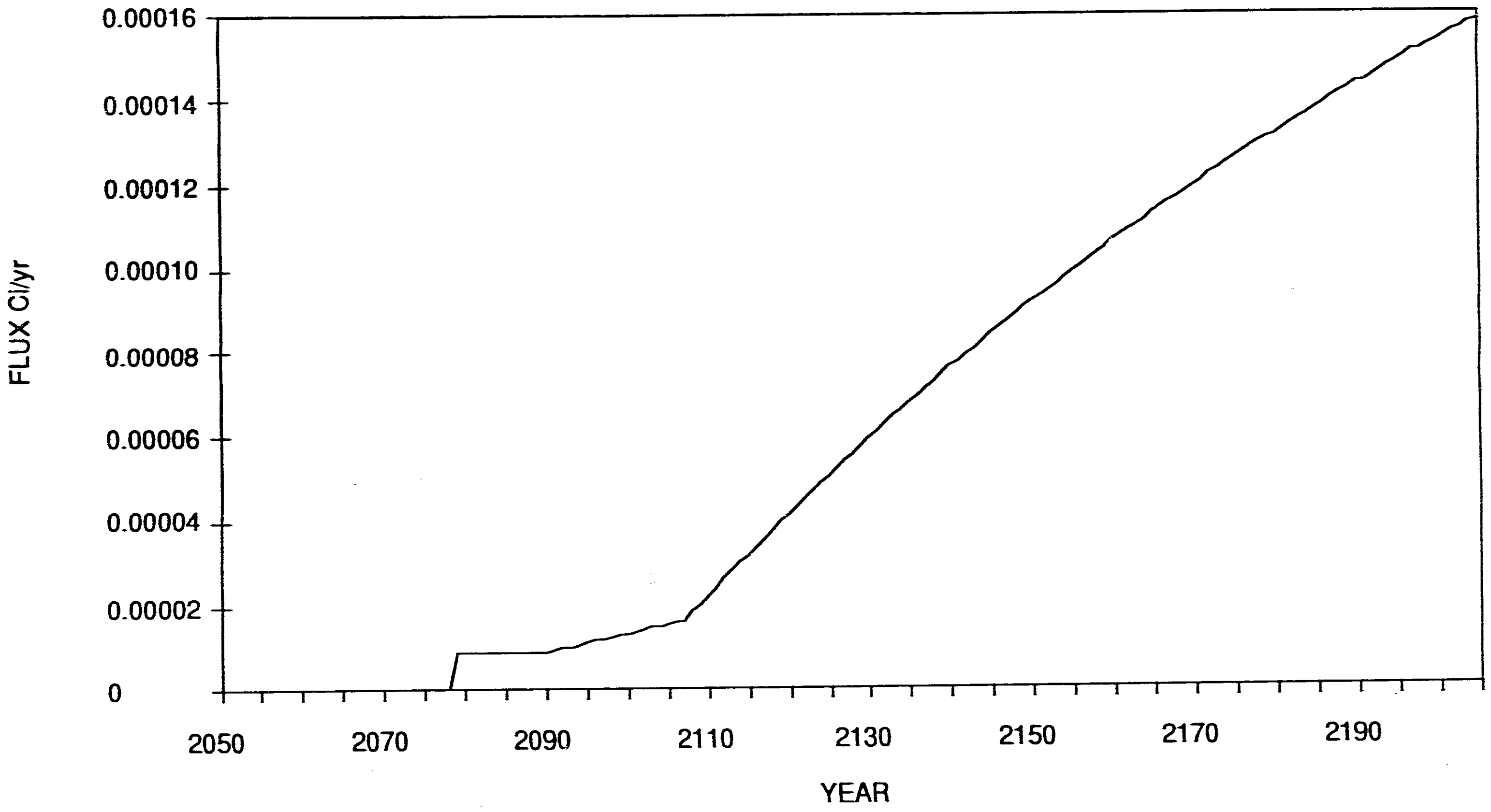


Fig. 5.57. Uranium-238 flux in groundwater across the WAG 6 boundary.

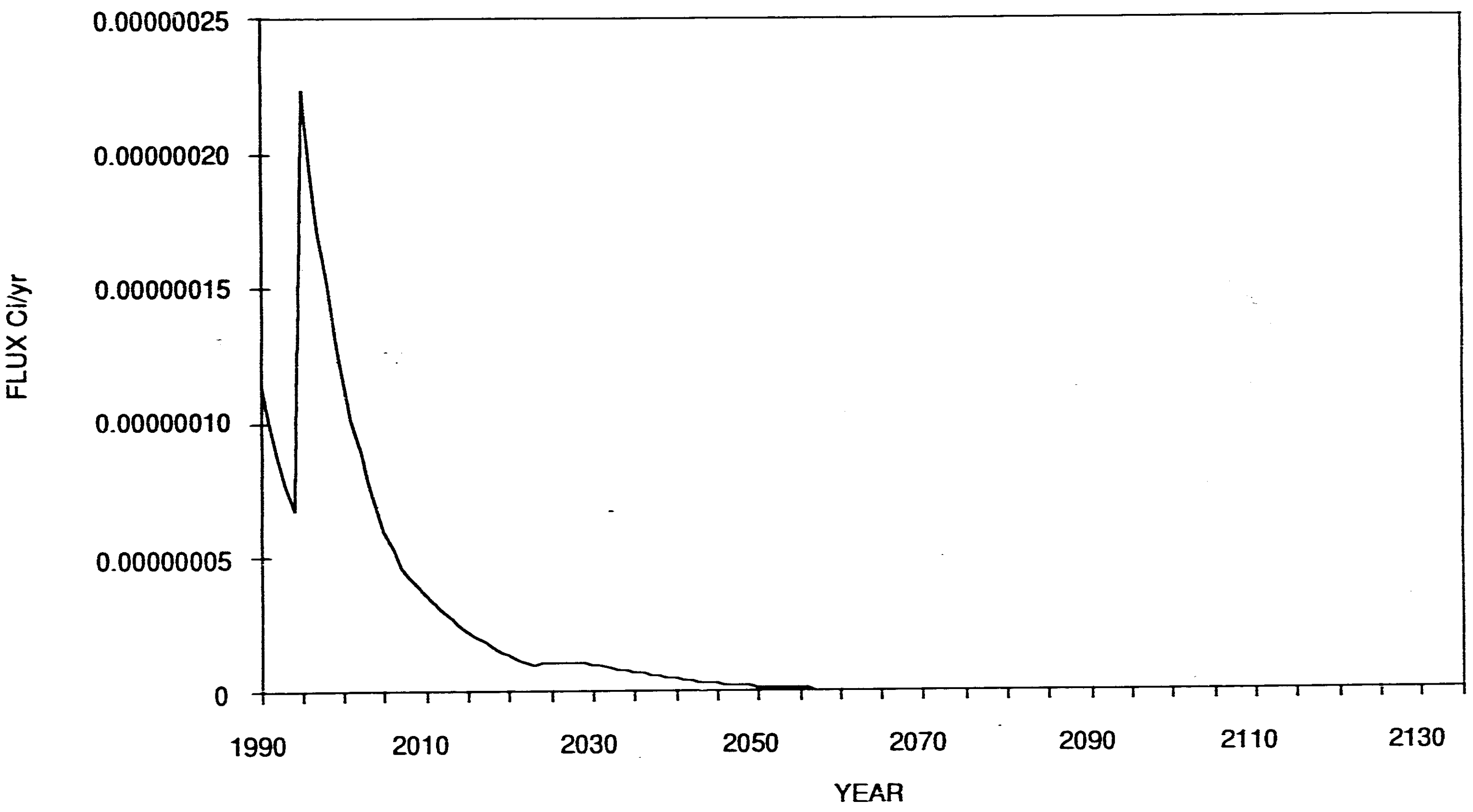


Fig. 5.58. Cobalt-60 flux in groundwater into the EWB.

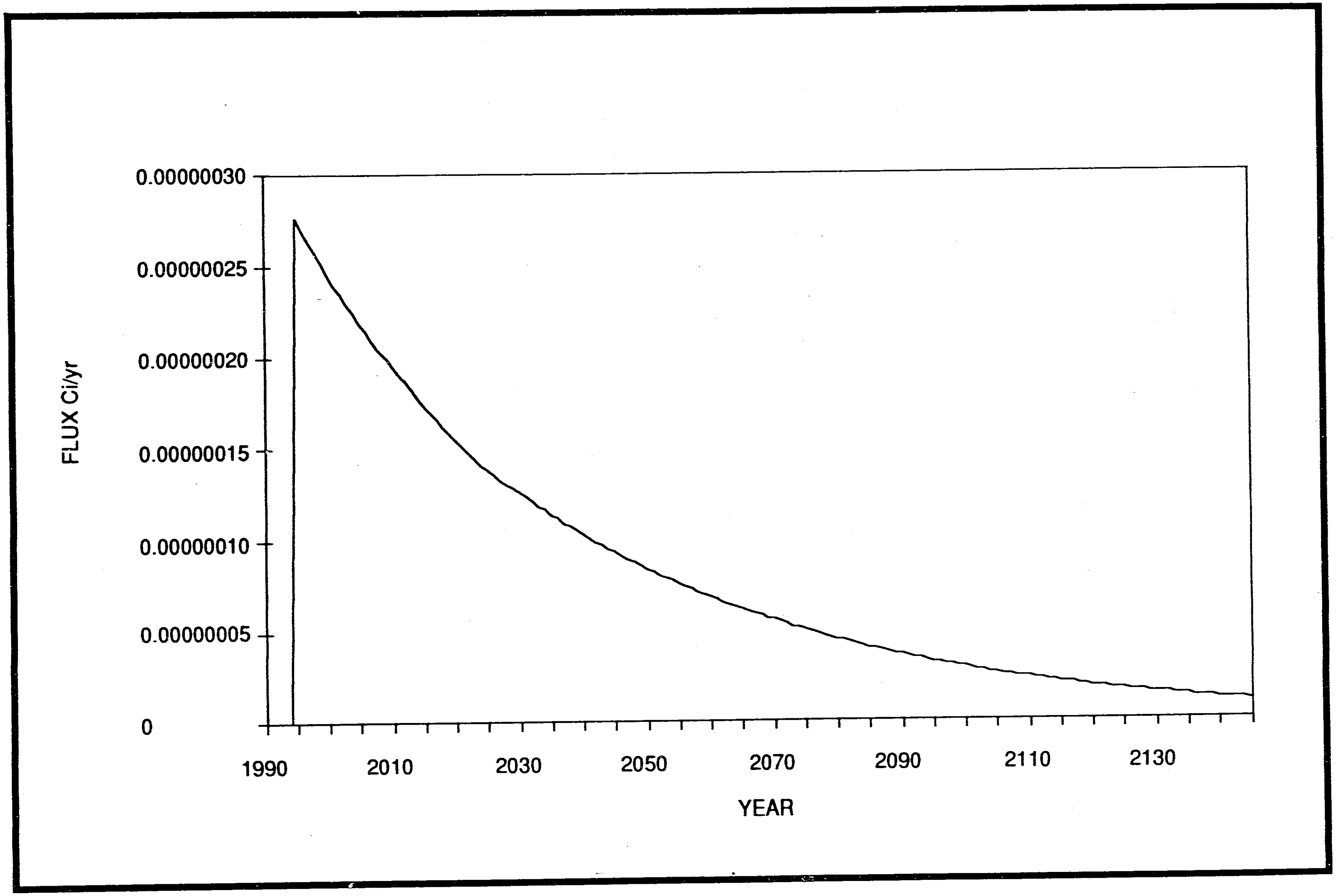


Fig. 5.59. Cesium-137 flux in groundwater into the EWB.

5.13D 4104.131

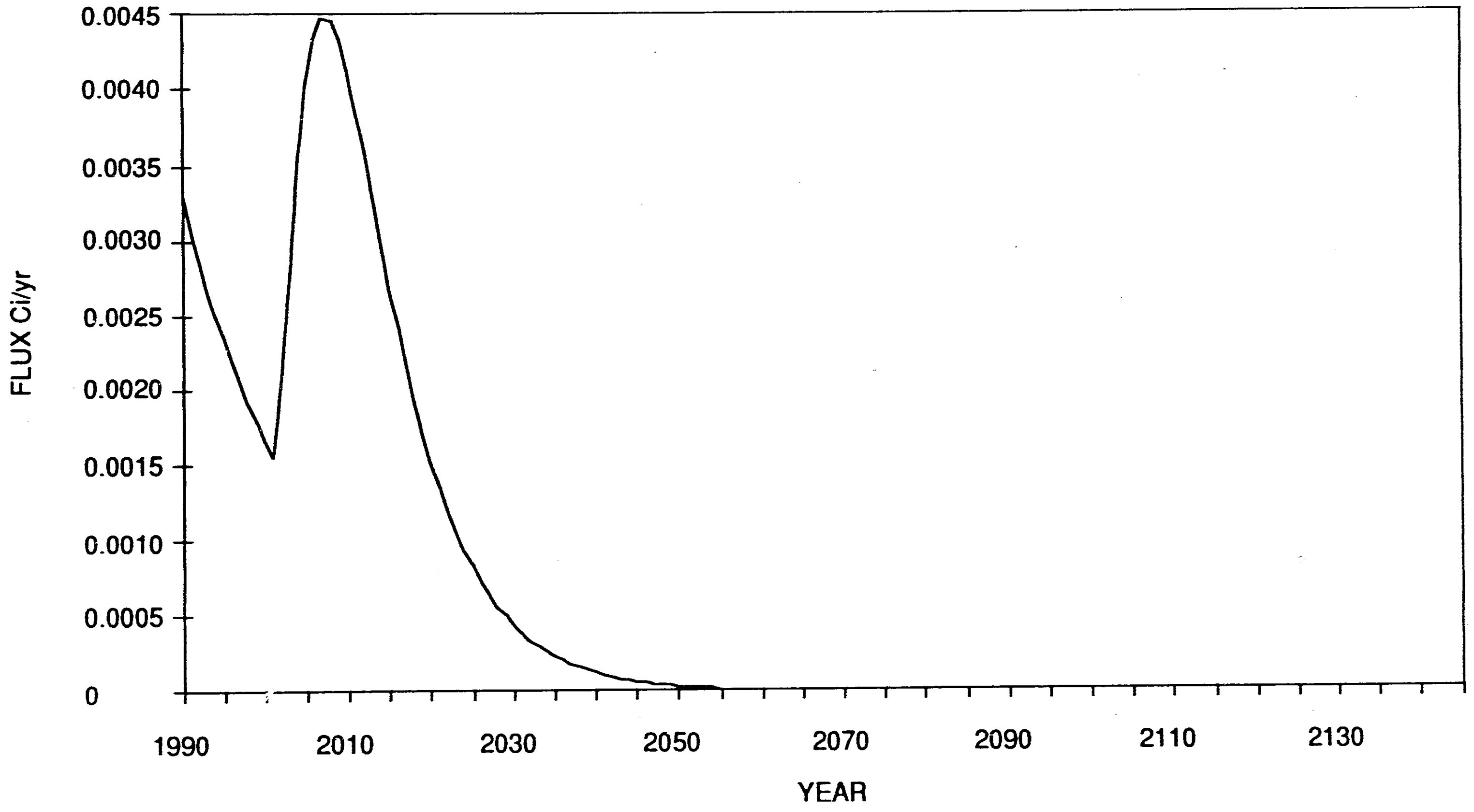


Fig. 5.60. Tritium flux in groundwater into the EWB.

5.18D 4104.132

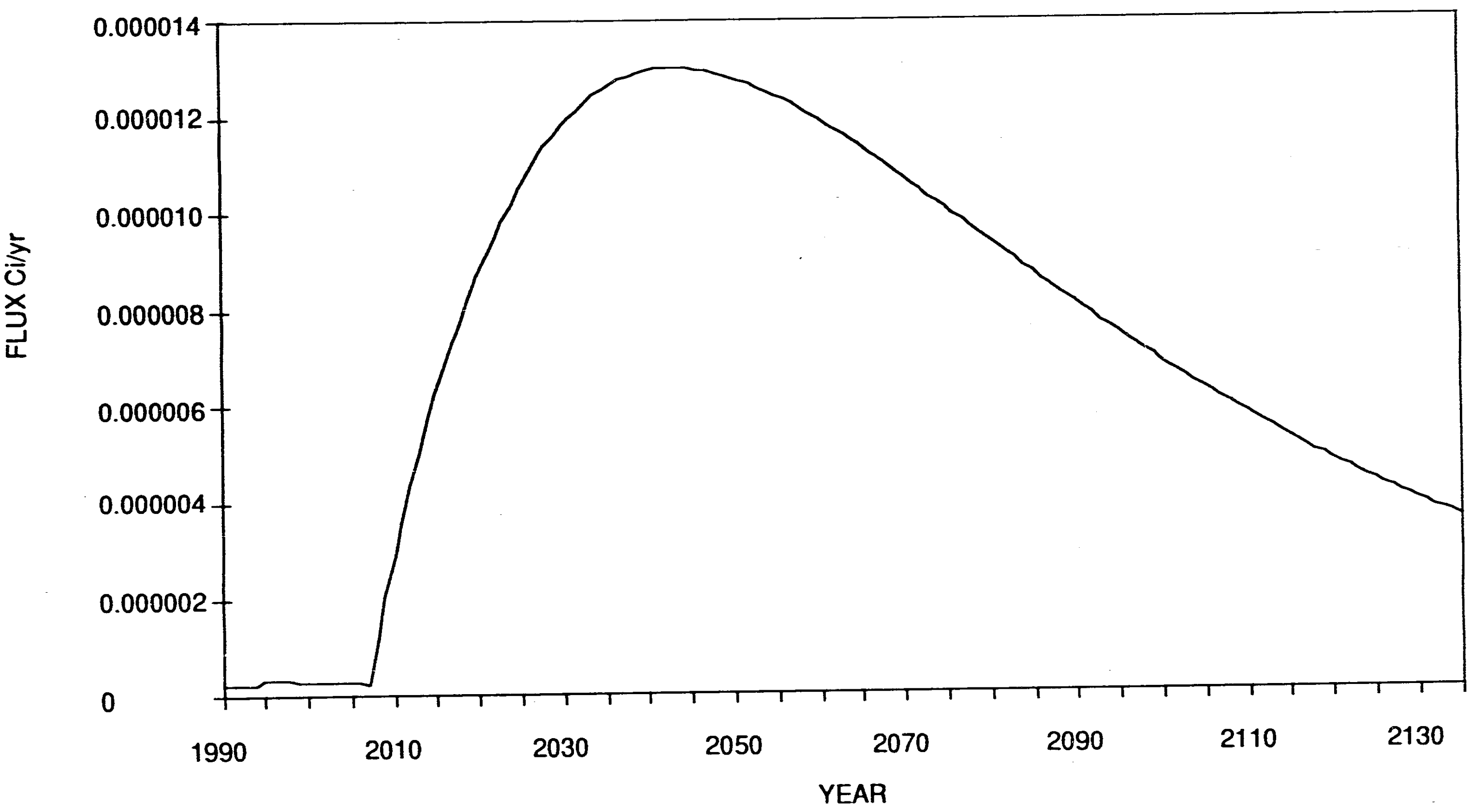


Fig. 5.61. Strontium-90 flux in groundwater into the EWB.

5.18D 4104.133

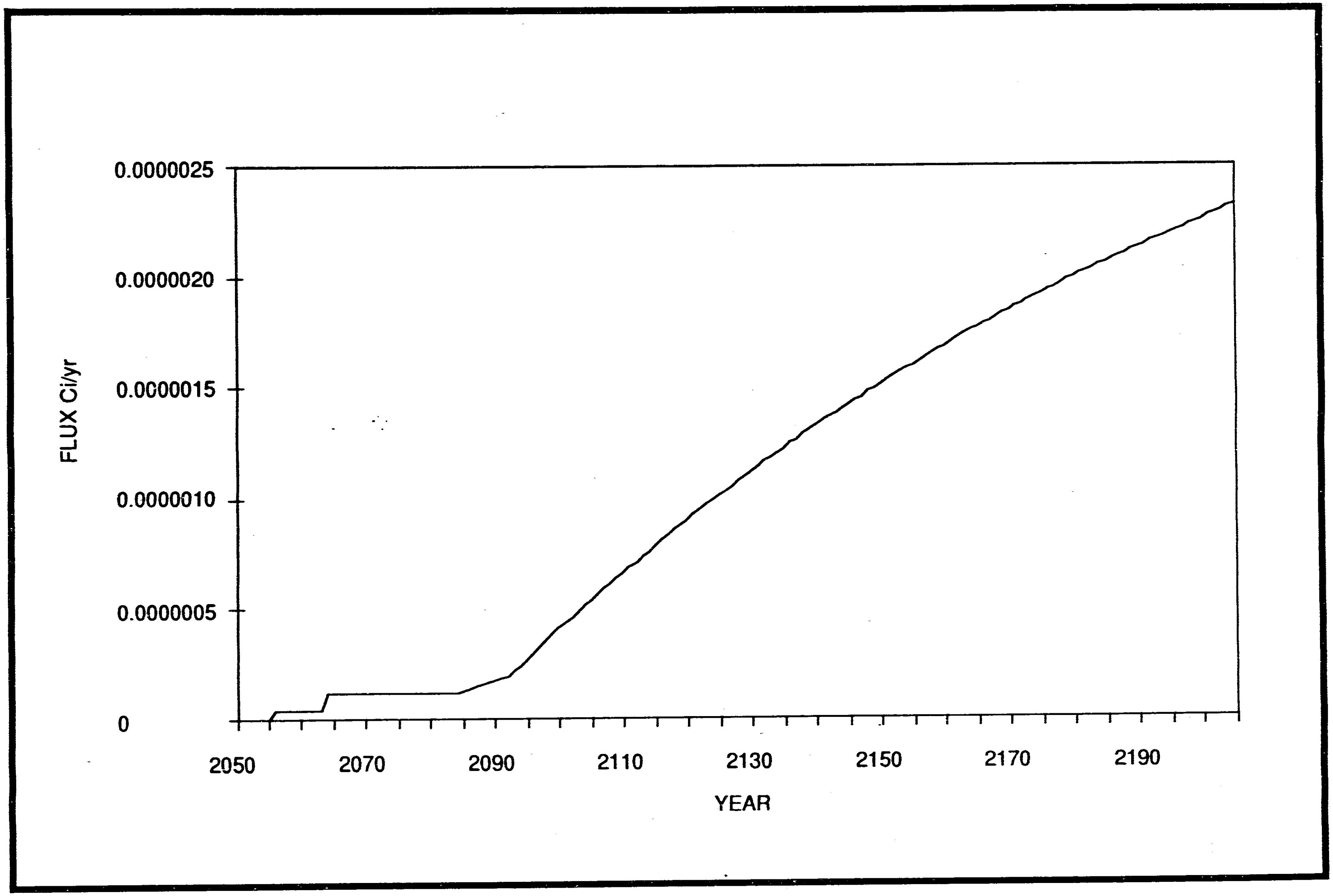


Fig. 5.62. Uranium-233 flux in groundwater into the EWB.

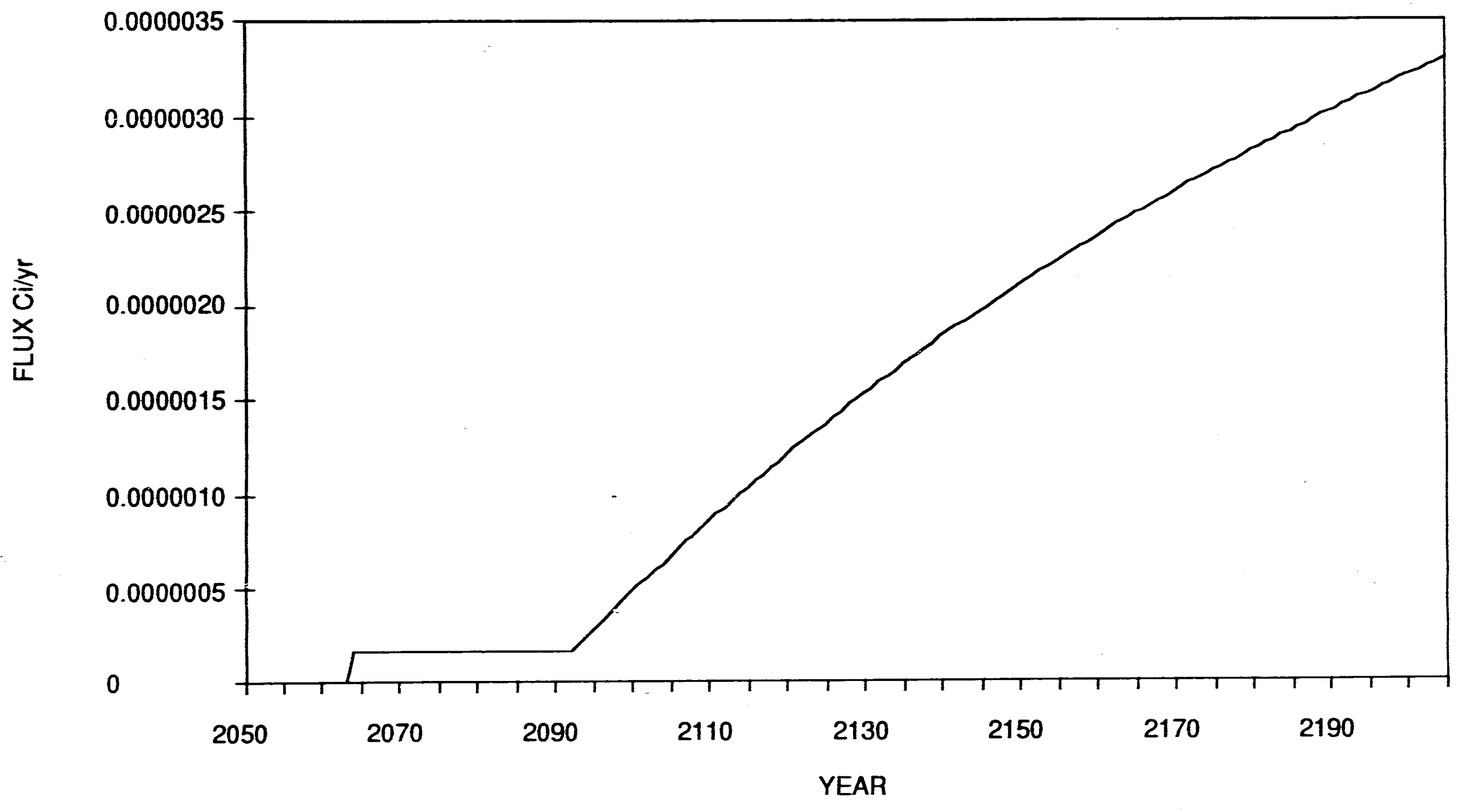


Fig. 5.63. Uranium-238 flux in groundwater into the EWB.

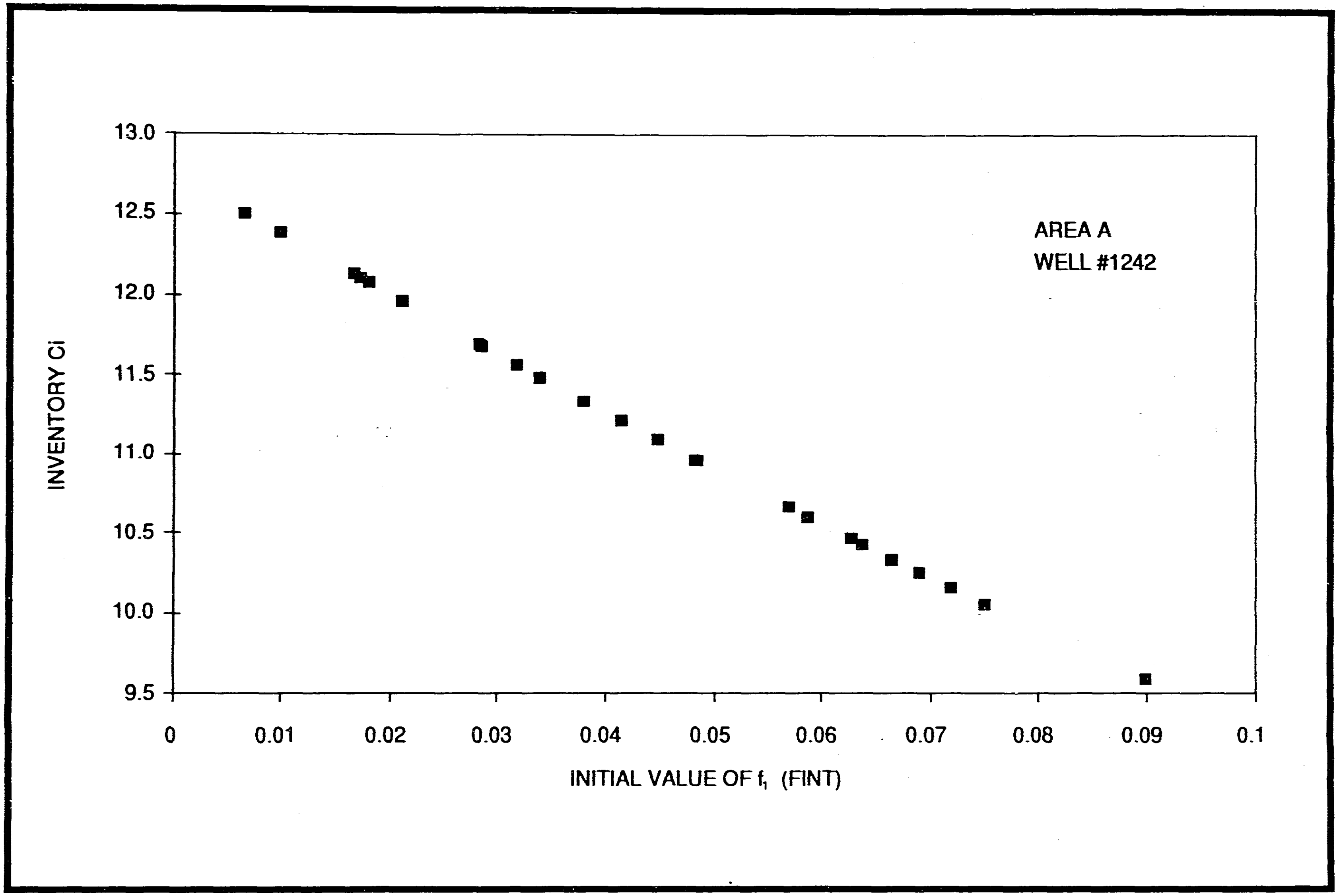


Fig. 5.64. Sensitivity analysis of WAG 6 contaminant transport model, tritium inventory — year 30.

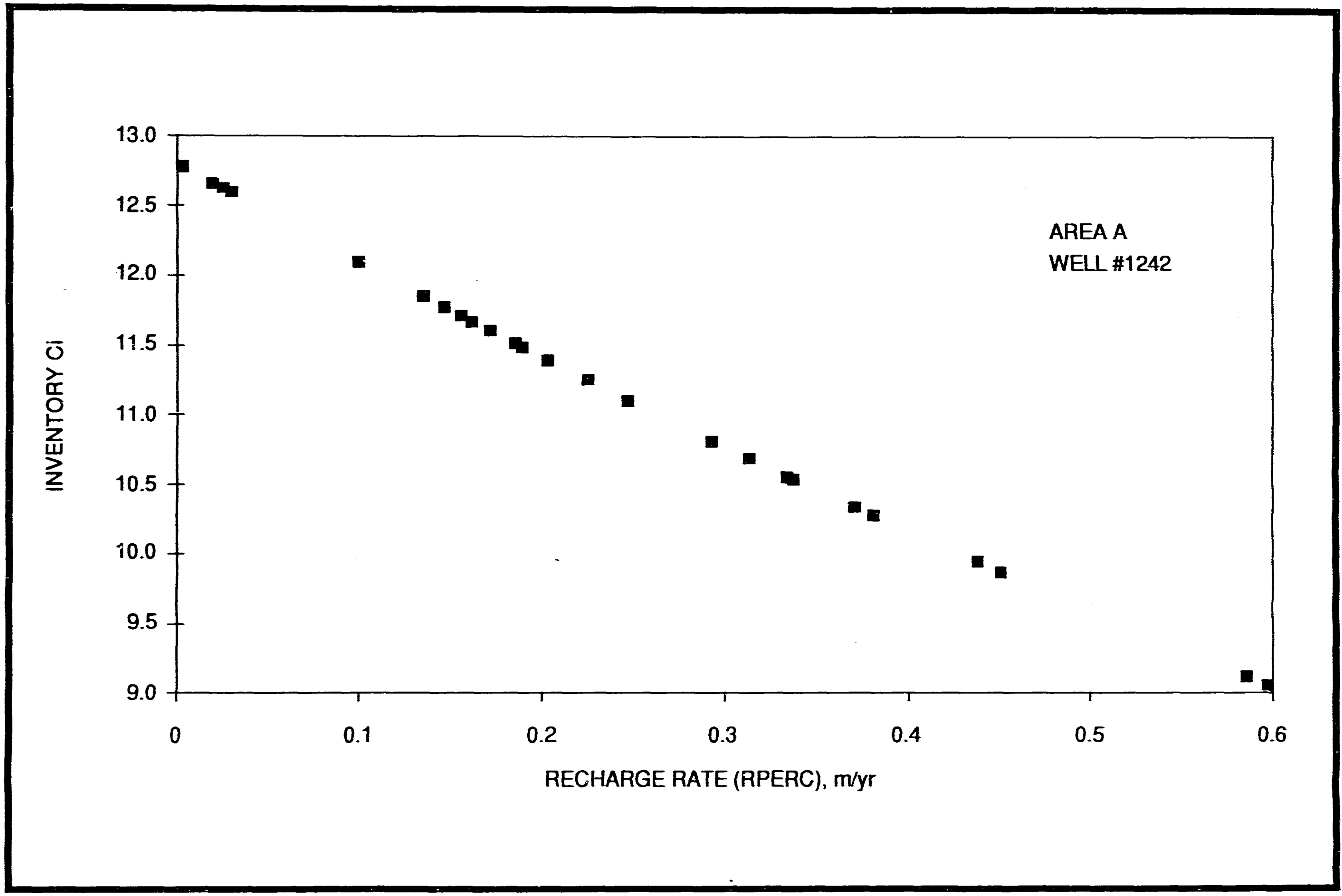


Fig. 5.65. Sensitivity analysis of WAG 6 contaminant transport model, tritium inventory — year 30.

5.18F 4104.149

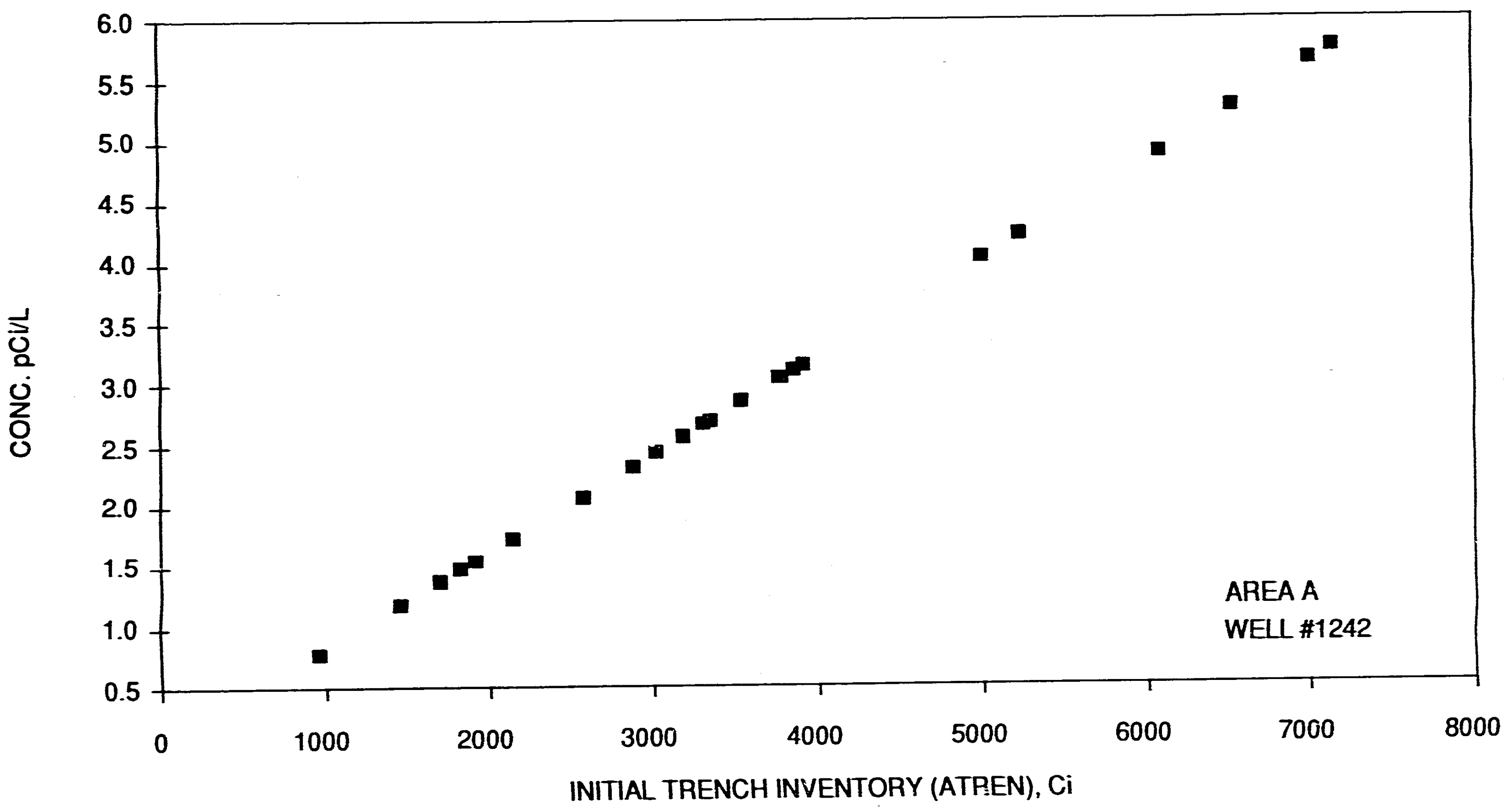


Fig. 5.66. Sensitivity analysis of WAG 6 contaminant transport model, cobalt-60 concentration — year 1.

5.18F 4104.150

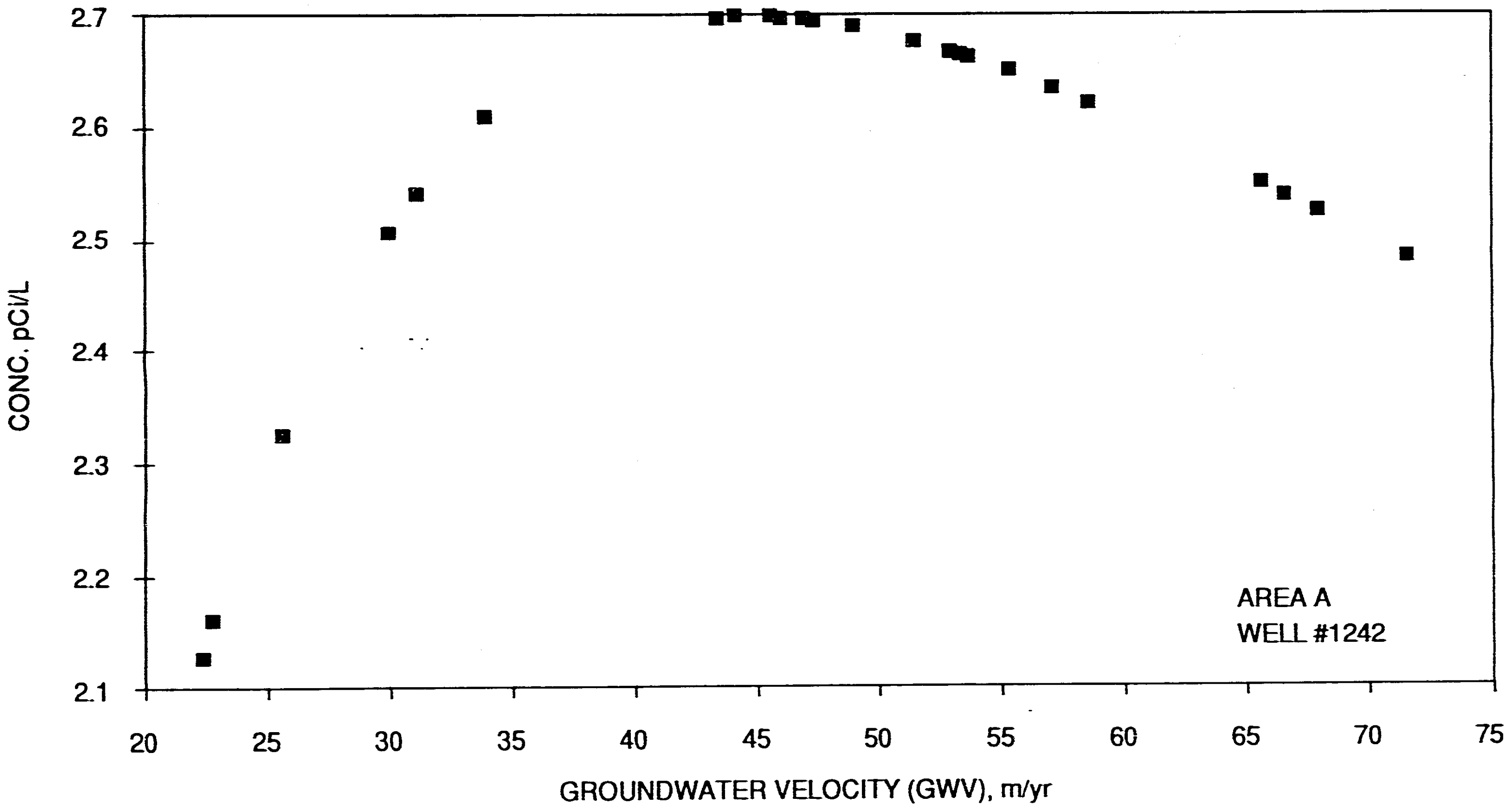


Fig. 5.67. Sensitivity analysis of WAG 6 contaminant transport model, cobalt-60 concentration — year 1.

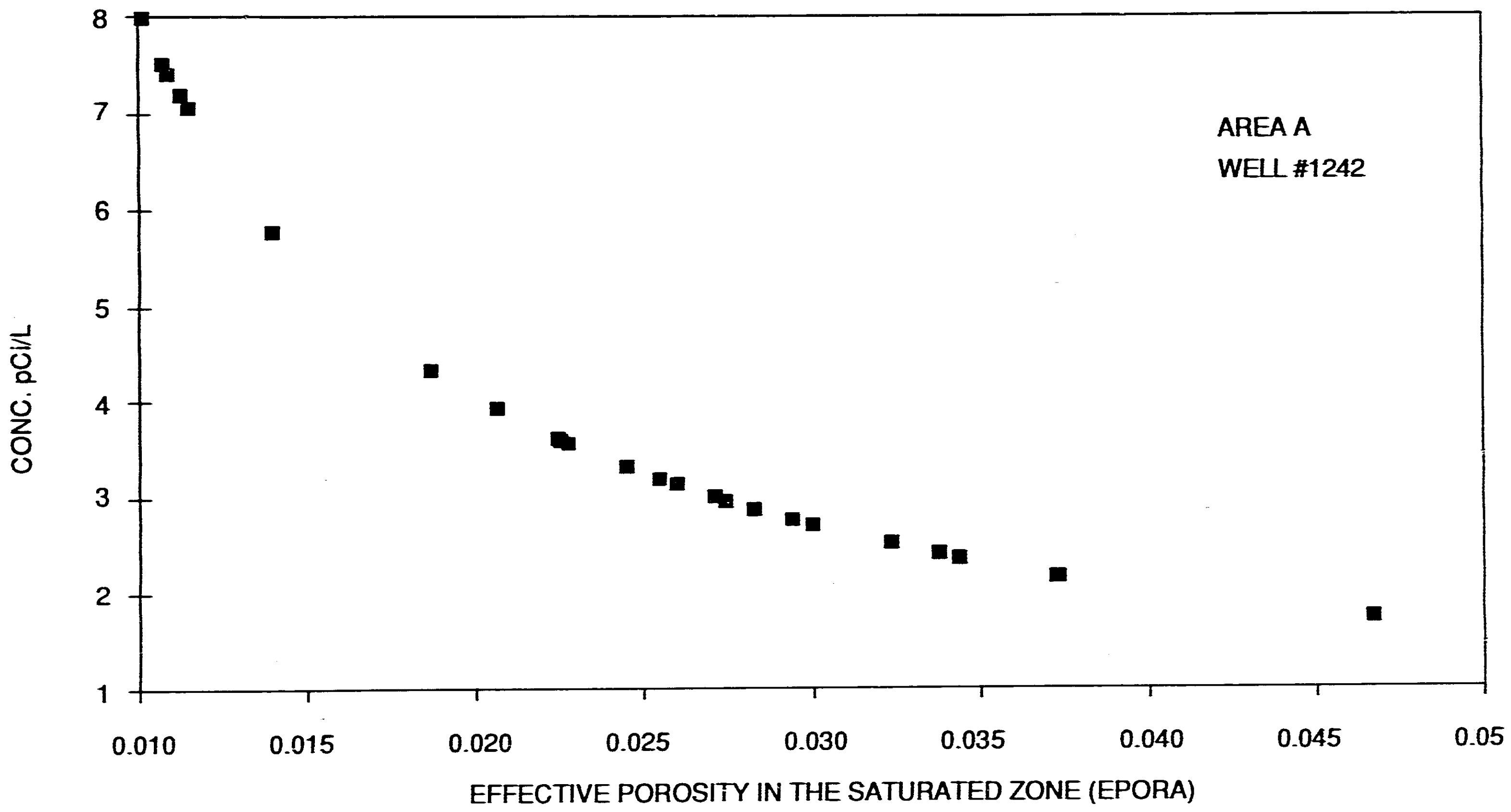


Fig. 5.68. Sensitivity analysis of WAG 6 contaminant transport model, cobalt-60 concentration — year 1.

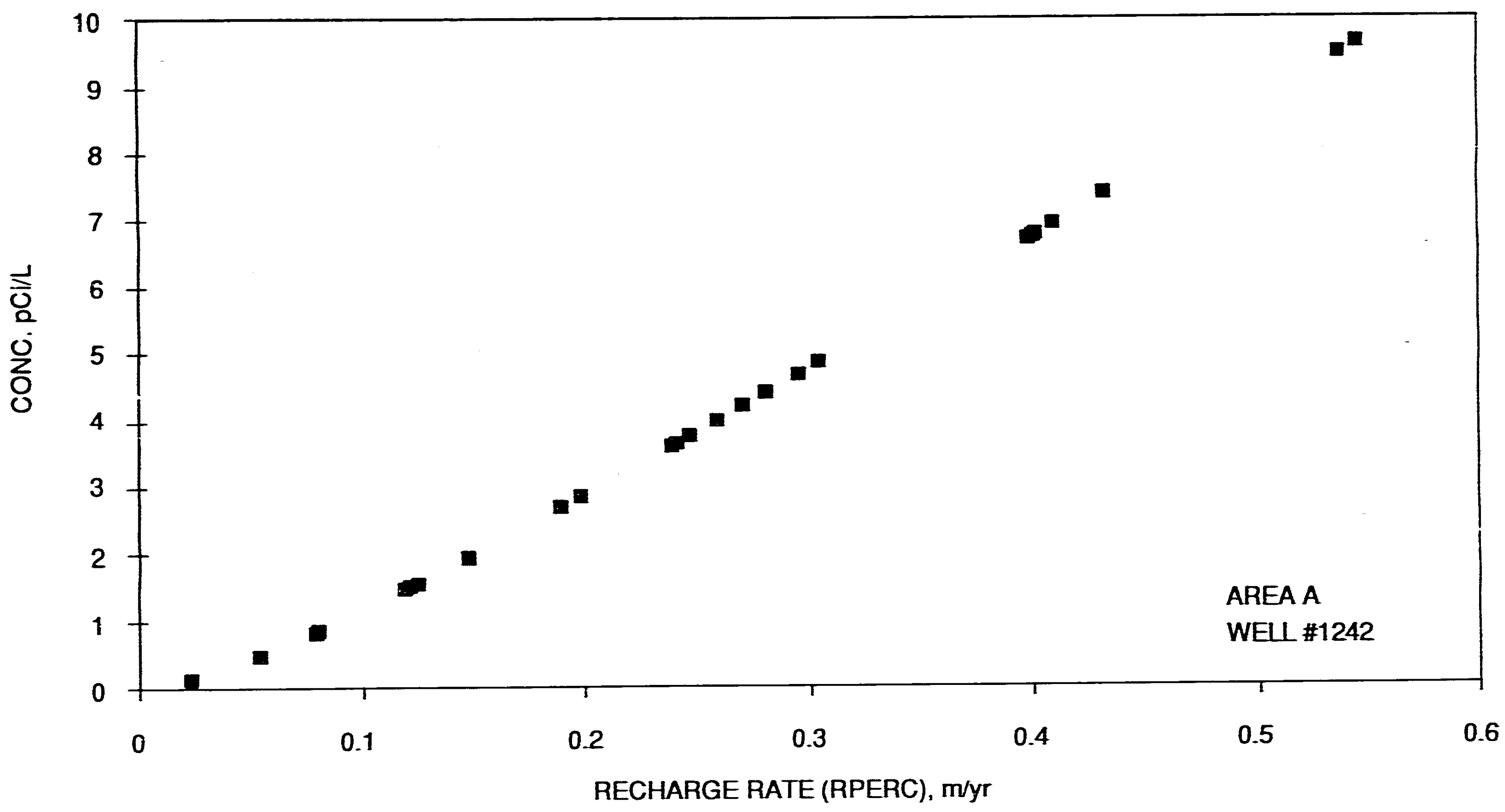


Fig. 5.69. Sensitivity analysis of WAG 6 contaminant transport model, cobalt-60 concentration — year 1.

5.18F 4104.153

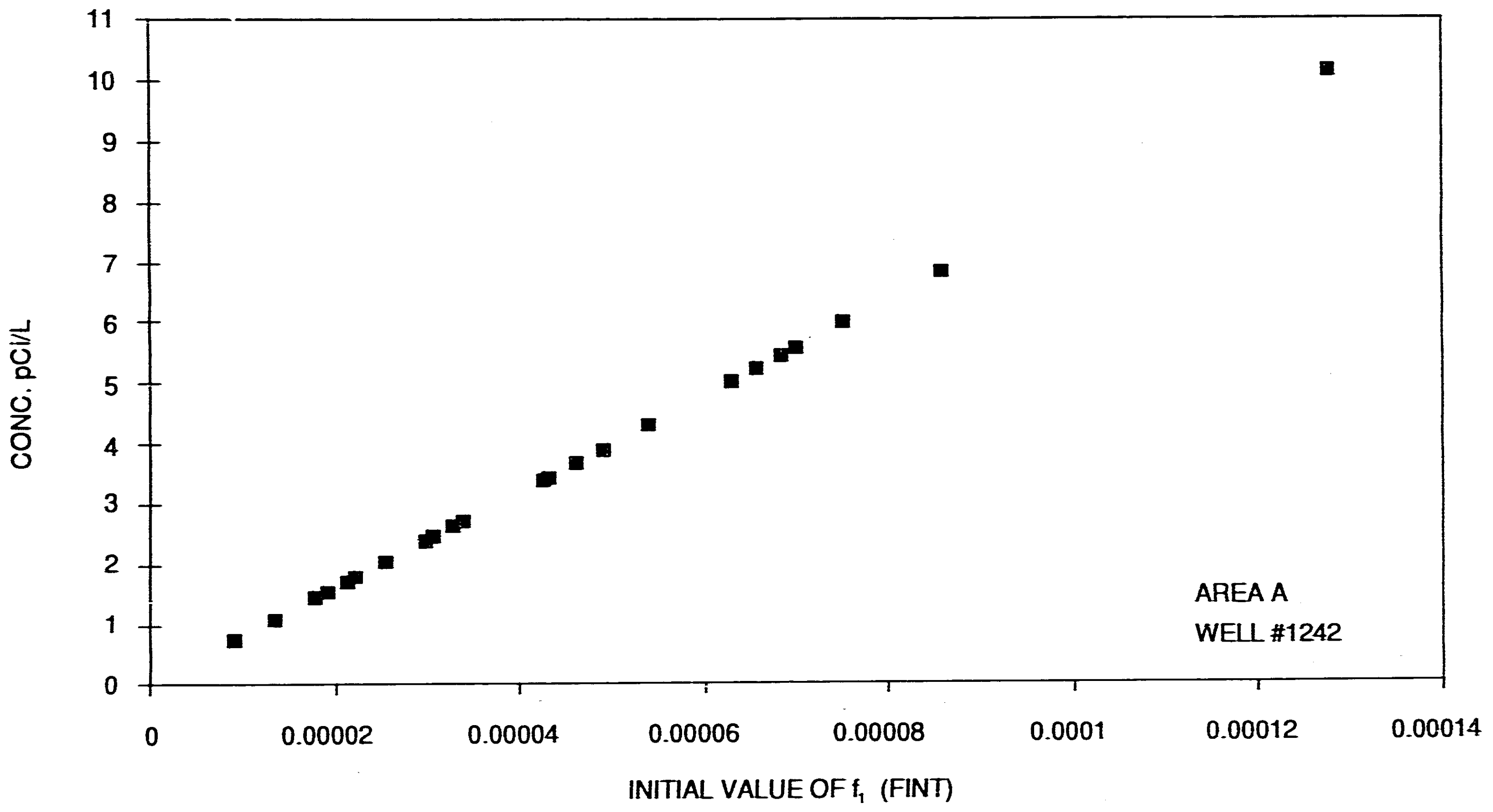


Fig. 5.70. Sensitivity analysis of WAG 6 contaminant transport model, cobalt-60 concentration — year 1.

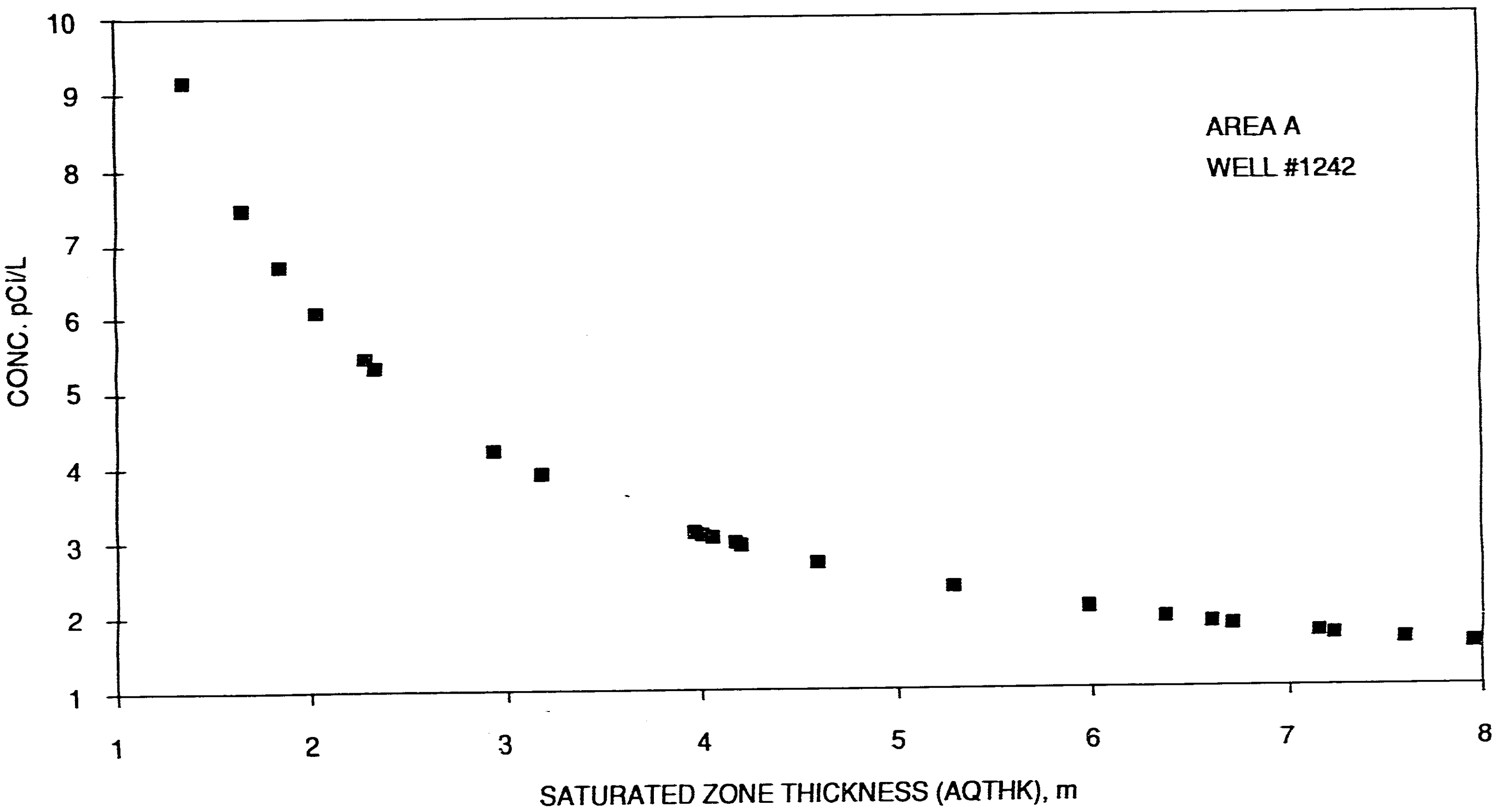


Fig. 5.71. Sensitivity analysis of WAG 6 contaminant transport model, cobalt-60 concentration — year 1.

5.18F 4104.155

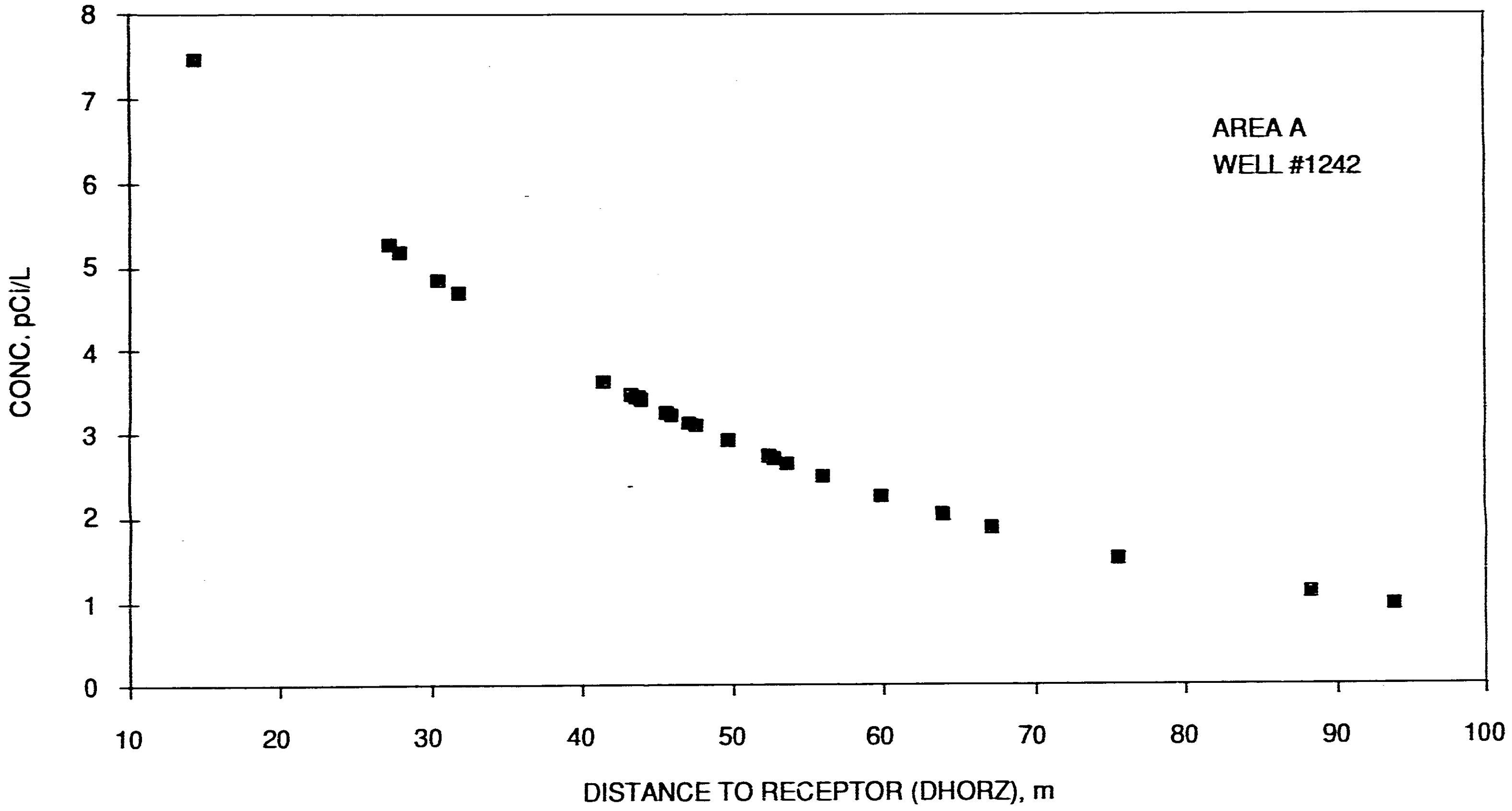


Fig. 5.72. Sensitivity analysis of WAG 6 contaminant transport model, cobalt-60 concentration — year 1.

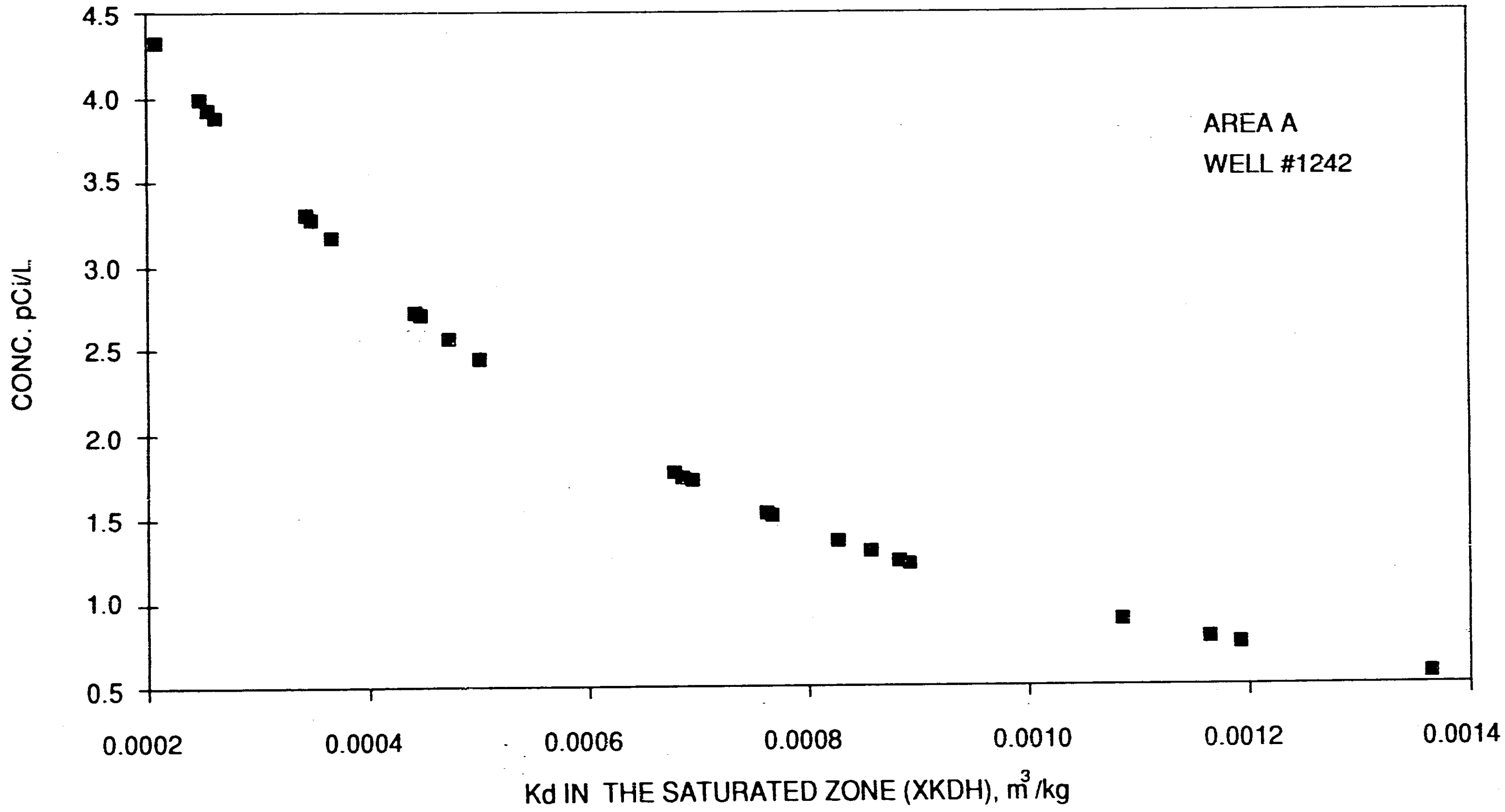


Fig. 5.73. Sensitivity analysis of WAG 6 contaminant transport model, cobalt-60 concentration — year 1.

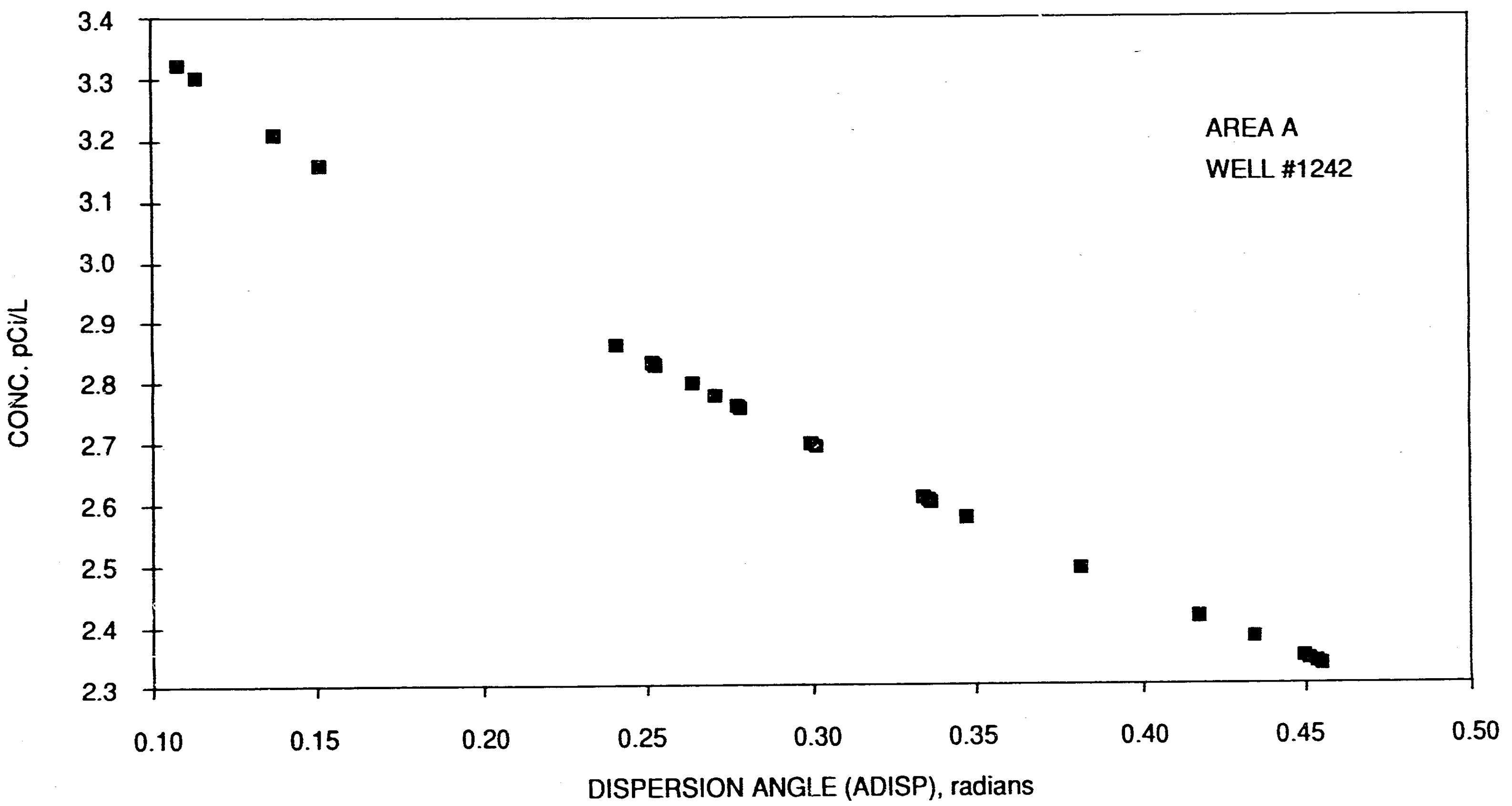


Fig. 5.74. Sensitivity analysis of WAG 6 contaminant transport model, cobalt-60 concentration — year 1.

5.18F 4104.158

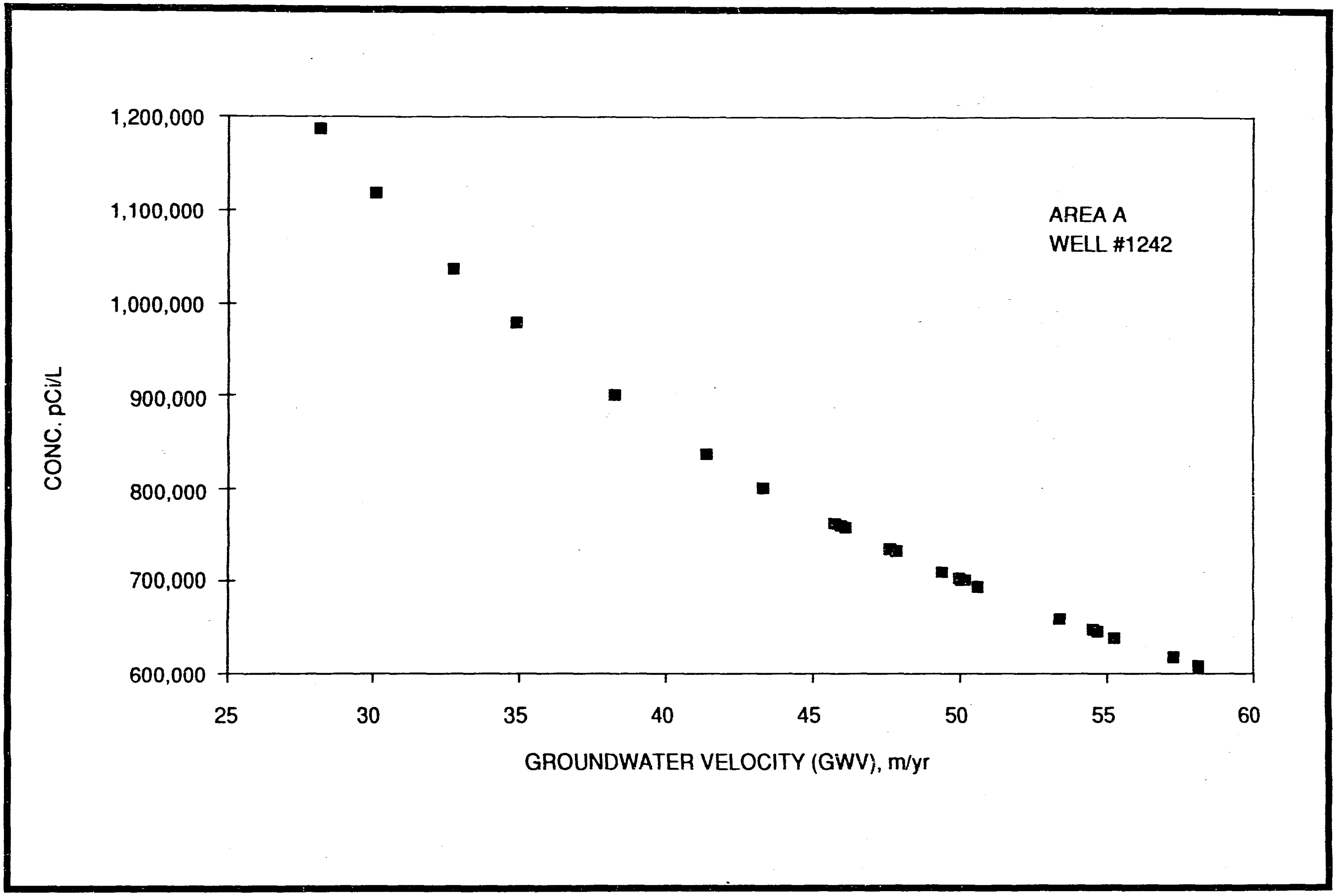


Fig. 5.75. Sensitivity analysis of WAG 6 contaminant transport model, tritium concentration — year 1.

5.18F 4104.159

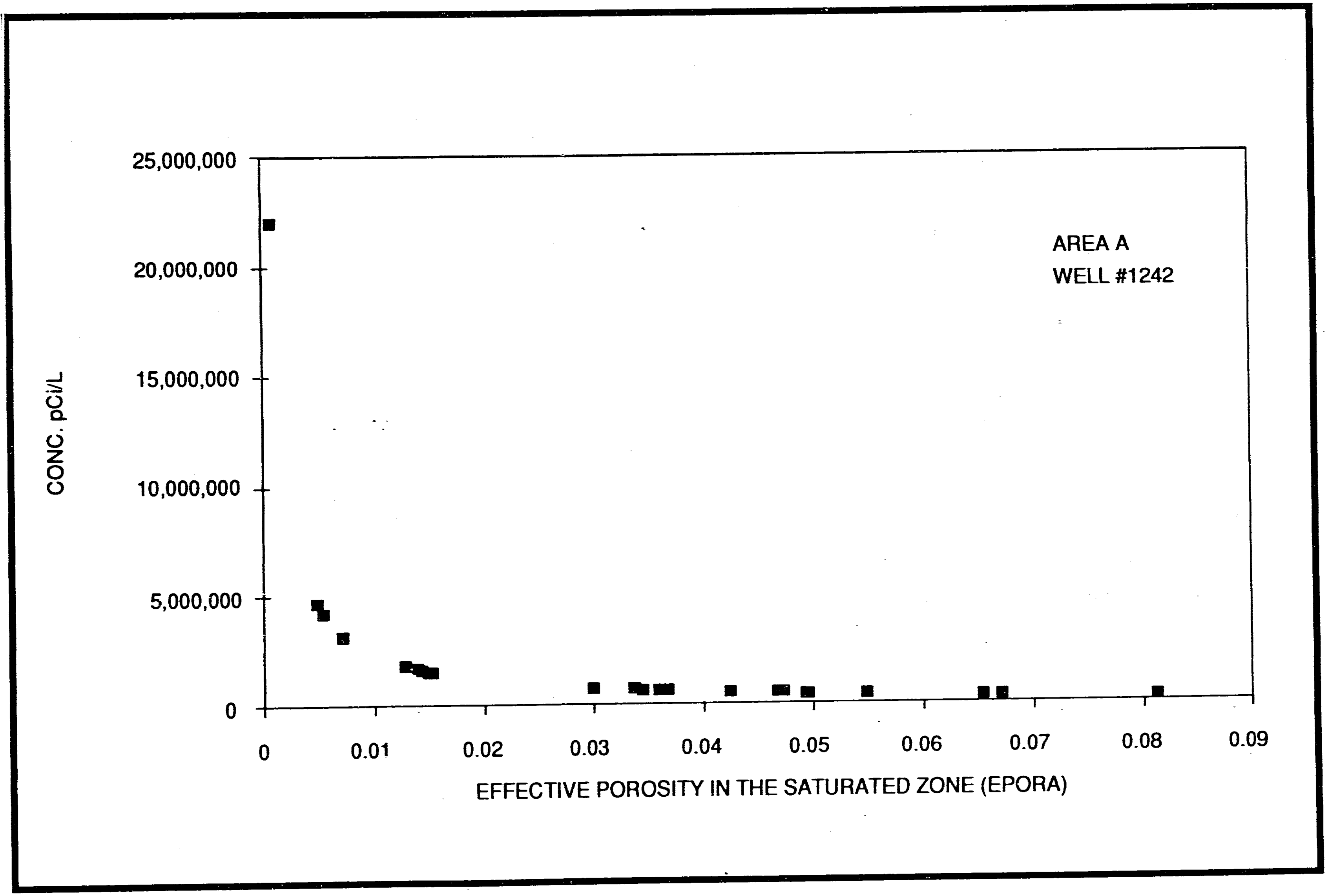


Fig. 5.76. Sensitivity analysis of WAG 6 contaminant transport model, tritium concentration — year 1.

5.18F 4104.160

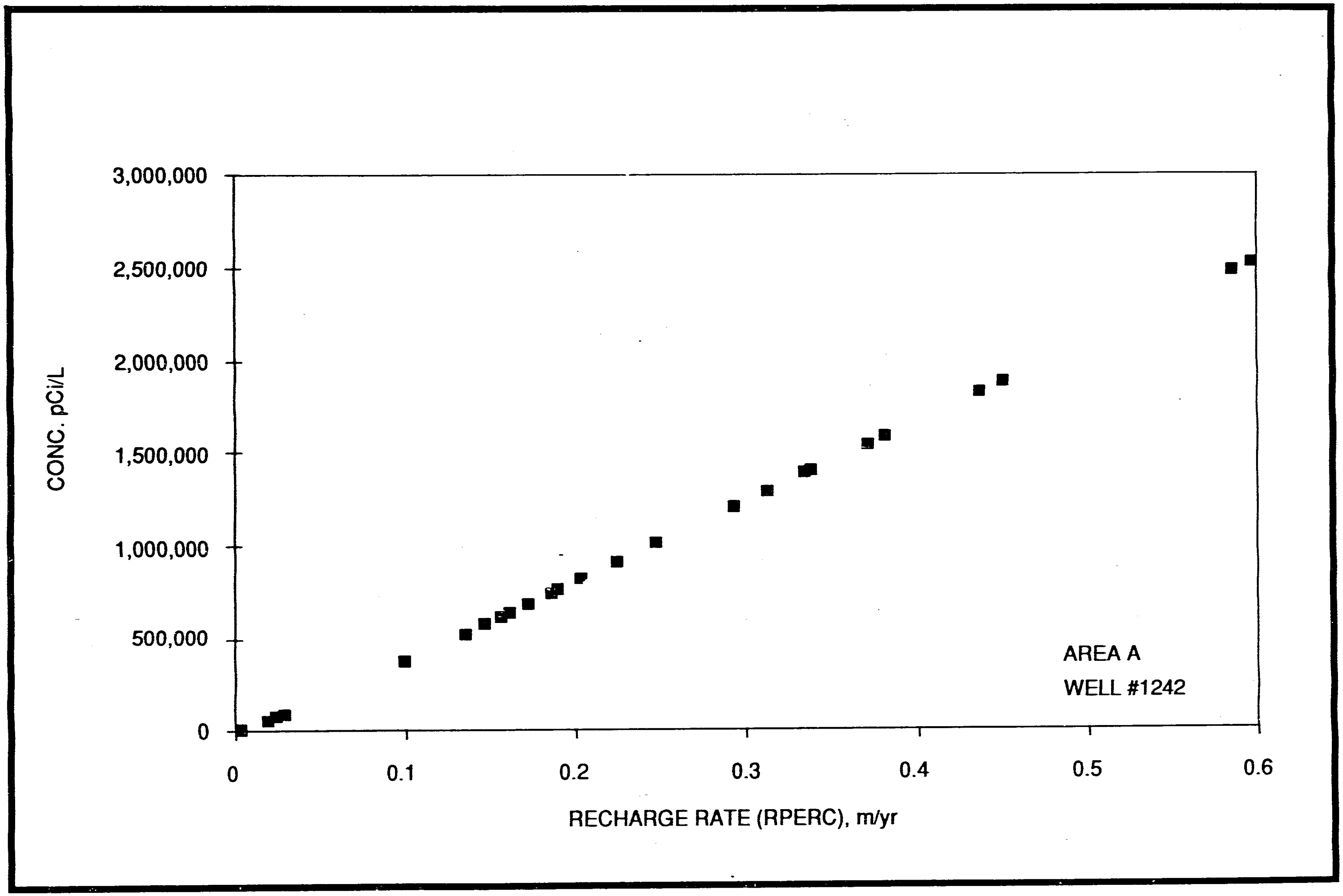


Fig. 5.77. Sensitivity analysis of WAG 6 contaminant transport model, tritium concentration — year 1.

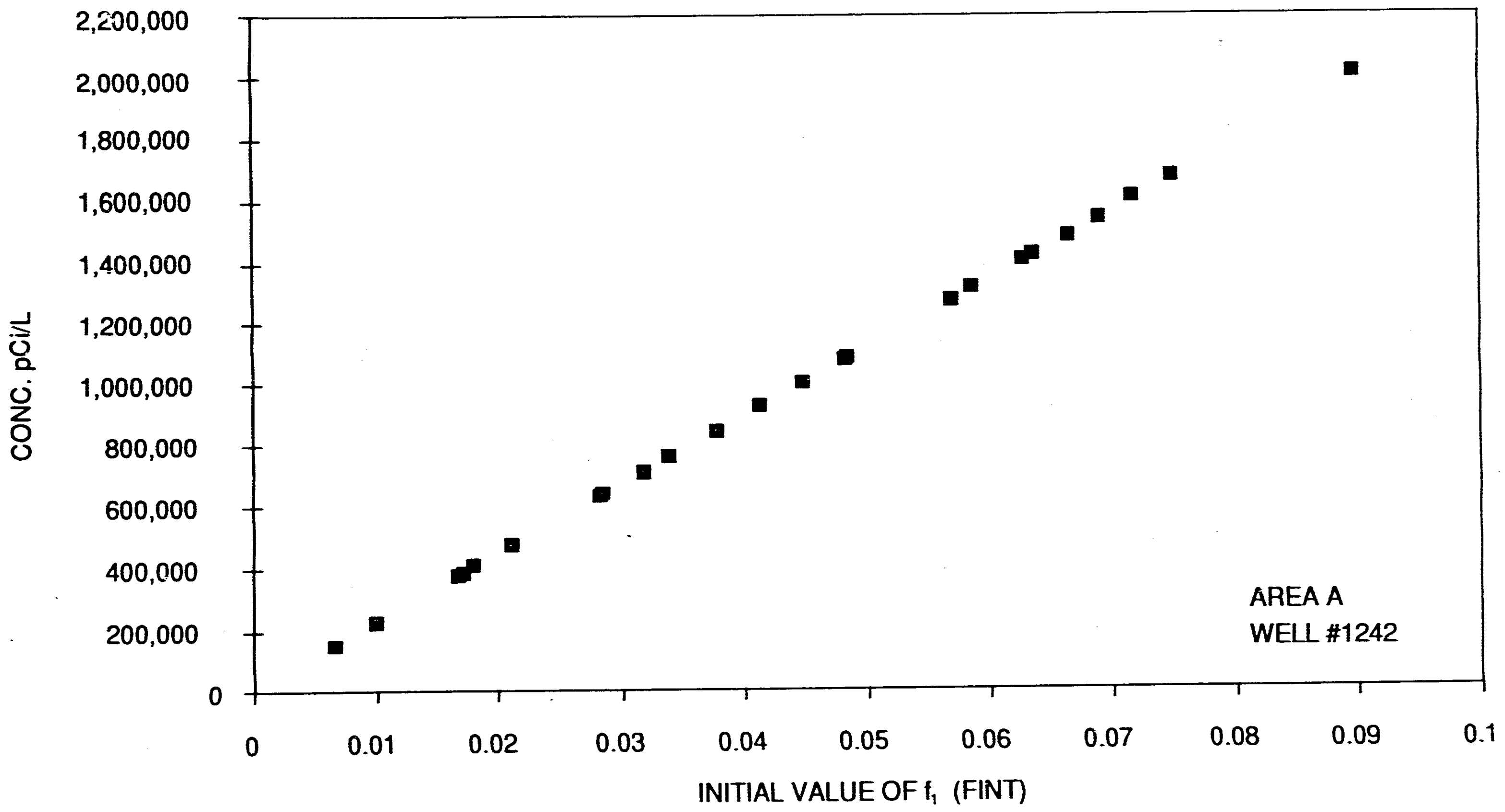


Fig. 5.78. Sensitivity analysis of WAG 6 contaminant transport model, tritium concentration — year 1.

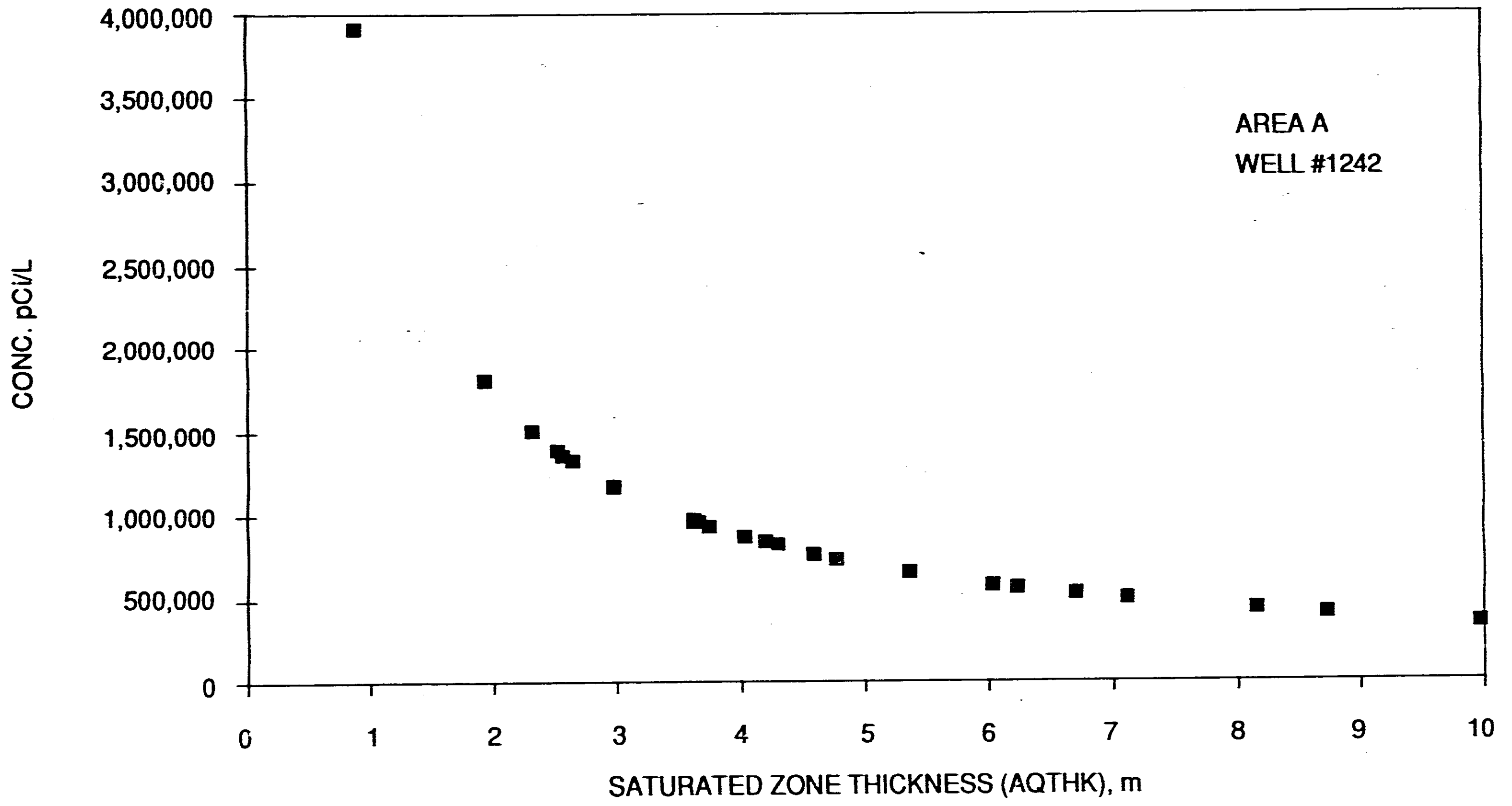


Fig. 5.79. Sensitivity analysis of WAG 6 contaminant transport model, tritium concentration — year 1.

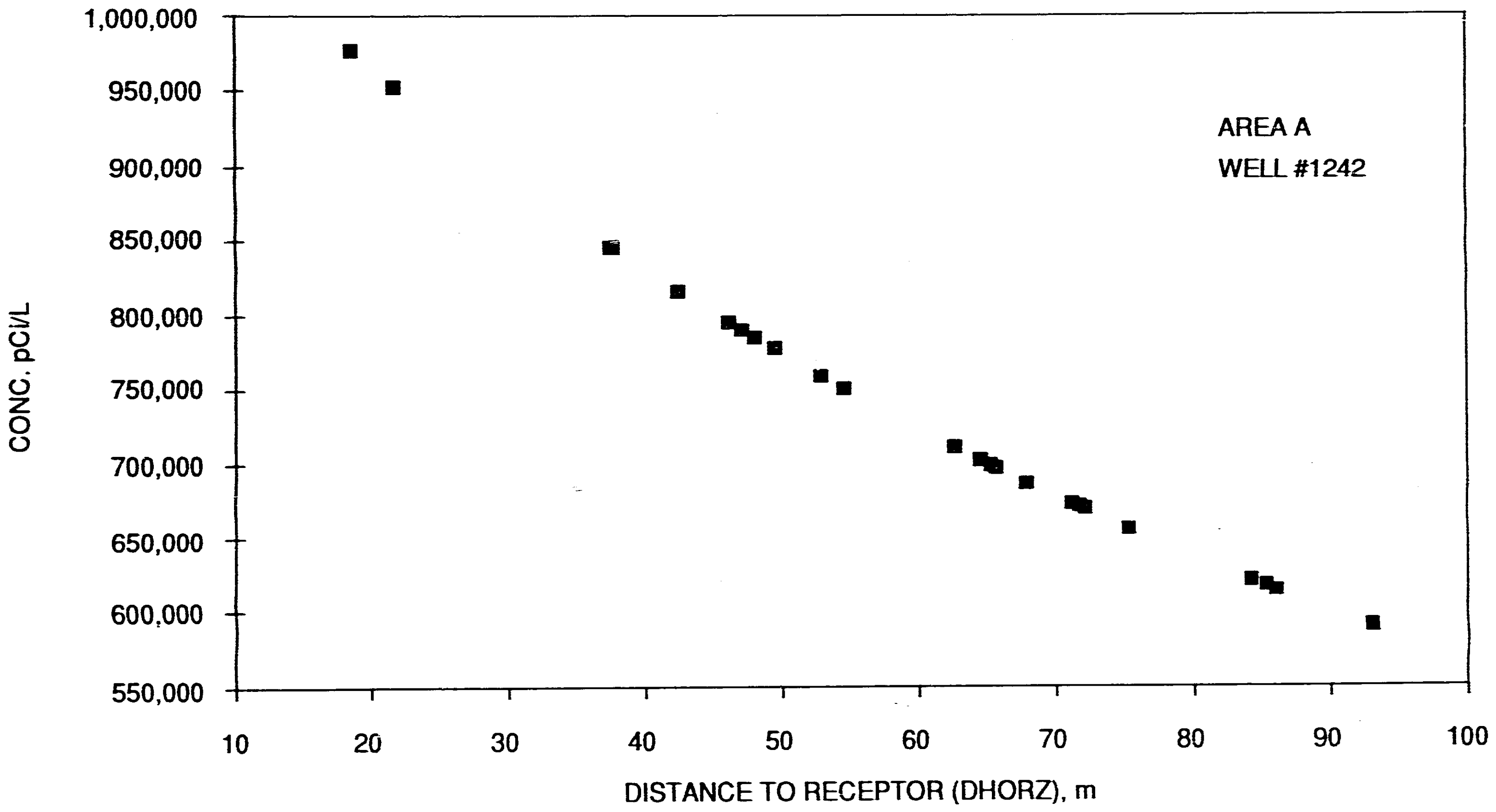


Fig. 5.80. Sensitivity analysis of WAG 6 contaminant transport model, tritium concentration — year 1.

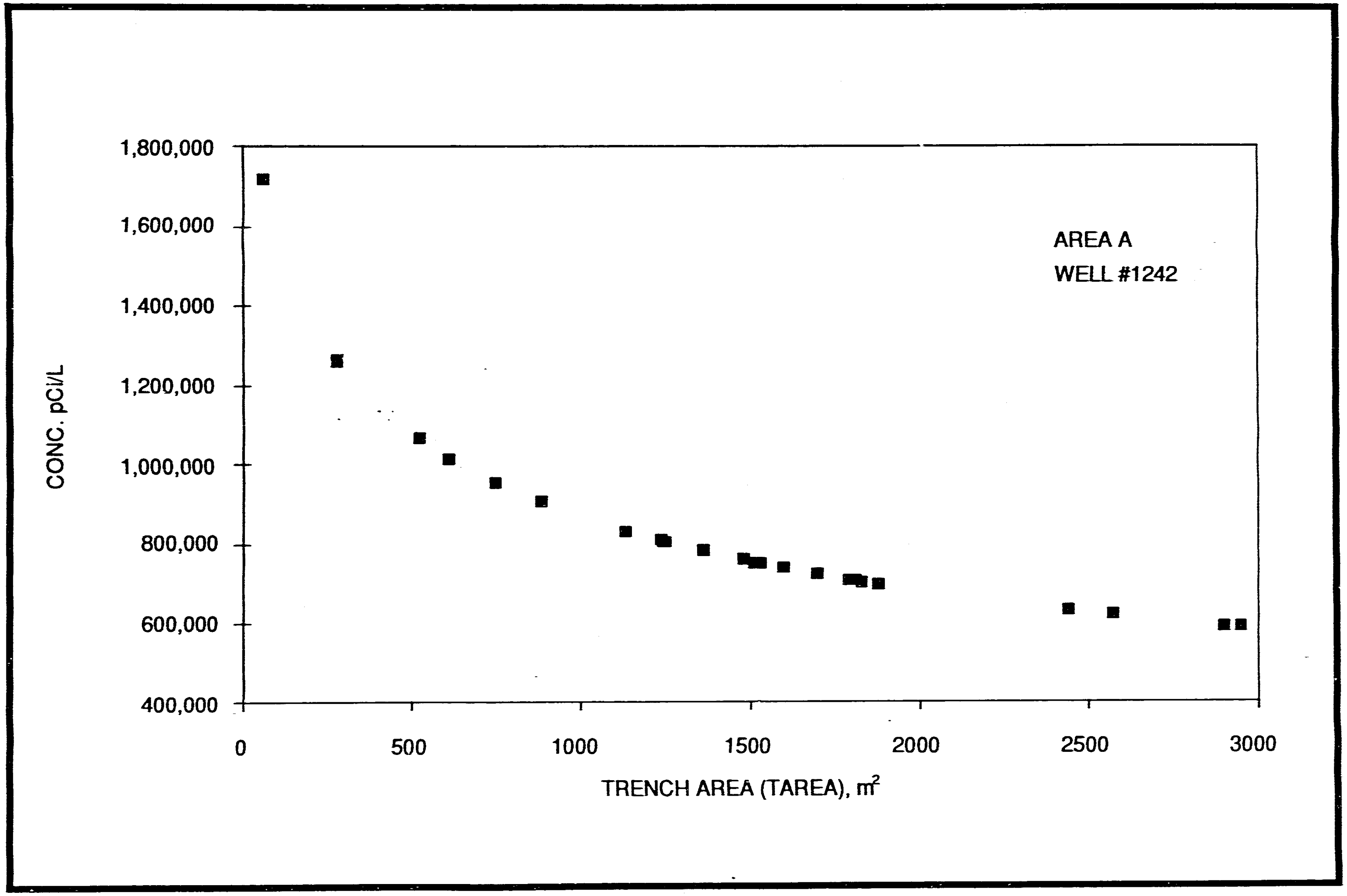


Fig. 5.81. Sensitivity analysis of WAG 6 contaminant transport model, tritium concentration — year 1.

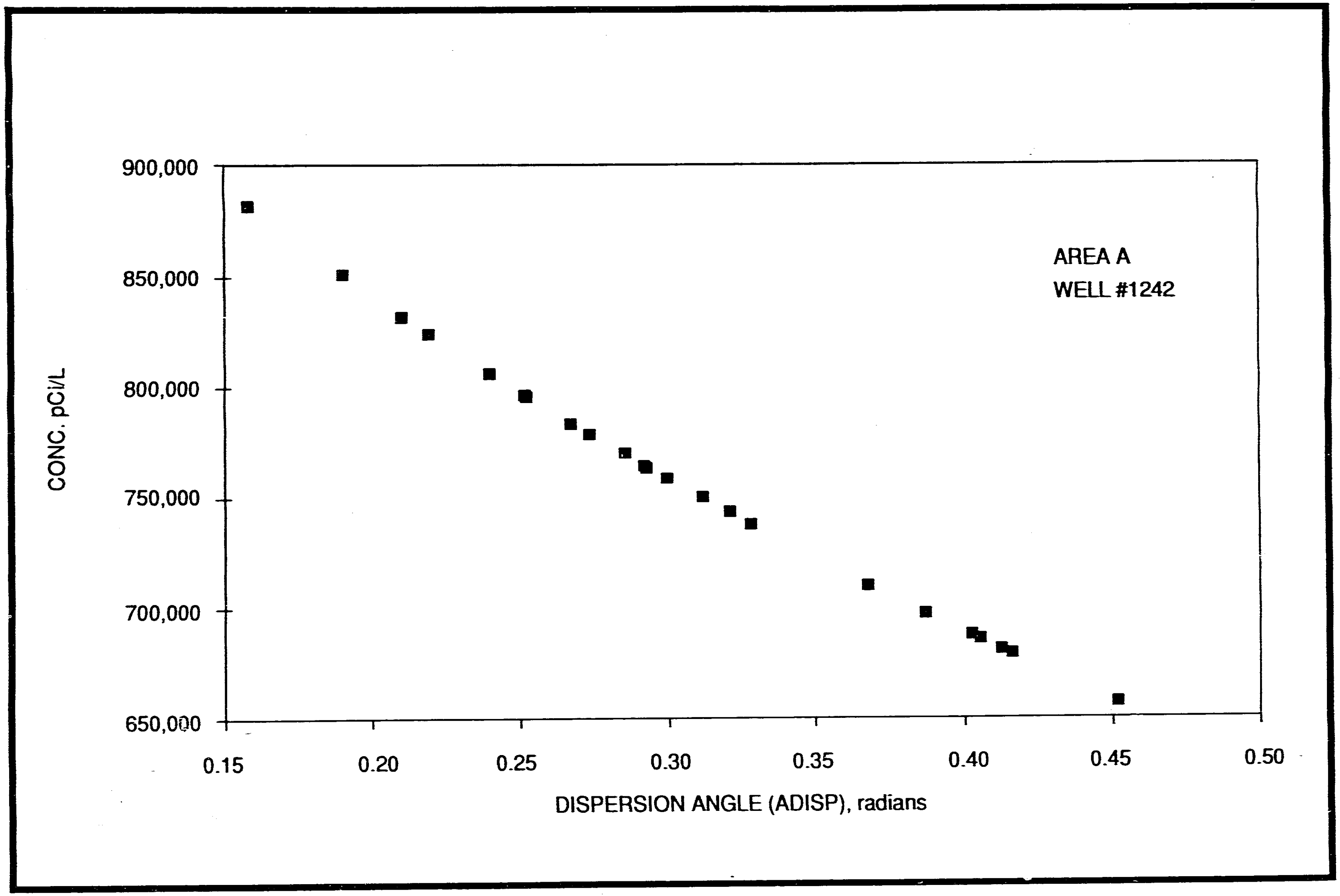


Fig. 5.82. Sensitivity analysis of WAG 6 contaminant transport model, tritium concentration — year 1.

5.18F 4104.166

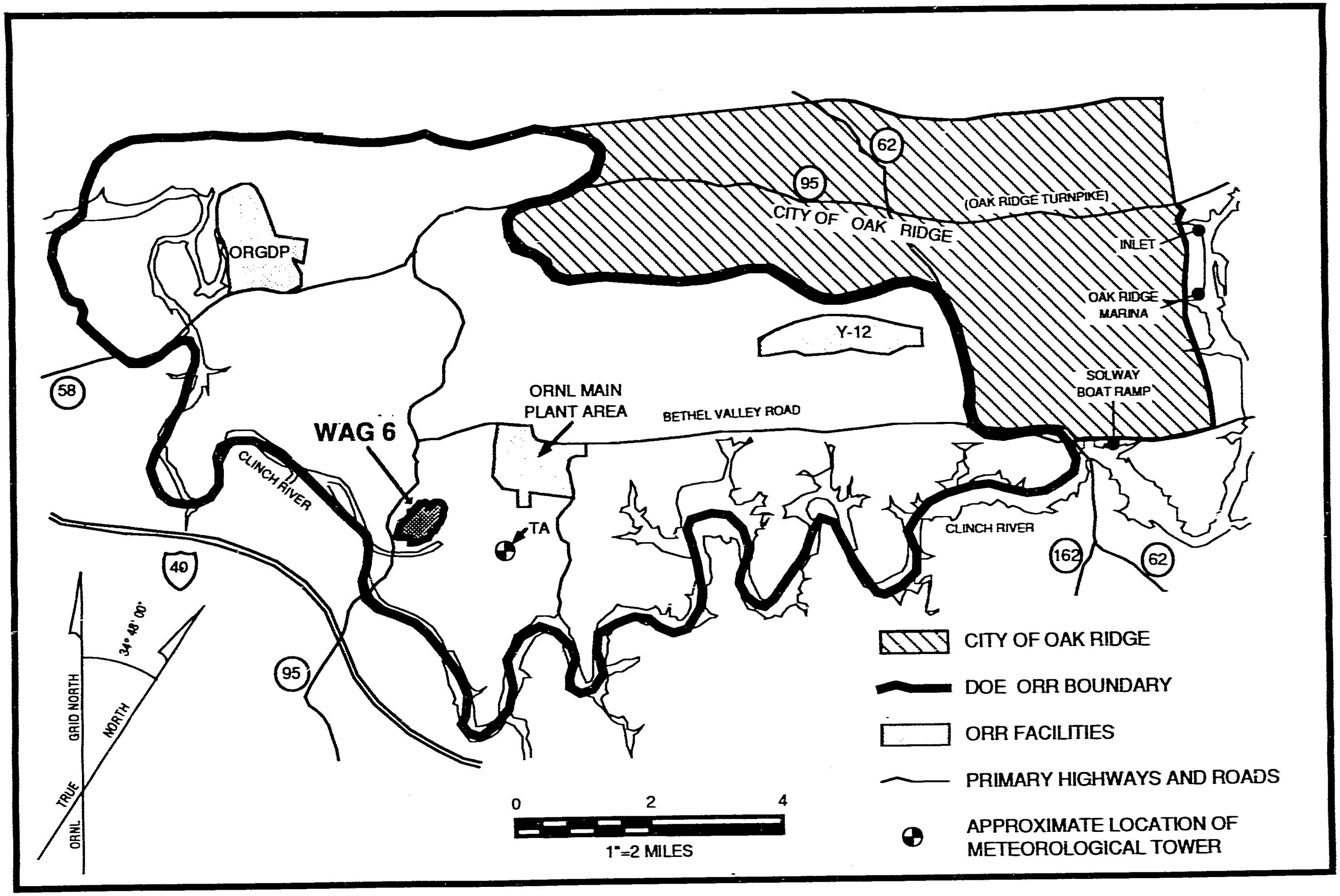
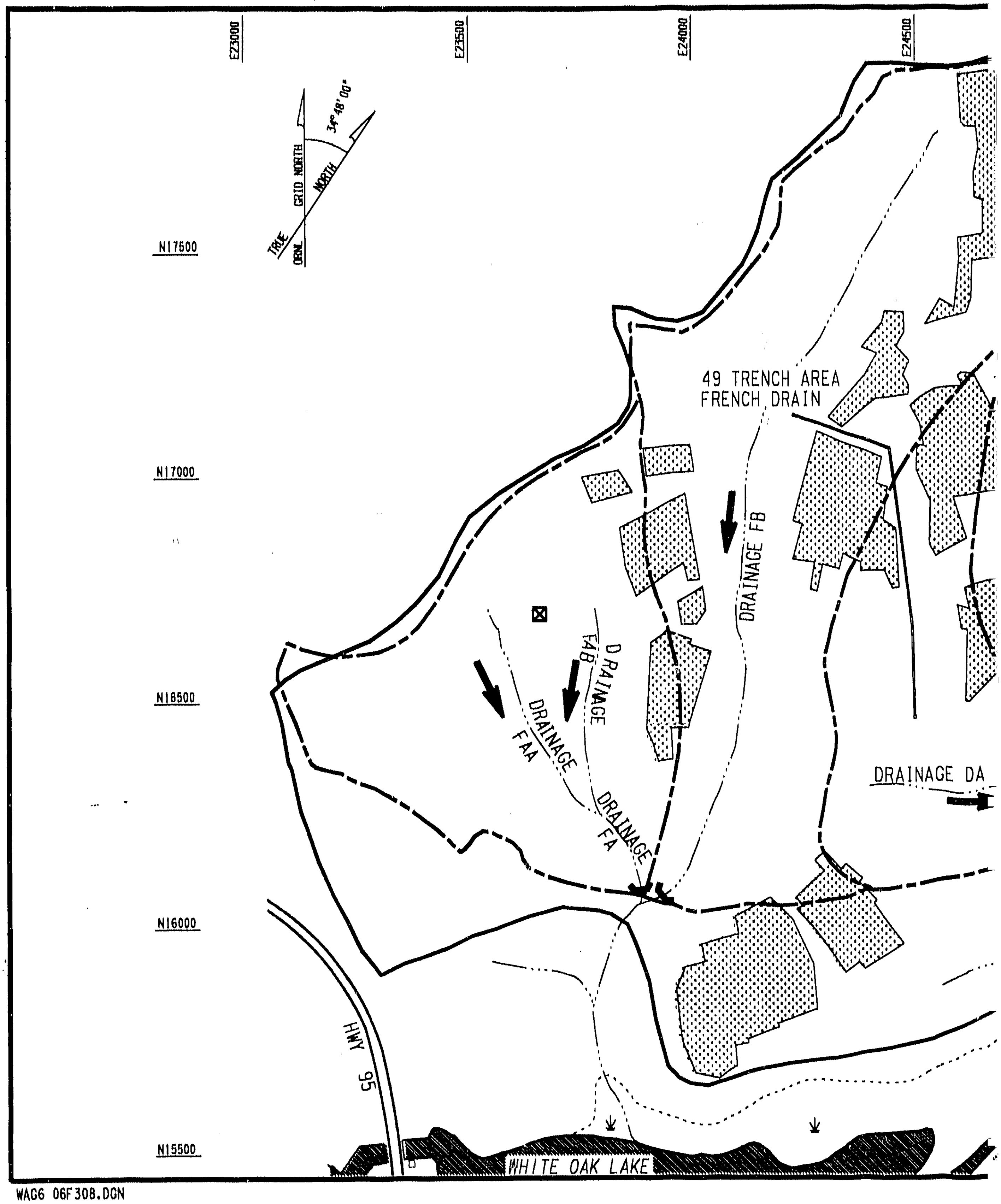


Fig. 5.83. Approximate location of meteorological tower TA.

5.18F 4104.177A



WAG6 06F308.DGN
9-4

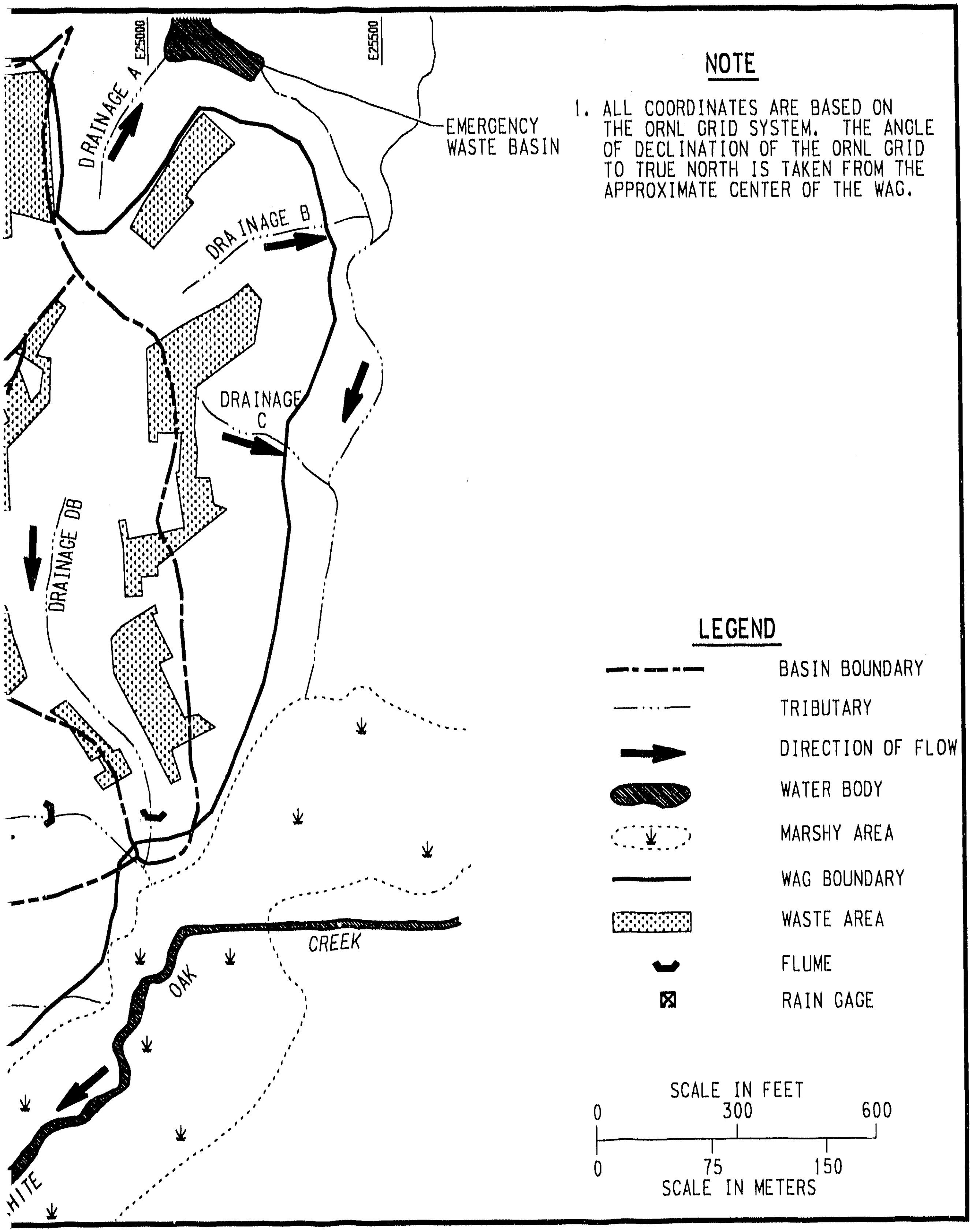
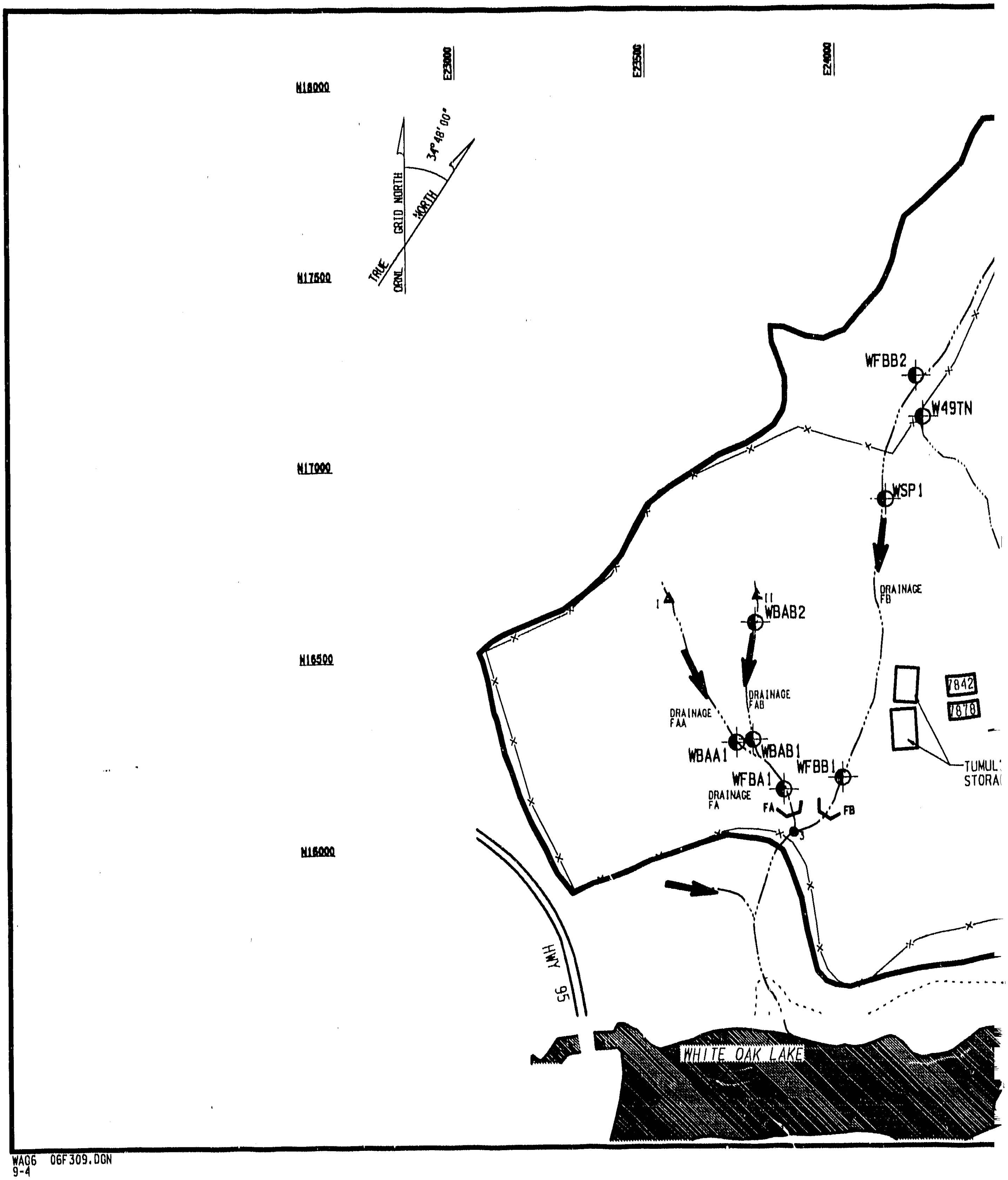


Fig. 5.84. Drainage patterns and waste areas at WAG 6.



WA06 06F309.DGN
9-4

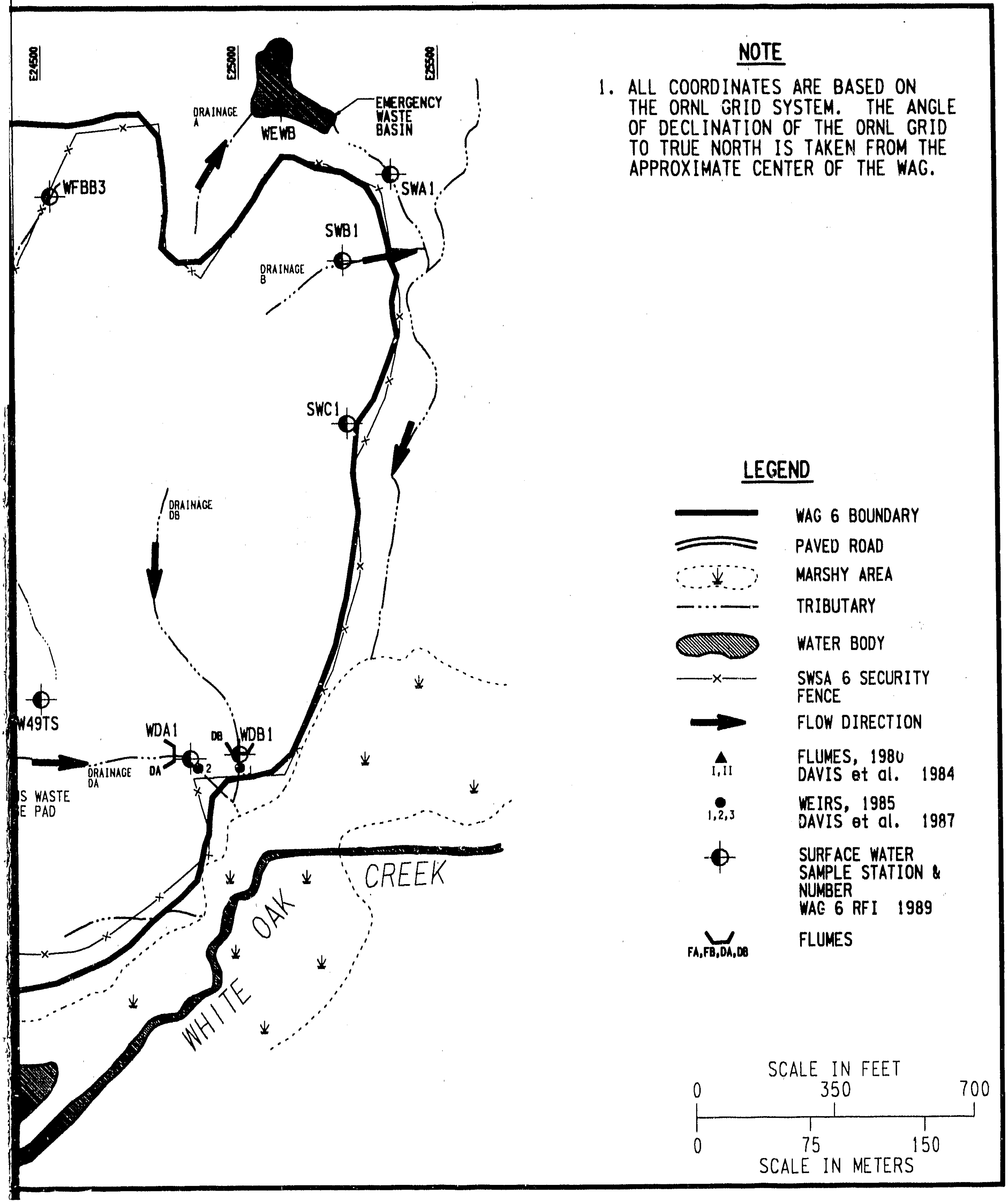
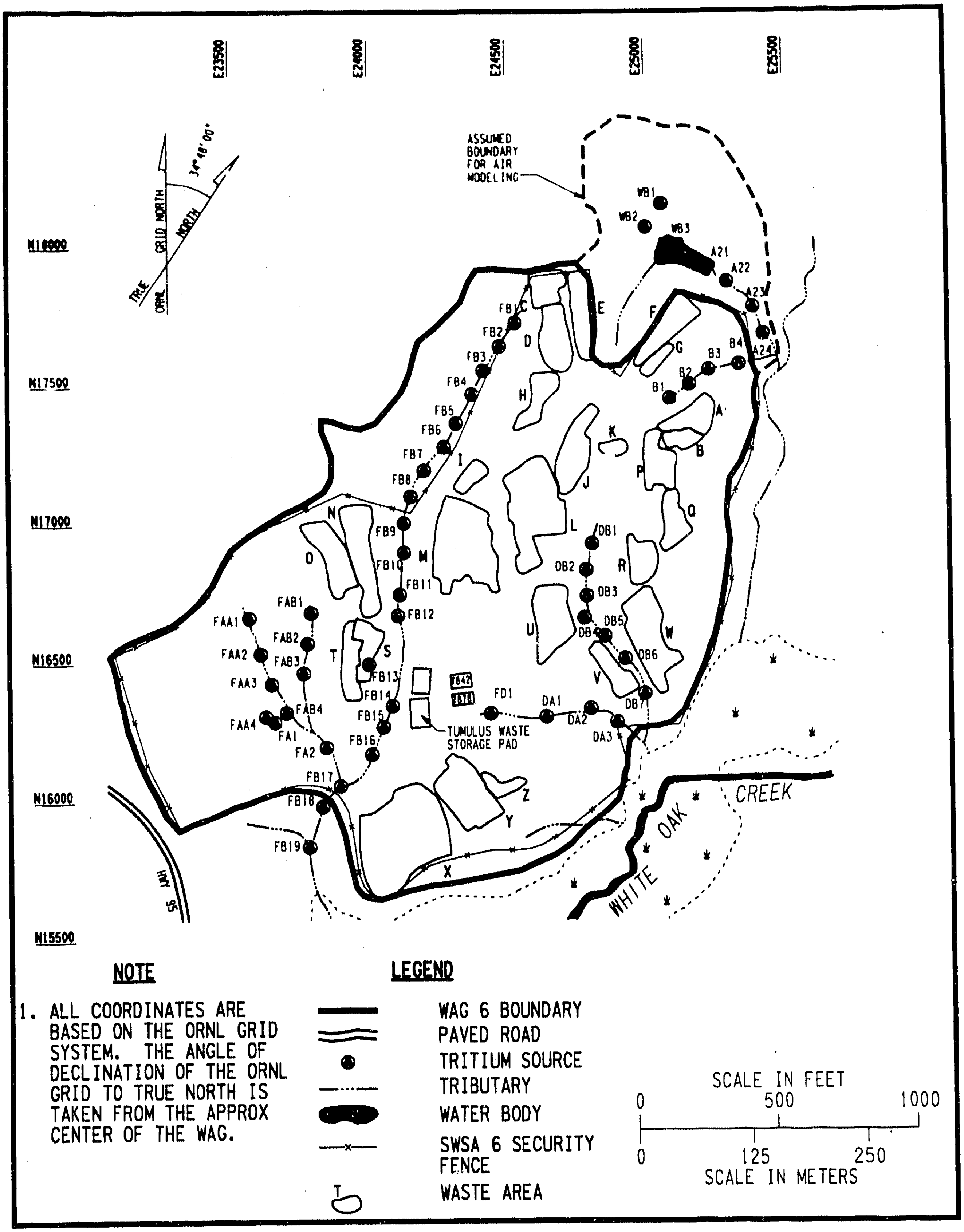


Fig. 5.85. WAG 6 surface water flow stations and sampling sites.

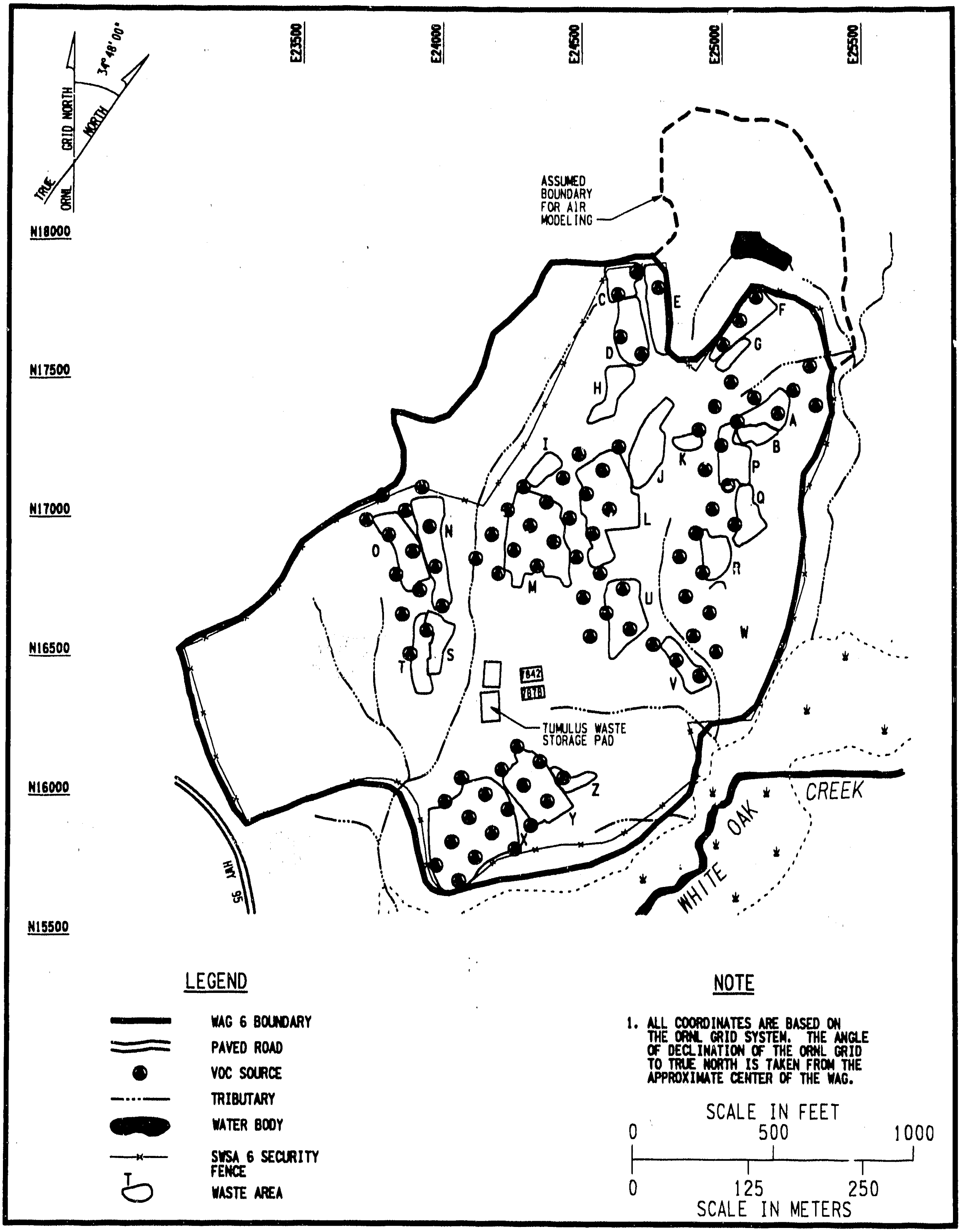


NOTE

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROX CENTER OF THE WAG.

WAG6 ORSRC.DGN
9-4

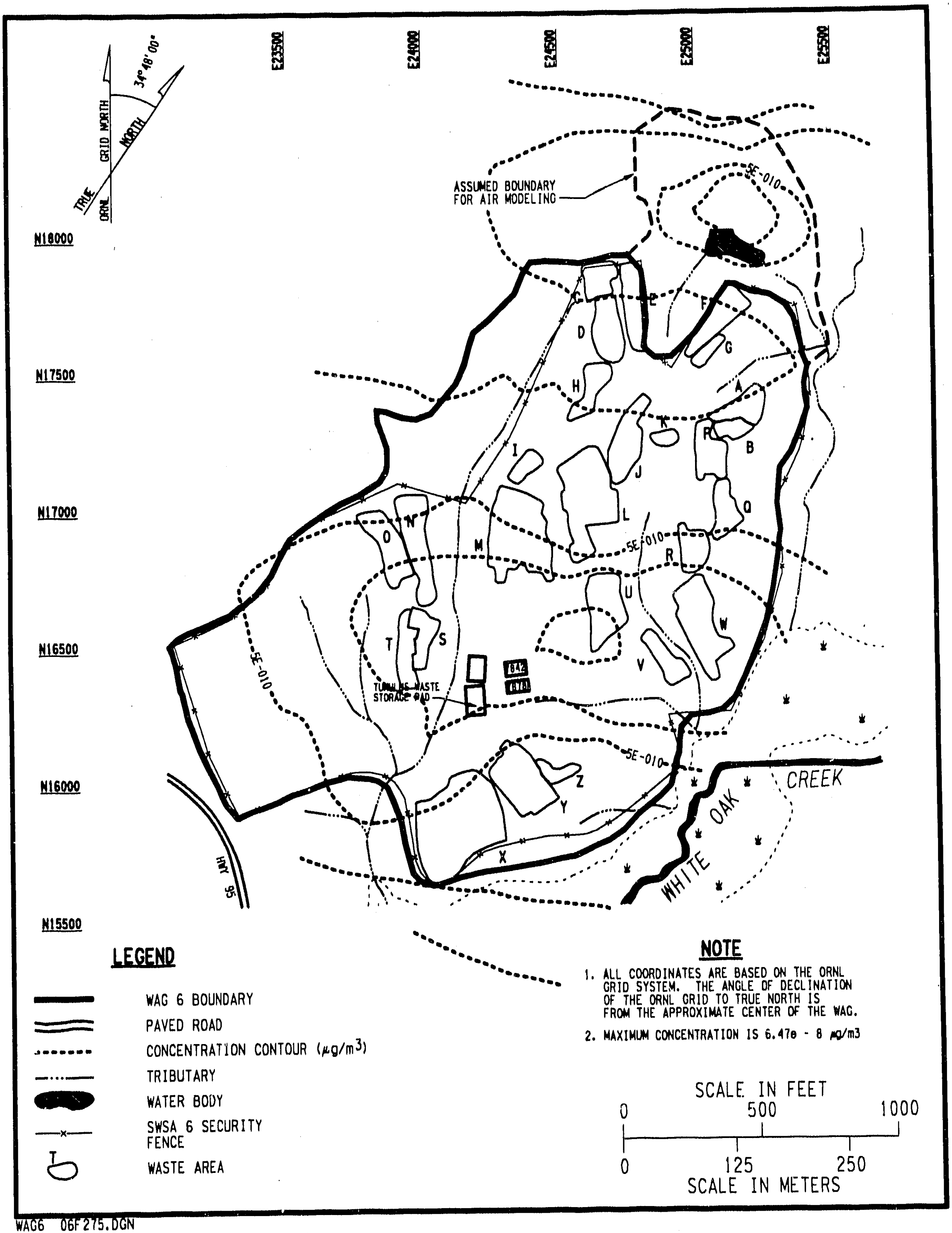
Fig. 5.86. ISCLT model source areas, 51 tritium sources.



1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.

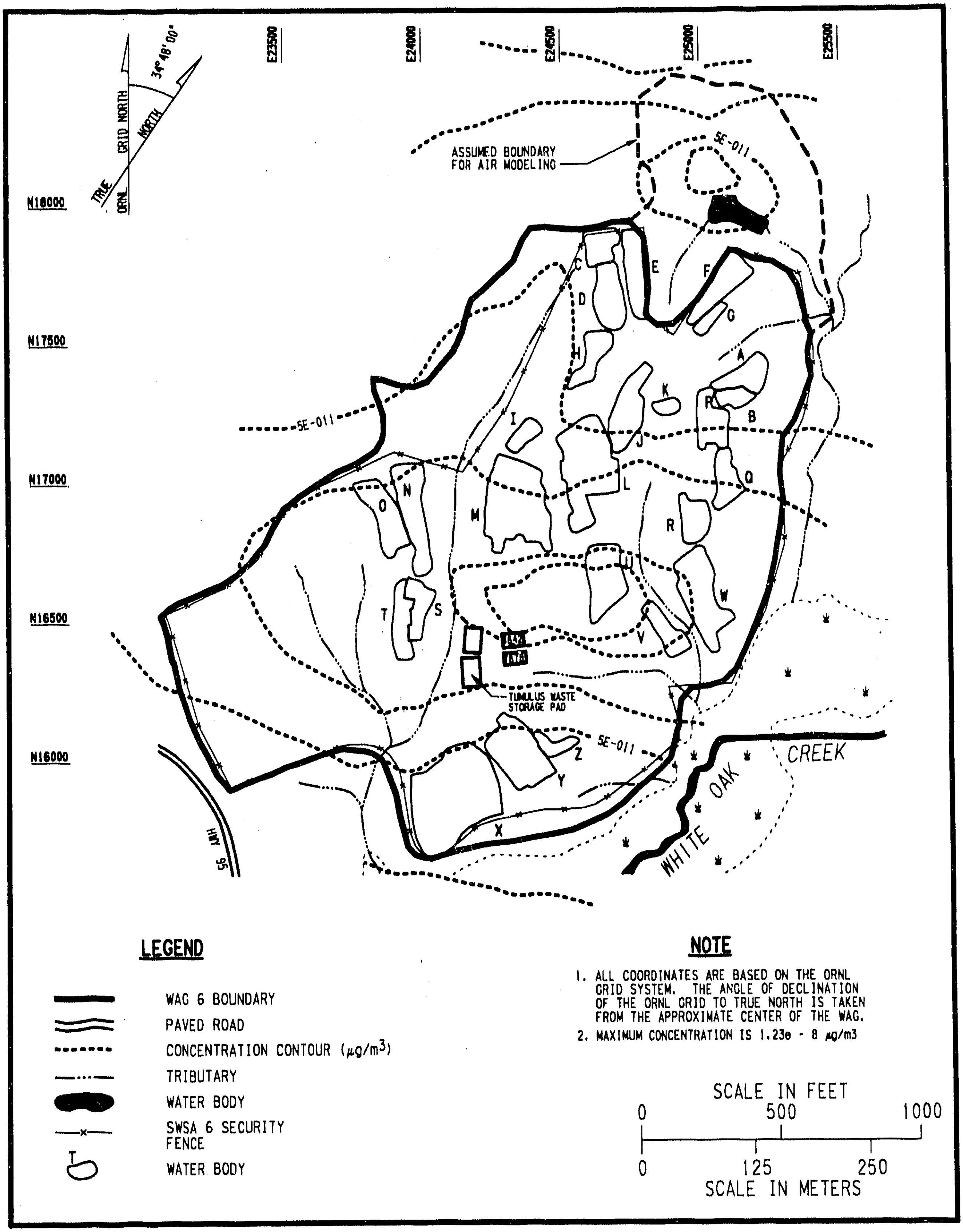
WAG6 ORSRCVOC.DGN
9-4

Fig. 5.87. ISCLT model source areas, 89 VOC sources.



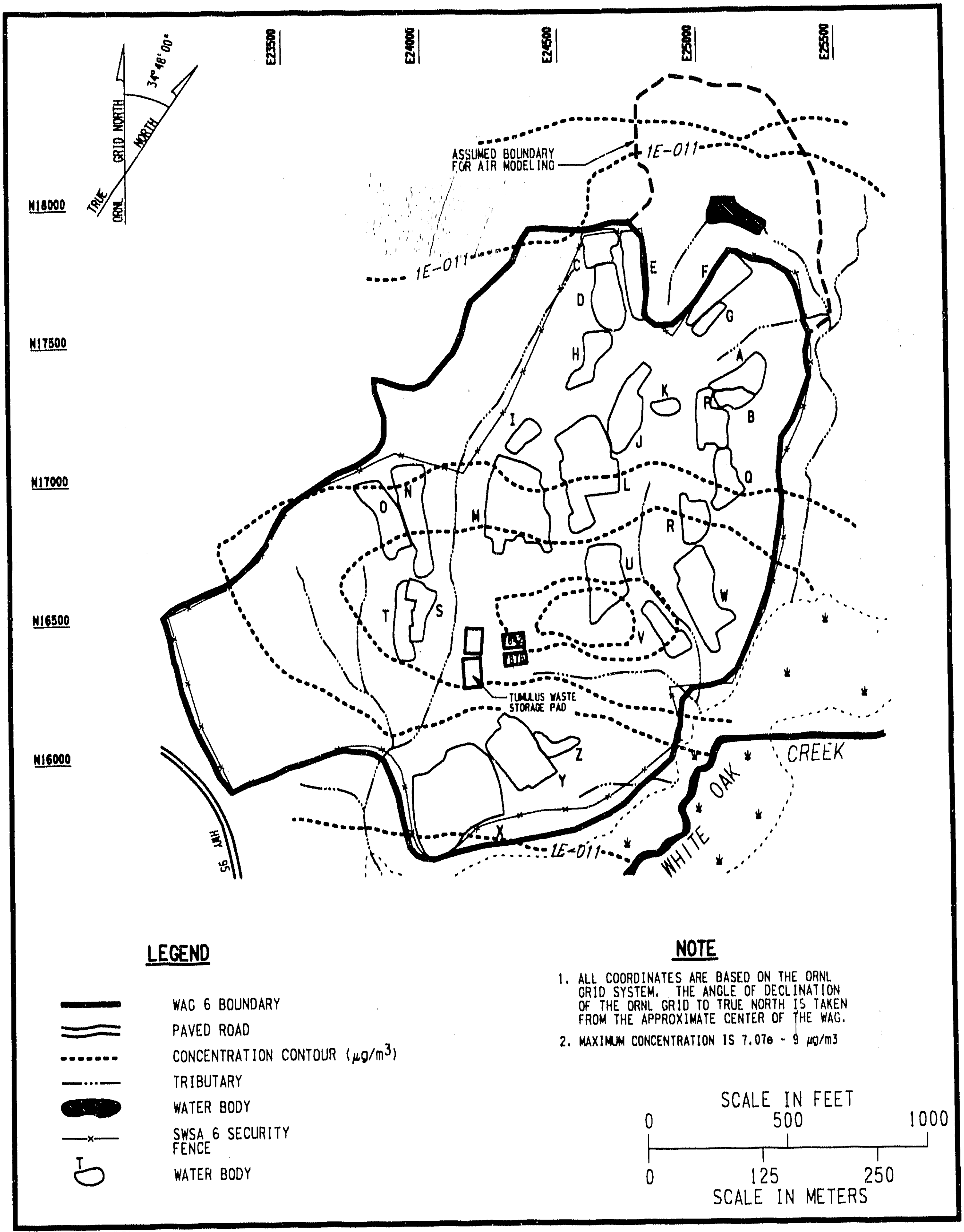
WAG6 06F275.DGN
9-5

Fig. 5.88. Tritium concentrations at present.



WAG6 06F276.DGN
9-5

Fig. 5.89. Predicted tritium concentrations, 1990-2019.

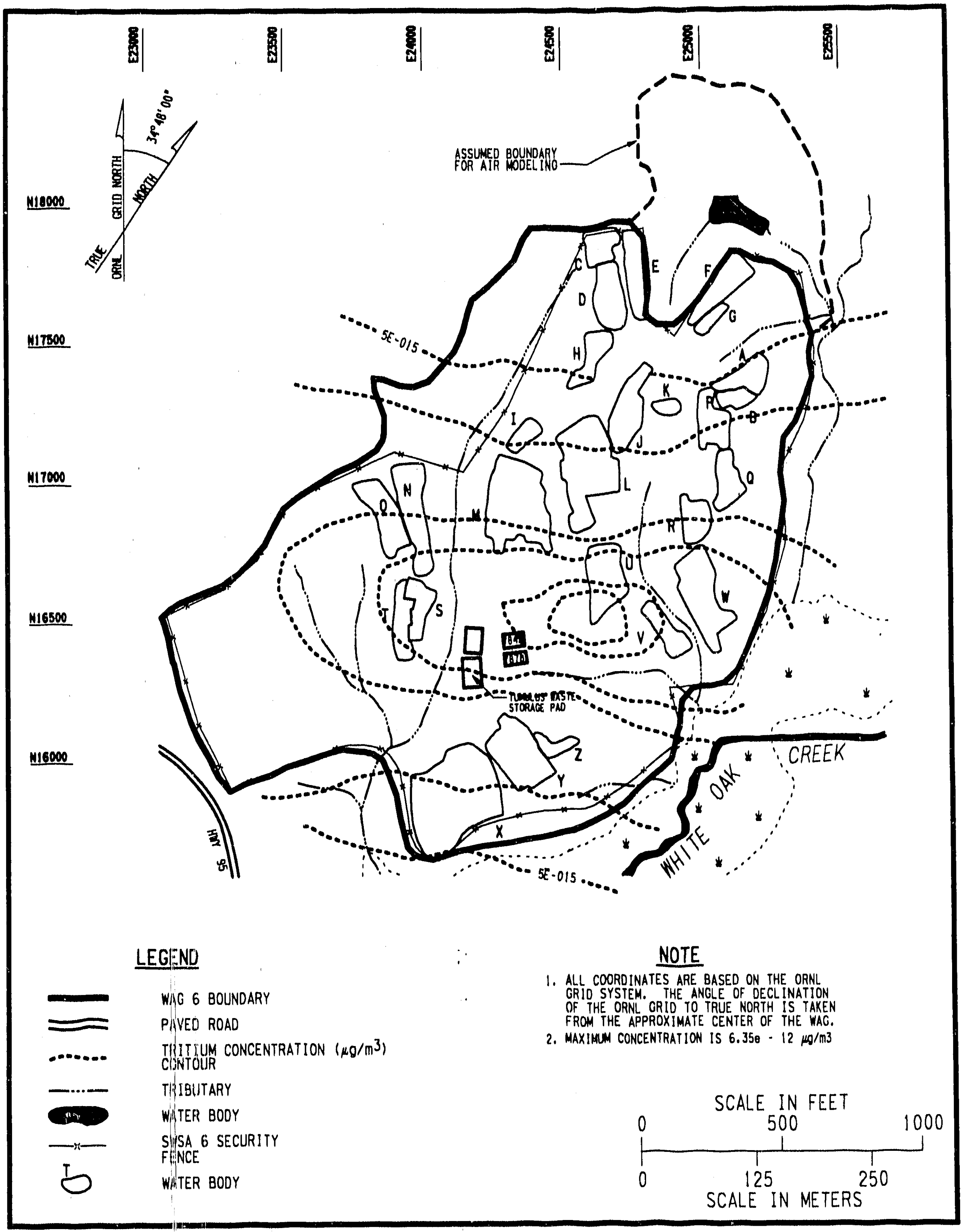


NOTE

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.
2. MAXIMUM CONCENTRATION IS 7.07e - 9 µg/m3

WAG6 06F277.DGN
9-5

Fig. 5.90. Predicted tritium concentrations, 1990-2059.

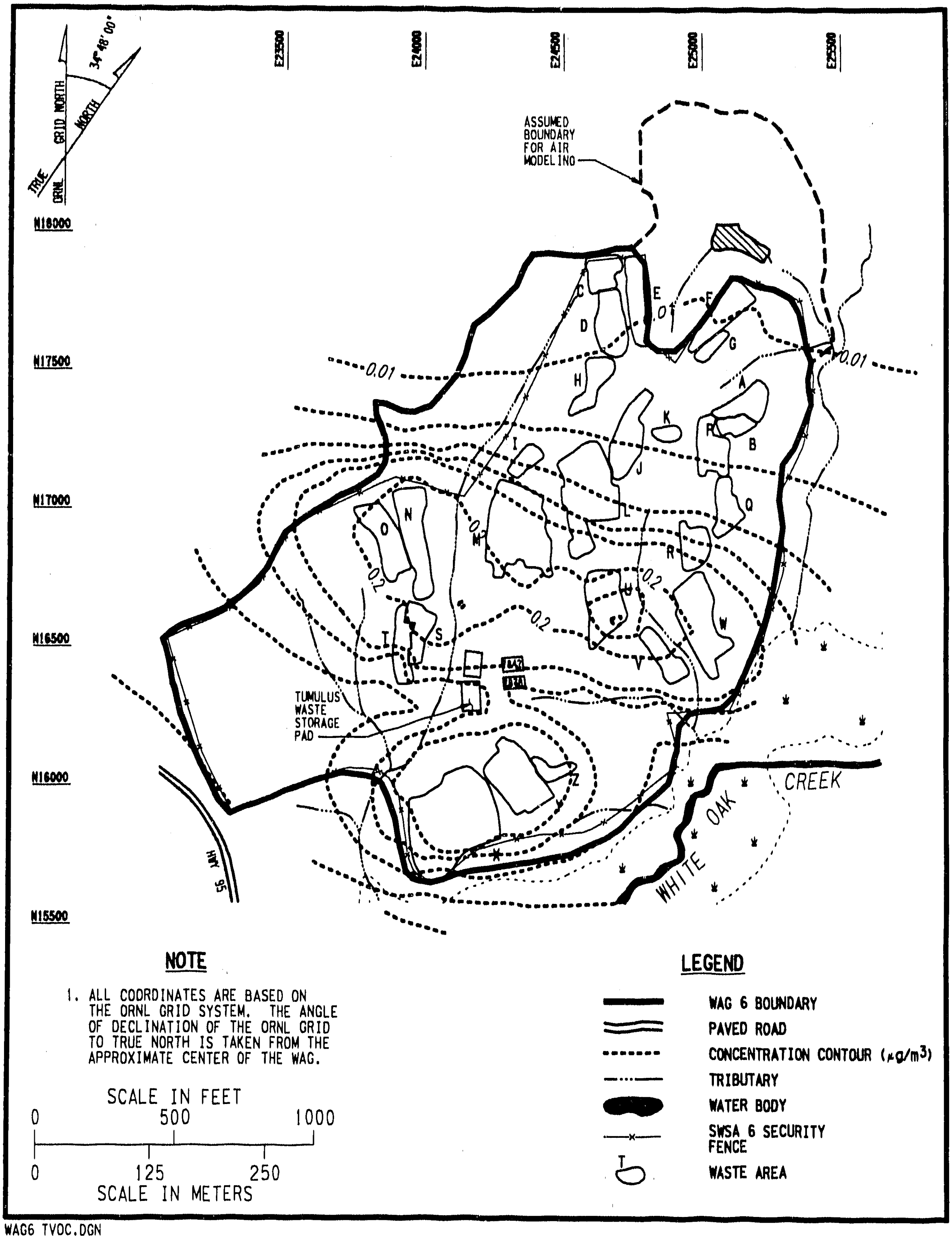


NOTE

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.
2. MAXIMUM CONCENTRATION IS 6.35e - 12 µg/m3

WAG6 06F278.DGN
9-5

Fig. 5.91. Predicted tritium concentrations, 2100-2129.



WAG6 TVOC.DGN
9-5

Fig. 5.92. Annual average base line VOC concentrations.

6. BASE LINE HUMAN HEALTH EVALUATION

6.1 INTRODUCTION

This section addresses the process and results of quantifying human health hazards associated with contamination at WAG 6 in terms of risk to the recipient. It is termed a *base line* human health evaluation because all data and analyses are not discounted in any way to reflect an improvement in conditions that might occur if remediation of the WAG were undertaken. The evaluation was developed using standard guidance documents, primarily the *Risk Assessment Guidance for Superfund, Vol. I: Human Health Evaluation Manual (Part A)* (EPA 1989b) and "Supplemental Region IV Risk Assessment Guidance" (EPA 1991a). The data on which it is based were collected during the RFI and are presented in Sects. 4 and 5 and in the Technical Memorandums published with this report.

Figure 6.1 illustrates the steps comprising the evaluation process. Section 6.2, Selecting Contaminants of Potential Concern, identifies the radionuclides and chemicals selected for study and describes the selection process. Section 6.3, Exposure Assessment, establishes a framework for quantifying human exposure to these contaminants by combining assumptions regarding conditions at the WAG (scenarios); example recipients (hypothetical receptors); concentrations of contaminants in environmental media and biota; and mechanisms by which the recipients would become exposed to the contaminants (exposure pathways). In Sect. 6.4, Toxicity Assessment, each contaminant of concern is classified as to the type of hazard it presents (e.g., carcinogenic or noncarcinogenic effects) and the toxicity factors typically used to quantify these hazards are specified. In Sect. 6.5, Risk Characterization, exposure values and toxicity parameters are combined to quantify the risk to a human resulting from each of the radionuclides and chemicals characterized. Section 6.6, Identification of Uncertainties, addresses uncertainty factors associated with each component of the human health evaluation and their potential influence on the estimates of risk presented in Sect. 6.5. Section 6.7, Summary and Conclusions, presents the results of the human health evaluation.

6.2 SELECTING CONTAMINANTS OF POTENTIAL CONCERN

This section describes the manner in which radioactive and chemical contaminants at WAG 6 were evaluated to identify those that are of potential concern to human health.

6.2.1 Radioactive Contaminants

In determining the radionuclides of concern to human health, no field sampling data were used as direct input to the evaluation. Sampling of the underground waste disposal areas in WAG 6 would have resulted in excessive worker exposure; instead, existing information on the WAG 6 waste disposal inventory was used as a source term. (Section 1 addresses WAG 6 waste disposal practices and the associated inventory information.) The inventory information identifies 82 different radionuclides and 69 daughters disposed of below ground, with a total activity (decayed to January 1991) of 571,710 curies (Table 6.1). Wastes in the aboveground disposal units contain 61 radionuclides with activity totaling 143 curies, approximately 0.025% of the below ground unit curie content. This below ground inventory includes all radionuclides present in wastes stored aboveground. Additional aboveground and below ground storage units are planned. Using the conservative assumption that wastes will be stored in these units at the same rate as in the past, their inventory was estimated to comprise 37 radionuclides with an activity of 6,600 total curies, or 1.1% of the below ground inventory. Table 6.2 summarizes the above information. The field sampling data, summarized in Sect. 4, contain radionuclides that are a subset of the below ground inventory information.

Because the inventory for the aboveground and future units totalled less than 2% of the below ground inventory, only the below ground inventory was used in the base line human health evaluation. Preliminary results indicated an unacceptable risk to selected receptors (the on-WAG homesteader) from this source alone. Moreover, the future inventory will, in fact, be significantly less than the estimate cited above since the disposal rate is projected to decrease over time. In addition, the future units will be constructed in concrete and consequently migration will be effectively eliminated for hundreds of years (NRC 1989a), barring catastrophic events.

The radionuclides in the below ground inventory were screened to identify those of potential concern. As the first step in the screening process, the relative dose from each radionuclide was estimated for three exposure routes: ingestion, inhalation, and external exposure from an infinitely thin area source. Appendix 6A details the calculation used. The potential relative dose was calculated for 1990 and 2100 using the WAG 6 radionuclide inventory decayed to these dates. This decay included the generation of radioactive decay products.

As the next step in the screening process, the radionuclides were ranked according to their potential relative doses. Those contributing the least (2.5% or less) were eliminated from further analyses. The remaining radionuclides, responsible for at least 97.5% of the potential dose, were considered to be preliminary contaminants of potential concern. These were americium-241, barium-137m, curium-244, cesium-137, cobalt-60, europium-152, europium-154, europium-155, radium-225, strontium-90, thorium-229, thorium-232, uranium-233, uranium-238, and yttrium-90.

All of the preliminary radionuclides of potential concern were selected for fate and transport modeling except barium-137m and yttrium-90, which are daughters of those selected and have a short half-life (assumed to be in secular equilibrium with the parent). These daughters were quantitatively evaluated in conjunction with the parent after the latter had been modeled for concentration at the point of ingestion, inhalation, or external exposure. Thorium-234, protactinium-234m, and protactinium-234 were daughters that were similarly evaluated in conjunction with parents even though they themselves were not among the radionuclides originally selected through the screening process.

Additionally these radionuclides were compared to those detected in groundwater and surface water samples on the WAG. This comparison led to the identification of H-3 (tritium) as an additional radionuclide of potential concern, because of the elevated concentrations detected and tritium's high mobility in the environment.

Modeling was completed for all selected radionuclides except for radium-225 and thorium-229. These radionuclides are decay products of uranium-233 and, because of their relative half lives, the activity of thorium-229 and radium-225 will increase for several thousand years. The fate and transport model could not account for their production from the decay of uranium-233, which has migrated beyond the waste disposal areas. In addition, the model could not be adequately calibrated for them, because uranium-233, thorium-229, and radium-225 were not detected in groundwater. Both daughters are not believed to be significant contributors to the risk to any of the current receptors. However, their importance to future receptors cannot be determined, because—while the activity of the currently important radionuclides will decrease—thorium-229 and radium-225 activity will continue to increase for several thousand years.

The 17 radionuclides of potential concern evaluated in the base line human health evaluation are listed in Table 6.3.

6.2.2 Chemical Contaminants

Because sampling of the underground waste disposal area was not possible and no chemical inventory information was available, only data from field sampling around waste disposal areas were used to identify chemicals of potential concern to human health (Sect. 4.1.2). Groundwater samples used were those from all sampling events at all of the RCRA compliance wells and RFI wells on WAG 6, except the deep wells (well numbers 1234, 1236, 1238, and 1239). Samples from the latter were not used because, as described in Sect. 3, active groundwater flow is limited to depths up to 65 ft in bedrock; a preponderance of evidence (e.g., aquifer geochemistry, age-dating, and actual sampling results from the deep wells) suggests that the deeper groundwater is not being affected by operational activities associated with waste disposal at the WAG.

Surface water samples used were those from the baseflow sampling events of February 1989, April 1989, and September 1989. Soil samples used were from the 0-6 ft depth interval or some smaller portion thereof. Soils from this interval were assumed to be best suited for evaluating exposures to contaminants that might result from direct contact with, or ingestion of, surface soil. Sediment samples used were taken from the EWB, where it was assumed that sediment becomes suspended in the surface water when disturbed by swimmers.

Seventy-three chemicals were detected in the above samples. These comprised 45 organics and 28 inorganics, each detected at least once. Many inorganics are naturally occurring chemicals and were detected frequently at WAG 6. The organics, however, are generally man-made and were detected less frequently, many only once.

All data used underwent review, as described in Appendix 4A. Samples were analyzed for TCL and TAL, or RCRA Appendix IX, constituents identified in Sect. 2. In accordance with standard guidance documents (EPA 1988c, 1989b, 1990a), TICs were also evaluated as part of the sample analyses (Appendix 6B).

All detected volatile organic chemicals except 2-hexanone were assumed to be of potential concern as a conservative measure to discount volatilization. Two-hexanone was not considered to be of potential concern because it was detected only once across all media and then at a concentration of 1 ppb. Semivolatile chemicals detected on-WAG in each of the four environmental media were statistically evaluated to determine which exhibited elevated concentrations with respect to reference concentrations. The statistically significant chemicals were identified as being of potential concern; the remainder underwent further analyses. Since these chemicals are generally man-made, they were included as being of potential concern if any of the following held true:

- A field sampling concentration was detected in excess of the CRDL
- The chemical was statistically significant in other media
- The chemical was a known WAG 6 contaminant (Teasley 1990)
- A field sampling concentration was detected in excess of a health-based criterion (identified in Sect. 4.1.2)

The results of evaluation against these four criteria are shown in Table 6.4.

Several inorganic chemicals were detected in field samples at concentrations statistically significantly different from reference concentrations. Of these, iron, magnesium, calcium, potassium, and sodium are trace elements essential in human nutrition; relatively high concentrations are required to produce toxic effects. These elements occur naturally in widely varying concentrations. Table 6.5 presents a comparison between the recommended dietary allowance of these nutrients and intakes calculated from concentrations detected in groundwater and soil on WAG 6. In each case the intake was far below the recommended dietary allowance and therefore these elements were not considered chemicals of potential concern. Inorganics that were detected in groundwater at twice reference concentrations were, however, classified as being of potential concern.

Table 6.6 lists the 41 chemicals of potential concern identified from the above evaluations. The list comprises 29 organics and 12 inorganics, most of which occurred only in groundwater.

6.3 Exposure Assessment

An individual's exposure (i.e., potential for intake of contaminants) is a function of the environmental conditions under which that person operates, the activities that person pursues, and the length of time that person conducts these activities within the environment. Consequently, to arrive at estimated intakes attributable to contaminants at WAG 6 assumptions (or scenarios) were established regarding the operation of the site, hypothetica receptors pertinent to each scenario were identified, and periods of evaluation were specifie (Table 6.7). For each receptor, exposure pathways (e.g., ingestion of contaminated water, inhalation of particulates) were selected; contaminant concentration values in environmental media (e.g., air, water) were derived from field data and modeled data and evaluated in the context of these pathways. Intakes are not presented in this section as end products in themselves, but are assessed in combination with toxicity values derived in Sect. 6.4 to characterize the overall health risk to each receptor in Sect. 6.5.

6.3.1 Exposure Scenarios

Two hypothetical scenarios were assumed for WAG 6. The first, termed the no-action scenario, assumed that DOE's current access restrictions for the site became ineffective immediately. Site fencing, warning signs, patrols, and institutional controls were assumed to disappear. Although this is not a reasonable scenario for the WAG, evaluation thereof is required under Section 300.430 (e)(6) of the National Contingency Plan (NCP). The second scenario assumed that DOE would continue to use WAG 6 as a disposal site for LLW for the next 10 years and that this operational period would be followed by a 100-year institutional control period. This scenario corresponds to DOE's operational and institutional control plans for WAG 6 for the next 110 years; it does not assume, however, that DOE will perform site closure or remediation other than maintenance of the existing soil cover and vegetation over the disposal units.

6.3.2 Hypothetical Receptors

Given the two scenarios described above, hypothetical receptors were selected. For the no-action scenario that assumed free access to the waste disposal areas, only a homesteader who lived in a house built directly on top of the burial grounds was assessed for current exposure since this would represent the greatest risk value for that scenario. (*TM 06-14 Demographic Data to Support WAG 6 RFI* showed that no residents live closer than 0.56 miles from the WAG boundary and that these residents live on the opposite bank of the Clinch River. Consequently, actual exposure estimates for these residents were not used since they would not be representative of exposure to a resident on the WAG.)

For the institutional control scenario, one hypothetical on-WAG receptor and three hypothetical off-WAG receptors were assessed for current exposure, the latter to represent exposure to the public in general. Since site controls would eliminate the feasibility of on-WAG homesteading, an on-WAG homesteader was assessed only for the future, post-control phase of this scenario (i.e., beginning in 2100). An ORNL employee conducting maintenance activities at the WAG was assessed because the WAG is still an operating disposal site. An individual hunting deer nearby was evaluated because hunting occurs at specified times of the year in designated areas within the ORR Wildlife Management Area south and west of the WAG. A person stationed at the fence that surrounds most of the WAG was assessed because certain sections of the fence are accessible to the public. Finally, a homesteader resident downstream along the Clinch River was examined because contamination could potentially be migrating from the WAG. These four individuals were not assessed for future exposure because the future exposure of the on-WAG homesteader would represent the greatest risk value for that phase of the institutional control scenario.

6.3.3 Exposure Pathways

Associated with each hypothetical receptor were a variety of potential pathways by which that individual could potentially be exposed to contaminants from WAG 6. An exposure pathway consists of four elements: a source; a transport medium by which contaminants move from the source to a receptor; a point of potential contact with the contaminated medium (e.g, on top of a waste disposal area); and an exposure route (e.g., ingestion) whereby contaminants may affect the receptor (Fig. 6.2). Potential pathways for the various receptors considered in this evaluation are listed in Table 6.8. Table 6.9 illustrates the significant receptor/pathway combinations selected for assessment and the periods for which they were evaluated. Assumptions made about each receptor and the pathways evaluated are presented in Table 6.10.

6.3.4 Quantification of Exposure

The final step in assessing radiological and chemical exposures to a hypothetical receptor was to determine the intake to the receptor from each pathway associated with that individual. This quantification of exposure was a two-stage process: first, the contaminant concentrations to which the receptor would be exposed in different media were estimated. Second, these concentrations were combined with other variables, such as exposure frequency and duration, to yield pathway-specific intakes.

When quantifying exposure for a given pathway variables used are selected so that the combination of all variables result in an estimate of the reasonable maximum exposure for that pathway (EPA 1989b). The goal of the reasonable maximum exposure is to combine upper-bound and average exposure factors so that the result represents an estimate of exposure that is both protective and reasonable, not the worst possible case (EPA 1991a). Thus not all variables used in this evaluation (e.g., body weight, contaminant concentrations) were at their individual maximums, but when used in combination with other variables resulted in reasonable estimates of maximum exposure.

6.3.4.1 Estimation of contaminant concentrations in media

To provide a reasonable estimate of maximum exposure, potential contact concentrations for each contaminant were calculated at the upper 95 percentile confidence interval. For radionuclides, 26 individual subareas within the WAG were studied. For chemicals, two study areas were defined: the entire WAG and the high-activity auger hole area in which 85 percent of the radioactivity at the WAG was disposed of. The results for radionuclides at the upper 95 percentile confidence interval around the mean are presented in Sect. 5.3.2.3. For chemicals, potential concentrations to which receptors could be exposed (contact concentrations) were calculated at the upper 95 percentile confidence interval on the median concentration for both the WAG-wide and auger hole study areas. The median (a nonparametric estimate of central tendency) was selected based on the non-normal distribution of the parameters detected. The 95 percentile upper confidence limit on the median was the concentration associated with the rank approximated in statistical tables (for sample sizes less than 20) or with the rank calculated (for sample sizes greater than 20) as described in Sect. 4.1.2. Appendix 6C lists the upper 95 percentile confidence limit on the median concentration of all chemicals of potential concern listed in Table 6.6.

Derivation of radionuclide concentrations. Figure 6.3 illustrates the process used to estimate concentrations of radionuclides for various routes of exposure. Concentrations in environmental media were estimated using fate and transport modeling.

The radionuclides modeled were those identified as contaminants of concern in Sect. 6.2.1. The two models used were a surface/subsurface water model (the water model) and an air model, both of which were discussed in Sect. 5 and associated appendixes. Groundwater and surface water contaminant concentrations from the water model were calibrated by comparing them with field sampling data from groundwater wells and surface water sources. Contaminant concentrations in soil were determined from the water model by accounting for particle adsorption to soil and subsequent decay of radionuclides. From these concentrations in soil, the concentrations of airborne radionuclide particulates were calculated. The air model used tritium concentrations in surface water from the water model as source data to calculate the maximum annual average concentration of tritium in air. Appendix 6D details the calculation of concentrations of radionuclides used in assessing exposures to hypothetical receptors.

Radionuclide concentrations in vegetables and fruits, beef, deer meat, milk, and fish were predicted from contaminant concentrations in surface water, groundwater, and soil, using a food chain transport model (Appendix 6D).

Off-WAG contaminant concentrations in the Clinch River resulting from migration of WAG 6 contaminants were computed by:

- Determining WAG 6 boundary fluxes of radionuclides
- Determining radionuclide concentrations in the White Oak Creek discharge (over White Oak Dam)
- Accounting for dilution of these concentrations in the Clinch River

Boundary fluxes of radionuclides were obtained from water modeling results presented in Sect. 5. Concentrations of radionuclides in the White Oak Creek discharge over the White Oak Dam were then computed by dividing the respective fluxes by the volume flow rate of the discharge. Finally, concentrations in the Clinch River were computed by multiplying the White Oak Dam discharge concentrations by a dilution factor. The dilution factor was derived from an advective-dispersion modeling in the Clinch River for 'dry' climatic conditions (ORNL 1991b). The concentrations of radionuclides in surface water at Clinch River Mile 9.5 are given in Appendix 6D.

The radionuclides quantified for the off-WAG homesteader were tritium, cobalt-60, strontium-90, cesium-137, europium-152, europium-154, thorium-232, uranium-233, and uranium-238, all of which were identified as being of potential concern. Two other radionuclides of potential concern, americium-241 and curium-244, were not included in the risk characterization for this receptor because the risk values for on-WAG receptors associated with these radionuclides were low or nonexistent. Off-WAG concentrations of radionuclides in soil and biota were calculated from surface water values for Clinch River Mile 9.5 (Appendixes 6D and 6E).

Derivation of chemical concentrations. Figure 6.4 illustrates the process used to estimate concentrations of chemicals for various exposure routes. Chemicals evaluated WAG-wide and in the high-activity auger hole area were those listed in Sect. 6.2.2 as being of potential concern. Field sampling results were used directly to calculate estimated concentrations for groundwater, surface water, sediment, and soil. Modeled data were used to determine concentrations of volatile organics in air across the entire WAG. The air model is discussed in Sect. 5.4.

Chemical concentrations in vegetables, fruit, beef, deer meat, milk, and fish were calculated using field data for soil, groundwater, and surface water and modeled data for air. Appendix 6F presents a detailed discussion of the calculation methods and results.

As with radionuclide concentrations, surface water concentrations downstream on the Clinch River were calculated using the surface water values at the WAG boundary and Clinch River dilution factors (ORNL 1991b). Boundary fluxes of the chemicals were computed by multiplying their concentrations in surface water on-WAG by the volume flow rate of water discharged from the WAG. The concentrations used represented the areal average of current concentrations for WAG 6 surface waters as determined by the median of the surface water sampling data (Sect. 4.4). Concentrations were quantified for six volatile organics (1,2-dichloroethylene, chloroform, acetone, tetrachloroethene, trichloroethene, and carbon disulfide) and one inorganic (silver). From the surface water results at Clinch River Mile 9.5, concentrations in soil, air (particulates), and biota were calculated. Appendix 6E presents the method for and results of calculating these concentrations in soil; Appendix 6F addresses concentrations in air and biota.

As stated in Sect. 6.2.2, no chemical source data were available. Therefore, future concentrations of chemicals in waste disposal areas could not be modeled and the conservative assumption was made that concentrations of chemicals in the environment would not change over time so that exposure intakes to a future on-WAG homesteader would be the same as those for the current on-WAG homesteader.

6.3.4.2 Calculation of pathway-specific intakes

Once contaminant concentrations were estimated for all environmental media and biota, reasonable maximum intakes were calculated for radioactive and chemical contaminants within identified pathways. Appendixes 6D and 6F detail calculation variables, methods, and results for radionuclides and chemicals, respectively. Upper-bound (90 percentile) and average values were used for equation variables to yield reasonable maximum intakes to each receptor. Results of intake calculations are discussed as part of the overall estimation of risk to receptors presented in Sect. 6.5, Risk Characterization.

6.4 TOXICITY ASSESSMENT

This section summarizes the toxicity assessment process, which consisted of two steps. First, hazards [illegible] identified and second, toxicity parameters were identified that quantified these hazards [illegible]ard identification is the process of determining whether or not exposure to an agent ca[illegible]e an increase in the incidence of a particular adverse health effect (e.g., cancer, birth de[illegible]). Toxicity parameters express a dose-response relationship (i.e., they quantify whether or not a relationship exists between the dose of a contaminant administered or received and the incidence of adverse health effects in the exposed population). These dose-response values were then used in combination with intake values (Sect. 6.3) to characterize risk (Sect. 6.5).

6.4.1 Radionuclide Toxicity Assessment

6.4.1.1 Hazard identification

Two broad classes of adverse health effects can result from exposure to radiation: stochastic and nonstochastic. Nonstochastic effects are those that have protective mechanisms that must be overcome before an observable adverse effect occurs. This concept is referred to as an "observable threshold." Nonstochastic effects increase in severity with increasing dose. Some examples of nonstochastic effects include cell death, clouding of the eye lens, cosmetically-unacceptable changes in the skin, and premature menopause. For radionuclides, these effects are observable only at relatively high doses. Human exposures below about 10 to 20 rem usually do not result in detectable nonstochastic effects.

Stochastic effects are those effects that are random (e.g., probabilistic) in nature and conservatively regarded as having no threshold. In the nonthreshold hypothesis (stochastic effects), no dose is thought to be risk free. Stochastic effects can be further divided into carcinogenic and noncarcinogenic effects.

Carcinogenic effects result in cancer. Among noncarcinogenic effects are genetic and teratological effects. The effects that are passed on through biologic mechanisms of inheritance are termed genetic effects. Teratological effects are commonly known as birth defects.

All radionuclides produce carcinogenic effects and have been formally classified as carcinogens by either the EPA Carcinogenic Assessment Group (EPA 1986c) or the International Agency for Research on Cancer (IARC 1982). These effects are assumed to be linearly proportional to dose by the International Commission on Radiological Protection (ICRP) and the National Council on Radiation Protection and Measurements (NCRP). No threshold associated with exposure to ionizing radiation is assumed, so that any exposure, no matter how small, might give rise to cancer.

Genetic and teratologic effects are not considered to be of primary concern relative to the WAG 6 human health assessment. The ICRP places the risk of genetic effects at approximately a factor of two lower than the risk of carcinogenic effects (e.g., 4×10^{-5} rem^{-1} versus 10^{-4} rem^{-1}) (ICRP 1977). Further, ICRP states that risk assessment focused on genetic effects should be made over the total population and most risk assessment scenarios do not include populations sufficiently large to represent a potential threat to the gene pool. For teratogens, current data on teratological effects are insufficient to predict a relationship at low doses (UNSCEAR 1977).

6.4.1.2 Dose-response evaluation for radionuclides

Toxicity parameters used to quantify the dose-response relationship associated with the 17 radionuclides of concern at WAG 6 (Table 6.3) were dose conversion factors and slope factors. Dose conversion factors (DCFs) have been derived from relating human exposure in an environmental medium to a resultant dose equivalent in the exposed individual. The DCFs for the three major exposure pathways at WAG 6 (inhalation, ingestion, and external exposure) are provided in Appendix 6G. However, because the factors converting from absorbed dose to dose equivalent were designed for control of occupational exposures, they are not readily applicable to the prediction of nonoccupational cancer risk. To compensate, an appropriate risk factor [risk per dose equivalent unit (rem)] can be multiplied by those dose equivalents (rem) to yield an estimate of risk. Nevertheless, these estimates do not take into account the observed cancer risk differences between high and low linear energy transfer (LET) radiation presented in Appendix 6G.

Slope factors are typically used in risk assessment studies to express cancer risk. These factors attempt to take the observed cancer risk differences between high and low LET radiation into account. They are computed using age- and sex-specific coefficients for individual organs receiving significant radiation doses. This information is then used along with organ-specific dose conversion factors to derive slope factors that represent the age-averaged lifetime excess cancer incidence per unit intake for radionuclides of concern. Slope factors for the three major exposure pathways at WAG 6 are provided in Appendix 6G.

None of the DCFs or slope factors were applicable to external exposure for contamination at depth (i.e., for subsurface contamination covered by a surface layer of clean soil). In this case, the dose-response relationship was expressed as the ratio of slope factor to DCF.

Because DCFs and slope factors have been based on internal dosimetry models developed for controlling occupational radiation exposure to adults, they do not account for metabolic differences between adults and children. DCFs published by DOE (DOE 1988) and EPA (EPA 1988d) are calculated for an adult with anatomical and physiological features defined as the "reference man" (ICRP 1974, 1980). The reference man is not representative of a child and, therefore, a child receptor was not evaluated as part of the radiological risk assessment for this human health evaluation.

6.4.2 Chemical Toxicity Assessment

6.4.2.1 Hazard identification

In identifying chemical hazards, potential contaminants of concern are separated into toxicological categories depending on whether or not they exhibit noncarcinogenic or carcinogenic effects. Exposure to noncarcinogenic chemicals may result in numerous health effects, ranging from effects on specific organ systems (e.g., neurotoxicity) to effects on reproduction and offspring (e.g., fetotoxicity). Chemicals considered to be potential carcinogens may also be capable of causing noncarcinogenic effects.

In Sect. 6.2, 41 chemicals of potential concern were identified for WAG 6 (Table 6.6). Of these, 12 are classified as known or probable human carcinogenic chemicals by either the EPA Carcinogenic Assessment Group or the International Agency for Research on Cancer, and 29 are noncarcinogenic chemicals (Table 6.11). Toxicity profiles for these chemicals are summarized in the attachment to Appendix 6G. These profiles address specific toxic effects associated with specific exposures, general toxicity information, cancer potential information, and symptoms of exposure. Nine of the chemicals found at WAG 6 exhibit both noncarcinogenic and carcinogenic effects.

6.4.2.2 Dose-response evaluation for chemicals

Toxicity parameters used to quantify hazards associated with chemicals were reference doses and slope factors. Reference doses were used to express the dose-response relationship for noncarcinogenic chemicals. The reference dose is an estimate of the maximum human chronic daily exposure level that is likely not to produce an appreciable risk of adverse effect during a lifetime. Since all of the exposure durations in this human health evaluation have been assumed to be 7 years or longer, chronic reference dose values have been used. The oral and inhalation RfDs for the potential chemicals of concern at WAG 6 are tabulated in Appendix 6G. Slope factors expressed the dose-response relationship for carcinogenic chemicals. EPA has assigned a weight-of-evidence classification to potential human carcinogens (EPA 1986c), which is tabulated in Appendix 6G. Of the chemical contaminants of potential concern at WAG 6 those classified as A, B, or C according to the EPA system were evaluated as carcinogens; slope factors used in these evaluations are tabulated in Appendix 6F.

Reference doses and slope factors were available for the inhalation and ingestion routes of exposure. In the absence of specific toxicity values based on absorbed doses for the dermal route of exposure, oral reference doses and slope factors were modified to express doses by assuming a 5% absorption efficiency [i.e., the oral reference dose was multiplied by 0.05 and the slope factor was divided by 0.05 (EPA 1989b)].

All chemicals were evaluated that had appropriate toxicity information available. Two organics and four inorganics had no toxicity values and were not considered further in the assessment. The organics were 1,2-dichloroethene (CAS 540-59-0) and 2-methylnaphthalene (CAS 91-57-6); the inorganics were aluminum (CAS 7429-90-5), titanium (CAS 7440-32-6), copper (CAS 7440-50-8), and cobalt (CAS 7440-48-4).

The toxicity value for chromium (VI) was used to represent chromium (total) for the water pathways at WAG 6. In the case of water this assumed that all of the total chromium was present in the form of chromium (VI). In freshwater, chromium can be present in both the chromium (III) (less toxic) and chromium (VI) (more toxic) forms (Towill et al. 1978); to be conservative in this human health evaluation the chromium (VI) form was assumed. In soil, chromium (VI) is easily converted to chromium (III) in the presence of organic matter, even at alkaline pH (Barlett and Kimble 1976). At WAG 6, the soil is low in organic matter, but has an acidic pH (Lietzke, Lee, and Tamura 1988). Therefore, it was assumed that the chromium (III) form was present in the WAG 6 soils.

6.5 RISK CHARACTERIZATION

In the risk characterization step of the base line human health evaluation, results from the exposure assessment (Sect. 6.3) and toxicity assessment (Sect. 6.4) were summarized and integrated into quantitative risk estimates. To quantify risk from exposure to carcinogenic contaminants, probabilities that a hypothetical receptor developed cancer over a lifetime of exposure were estimated from projected intakes and contaminant-specific dose-response information. In addition, risks from exposure to naturally occurring background radiation were quantified. To quantify risk from exposure to noncarcinogenic chemicals, direct comparisons were made between projected exposure levels and toxicity values. The following sections address risk from carcinogenic and noncarcinogenic contaminants for the hypothetical receptors associated with WAG 6. As stated in Sect. 6.4, carcinogenic risk was quantified for the radionuclides of potential concern; both types of risk were calculated for chemical contaminants of potential concern.

In the automated calculations used in quantifying the risk estimates presented in this section, chemicals for which all detects were "J-flagged" (estimated value) were inadvertently omitted from the chemicals pertinent to the groundwater pathway. However, subsequent manual calculations that included these chemicals for the most significant WAG-wide pathways (ingestion of groundwater, vapor phase inhalation while showering, and dermal contact with groundwater) resulted in negligible changes to the overall risk values and hazard indices. Risk values and hazard indices that did not exceed acceptable criteria based on the incomplete data remained below the acceptable limits with the addition of the "J-flagged" chemicals; four values that already exceeded acceptable levels increased slightly, as footnoted in Table 6-17. Results of the manual calculations are fully presented in Appendix 6F. The "J-flagged" data will be included in the automated calculations in a future revision of this document, consistent with EPA's recommendation to include such data in risk characterization (EPA 1989b). This approach was agreed upon with the Energy Systems Central Risk Assessment Council.

6.5.1 Quantification of Carcinogenic Risk

For carcinogenic contaminants, risks were estimated as the excess probability of an individual developing cancer over a lifetime as a result of exposure to a potential carcinogen via a specific pathway. A slope factor was used to convert estimated daily intakes averaged over a lifetime to a value expressing incremental risk. For the inhalation and ingestion pathways, risk associated with each radionuclide or chemical was estimated by multiplying the appropriate slope factor by the estimated intake of the substance in question. Risk from the external exposure pathway (relevant only to radionuclides) was computed as the product of the slope factor, the concentration in soil, and the duration of exposure to the radionuclide under study.

Simultaneous exposures to several radionuclides and chemicals via more than one pathway were also assessed. The risk of developing cancer from simultaneous exposure to several contaminants is based on the assumption that carcinogenic risks are additive (EPA 1989b). Appendix 6D presents calculations that express risk from exposure to multiple radionuclide carcinogens across several pathways and the values resulting from these calculations. Appendix 6F presents similar information for chemical carcinogens.

The on-WAG ORNL employee and the boundary receptor were exposed to only undisturbed subsurface contamination. Since none of the available slope factors were applicable to external exposure for such radioactive contamination, a radiation shielding model was employed to quantify dose; risk was then calculated using the ratio of slope factor to DCF for each of the radionuclides as the toxicity parameter. As suggested by EPA (1989b), radiation dose values were also calculated for each of the other receptors for the current and future time periods; these results are also presented in Appendix 6D.

Risk to the on-WAG homesteader was characterized several ways: using data from the entire WAG; data from the high-activity auger hole area (i.e., northeast auger hole area as illustrated in Fig. 1.6) and neighboring EWB; and data exclusively from the EWB, which has reportedly never been used as a holding basin for wastes. For risk from exposure to radionuclides, contaminant concentrations calculated at the 95 percentile confidence interval for each of the 26 study areas were averaged to yield a "WAG-wide" risk. Similarly, the radionuclide contaminant concentrations calculated for the study areas comprising the auger hole area and the EWB were averaged to estimate risk specific for these areas.

EPA has established a target risk range of 10^{-6} to 10^{-4} for risks from exposure to radionuclides and chemicals (EPA 1990c). When a base line risk assessment indicates that an individual's cumulative risk, derived using reasonable maximum exposure assumptions for either current or future land use, exceeds 10^{-4} (i.e., 1 in 10,000 chance of developing cancer), action is generally warranted at the site. For sites where this cumulative risk to an individual is less than 10^{-4}, action generally is not warranted. However, EPA has expressed a preference for remediation to achieve the more protective end of the target risk range (10^{-6} or a 1 in 1,000,000 chance of developing cancer) (EPA 1991f). General guidance for exposure to workers has been established in the *Federal Radiation Protection Guidance for Occupational Exposure*: "the effective dose equivalent received in any year by an adult worker should not exceed 5 rems" (52FR2822, January 27, 1987).

Risks calculated for the WAG 6 receptors were compared with EPA's target range. As shown in Fig. 6.5, the current day receptors with the highest cancer risk from radionuclides were the on-WAG homesteaders, whose risks were characterized based on data from the entire WAG. These individuals exhibited a risk of 1 (i.e., a 100 percent probability of developing cancer). Cancer risk for the off-WAG homesteader was below the upper limit of the EPA's target range at 6 x 10^{-5}, and for the hunter and boundary receptor was less than 10^{-6}. Risk to the on-WAG homesteader using data exclusively from the EWB was 4 x 10^{-6} (Table 6.12). The future on-WAG homesteader whose risk was characterized using data from the auger hole area exhibited a risk of 9 x 10^{-1}, while the future on-WAG homesteader characterized using data from the entire WAG exhibited a lesser risk of 10^{-1}. The on-WAG ORNL employee showed a risk of 10^{-3} (or 2 rem) over a 30-year exposure period, which exceeded the upper limit of EPA's target range, but was well below the maximum 5 rem/yr occupational exposure limit cited above. Tables 6.13 and 6.14 illustrate the exposure pathways contributing to these risks above or within EPA's target range.

Exposure pathways for receptors with risks below the lower end of the risk range are presented in Appendix 6D.

The assumption that waste is exhumed and placed in the garden and yard had a significant impact on the risk values from radionuclides to the on-WAG homesteaders. To put the importance of this assumption in perspective, risk values were calculated for the current and future on-WAG homesteaders assuming no waste exhumation and also construction of the homesteader's residence on a concrete slab over the wastes (Table 6.15). Three pathways were evaluated under these assumptions:

- External exposure from radionuclides in subsurface soil
- Ingestion of radionuclides in groundwater
- Ingestion of biota that consumed groundwater contaminated with radionuclides

Risk values were still above the upper limit of EPA's target range, but considerably less (three orders-of-magnitude) than those calculated assuming waste exhumation (Table 6.16.).

For perspective, the radionuclide risk results presented above may be compared to the probability of an individual developing cancer from exposure to naturally occurring background radiation. Unlike many risks, exposure to natural radiation is largely unavoidable. The EPA evaluates risks from exposure to average levels of background radiation in its *Background Information Document for the Environmental Impact Statement for NESHAPS Radionuclides,* (EPA 1989f). The EPA distinguished two components of background radiation: low-LET radiation (gamma and x-radiation from cosmic and terrestrial sources plus radionuclides internal to the body) and high-LET radiation (from alpha emitters in the body).

The average dose to the U.S. population from low-LET radiation was estimated to be 87 mrad/yr, with 80% of the population receiving doses between 75 and 115 mrad/yr. The risk of fatal cancer for lifetime exposure (70.7 years) was estimated to be 2.4×10^{-3} or approximately 1.5% of all U.S. cancer deaths.

The average dose from alpha emitters (high-LET radiation) other than radon decay products in lung tissues was estimated to be 0.51 mrad/yr, which corresponds to a lifetime cancer risk of 2.0×10^{-5}. The average dose to endosteal surfaces of bone from naturally occurring alpha emitters was estimated to be 6 mrad/yr, which corresponds to a lifetime risk of fatal bone cancer of 8.5×10^{-5}. The average residential exposure to radon decay products was estimated to be 0.2-0.3 WLM/yr, which (using 0.25 WLM/yr) corresponds to a lifetime lung cancer risk of 6.4×10^{-3}. Therefore, the risk from high-LET radiation corresponds to approximately 4% of all U.S. cancer deaths.

The total risk of fatal cancer due to background radiation is therefore 8.9×10^{-3}. To render this risk comparable to those calculated for exposure to radionuclides from WAG 6, it must be converted to the lifetime risk of cancer incidence. From EPA's risk factors for low-LET radiation, a ratio of cancer incidence to fatal cancers is calculated to be 1.6. Using this ratio, the lifetime risk of cancer incidence is approximately 1.4×10^{-2} or 100 times greater than the upper bound of EPA's target risk range (10^{-4}).

As shown in Fig. 6.5, the adult on-WAG homesteader exhibited a cancer risk from chemicals of 3 x 10^{-4} based on data from the entire WAG and a risk of 3 x 10^{-4} based on data from the high-activity auger hole area. These values and those for the child homesteader in both cases slightly exceed the upper limit (10^{-4}) of EPA's target range. Table 6.17 illustrates the exposure pathways contributing to these risks. Risks to the other receptors based on WAG-wide and auger hole area chemical data were all less than 10^{-6}; tables illustrating the pathways contributing to these risks are presented in Appendix 6F. Chemical risk to the adult on-WAG homesteader based on data from the EWB was 7 x 10^{-8}; for the child it was 3 x 10^{-8} (Table 6.18).

6.5.2 Quantification of Noncarcinogenic Risk

The potential for noncarcinogenic effects on health was evaluated by comparing an exposure level with a reference dose. This ratio of exposure to toxicity is called a hazard quotient. The hazard quotient assumes that a level of exposure exists [reference dose (RfD)] below which it is unlikely for a receptor to experience adverse health effects. If the exposure level (E) exceeds this threshold (i.e., if E/RfD exceeds unity), the potential exists for noncarcinogenic effects to develop.

Exposures to noncarcinogenic chemicals may involve potential exposure to several chemicals simultaneously for a critical effect. To assess the overall potential for noncarcinogenic effects in such circumstances, a hazard index (HI) specific to a critical effect is used. The HI is equal to the sum of the hazard quotients. Thus, the HI can exceed unity even if no single chemical exposure exceeds the RfD for that chemical. Naturally, any single chemical with an estimated daily intake greater than the corresponding RfD will cause the HI to exceed unity. When the HI exceeds unity, adverse critical effects on human health may develop.

Figure 6.6 shows that the receptors whose hazard index for a critical effect exceeded unity were the on-WAG homesteader child whose risks were characterized for the entire WAG and the on-WAG homesteader adult and child whose risks were characterized for the high-activity auger hole area. The hazard index for total effects exceeded or approached unity for the adult homesteader whose risk was characterized WAG-wide; the hazard index for this receptor never exceeded unity for a specific critical effect. Table 6.19 details the exposure pathways and chemicals contributing to the health hazards that had a critical effect-specific hazard index exceeding 1. Exposure pathways for receptors with effect-specific hazard indices less than 1, and total hazard indices are presented in Appendix 6F. Data exclusively from the EWB yielded hazard indices well below unity for both adult and child on-WAG homesteaders (Table 6.20).

6.6 IDENTIFICATION OF UNCERTAINTIES

Uncertainty factors were associated with each component of the human health evaluation: selection of contaminants of potential concern, exposure assessment, toxicity assessment, and risk characterization. A qualitative approach was used to identify variables and assumptions that contributed to uncertainty and to evaluate their influence on final risk estimates for receptors associated with WAG 6 (Table 6.21).

6.7 SUMMARY AND CONCLUSIONS

The base line human health evaluation quantified potential carcinogenic and noncarcinogenic effects on hypothetical receptors of exposure to WAG 6 radionuclides and chemicals. The objective of the evaluation was to determine whether or not human health is threatened if no remedial action is undertaken at the WAG. Incremental lifetime cancer risks were estimated on the basis of total dose received from all possible exposure routes. However, these calculated "excess" lifetime cancer risks do not reflect, in any manner, actual risks to the exposed population in the vicinity of the WAG. Rather, they are conservative estimates of the reasonable maximum risk to hypothetical receptors, using assumptions within the framework of accepted health risk assessment methodologies.

Data on radionuclide and chemical contaminants at the WAG were summarized for groundwater, surface soil, surface water, and sediment. Information on the WAG 6 radionuclide inventory was screened to identify radionuclides of potential concern. Models were then used to calculate current and future concentrations of these radionuclides in groundwater, soil, surface water, and air. Statistical methods were used to identify potential chemicals of concern in addition to organics, which were evaluated further. Models were used to determine concentrations of chemicals in surface water and concentrations of volatile organic chemicals in air.

An on-WAG homesteader, an on-WAG ORNL employee, a hunter near the WAG, a boundary receptor at the fenceline of the WAG, and an off-WAG homesteader were evaluated as hypothetical receptors. Exposure routes analyzed for these receptors were ingestion of groundwater, surface water, fruit, vegetables, milk, beef, and deer meat; incidental ingestion of surface water, soil, and sediment; inhalation of suspended particulates and volatiles from ambient air and showering; dermal contact with soil, surface water, sediment, and groundwater; and external exposure from soil. Intakes of contaminants from these exposure pathways were then used in conjunction with toxicity parameters (or dose-response values) for the potential contaminants of concern to quantify the risk of developing carcinogenic and noncarcinogenic health effects.

6.7.1 Carcinogenic Risk

Carcinogenic risk was quantified for the 17 radionuclides of concern and the 12 chemicals classified as carcinogens. Under the toxicological and exposure assumptions used in the evaluation, the radionuclide-related excess lifetime cancer risk of the on-WAG homesteader and on-WAG ORNL employee exceeded the upper limit of EPA's target range (10^{-4}), but for the employee was only 1.3% of the limit recommended in the guidance for occupational exposure. These risks were also well below the total risk of fatal cancer resulting from background radiation (1.4×10^{-2}).

The off-WAG homesteader and the on-WAG homesteader whose risk was characterized using data exclusively from the EWB exhibited radionuclide-related excess lifetime cancer risk below the upper limit of EPA's target range. Risks calculated for the hunter and boundary receptors were both less than 10^{-8}.

Chemical-related excess lifetime cancer risk was highest for the on-WAG homesteader, slightly exceeding the upper limit of EPA's target range. The on-WAG homesteader whose risk was characterized using data exclusively from the EWB exhibited a risk less than 10^{-8}. The on-WAG ORNL employee, hunter near the WAG, boundary receptor, and off-WAG homesteader all exhibited risks less than 10^{-10}.

6.7.2 Noncarcinogenic Risk

In the case of noncarcinogens, the risk of developing adverse health effects was expressed by the degree to which the effect-specific HI exceeded one. Only the on-WAG homesteader in the auger-hole subarea exhibited a critical effect specific hazard index exceeding one.

Section 6 Tables

Table 6.1. WAG 6 radionuclide inventory (decayed to 1 January 1991)

Radionuclide	Curies	Radionuclide	Curies
Ac-225	1.48E-1	AC-227	9.08E-6
Ac-228	1.75E+0	Ag-109m	1.15E-3
Ag-110	1.70E-4	Ag-110m	1.13E-2
Ag-111	2.69E-37	Am-241	3.69E+0
Am-243	2.08E-2	Am-245	3.09E-9
Ar-39	1.29E-3	At-217	1.48E-1
Ba-137m	3.80E+4	Be-10	7.12E+2
Bi-210	1.04E-2	Bi-211	8.79E-6
Bi-212	1.39E+0	Bi-213	1.48E-1
Bi-214	1.04E-2	Bk-249	3.09E-6
Bk-250	9.80E-6	C-14	2.69E+2
Cd-109	1.15E-3	Cd-113	4.00E+3
Ce-141	1.05E-14	Cf-249	1.29E-2
Cf-250	2.49E-2	Cl-36	3.06E-1
Cm-242	3.34E-15	CM-243	9.60E-6
Cm-244	2.24E+1	Cm-245	3.47E-6
Cm-248	2.40E-4	Co-57	1.86E-3
Co-58	4.47E-1	Co-60	2.10E+4
Cr-51	7.49E-10	Cs-134	4.95E+0
Cs-137	4.01E+4	Es-253	2.90E-31
Es-254	9.80E-6	Eu-152	2.39E+5
Eu-154	1.71E+5	Eu-155	4.23E+4
Eu-156	2.27E-25	Fe-55	7.79E+1
Fe-59	2.41E-2	Fr-221	1.48E-1
Fr-223	1.27E-7	Ga-68	1.19E-2
Gd-152	2.19E-9	Gd-153	9.60E+1
Ge-68	1.19E-2	H-3	4.92E+3
Hg-203	4.08E-11	I-125	5.14E-5
I-131	1.45E-14	Ir-191m	1.00E-7
Ir-192	8.43E-1	K-40	4.68E-3
Kr-85	1.52E+0	Mn-54	1.21E+1
Na-22	2.70E-2	Nb-93m	9.56E+0
Nb-95	9.11E-8	Nb-95m	3.95E-10
Ni-63	1.95E+0	Np-237	9.07E-3
Np-239	2.08E-2	Np-240	7.05E-15
Np-240m	7.05E-12	Os-191	1.00E-7
P-32	5.35E-11	Pa-231	8.61E-5
Pa-233	9.07E-3	Pa-2324	2.05E-1
Pa-234m	2.05E+2	Pb-207m	7.88E-1
Pb-209	1.48E-1	Pb-210	1.04E-2
Pb-211	8.79E-6	Pb-212	1.39E+0
Pb-214	1.04E-2	Pd-103	6.88E-5
Pd-107	1.30E-1	Pm-147	4.07E+1
Po-210	1.03E-2	Po-211	2.64E-8
Po-212	8.89E-1	Po-213	1.45E-1
Po-214	1.04E-2	Po-215	8.79E-6

Table 6.1. (continued)

Radionuclide	Curies	Radionuclide	Curies
Po-216	1.39E+0	Po-218	1.04E-2
Pu-238	3.24E-2	Pu-239	6.04E-1
Pu-240	2.33E-2	Pu-241	2.67E-7
Pu-244	7.06E-12	Ra-223	8.79E-6
Ra-224	1.39E+0	Ra-225	1.49E-1
Ra-226	1.04E-2	Ra-228	1.75E+0
Re-188	1.82E-6	Rh-103m	6.89E-5
Rh-106	3.10E+1	Rn-219	8.79E-6
Rn-220	1.39E+0	Rn-222	1.04E-2
Ru-106	3.10E+1	S-35	2.05E-6
Sm-147	4.79E-9	Sm-151	1.37E+2
Sn-119m	5.23E-3	Sn-121m	7.16E+1
Sn-123	5.44E-10	Sr-85	1.97E-6
Sr-89	5.07E+0	Sr-90	6.55E+3
Ta-179	1.08E-1	Tc-95	2.06E-10
Tc-95m	5.22E-9	Tc-99	3.83E+1
Th-227	8.77E-6	Th-228	1.39E+0
Th-229	1.50E-1	Th-230	6.10E-3
Th-231	5.92E-1	Th-232	2.66E+0
Th-234	2.05E+2	Tl-204	3.54E-2
Tl-207	8.77E-6	Tl-208	5.00E-1
Tl-209	3.25E-3	U-232	2.08E-2
U-233	2.28E+2	U-234	3.73E-3
U-235	5.92E-1	U-235m	6.04E-1
U-236	7.00E-5	U-238	2.05E+2
U-240	7.05E-12	W-188	1.80E-6
Xe-131m	6.66E-13	Y-90	6.55E+3
Zn-65	1.11E+0	Zr-93	2.26E+1
Zr-95	4,14E-8		
Total Activity			5.72E+5

Table 6.2. Summary of WAG 6 waste disposal details

Unit type	Unit description; disposal dates	Total curies	Percentage of below ground units
Below ground	Auger holes, silos, silo wells, trenches; January 1962 - December 1990	571,710[a]	100
Aboveground	Tumulus I and II, Hill Cut Disposal Test Facility; 1981-December 1990	143	0.025
Future	Tumulus II, silos, silo-wells; December 1990-September 1992	~6,600[b]	1.1

[a] Delayed to 1 January 1991.

[b] Assuming that the same radioactive waste is disposed of in the future and at the same rate as in the past.

Table 6.3. Radionuclides of potential concern at WAG 6

Radionuclide	Radionuclide
Am-241	Pa-234[a]
Ba-137m[a]	Pa-234m[a]
Cm-244	Sr-90
Co-60	Th-232
Cs-137	Th-234[a]
Eu-152	U-233
Eu-154	U-238
Eu-155	Y-90
H-3	

[a] Short-lived daughter assumed to be in equilibrium with parent and therefore also considered in the quantitative analysis of risk.

Table 6.4. Results of further analysis of statistically nonsignificant semivolatile organics

Chemical[a,b]	CAS number	Concentration exceeded CRDL	Chemical statistically significant in other media	Concentration exceeded health-based criterion	Known WAG contaminant
Groundwater					
Diethylphthalate	84-66-2	X			
Naphthalene	91-20-3	X		X	X
2-Methylnaphthalene	91-57-6	X			
2,4,5-TP	93-72-1	X			
2,4,5-T	93-76-5	X			
2,4-D	94-75-7	X			
Benzyl alcohol	100-51-7	X			
Soils					
Benzyl alcohol	100-51-6	X			

[a] Only chemicals meeting one or more of the four selection criteria shown in this table.
[b] Includes some pesticides and herbicides.

Table 6.5. Intakes of essential nutrients from media at WAG 6 compared with recommended dietary allowances

		Intake calculated from WAG 6 sampling results[a]	
Element	Recommended dietary allowance (mg/d)[b]	Groundwater (mg/d)[b]	Soil (mg/d)
Calcium	400-1200	200	
Magnesium	40-400	30	
Iron	6-30	17	4.0
Sodium	120-500	18	
Potassium	500-2000		0.2

[a] Intakes calculated using average nutrient concentration detected and assuming consumption of 2 L water per day and 100 mg of soil per day.

[b] Includes adult males and females and sensitive individuals (i.e., children, infants, pregnant and lactating females).

Source: National Research Council. 1989b. *Recommended Dietary Allowances*, 10th Edition, Subcommittee on Recommended Dietary Allowances, Food and Nutrition Board; Committee on Life Sciences, National Academy Press, Washington, D.C.

Table 6.6. Rationale for designation as chemical contaminant of concern at WAG 6[a]

Chemical	CAS number	Medium				
		Soil	Groundwater	Sediment[b]	Surface water	Air
Organics						
Carbon Tetrachloride	56-23-5		V			
Acetone	67-64-1	V		V	V	V
Chloroform	67-66-3		V	SS	V	V
Benzene	71-43-2		V			V
Chloromethane	74-87-3		V			
Ethyl Chloride	75-00-3					V
Vinyl Chloride	75-01-4		V			
Methylene Chloride	75-09-2	V		V		V
Carbon Disulfide	75-15-0		V		V	
1,1-Dichloroethane	75-34-3		V			V
Trichlorofluoromethane	75-69-4			V		
2-Butanone	78-93-3			SS		V
Trichloroethene	79-01-6	V	V	V	V	V
1,1,2,2-Tetrachloroethane	79-34-5					V
Diethylphthalate	84-66-2		O			
Di-n-Butylphthalate	84-74-2		O			
Naphthalene	91-20-3		O			V
2-Methylnaphthalene	91-57-6		O			
2,4,5-TP	93-72-1		O			
2,4,5-T	93-76-5		O			
2,4-D	94-75-7		O			
Ethylbenzene	100-41-4	V	SS			V
Benzyl Alcohol	100-51-6	O				V
2,4-Dimethylphenol	105-67-9					V
1,2-Dichloroethane	107-06-2		V			V
Toluene	108-88-3	V	V	V		V
Tetrachloroethene	127-18-4		V		V	V
1,2-Dichloroethene	540-59-0	V	V		V	
Xylenes (total)	1330-20-7	V	V			V
Inorganics						
Aluminum	7429-90-5	SS	O			
Manganese	7439-96-5		SS			
Silver	7440-22-4		O	SS	SS	
Thallium	7440-28-0	SS				
Titanium	7440-32-6		SS			
Arsenic	7440-38-2		O			

Table 6.6. (continued)

Chemical	CAS number	Medium				
		Soil	Groundwater	Sediment[b]	Surface water	Air
Barium	7440-39-3		O			
Chromium	7440-47-3	SS	O			
Cobalt	7440-48-4		O			
Copper	7440-50-8		O			
Vanadium	7440-62-2	SS	SS			
Zinc	7440-66-6		O			

[a] Coding in table is as follows: SS-statistically significant, V-volatile, O-other (in excess of contract required detection limit; statistically significant in other media; in excess of health-based criterion; in excess of reference concentration; known WAG contaminant).

[b] For EWB only.

Table 6.7. Scenarios, receptors, and periods of evaluation studied

Scenario	Hypothetical receptor	Evaluation period
No action	On-WAG homesteader[a]	Current[b]
Institutional control	On-WAG homesteader[a]	Future[c]
Institutional control	On-WAG ORNL employee	Current
Institutional control	Hunter near WAG	Current
Institutional control	Boundary receptor	Current
Institutional control	Off-WAG homesteader[a]	Current

[a] Homesteader assumes adult receptor for exposure to radionuclides and assumes adult and child receptors for exposure to chemicals.

[b] Beginning in 1990.

[c] Beginning in 2100.

Table 6.8. Potential exposure pathways for WAG 6 receptors

Receptor	Exposure medium	Exposure route	Pathway evaluated	Reason for selection or exclusion
Hunter	Deer	Ingestion (R,C)[a]	Yes	Contaminants detected in deer killed on ORR
Boundary receptor	Air	Inhalation - Vapors (R,C)	Yes	Volatile organics and tritium present in soil on WAG
		- Particulates (R,C)	No	No surface contamination since waste areas covered by 1 ft. of clean cover material
	Soil	External Exposure (R)	Yes	Gamma emitters present in disposal units
On-WAG ORNL employee	Air	Inhalation-Vapors (R,C)	Yes	Volatile organics present in soil on WAG; tritium present in surface water on WAG
		- Particulates (R,C)	No	No surface contamination since waste areas covered by 1 ft. of clean cover material
	Soil	External Exposure (R)	Yes	Gamma emitters present in disposal units
		Dermal contact (C)	No	No surface contamination since waste areas covered by 1 ft. of clean cover material
	Surface water	Dermal contact (C)	No	Not in contact with surface water bodies
	Sediment	External exposure (R)	No	Not in contact with surface water bodies
		Dermal contact (C)	No	Not in contact with surface water bodies
On-WAG homesteader	Air	Inhalation-Vapors (R,C)	Yes	Volatile organics present in soil on WAG; tritium present in surface water on WAG
		- Particulates (R,C)	Yes	Waste material present in garden and yard

Table 6.8. (continued)

Receptor	Exposure medium	Exposure route	Pathway evaluated	Reason for selection or exclusion
	Soil	External exposure (R)	Yes	Gamma emitters present in waste material in garden and yard
		Incidental ingestion (R,C)	Yes	Waste material present in garden and yard
		Dermal contact (C)	Yes	Waste material present in garden and yard
	Groundwater	Inhalation-Vapors while showering (C)	Yes	Volatile organics present in groundwater
		Ingestion (R,C)	Yes	Contaminants present in groundwater
		Dermal contact (C)	Yes	Waste materials present in groundwater
	Surface water	Incidental ingestion (C)	Yes	While swimming in EWB
		Dermal contact (C)	Yes	While swimming in EWB
	Sediment	External exposure (R)	No	Insignificant compared to soil exposures
		Incidental ingestion (C)	Yes	While swimming in EWB
		Dermal contact (C)	Yes	While swimming in EWB
	Biota	Ingestion (R,C)	Yes	Vegetables/fruits from garden; milk/meat from cows
Off-WAG homesteader	Air	Inhalation-Vapors (R,C)	No	Volatile organics insignificant in surface water
		- Particulates (R,C)	Yes	Contaminants present in garden and yard
	Soil	External exposure (R)	Yes	Gamma emitters present in garden and yard
		Incidental ingestion (R,C)	Yes	Contaminants present in garden and yard
		Dermal contact (C)	Yes	Contaminants present in garden and yard
	Surface water	Inhalation-Vapors while showering (C)	No	Volatile organics insignificant in surface water

Table 6.8. (continued)

Receptor	Exposure medium	Exposure route	Pathway evaluated	Reason for selection or exclusion
		Ingestion (C)	Yes	Contaminants present in surface water
		Incidental ingestion (R,C)	No	Insignificant compared to ingestion of surface water
		Dermal contact (C)	Yes	While swimming in Clinch River and showering
	Sediment	External exposure (R)	No	Insignificant compared to soil exposures
		Incidental ingestion (R,C)	No	Insignificant compared to soil exposures
		Dermal contact (C)	No	Insignificant compared to soil exposures
	Biota	Ingestion (R,C)	Yes	Vegetables/fruits from garden; milk/meat from cows

[a] (R) - radionuclides; (C) - chemicals

Table 6.9. Receptor/pathway combinations studied for WAG 6

Potential receptors	Pathway evaluated for current assessment[a] 1990		Pathway evaluated for future assessment 2100
	On-WAG	Off-WAG	WAG 6 area[b]
Hunter receptor		Ingestion-deer (R,C)	
Boundary receptor[c]		Inhalation-air (R,C) External exposure-soil (R)	
On-WAG ORNL employee[d]	Inhalation-air[e] (R,C) External exposure-soil (R)		
On-WAG homesteader[f,g]	External exposure-soil (R) Inhalation-air (R,C); SW (R); GW (C) Ingestion-GW, biota (R,C) Incidental ingestion-soil (R,C); SW, sediment (C) Dermal contact-soil, GW, SW, sediment (C)		External exposure-soil (R) Inhalation-air (R,C); SW (R); GW (C) Ingestion-GW, biota (R,C) Incidental ingestion-soil (R,C); SW, sediment (C) Dermal contact-soil, GW, SW, sediment (C)
Off-WAG homesteader[f,h]		External exposure-soil (R) Inhalation-air[i] (R,C) Ingestion-SW, biota (R,C) Incidental ingestion-soil (R,C) Dermal contact-soil, SW (C)	

[a] GW means groundwater; SW means surface water; R means assessed for radionuclide components; C means assessed for chemical components.

[b] WAG 6 area in future, no WAG boundaries exist; assumes institutional controls of 10-year operational period plus 100-year post-operational period.

[c] Only where WAG does not adjoin other WAGs (i.e., along its western and northern limits).

[d] Assumes normal working conditions and no accident scenarios.

[e] Includes inhalation of volatiles and particulates for air exposure routes.

[f] For radiological assessment evaluated adult only; for chemical assessment evaluated adult and child.

[g] Represents concentrations across the entire WAG and at the high-activity auger hole area.

[h] Off-WAG receptor located downstream at Clinch River Mile 9.5; includes contamination of specific media through irrigation using contaminated Clinch River water.

[i] Assessed for particulates only.

Table 6.10. Assumptions about receptors and pathways evaluated

I. <u>On-WAG Homesteader-Adult and Child (Current and Future)</u>
- Assumptions:
    - homesteader family comprised of two adults and one child [average family size for area surrounding WAG 6 (1990 Census for Roane County)]
    - adult homesteader evaluated for exposure to both radioactive and chemical contaminants; child homesteader evaluated for exposure to chemical contaminants only
    - waste exhumed during construction of a house on the waste disposal area is placed in the garden and yard; waste in these locations is indistinguishable from native soil
- Pathways:
    - ingestion of contaminated drinking water from a groundwater well near the waste disposal areas
    - ingestion of vegetables/fruit grown in contaminated soil
    - ingestion of milk and beef from dairy and beef cattle that drink contaminated groundwater
    - incidental ingestion of contaminated soil from the vegetable/fruit garden and yard
    - incidental ingestion of contaminated water and sediment while swimming in the EWB
    - external exposure from contaminated soil in the garden and yard
    - inhalation of suspended particulates and volatiles from contaminated soil in the garden and yard
    - inhalation of vapors from groundwater while showering
    - inhalation of tritium from surface water (i.e., on-WAG)
    - dermal contact with soil in the yard and garden
    - dermal contact with ground water while showering
    - dermal contact with surface water and sediment while swimming in the EWB

II. <u>Off-WAG Homesteader-Adult and Child</u>
- Assumptions:
    - homesteader family comprised of two adults and one child [average family size for area surrounding WAG 6 (1990 Census for Roane County)]
    - adult homesteader evaluated for exposure to both radioactive and chemical contamination; child homesteader evaluated for exposure to chemicals contaminants only
    - homesteader located at Clinch River Mile 9.5 downstream of WAG 6
    - contamination at this location is exclusively from surface water leaving WAG 6
    - homesteader uses Clinch River water to:
        - drink
        - irrigate his vegetable/fruit garden and yard
        - provide water for his cattle to drink
        - swim and fish in
- Pathways:
    - external exposure from contaminated soil in garden and yard
    - ingestion of surface water for drinking water
    - ingestion of fruit and vegetables grown in garden irrigated with contaminated surface water
    - ingestion of milk and beef from cattle drinking contaminated surface water
    - ingestion of fish from Clinch River
    - incidental ingestion of contaminated soil from garden and yard

Table 6.10. (continued)

- inhalation of particulates from contaminated soil from garden and yard
- dermal contact with soil in the yard and garden
- dermal contact with surface water while showering

III. On-WAG ORNL Employee

- Assumptions:
    - employee does maintenance and surveillance work only (i.e., not involved with disposal of waste materials during the 10-year operational period)
    - employee is not a radiation protection worker (i.e., protective equipment is not required)
    - normal working conditions and no accident scenarios
    - employee is evaluated on waste disposal areas (i.e., not on areas of WAG where no waste disposal occurred)
    - all waste material is covered by a minimum of 1 ft of clean material
- Pathways:
    - external exposure from soil
    - inhalation of volatiles (including tritium)

IV. Boundary Receptor at Fenceline

- Assumptions:
    - adult only since exposure to child is unlikely due to remote location of WAG
    - receptor is located only along portions of WAG that are not adjacent to another WAG (e.g., WAG 2)
    - exposures are only from waste disposal areas nearest fence (i.e., average value from entire WAG will not be used)
    - waste disposal areas are covered by a minimum of 1 ft of clean material
- Pathways:
    - external exposure from soil on-WAG
    - inhalation from volatiles (including tritium)

V. Hunter - Off-WAG

- Assumptions:
    - hunter kills deer that drink contaminated water exclusively from EWB and other contaminated ephemereal surface water streams on WAG 6
    - hunter is at least 100 yards from WAG 6; therefore, is not exposed directly to WAG 6 contaminants at the boundary
- Pathway:
    - ingestion of contaminated deer meat

Table 6.11. Chemicals of potential concern exhibiting carcinogenic effects and noncarcinogenic effects

Chemical	CAS number	Carcinogen	Noncarcinogen
Organics			
Carbon Tetrachloride	56-23-5	X	X
Acetone	67-64-1		X
Chloroform	67-66-3	X	X
Benzene	71-43-2	X	
Chloromethane	74-87-3	X	
Ethyl Chloride	75-00-3		X
Vinyl Chloride	75-01-4	X	
Methylene Chloride	75-09-2	X	X
Carbon Disulfide	75-15-0		X
1,1-Dichloroethane	75-34-3		X
Trichlorofluoromethane	75-69-4		X
2-Butanone	78-93-3		X
Trichloroethylene	79-01-6	X	
1,1,2,2-Tetrachloroethane	79-34-5	X	
Diethylphthalate	84-66-2		X
Di-n-Butylphthalate	84-74-2		X
Naphthalene	91-20-3		X
2-Methylnaphthalene[a]	91-57-6		
2,4,5-TP	93-72-1		X
2,4,5-T	93-76-5		X
2,4-D	94-75-7		X
Ethylbenzene	100-41-4	X	
Benzyl Alcohol	100-51-6		X
2,4-Dimethylphenol	105-67-9		X
1,2-Dichloroethane	107-06-2	X	
Toluene	108-88-3		X
Tetrachloroethylene	127-18-4	X	X
1,2-Dichloroethylene[a]	540-59-0		
Xylenes (total)	1330-20-7		X
Inorganics			
Aluminum[a]	7429-90-5		
Manganese	7439-96-5		X
Silver	7440-22-4		X
Thallium	7440-28-0		X
Titanium[a]	7440-32-6		
Arsenic	7440-38-2	X	X
Barium	7440-39-3		X

Table 6.11. (continued)

Chemical	CAS number	Carcinogen	Noncarcinogen
Chromium VI	18540-29-9	X	X
Cobalt[a]	7440-48-4		
Copper[a]	7440-50-8		
Vanadium	7440-62-2		X
Zinc	7440-66-6		X

[a] No EPA-approved dose-response relationship available; not further evaluated in risk characterization.

Table 6.12. Carcinogenic risk to the adult on-WAG homesteader from radionuclides in the EWB

Pathway	Risk	Dose	Percentage of total dose
Inhalation	2E-9	1E-6	<1
Incidental ingestion of surface water	4E-6[a]	5E-3	99
Incidental ingestion of sediment	1E-8	4E-5	<1
Pathway total	4E-6	5E-3	

[a] 96% of risk from this pathway results from exposure to tritium; remainder results from exposure to strontium-90 and cesium-137.

Table 6.13. Radionuclide risk and dose estimated for the on-WAG homesteader under the no-action scenario

	Homesteader WAG-wide			Homesteader auger-hole area		
Pathway	Risk[a]	Dose[b]	Percentage of total dose[c,d]	Risk[a]	Dose[b]	Percentage of total dose[c,d]
External exposure	1E+2	2E+5[f]	99	7E+2	1E+6[g]	98
Inhalation	6E-1	3E+3	1	4E+0	2E+4	2
Ingestion of food[e]	3E-3	3E+1	<1	2E-2	2E+2	<1
Ingestion of groundwater	6E-4	1E+0	<1	8E-4	1E+0	<1
Ingestion of soil	5E-3	9E+1	<1	3E-2	7E+2	<1
Total	1E+2	2E+5		7E+2	1E+6	
Total one-hit risk	1E+0[h]			1E+0[h]		

[a] Excess probability of individual developing cancer calculated using linear low-dose cancer equation.

[b] Expressed in units of rem.

[c] Dose attributable to the specific pathway divided by receptor's total dose.

[d] Sum of percents may be greater than 100 due to rounding.

[e] Includes ingestion of fruits and vegetables, milk, and meat.

[f] Dose of 1E+5 rem from europium-152 and 9E+4 rem from europium-154.

[g] Dose of 8E+5 rem from europium-152 and 5E+5 from europium-154.

[h] Calculated using one-hit equation for high carcinogenic risk levels.

Table 6.14. Radionuclide risk and dose estimated for the on-WAG ORNL employee, on-WAG homesteader, and off-WAG homesteader under the institutional control scenario

	On-WAG ORNL employee			On-WAG homesteader, WAG-wide			On-WAG homesteader, auger hole area			Off-WAG homesteader		
Pathway	Risk[a]	Dose[b]	Percent[c,d]	Risk[a]	Dose[b]	Percent[c,d]	Risk[a]	Dose[b]	Percent[c,d]	Risk[a]	Dose[b]	Percent[c,d]
External exposure	1E-3	2E+0[e]	98	4E-1	6E+2	18	2E+0	4E+3	15	6E-5	1E-1[f]	90
Inhalation	7E-9	5E-6	2	6E-2	3E+3[g]	79	5E-1	2E+4[h]	83	4E-8	6E-4	<1
Ingestion of food[i]				3E-4	2E+1	<1	2E-3	2E+2	<1	1E-6	5E-3	4
Ingestion of water				3E-4	7E-1	<1	2E-4	8E-1	<1	3E-6	6E-3	5
Ingestion of soil				1E-3	8E+1	2	6E-3	7E+2	2	2E-8	7E-5	<1
Total	1E-3	2E+0		4E-1	4E+3		3E+0	3E+4		6E-5	1E-1	
Total one-hit risk				3E-1[j]			9E-1[j]					

[a] Excess probability of an individual developing cancer calculated using the linear low dose cancer equation.
[b] Expressed in units of rem.
[c] Dose attributable to the specific pathway divided by receptor's total dose.
[d] Sum of percents may be greater than 100 due to rounding.
[e] Dose is attributable to europium-152, europium-154, europium-155, cobalt-60 and cesium-137.
[f] Dose of 6E-2 rem from cobalt-60 and 3E-2 rem from cesium-137.
[g] Dose of 3E+3 rem from thorium-232.
[h] Dose of 2E+4 rem from thorium-232.
[i] Includes ingestion of fruits and vegetables, milk, and meat.
[j] Calculated using the one-hit equation for high carcinogenic risk levels.

Table 6.15. Radionuclide risk estimated for the on-WAG homesteader with no waste exhumed[a]

Pathway	Current day homesteader risk[b]	Future day homesteader risk[b]
External exposure	8E-3[c]	8E-5[d]
Inhalation of tritium	5E-8	2E-11
Ingestion of meat and milk	1E-4	2E-5
Ingestion of groundwater	6E-4	3E-4
Total	9E-3	4E-4

[a] Potential exposure pathways are for the on-WAG homesteader which does not have wastes exhumed and mixed in the garden and yard.

[b] Excess probability of an individual developing cancer calculated using the linear low dose cancer equation.

[c] Value is calculated from on-WAG worker's external exposure pathway risk adjusted for a homesteader living on-WAG 365 d/yr, 8 h/d for 30 years.

[d] Estimated from the current day homesteader's risk from the external exposure pathway, and decreasing the risk by 1E-2 based on the decrease in the on-WAG homesteader's risk from the external exposure pathway when the waste is exhumed (Appendix 6D).

Table 6.16. Comparison of risks for the on-WAG homesteader when waste is and is not exhumed

Period	Risk to on-WAG homesteader with wastes exhumed[a]	Risk to on-WAG homesteader with wastes not exhumed[a]
Current	1[b]	9E-3[b]
Future	3E-1[c]	4E-4[c]

[a] Excess probability of an individual developing cancer.

[b] Calculated using the linear low-dose equation.

[c] Calculated using the one-hit equation.

Table 6.17. Chemical risk for the on-WAG homesteader

Pathway	WAG-wide, adult		WAG-wide, child		Auger hole area, adult		Auger hole area, child	
	Risk[a]	Percent[b,c]	Risk[a]	Percent[b,c]	Risk[a]	Percent[b,c]	Risk[a]	Percent[b,c]
Ingestion of soil and sediment	2E-10	<1	4E-10	<1	3E-11	<1	6E-11	<1
Vapor phase and particulate inhalation	7E-8	<1	4E-8	<1	7E-8	<1	4E-8	<1
Vapor phase inhalation while showering	3E-5[d]	9	3E-5[d]	18	1E-4	36	1E-4[e]	52
Ingestion of groundwater	3E-4[f]	87	1E-4[g]	82	2E-4[h]	61	1E-4[i]	43
Ingestion of food[j]	1E-6	<1	1E-6	<1	9E-9	<1	7E-9	<1
Dermal contact with groundwater	8E-6	3	3E-6[k]	2	6E-6	2	2E-6	1
Dermal contact with soil and sediment	8E-8	<1	6E-8	<1	7E-8	<1	3E-8	<1
Total	3E-4		2E-4		3E-4		2E-4	

[a] Excess probability of an individual developing cancer.

[b] Risk attributable to the specific pathway divided by receptor's total risk.

[c] Sum of percents may be greater than 100 due to rounding.

[d] Risk changes to 4E-5 when "J-flagged" chemicals are included in calculations.

[e] Risk of 5.3E-5 from carbon tetrachloride and 3.4E-5 from trichloroethene.

[f] Risk of 2.3E-4 from vinyl chloride.

[g] Risk of 1.2E-4 from vinyl chloride. Total risk changes to 2E-4 when "J-flagged" chemicals are included in calculations.

[h] Risk of 1.2E-4 from carbon tetrachloride and 4.8E-5 from trichloroethene.

[i] Risk of 5.9E-5 from carbon tetrachloride and 2.5E-5 from trichloroethene.

[j] Includes ingestion of fruits and vegetables, milk, and meat.

[k] Risk changes to 4E-6 when "J-flagged" chemicals are included in calculations.

Table 6.18. Carcinogenic risk to the on-WAG homesteader from chemicals in the EWB

Pathway[a]	Adult		Child	
	Risk	Percent	Risk	Percent
Incidental ingestion of sediment	3E-11	< 1	6E-11	< 1
Dermal contact with sediment	7E-8	> 99	3E-8	> 99
Pathway Total	7E-8		3E-8	

[a] Surface water samples were analyzed, but no chemical contaminants were detected.

Table 6.19. Noncarcinogenic effects for the on-WAG homesteader

Pathway	Auger hole area, adult[a]		Auger hole area, child[a]		WAG-wide, child[b]	
	Hazard index	Percent[c,d]	Hazard index	Percent[c,d]	Hazard index	Percent[c,d]
Ingestion of soil and sediment	3E-7	<1	3E-6	<1	9E-2	6
Ingestion of groundwater	3E+0[e]	97	7E+0[f]	97	8E-1[g]	60
Ingestion of food[h]	2E-4	<1	7E-4	<1	1E-1	9
Dermal contact with groundwater	1E-1	3	2E-1	2	2E-2	1
Dermal contact with soil and sediment	8E-4	<1	1E-3	<1	3E-1	22

[a] Critical effect is to the liver.

[b] Critical effect is to the skin.

[c] Critical effect hazard index attributable to the subject pathway divided by the receptor's total critical effect hazard index.

[d] Sum of percents may be greater than 100 due to rounding.

[e] Hazard index from carbon tetrachloride.

[f] Hazard index from carbon tetrachloride.

[g] Hazard index of 6E-1 from arsenic and 2E-1 from silver.

[h] Includes ingestion of fruits and vegetables, milk, and meat.

Table 6.20. Noncarcinogenic effects to the on-WAG homesteader from chemicals in the EWB

	Adult		Child	
Pathway[a]	Hazard Index	Percent	Hazard Index	Percent
Incidental ingestion of sediment	1E-4	1	1E-3	4
Dermal contact with sediment	2E-2	99	3E-2	96
Pathway Total	2E-2		3E-2	

[a] Surface water samples were analyzed, but no chemical contaminants were detected.

Table 6.21. Qualitative assessment of uncertainty factors specific to the WAG 6 base line human health evaluation

	Effect of uncertainty		
Uncertainty factor	May over-estimate risk	May under-estimate risk	Either
Selection of contaminants of potential concern			
Inventory information:			
Reliability of radionuclide inventory information (missing information estimated from existing inventory)			X
Assumption that inventory represents waste disposed of	X		
Sampling and analysis:			
Variability in sampling and analysis			X
Representativeness of field samples (could not directly sample source areas)		X	
Nonexistent chemical inventory information (field data used to represent source data)		X	
Possible loss of contaminant VOCs in media sample		X	
Use of detection limit for undetected results	X		
Statistical Method:			
Use of a 0.05 level of significance for identifying contaminants of concern			X

Table 6.21. (continued)

Uncertainty factor	Effect of uncertainty May over-estimate risk	 May under-estimate risk	 Either
Elimination of chemicals from the list of potential contaminants of concern when concentrations were comparable with background concentrations		X	
Inclusion of all chemical volatiles as contaminants of potential concern (even if not statistically significant)		X	
Exposure assessment			
Receptors and Exposure Pathways:			
Radioactive waste exhumed and mixed in the garden and yard; indistinguishable from native soil	X		
Hypothetical exposure assumptions (do not reflect actual characteristics of the exposed population)	X		
Assumption of residential development on the WAG (current conditions at the site preclude homesteading)	X		
Assumption that hunter ingests meat from deer that have drunk contaminated water exclusively from WAG 6	X		
Assumption of lifetime exposure duration for the boundary receptor	X		

Table 6.21. (continued)

Uncertainty factor	Effect of uncertainty May over-estimate risk	 May under-estimate risk	 Either
Conservative assumptions for exposure time and frequency for the on-WAG ORNL employee	X		
Assumption that off-WAG homesteader irrigates his garden and yard and supplies his cows with contaminated water from the Clinch River	X		
Concentrations of contaminants in media:			
Quantification of the maximum annual average air contaminant concentrations	X		
Nonuniform distribution of radionuclide concentrations in the wastes			X
Assumption that chemical concentrations remain constant over time	X		
Use of k_{ow} values in estimating concentrations of contaminants in biota	X		
Analysis of total metals only (metals assumed in their most toxic form)	X		
Estimating intake:			
Use of upper bound values to estimate reasonable maximum exposures for many intake factors	X		

Table 6.21. (continued)

Uncertainty factor	Effect of uncertainty May over-estimate risk	May under-estimate risk	Either
Toxicity Assessment			
Carcinogens:			
Derivation of slope factors and reference doses primarily from laboratory studies; extrapolation of data from high doses to low doses and from one species to another			X
Predicted effects from low doses of radiation (not as well known as effects from high doses)			X
Noncarcinogens:			
Use of chronic-exposure toxicity values	X		
Lack of toxicity values for all chemicals (led to exclusion of such chemicals from risk characterization)		X	
Application of uncertainty and modifying factors to reference doses from NOAEL or LOAEL			X
Unavailability of dermal toxicity factors (used oral toxicity factors, which are administered not absorbed)	X		
Risk Characterization			
Carcinogens:			
Estimation of risk to the on-WAG ORNL employee and boundary from gamma radiation using a radiation shielding model (assumes clean soil over waste disposal area)			X

Table 6.21. (continued)

Uncertainty factor	Effect of uncertainty May over-estimate risk	 May under-estimate risk	 Either
Assumption that carcinogenic risks are additive for multiple contaminants and pathways for the same receptor and exposure period (risks may not be additive due to synergistic or antagonistic effects)			X
Noncarcinogens:			
Assumption that noncarcinogenic hazard indices are additive by effect for multiple chemicals and pathways for the same receptor and exposure period (may not be additive due to synergistic or antagonistic effects)			X
Use of hazard quotient (HQ) values for multiple effects in calculating the hazard index (HI) for each effect (e.g., the HQ for effects to the kidney and liver is considered twice, once as part of the HI for the kidney and again as part of the HI for the liver, with no attempt to segregate the effects)	X		

Section 6 Figures

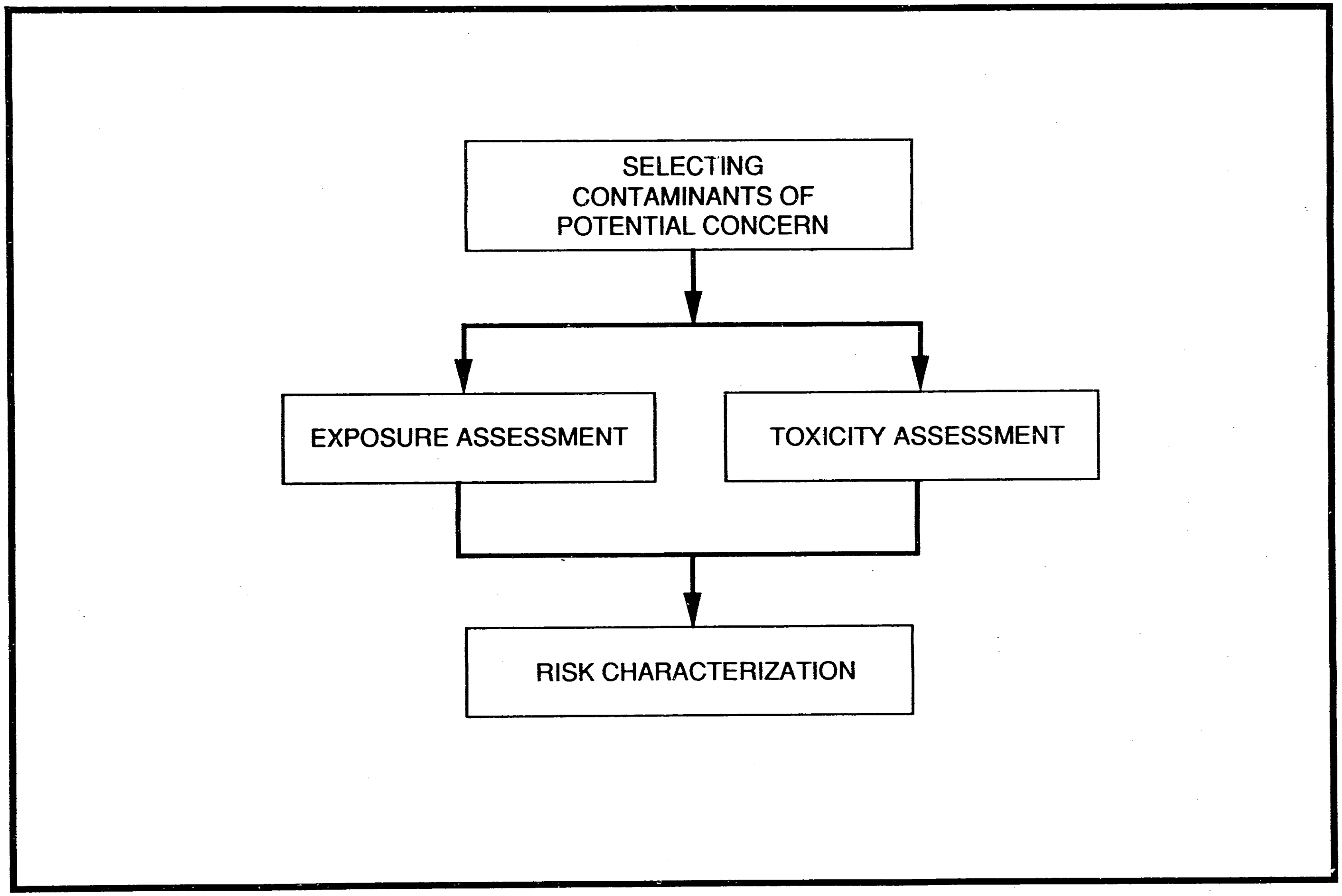


Fig. 6.1. Steps in the human health evaluation process.

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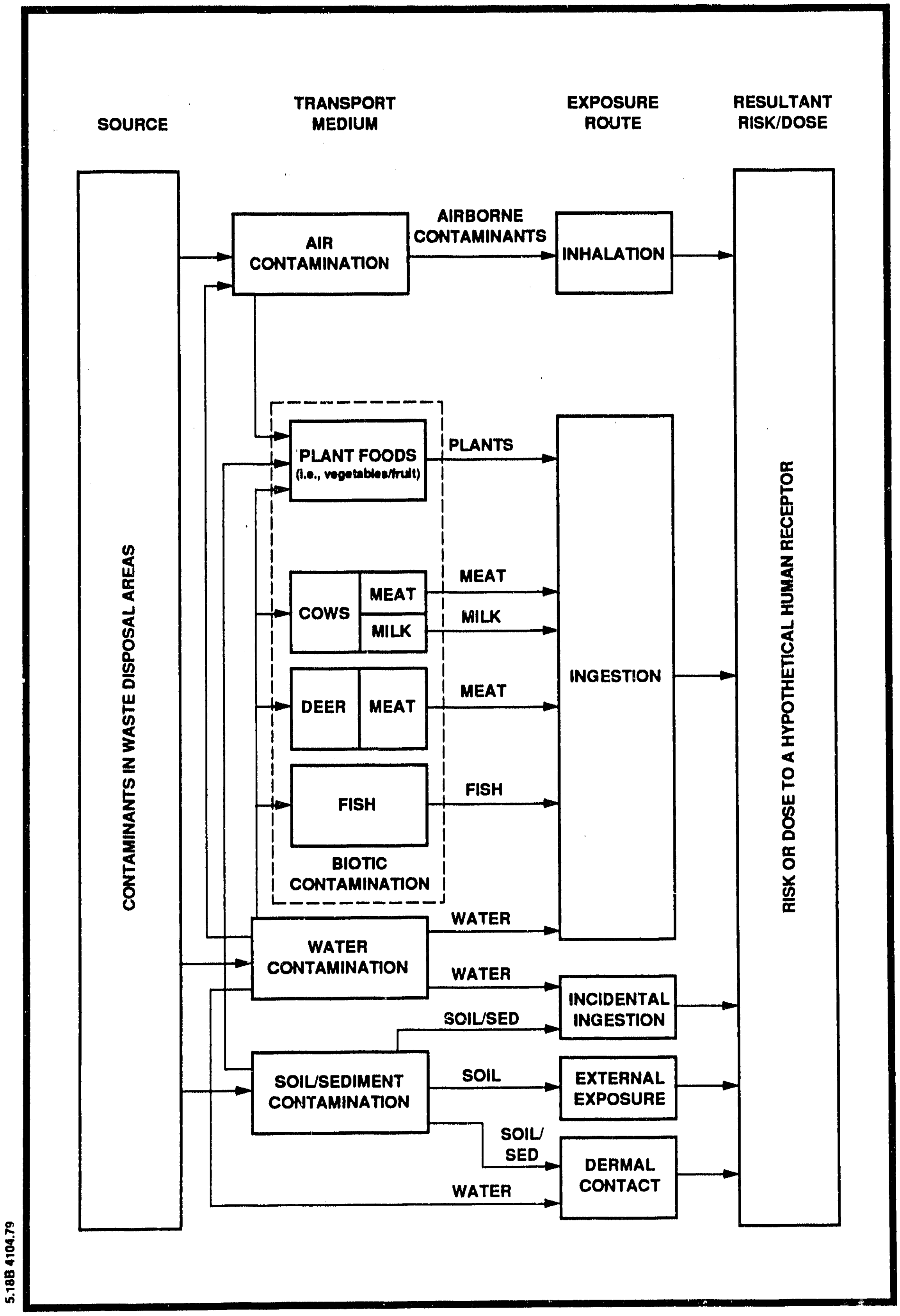

Fig. 6.2. Flow diagram of exposure pathways for WAG 6.

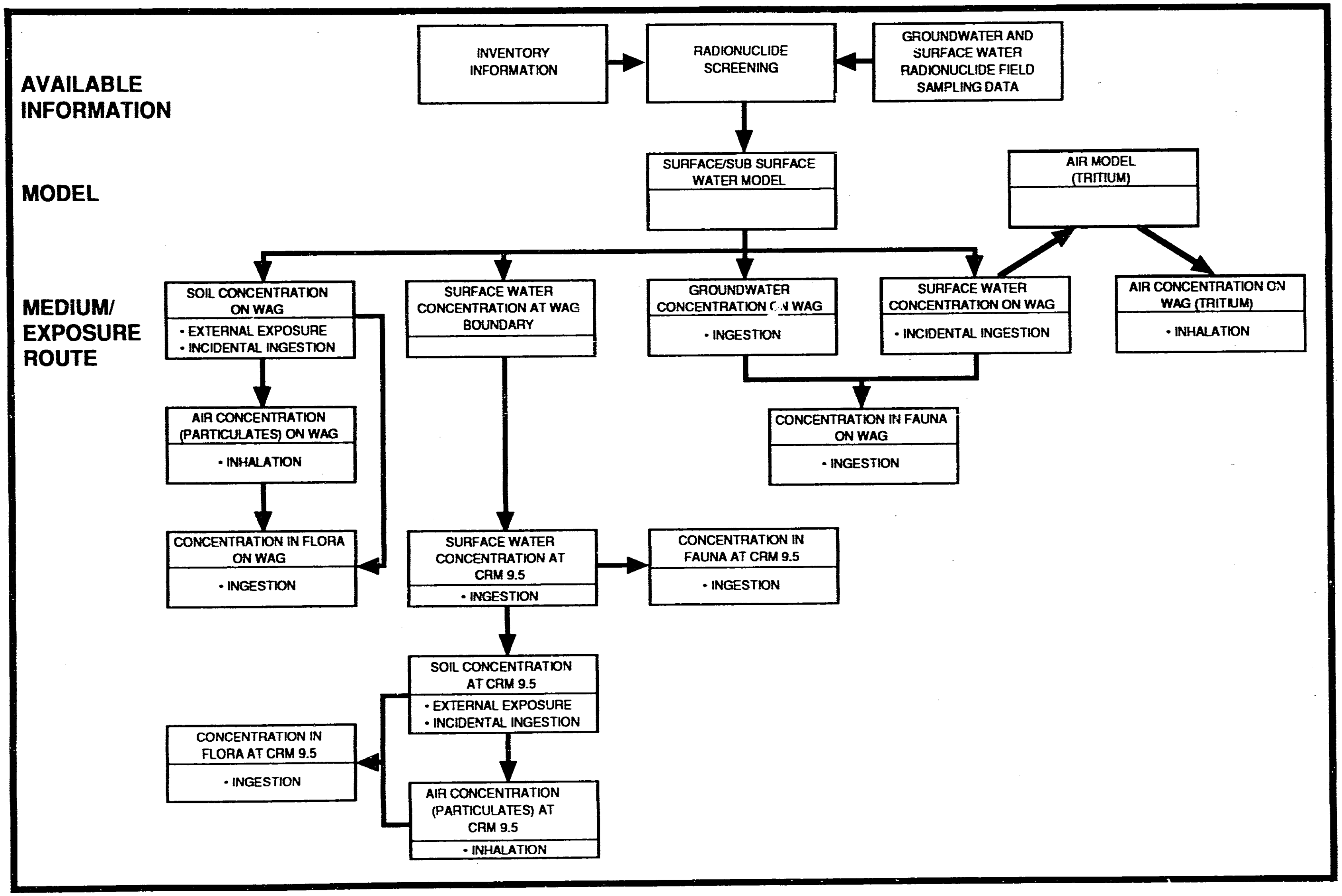

Fig. 6.3. Process used to estimate media concentrations of radionuclides for various routes of exposure.

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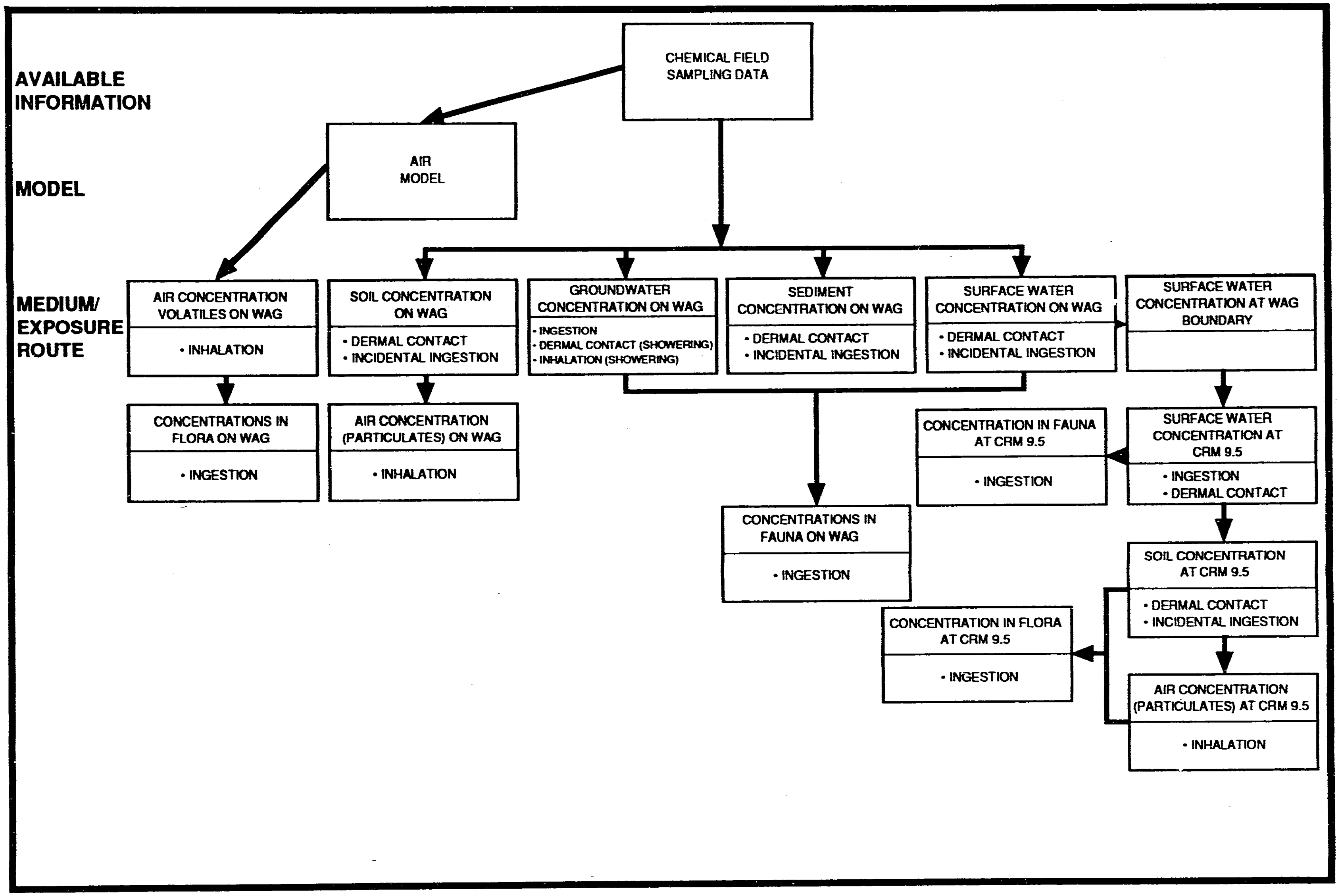


Fig. 6.4. Process used to estimate media concentrations of chemicals for various routes of exposure.

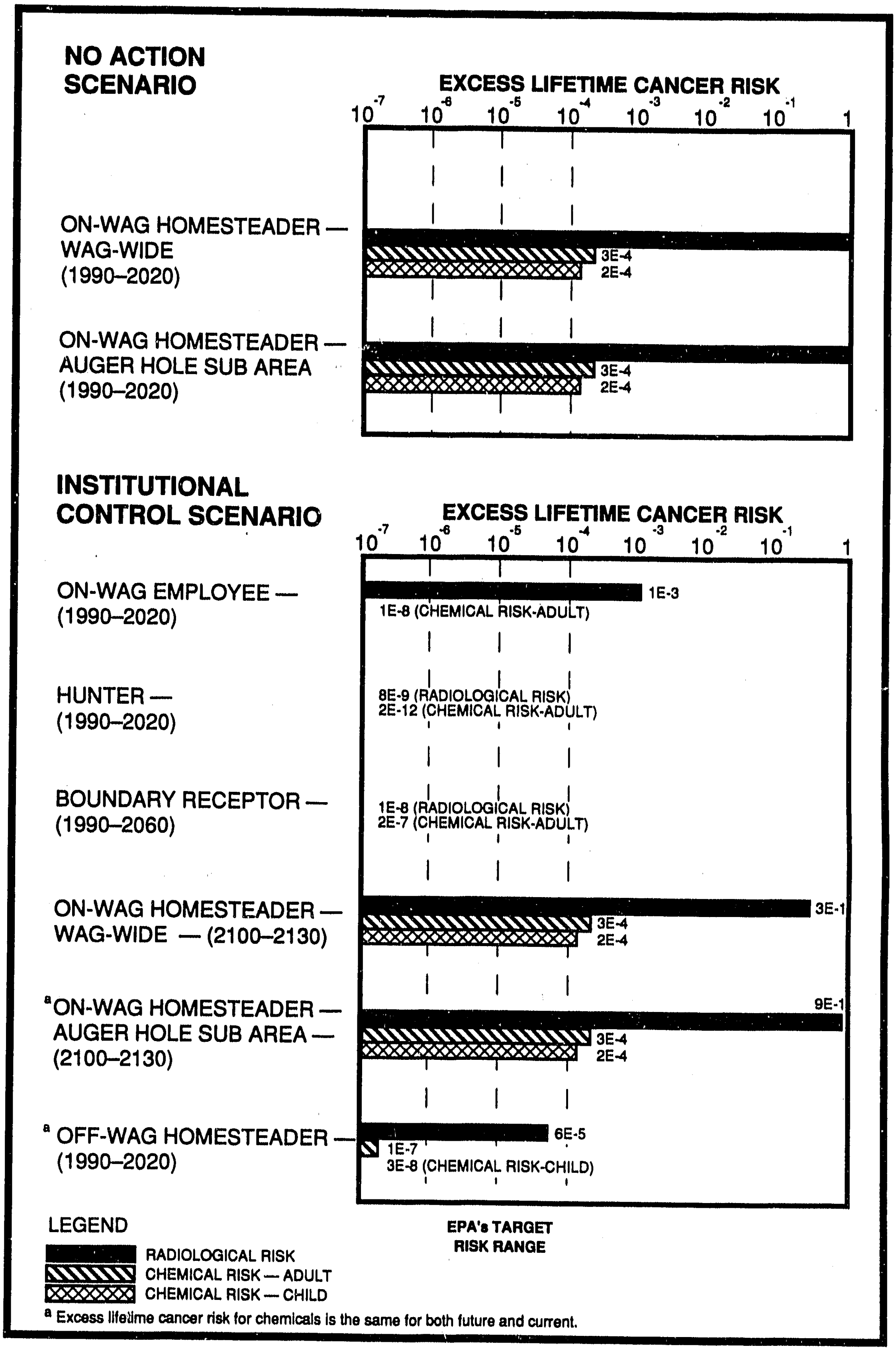
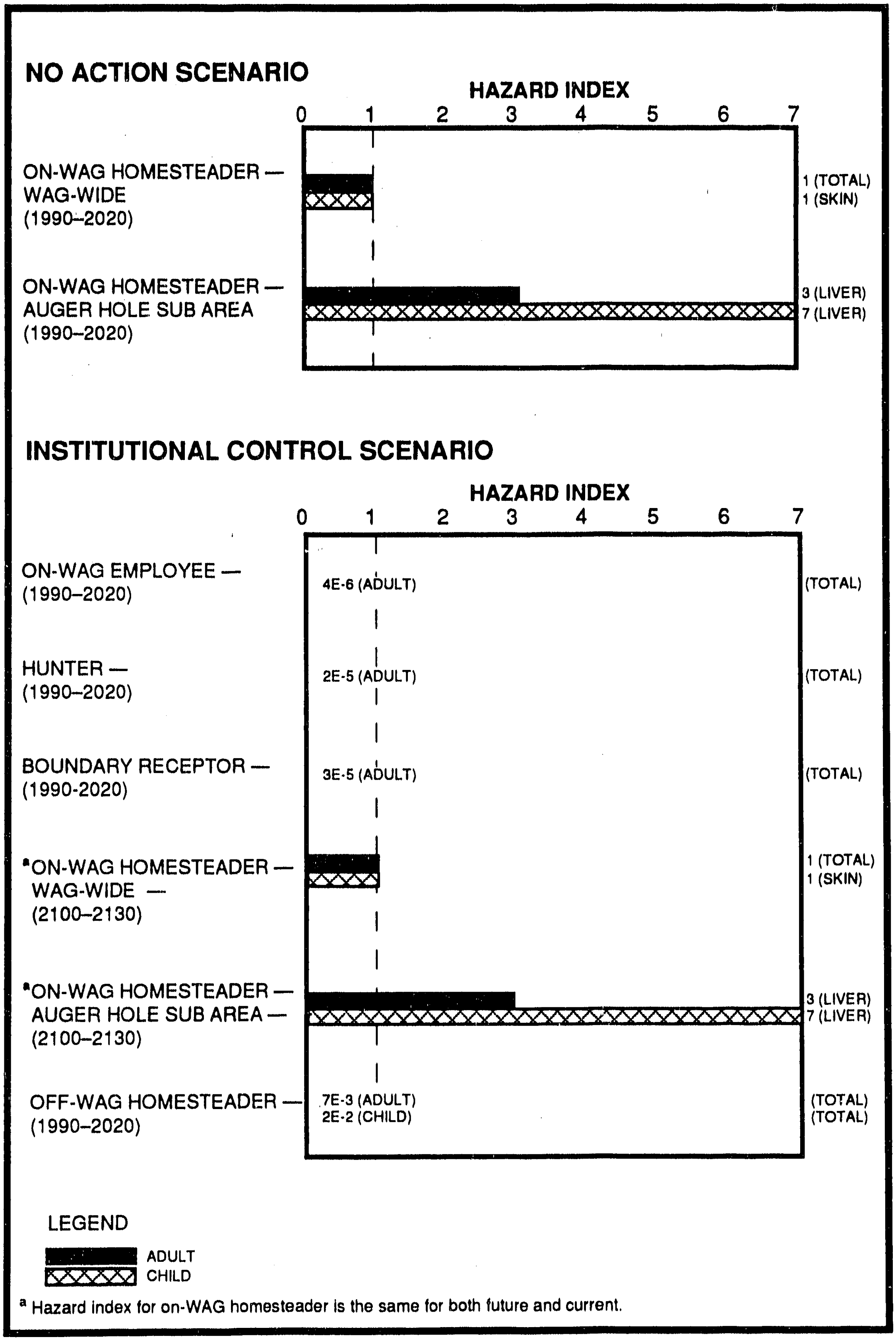


Fig. 6.5. Summary of carcinogenic risk values.



[a] Hazard index for on-WAG homesteader is the same for both future and current.

Fig. 6.6. Summary of noncarcinogenic effects.

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7. BASE LINE ENVIRONMENTAL EVALUATION

7.1 INTRODUCTION

Protection of wildlife, fisheries, threatened and endangered species, and important habitats is required by CERCLA as amended by SARA and other statutes. An environmental evaluation, as the second component of the base line risk assessment, is the avenue through which the type and magnitude of risks to ecological receptors associated with a given site are assessed. Guidelines for conducting this environmental evaluation are provided in EPA 1989c.

Three considerations influenced the methodology employed in this environmental evaluation. First, because large areas of WAG 6 have been cleared for waste disposal, very little natural wildlife habitat remains. Wildlife that might be expected to occupy cleared land has been discouraged from doing so by regular mowing and site operations. Second, due to the large inventory and volume of wastes at WAG 6, the probability of detrimental environmental effects under a no-action scenario is presumed likely. Third, the human health evaluation (Sect. 6) has previously demonstrated the need for continued control of the site as well as the need for progress toward site closure and remediation.

Based on these considerations, the objective of the environmental evaluation was not to characterize all potential impacts to the flora and fauna on WAG 6, but rather to use selected representative target species and a limited number of contaminants to demonstrate that there is a present or potential risk of detrimental effects to the environment in the absence of any remediation under the no-action scenario.

7.2 ORGANIZATION OF THE BASE LINE ENVIRONMENTAL EVALUATION

Section 7.3 presents the methodology and approach used for the base line environmental evaluation. Section 7.4 discusses potentially affected habitats and exposed populations in the environmental setting, and Sects. 7.5 through 7.8 summarize the likely impacts to terrestrial flora and fauna and aquatic vertebrates and invertebrates. Section 7.9 presents a summary of these likely impacts, and Sect. 7.10 discusses the limitations of this evaluation. The results of the ORNL ESD survey of threatened and endangered species are contained in Appendix 7A. Tables detailing the toxicity of the selected contaminants on aquatic species are contained in Appendix 7B.

7.3 METHODOLOGY AND APPROACH FOR WAG 6

Aquatic and terrestrial surveys were conducted for threatened and endangered species, and a wetlands survey was conducted. (No threatened and endangered species were identified, and no wetland community development was found.) Based on knowledge of the environment surrounding WAG 6, hypothetical target species were selected to evaluate potential impacts under a no action scenario. For each of the hypothetical target species, exposure pathways were identified and contaminants of potential concern were selected. For each target species, impacts were estimated to the extent feasible given the available data. The evaluation estimated potential effects on the WAG 6 environment under hypothetical circumstances where fences were down and animals were free to move off- and on-WAG at will.

An environmental evaluation can be either a qualitative or quantitative appraisal of the actual or potential effects of a hazardous waste site on plants and animals other than humans and domesticated species. This environmental evaluation is primarily qualitative because of a lack of quantitative information. Therefore, it was necessary to make inferences and extrapolations from similar species and other locations. For example, results of a laboratory study using white rats might serve as the basis for assuming similar responses in wild rodents.

With the exception of the Federal Ambient Water Quality Criteria, the regulatory standards to which the contaminant levels are compared are those for the protection of human health and not the environment. It was necessary to use these because of a lack of specific standards for the protection of terrestrial flora and fauna. However, in using these standards, the following assumptions were required: (1) nonhuman species will receive the same time of exposure and the same dose as humans, and (2) nonhuman species exhibit the same sensitivity to the contaminants as do humans.

With more than 1500 vascular plants and 250 vertebrate animal species present on the ORR, it was impractical to consider all exposure pathways or potential receptors. Therefore, representative exposure pathways were selected that included different aquatic and terrestrial trophic levels that were composed of species likely to be found in the WAG 6 environment.

This evaluation encompassed an extensive review of available literature, which included examination of hundreds of bibliographic entries and the subsequent in-depth review of approximately 400 scientific reports of special relevance. This literature review encompassed several tiers: (1) information specific to WAG 6, (2) information from the ORR, (3) studies conducted at other DOE facilities, (4) studies from other contaminated sites, and (5) general scientific literature.

Because this environmental evaluation was limited by a lack of site-specific quantifiable data and toxicological data, assumptions and extrapolations were made using data available in the literature. Generally, these data in the literature were derived using laboratory animals or agricultural crop species. Plants and animals in the natural environment tend to have longer exposures than those in the laboratory due to life span, multiple exposure pathways, and differences in metabolism. Therefore, a conservative approach was taken for this environmental evaluation.

Certain specific factors influenced this assessment, most importantly, that WAG 6 is:

- A highly disturbed site with limited natural habitat
- An active waste disposal site
- An active center for environmental restoration research and development operations

Because of these factors, the collection of biota and analysis of contaminant body burdens were not included as part of the environmental evaluation for WAG 6. Therefore, the general approach to this evaluation was to conduct a review of pertinent literature, conduct aquatic and terrestrial field surveys for threatened and endangered species, determine representative exposure pathways for target species, select contaminants of concern, delineate expected responses based upon contaminant levels, and characterize the likely environmental risk impacts. This evaluation is ultimately an integrated evaluation of historical, chemical, analytical, environmental, and toxicological data. Each step in the environmental evaluation is biased toward conservative estimations using maximum concentrations.

7.3.1 Hypothetical Target Species

Several factors influenced the selection of the target species for this environmental evaluation for WAG 6. Species were chosen to represent different trophic levels and various exposure pathways. The tulip poplar was selected to represent terrestrial flora because it is ubiquitous on the ORR, it is commonly found in disturbed areas such as WAG 6, and it has been used in studies of radioisotope uptake, accumulation, and recycling. The white-tailed deer is a large herbivorous mammal that is not only common on the ORR but is also subject to managed hunts on portions of the reservation. They are excellent biological monitors, inhabit both upland and bottomland forests, and have a remarkably adaptable diet (Wren 1986; Kitchings and Mann 1976). The red-tailed hawk is a carnivorous raptor that does not supplement its diet with any vegetation. The lower trophic levels in this food chain are represented by Eastern cottontail rabbits, which are herbivores, and shorttail shrews, which are small, burrowing, carnivorous mammals feeding on earthworms and insects. The raccoon is omnivorous, consuming vegetation and both aquatic and terrestrial prey. For the purpose of this environmental evaluation, its prey is considered to be the terrestrial, herbivorous vole and aquatic organisms such as crayfish and fish. Besides selecting species to represent different trophic levels and various exposure pathways, another important factor in choosing these target species was the availability of relevant information in the literature regarding effects of the selected contaminants on the same or similar organisms. It was also important that at least some portion of WAG 6 have appropriate habitat for these species to live or hunt.

Bluegill sunfish and fathead minnows were selected as key species representing aquatic vertebrates because both are common in the WOC watershed and because there is substantial literature on the toxicity of contaminants for these two species. Benthic macroinvertebrates are relatively good integrators of contaminant releases into aquatic ecosystems because of their abundance, relatively sedentary nature, close association with bottom substrates, and lifespans of a few weeks to a year or more. For assessing potential toxicity of contaminants at WAG 6, four orders of aquatic insects were emphasized:

mayflies (Ephemeroptera), caddisflies (Trichoptera), stoneflies (Plecoptera), and true flies (Diptera). Two crustaceans and aquatic worms were also selected because of their relative abundance in tributaries at WAG 6. The crustaceans are relatively sensitive to metal and organic contaminants, whereas many oligochaetes are relatively resistant.

7.3.2 Contaminants Considered in the Environmental Evaluation

Approximately 65 contaminants of concern were identified for WAG 6, but a reduced number was selected for this environmental evaluation. Factors influencing the selection of contaminants were: (1) their known or suspected potential to cause ecological effects, (2) their known or suspected levels based on site characterization data collected during the RFI and/or historical records about quantities disposed there, (3) regulatory standards and guidelines, and (4) the availability of information on uptake and potential adverse effects in the literature for the target species or similar species, as well as the species from lower trophic levels that compose the food chains selected for this environmental evaluation.

Results from RFI sampling events provided recent information about the contaminants in the media and at the locations sampled. Because source areas are expected to release contaminants unpredictably at different times and in different areas of the site in the future, including contaminants not yet detected in environmental media, data presented in management records and earlier characterization studies were considered.

For terrestrial species, three radionuclides (strontium-90, cesium-137, and tritium), four VOCs (trichloroethene, tetrachloroethene, toluene, xylene), and two metals (cadmium and chromium) were selected. For aquatic species, a reduced number of contaminants was considered: two radionuclides (strontium-90 and cesium-137), two VOCs (toluene and xylene), and two metals (cadmium and copper). Concentrations of contaminants considered in this evaluation included those detected during site sampling. As with the human health evaluation, the environmental evaluation assumed a steady-state condition for chemicals. The chemical concentrations in the environment were presumed constant over time so that risk from exposure to chemicals would not change.

Radionuclides present potential problems to ecosystems because they can bioaccumulate in target tissues, causing deleterious effects, as well as become sources of internal and external ionizing radiation, causing mutagenic, teratogenic, and carcinogenic effects. Of the three radionuclides selected for this environmental evaluation, maximum concentrations of strontium-90 exceeded 10,000 pCi/L, which is many orders of magnitude greater than the SDWA MCL of 8 pCi/L. As a metallic analog of calcium, this contaminant is a bone-seeking radionuclide that is known to cause leukemia and bone tumors. Its adverse biological effects have been studied by many researchers, thereby providing information in the scientific literature about effects likely to occur to the flora and fauna inhabiting WAG 6 in the absence of any remediation under the no-action scenario. Cesium-137 is an analog of potassium and affects soft tissue, especially liver and kidney. It is not as mobile in the environment as strontium-90 or tritium, becoming bound to illitic clays where it is ingested by earthworms or small burrowing animals. These animals can also serve as an important pathway for contaminant transport to the surface at disposal sites. Cesium-137 was detected in soil/sediment and surface water samples from WAG 6, and there is information in the literature to support its transport through the ecosystem. Tritium does not bioaccumulate, quickly reaches equilibrium in organisms, and has a short half-life. Its potential for ecological harm results from ionizing radiation. Tritium was ubiquitous at WAG 6, reaching concentrations as high as 14,000,000 pCi/L in surface water.

Cadmium and chromium were selected as metal contaminants for WAG 6. Cadmium, which has a strong affinity for soft body tissues, can produce toxic effects at sufficient concentrations. Maximum concentrations detected at WAG 6 were 30 mg/kg in surface water, compared to a SDWA MCL of 5 mg/kg. There are species differences in accumulation related to diet and trophic level. Shrews (one of the prey species in the food chain culminating in the red-tailed hawk) have been shown to bioconcentrate cadmium in liver and kidney tissue 15 to 33 times greater than in the surrounding soil. Chromium primarily occurs in the natural environment in either the trivalent or hexavalent state, with the hexavalent state being much more damaging to biota because it easily crosses biological membranes. Chromium is mutagenic, teratogenic, and carcinogenic to a variety of organisms. Chromium was present at WAG 6 in levels far exceeding the Federal Ambient Water Quality Criteria for the protection of freshwater organisms. Copper is a required minor nutrient for both plants and animals; however, at concentrations over a few parts per billion, it can be toxic to aquatic plants and animals. Copper was present in surface water at WAG 6 in concentrations above the Federal Ambient Water Quality Criteria, and there is information in the literature about its toxic effects to the aquatic species selected for this environmental evaluation.

The third category of contaminants considered for this environmental evaluation was VOCs. Tetrachloroethene and trichloroethene both exceeded the SDWA MCLs, and xylene and toluene are known to have been disposed there in large quantities. All are listed as hazardous substances according to federal statutes. Studies on humans and laboratory animals have shown that these contaminants have the potential to produce toxic effects on components of the ecosystem.

7.3.3 Selection of Exposure Pathways

Exposure pathways incorporate the affected media, potential routes of exposure, and the ecological receptors.

Certain criteria were used in selecting exposure pathways for the environmental evaluation. First, the target species selected represent different trophic levels and various exposure pathways. Second, it was essential that relevant information be available regarding effects of the selected contaminants of concern on the selected organisms.

The media affected by WAG 6 disposal units were determined to be groundwater, surface water, sediments, soil, and, to a much lesser extent, air. The evaluation assumed that the tulip poplar grew on WAG 6 and that the raccoon and deer had established home ranges on-WAG. The evaluation also assumed that the hawk consumed prey on-WAG and that the fathead minnow, bluegill, and benthic macroinvertebrates were found inhabiting WAG 6 tributaries or the EWB.

7.4 ENVIRONMENTAL SETTING

7.4.1 Potentially Affected Habitats

Several WAG 6 habitats have the potential to be affected by site contamination. WAG 6 consists primarily of fields planted in fescue and second growth forests along drainages. Both the fields and wooded areas are habitats subject to contamination by discharge of contaminated groundwater to surface water and by surface runoff. Another mode of transport that will affect open areas through soil displacement is animal burrowing. Arthur and Markham (1983) reported that small burrowing mammals displaced annually about 27 pounds of subterranean soil to the surface at an 89-acre solid radioactive waste disposal area. This mode of transport is of particular importance to WAG 6 because of the potential for shrews, voles, and groundhogs inhabiting the site.

The aquatic habitat at WAG 6 is limited to the EWB and two intermittent streams, Drainages DB and FA, draining the western and east-central portions of the area. These streams are discussed in Sect. 3.4.1 and shown in Fig. 3.1. They drain directly into WOL. Runoff is seasonal, with highest flows in the winter. (See Sect. 3.1.2.1.) Investigations at WAG 6 have determined that leaching of contaminants from disposal areas and subsequent surface or shallow subsurface flow represent the primary mode of contaminant transport to the WAG 6 streams (BNI Team 1990).

Drainage DE lacks adequate habitat for fish (Ryon 1990); although Drainage FA contains pools, undercut banks, and pool-riffle sections suitable for minnows native to the area, no fish were found during an electroshocking survey (Ryon 1990). Both drainages support benthic macroinvertebrate communities, at least in sections containing water much of the year. The EWB reportedly has never been used to store wastes; thus, its muddy bottom should support macroinvertebrates. During an electroshocking survey of the EWB in September 1990, five bass were collected (Ryon 1991).

7.4.2 Potentially Exposed Populations

Under a no action scenario, WAG 6 would provide habitats for a diversity of terrestrial and aquatic flora and fauna. Thirty-seven species of mammals, 151 species of birds, and 61 species of reptiles and amphibians have been observed on the ORR (Kitchings and Mann 1976; Boyle et. al. 1982). It is assumed that most of these species could be present at some time on WAG 6, if only temporarily. The bird, mammal, reptile, and amphibian species found on the ORR are listed in Appendix 3E.

In the WAG 6 wooded areas, the plant populations that may be exposed to contamination are the mixed mesophytic association of trees, including such predominant overstory species as oak (*Quercus* spp.), tulip poplar (*Liriodendron tulipifera*), beech (*Fagus grandiflora*), sugar maple (*Acer saccharum*), red maple (*Acer rubrum*), magnolia (*Magnolia* spp.), buckeye (*Aesculus octandra*), and basswood (*Tilia americana*). The understory species include dogwood (*Cornus florida*), red maple, black cherry (*Prunus sylvatica*), and sweet gum (*Liquidamber styraciflua*). Examples of herbaceous species growing in association with the upland hardwood forests found on WAG 6 are greenbriar (*Smilax* spp.), honeysuckle (*Lonicera japonica*), bracken fern (*Pteridium aquilinum*), and strawberry bush (*Euonymus* spp.). In the maintained areas, the predominant grass cover is fescue (*Festuca elatior*).

The populations of fauna that may inhabit the forested areas on WAG 6 include small mammals, such as the white-footed mouse (*Peromyscus leucopus*), chipmunk (*Tamias striatus*), vole (*Microtus pinetorum*), golden mouse (*Ochrotomys nuttalli*), shorttail shrew (*Blarina brevicauda*), and house mouse (*Mus musculus*). They provide prey for predators such as raccoons (*Procyon lotor*), red and grey foxes (*Vulpes fulva* and *Urocyon cinereoargenteus*, respectively), striped skunks (*Mephitis mephitis*), bobcats (*Lynx rufus*), and weasels (*Mustela frenata*). White-tailed deer (*Odocoileus virginianus*) also inhabit the forested areas and graze along the edges of fields where food is plentiful and tree cover is available.

Different bird species prefer different habitats, and the WAG 6 wooded areas and their edges provide many habitat types that can support the avian species presumed to be present. Examples of species of birds that prefer habitats in the wooded areas are yellow-shafted flickers (*Colantes auratus*) red-bellied woodpeckers (*Centurus carolinus*), hairy and downy woodpeckers (*Dendrocopes villosus* and *D. pubescens*, respectively), blue jays (*Cyanocitta cristata*), Kentucky warblers (*Oporonis formosus*), and ovenbirds (*Seiurus aurocapillus*). Species that prefer the shrubby forest edges are brown thrashers (*Toxostoma rutum*), rufous-sided towhees (*Pipilo erythrophthalmus*), and Carolina wrens (*Thryothorus ludovicianus*). Predatory birds that may utilize the woodlands for nesting and hunting are the red-tailed hawk (*Buteo jamaicensis*), broad-winged hawk (*Buteo platypterus*), great horned owl (*Bubo virginianus*), screech owl (*Otus asio*), and barred owl (*Strix varia*).

According to Kitchings and Mann (1976), there are several taxonomic groups of forest arthropods on the ORR that might be affected by WAG 6 site contamination: spiders (Aranoe), centipedes (Chilopoda), beetles (Coleoptera), millipedes (Diplopoda), flies and mosquitos (Diptera), bees and ants (Hymenoptera), butterflies (Lepidoptera), and grasshoppers (Orthoptera). According to Johnson (1964), anurans (frogs and toads) and most of the reptiles are found in old field areas on the ORR. Reptiles and amphibians that may inhabit WAG 6 forested areas include frogs (*Bufo* spp.), salamanders (*Eurycea* spp.), eastern box turtles (*Terrapene carolina*), northern copperheads (*Agkistrodon contortix*), timber rattlesnakes (*Crotalus horridus*), black rat snakes (*Elaphe obsoleta*), and fence lizards (*Sceloporus undulatus*).

In the grassy areas, the mammal species are similar to those inhabiting old field areas, whether the vegetative cover is tree seedlings and shrubs or fescue grass such as that covering portions of WAG 6. Small mammals on these grassy fields include cotton rats (*Sigmodon hispidus*), white-footed mice, shorttail shrews, and eastern harvest mice (*Reithrodontomys humulis*). Another larger mammal that may inhabit the expanse of WAG 6 fields is the groundhog (*Marmota monax*). White-tailed deer also graze the fields. Use of field areas by bird species is based on the presence of shrubs and low-growing herbaceous plants that thrive on the edges of fields and forests. These herbaceous species provide food and nesting areas for birds such as sparrows, towhees, blue grosbeaks (*Guiraca caerulea*), and brown thrashers. The maintained fields in WAG 6 may be used for courtship displays and breeding purposes by some birds such as quail (*Colinus virginianus*) or woodcocks (*Philohela minor*). Raptors may also hunt in those areas.

Fish common in the upper reaches of the WOC watershed primarily include minnows (Cyprinidae) and sunfish (Centrarchidae). Creek chubs (*Semotilus atromaculatus*) and blacknose dace (*Rhinichthys atratulus*) are two common native minnows. Fathead minnows (*Pimephales promelas*), a widely introduced species, are also common in WOC. All three species inhabit small streams with pool-riffle areas and spawn during the summer. The fathead minnow is more tolerant of turbidity, high temperatures, and low dissolved oxygen than the chub or dace (Pflieger 1975, Clay 1975). Several species of sunfish also inhabit the WOC watershed, including the redbreast sunfish (*Lepomis auritus*), largemouth bass (*Micropterus salmoides*), and bluegill sunfish (*L. macrochirus*).

The smaller headwater tributaries of the WOC watershed contain a wide variety of benthic macroinvertebrates, predominantly insect larvae, crustaceans, oligochaetes, and molluscs. During a May 1990 ESD survey of the DB and FA drainages in WAG 6, nine orders of Insecta, four orders of Crustacea, molluscs, and aquatic worms (Oligochaeta) were collected. In Drainage DB, oligochaetes, the isopod *Lirceus* (Crustacea), and several dipteran groups accounted for most of the benthic organisms collected (Smith 1990). In Drainage FA, the isopod *Lirceus* accounted for 62 to 92% of the total population at the three stations sampled. The amphipod *Crangonyx* and stonefly *Amphineura* were abundant at one station.

In conjunction with the WAG 6 RFI environmental evaluation, surveys for threatened and endangered species and sensitive environments were conducted in WAG 6. Results of these surveys are provided in Appendix 7A. The fish survey consisted of qualitative electrofishing sampling and assessing the presence of potential fish habitat. No fish were found in the tributaries within the SWSA 6 boundaries, possibly due to the intermittent nature of the streams and their isolation. Drainage DB did not contain sufficient quality habitat to maintain fish populations; however, the two branches of Drainage FA did contain suitable habitat for species such as creek chubs and black nose dace. Electrofishing at the EWB, as cited earlier, produced five bass (*Micropterus* spp.).

Surveys of macroinvertebrates revealed no threatened or endangered species within WAG 6. Two relatively undisturbed reference streams were also sampled: upper Mitchell Branch, located east of the Oak Ridge K-25 site, and a small, unnamed tributary of upper Melton Branch, located east-southeast of the High Flux Isotope Reactor at ORNL (Smith 1991). The most notable difference between samples collected from streams within WAG 6 and those from reference areas was the low relative abundance or absence of Ephemeroptera, Plecoptera, and Trichoptera (EPT) from WAG 6 sites. Because most EPT taxa are very sensitive to adverse changes in water quality, this lack of abundance or absence is an indication that streams in WAG 6 are impacted. The most obvious cause of impact is from past physical disturbances (e.g., stream channel modification, vegetation removal/alteration, and siltation), although degraded water quality due to contaminants from waste disposal sites in WAG 6 cannot be excluded as a possible cause for the observed impacts (Smith 1991). Survey results and maps showing study sites and reference areas are presented in Appendix 7A.

A brief survey was conducted for threatened and endangered bird and mammal species at WAG 6, and none were found. Thirty-three bird and one mammal species were actually observed, although many other species probably occur. The site also appears to be relatively unimportant for species listed by the State of Tennessee as threatened, endangered, or in need of management (e.g., Cooper's hawk, sharp-shinned hawk, black vulture, and red-shouldered hawk). These species may occasionally hunt for prey at the site, but SWSA 6 does not appear to have suitable habitat for nesting (Kroodsma 1990).

A survey was conducted for rare plant species known to exist on the ORR as well as any unknown taxa, but no rare species were found (Cunningham 1990a). Similarly, a survey for wetland communities within the fenced area revealed a few wetland species (Scirpus and Carex) but essentially no wetland community development (Cunningham 1990b).

7.4.3 Potential Future Setting

Under a no-action scenario with all fences removed, WAG 6 will experience the ecological successional stages that are typical for the temperate climate and mixed mesophytic conditions that exist on the ORR. The currently maintained grassy fields will gradually succeed into a stage that is dominated by a thick growth of shrubs such as blackberry and vines of honeysuckle, smilax, and Virginia creeper. Seedlings of red maple, yellow poplar, oaks, and pines such as Virginia pine and shortleaf pine will begin to grow in open areas among the vines, shrubs, and other vegetation. Small mammals such as mice, voles, shorttail shrews, and rabbits will likely inhabit this early stage of succession. Raccoons will use the intermittent streams and other wet areas for food. Foxes and skunks will hunt for prey there, and predatory birds such as screech owls and red-tailed hawks will also likely hunt there. Other birds likely to be found there are Eastern phoebes, Carolina wrens, common yellowthroats, yellow-breasted chats, brown-headed cowbirds, field sparrows, white-throated sparrows, and song sparrows (Hamel et al. 1982).

Yellow poplars are a pioneer species that rapidly inhabit a disturbed site and grow quickly to shade out competing species. In the potential future setting of WAG 6, as yellow poplar seedlings develop into saplings, they will soon dominate the site, shading out species for which sunlight is a critical factor. In old field situations, it is not uncommon in East Tennessee to find pure stands of even-aged yellow poplar trees with a sparse understory composed of dogwood, sourwood, red maple, redbud, and small oaks and hickories. Small mammals will continue to inhabit this young forest and be preyed upon by skunks, weasels, foxes, and hawks and owls.

As this predominantly yellow poplar forest matures, slower-growing, shade-tolerant species of oak and hickory will begin to dominate. Their large crowns will gradually shade out the yellow poplars. Their expanding boles will begin to require more of the available nutrients and moisture. The mature yellow poplars will begin to die, creating more space for the slow-growing oaks and hickories. An oak-hickory climax forest is typical for disturbed, upland sites, and this forest will dominate until natural or human events occur that may eliminate this forest or change its composition.

Mammals that may inhabit this type of forest are mice, chipmunks, shrews, voles, foxes, skunks, bobcats, weasels, and white-tailed deer. Avian species will likely include wild turkeys, ruffed grouse, owls, hawks, woodpeckers, Eastern pewees, bluejays, chickadees, nuthatches, Carolina wrens, wood thrushes, red-eyed vireos, and ovenbirds (Hamel et al. 1982).

Succession in the intermittent streams in WAG 6 will probably not develop because of their ephemeral nature, and the EWB will likely never support large populations of aquatic life because of its limited size.

7.5 TERRESTRIAL FLORA—TULIP POPLAR

7.5.1 Exposure Pathways—Tulip Poplar

The tulip poplar tree is endemic to Tennessee and is common throughout the eastern United States. It is a fast-growing pioneer species that quickly inhabits a disturbed site, such as an old field, and matures rapidly to full height (60 ft) and diameter (3 to 4 ft) (Brockman 1968). It dominates the forest canopy until shaded out by slower growing, shade-tolerant species such as oaks, hickories, and hemlocks

As shown in Fig. 7.1, the four routes of contamination to a tulip poplar tree growing on WAG 6 are assimilation from the soil, gas exchange with the atmosphere, bioaccumulation from surface water, and surface deposition (IT Corp. 1990). Important pathways of transfer to the forest floor from tree boles are rain leaching and passage through the roots. In trees tagged with cesium-134, translocation of the contaminant was very rapid (Witherspoon 1964).

7.5.2 Exposure and Toxicity Assessments—Tulip Poplar

Much of the information for this evaluation was obtained from a literature review. The majority of data available for the selected contaminants of WAG 6 flora characterize uptake, accumulation, transport, distribution, and recycling throughout trees in a compartmentalized fashion (Witherspoon, Auerbach, and Olson 1962; Witherspoon 1964; Witkamp and F 1964; Waller and Olson 1967). Dr. Glenn Suter II from ORNL provided sources of phytotoxicity data, and several authorities were questioned about effects data or specific toxicity data that could be applied to an analysis of contamination of WAG 6. Those people included the following: Dr. Frank McCormick, Ecology Program, University of Tennessee; J. Thomas Kitchings and Keith Owenby, ERCE, Denver, Colorado; Jackie Cunningham, ORNL; Dr. Liz Etnier, ORNL; C.T. Garten, ORNL; Ann West CH2M HILL, Fairfax, Virginia; Charles Murphy, SRL, Aiken, South Carolina; Elizabeth Deming, Agricultural Engineering, University of Tennessee; Diane Beeman, U.S. Fish and Wildlife Service, Raleigh, North Carolina; Earl Leming, Tennessee Division of Solid Waste Management; and Dr. Art Newell, New York Department of Environmental Conservation, Stony Brook, New York.

One of the WAG 6 selected contaminants, cesium-137, was used extensively in early studies on the ORR to trace the route of contamination in tulip poplar trees (Waller and Olson 1967). Tulip poplar trees tagged with cesium-137 showed that the important paths of transfer to the forest floor are rain leaching and passage through the roots. Cesium-137 moves rapidly throughout the tree with 40% of the inoculant found in the litter, soil, and smaller root components 5 months after initial injection into the trees. The maximum burden (27%) was found in the canopy about 4 weeks after inoculation, and this was reduced to 12.5% 3 months later. Of the maximum burden, the remaining two-thirds moved back into woody tissue (stem and roots) before leaf fall. Losses of radiocesium from the foliage and stems through litterfall and rain leaching accounted for almost 8% of total cesium-137 introduced into the tree boles.

When white oak trees were inoculated with cesium-134, loss of the radionuclide began with the first rains after inoculation. However, the cesium-134 transferred from the trees via annual leaf fall was two times greater than that exported by rain. Translocation of cesium-134 through trees, litter, and roots was found to be very rapid, and 92% of the total soil cesium-134 remained in the top 4 in. of soil under inoculated trees 19 months after inoculation (Witherspoon 1964).

In studies on agricultural crops, cesium-137 was found in greater concentrations in sandy soil than in loamy or Chernozem soils (Kabata-Pendias and Pendias 1984). This is because cesium-137 released into the atmosphere becomes strongly adsorbed by clay minerals and also by organic matter soils (Kabata-Pendias and Pendias 1984). Illitic clays such as those on the ORR have been shown to reduce the bioavailability of cesium-137 because it becomes tightly bound to the clay fraction of the soil (Fredriksson and Ericksson 1958; Crossley, Reichle, and Edward 1971). Dahlman and Van Voris (1976) measured plant:soil ratios of 0.03 where cesium-137 concentrations were in excess of 20,000 pCi/g in soil. Crossley and Reichle (1969) found a similar concentration factor of 0.027 for cesium-137.

Strontium-90, another WAG 6 contaminant of concern, is found in WAG 6 soils in amounts exceeding background levels for soils (25.5 pCi/g maximum concentration; 0.70-0.80 pCi/g background level; see Table 7.2), for surface water (10,222 pCi/L maximum concentration; 8 pCi/L is the MCL; see Table 7.2), and for groundwater (5400 pCi/L; 8 is the MCL; see Table 7.2). It is one of the most important fission products that can be readily taken up by plants, according to Oakes et al. (1982) and Kabata-Pendias and Pendias (1984). It is a bone-seeking radionuclide and poses potential harm to the organisms that consume contaminated plants. According to Kabata-Pendias and Pendias (1984), there are few reports of strontium toxicity in plants, and plants vary in their tolerance to this element. Shacklette et al. gave the toxic strontium level for plants as 30 ppm AW (ashed weight). However, there were no levels given for strontium-90, the contaminant of concern. There are data from nearby SWSA 5 that indicate that the maximum concentration of strontium-90 in herbaceous vegetation is 90 nCi/g and averaged as high as 30 and 19 nCi/g in honeysuckle and blackberry shoots, respectively, at two seeps (Garten and Lomax 1987). The significance of these levels will be discussed in Sect. 7.6.1.3.

The influence of soils on plant uptake of radionuclides has been demonstrated throughout the literature. Baes, Garten, and Tolbert (1987) reviewed existing literature on the plant:soil concentration ratios of various radionuclides and determined that the range of reference mean values for the plant:soil concentration ratio of strontium-90 was 0.077 to 17, with a geometric mean of 2.7. Variability in the ratio arises from regional variation in soil parameters affecting strontium-90 uptake by plants. The soil property that most influences the plant uptake of strontium-90 is the exchangeable calcium status (Francis 1978). For some soils, other properties such as pH, texture, and organic matter dominate strontium-90 uptake processes. WAG 6 soils are generally acidic and heavily reworked. Because the site is an active waste disposal site, native soil properties have been altered by the importation of other soils as backfill. Kabata-Pendias and Pendias (1984) recognize that strontium-90 as a pollutant is easily mobile in light soils and, therefore, is readily taken up by plants. Although the maximum level for strontium-90 on WAG 6 is 25.50 pCi/g (more than 24 pCi/g above background levels), the maximum level for strontium-90 in soils in WAG 6 is 25.5 pCi/g (more than 24 pCi/g above background levels). This converts to 1.8 x 10^{-5} ppm. Strontium is toxic to plants at 30 ppm. Therefore, toxic effects on plants are highly unlikely (Suter 1991). The cause for concern is in the high levels of strontium-90 in groundwater, surface water, and soil in the levels found in vegetation in nearby SWSA 5.

Tritium is the third radionuclide of concern on WAG 6. There are numerous potential modes of tritium uptake by organisms (Elwood 1971). Plants can obtain tritium from both foliar deposition and the transpirational process. The initial source of tritium exposure is primarily tritiated water, although exposure to tritiated vapor does occur. Uptake by plants in the transpirational stream is quite rapid. In one study, tritium was detected in transpired water from small oak (*Quercus* spp.) trees 4 h after tritiated water was placed in the soil (Woods and O'Neal 1965). In another study, tritium was detected in plant water collected from all parts of the plants including leaves, stems, and roots after exposure of leaves to tritiated vapor for 5 h. The highest concentrations measured were in the exposed leaves (Elwood 1971). Although tritium is taken up by terrestrial vegetation, it apparently does not bioaccumulate in the environment or in food chains (Rohwer and Wilcox 1976).

Cadmium and chromium are the heavy metal contaminants selected for WAG 6. Both metals have been studied extensively, however, the biological behavior and environmental cycling of both, particularly chromium, are poorly characterized. In an exposure assessment of terrestrial flora, cadmium is primarily associated with the soil. It is a metal whose activity is strongly affected by soil pH, and it is most mobile in acidic soils within the range of pH 4.5 to 5.5 (Kabata-Pendias and Pendias 1984). A shifting of soil pH toward the acidic range not only results in greater plant uptake of cadmium, but also results in more acute phytotoxic symptoms (Lepp 1981). Under man-induced conditions, cadmium is likely to be built up in surface soils (Lepp 1981).

A study on uptake and movement of cadmium was conducted by Witherspoon and Gaskins (1972) in which the trunk of a cedar tree (*Juniperis virginianus*) was inoculated with cadmium, and the resultant activity was determined in the various parts of the tree and in the understory vegetation, litter, and soil. About 65% of the cadmium was transported to the crown. Over an 8-week period, only 0.1% of the cadmium in the crown was leached out to the forest floor, whereas about 68% remained in the litter under the canopy. Uptake by the understory vegetation was low, about 3%, and the remaining 29% entered the soil, where most of it remained in the top 2 cm.

Studies on legumes, spinach, radish, carrots, and oats demonstrate that effects of cadmium toxicity are brown leaf margins; chlorosis; reddish veins and petioles; curled leaves; and brown, stunted roots (Kabata-Pendias and Pendias 1984). Brooks (1972) reports that the general toxicity of cadmium to plants is moderate, and Allaway (1968) notes that, in general, 3 ppm cadmium in the tissue of plants depressed growth. The metal is a cumulative poison with the greatest concentration found in leaves, then fruit, then seeds (Lepp 1981).

Chromium is a transitional element that occurs in two environmentally important oxidation states: trivalent and hexavalent. Trivalent chromium does not migrate significantly and thus should only be a problem at the source; conversely, hexavalent chromium migrates quite readily. Furthermore, trivalent chromium is poorly absorbed by biota and is considered biologically insignificant, but hexavalent compounds are toxic when ingested, absorbed, or inhaled. Hexavalent chromium is generally not available for incorporation into plant tissues (Taylor 1983; Kabata-Pendias and Pendias 1984). Only 1% of the chromium deposition mass was incorporated into foliage, with the remaining fraction being transferred to the soil-litter compartment by weathering, plant fragmentations, or direct deposition. Approximately 5% of the hexavalent chromium reaching the soil compartment is available to vegetation by root uptake. The transport of chromium up through roots is slow, accounting for low levels in the tops of plants (Lepp 1981). Toxic effects of chromium on vegetation are chlorosis of new leaves and injured root growth (Kabata-Pendias and Pendias 1984). The majority (94%) is unavailable for incorporation into plant tissues (Taylor 1983), and chronic deposition or elemental pools do not pose a serious threat for the terrestrial ecosystem because hexavalent chromium is essentially nonbioavailable.

Other WAG 6 selected contaminants that are of significant concentrations to potentially impact target species are the volatile organic chemicals: xylene, toluene, tetrachloroethene, and trichloroethene. However, no data were found in the literature from which to draw conclusions or inferences about effects of these contaminants on vegetation.

7.5.3 Evaluation of Potential Impacts—Tulip Poplar

Of the three radionuclides selected for analysis in this environmental evaluation, it is possible that strontium-90 will have adverse impacts on vegetation growing on WAG 6. Cesium-137 is tightly bound to the clay fraction in the soil and is not expected to produce effects; tritium does not bioaccumulate in the environment. However, strontium-90 is found at very high levels in groundwater on WAG 6, in surface water, and in soil (5400 pCi/L, 10,000 pCi/L, and 25 pCi/g, respectively). It is also readily taken up by vegetation, particularly by tulip poplars that grow rapidly and require well-drained soils from which they can derive a lot of moisture. It is conceivable that although toxic effects from strontium-90 at 30 ppm in soil are highly unlikely (Suter 1991), the combination of high concentrations in soil, groundwater, and surface water may cause adverse impacts on trees growing in areas of these concentrations.

Depending on soil pH, cadmium may impact tulip poplars on WAG 6. It is found in soils and sediments on WAG 6 at 4 ppm; however, that level is close to background levels for soils and sediments (Table 7.2). Kabata-Pendias and Pendias (1984) report that "phytotoxically excessive levels in surface soils" for cadmium fall in the range of 3 to 8 ppm. Hence, the WAG 6 soils may be marginally phytotoxic due to cadmium (Suter 1991). On more acidic soils, tulip poplars may take up cadmium in greater concentrations and at a faster rate than on alkaline soils, thereby accumulating the metal in its foliage. Chromium, found in concentrations as high as 42.4 ppm in soils on WAG 6, may accumulate in the roots of vegetation, but will be essentially unavailable for incorporation into plant tissues (Taylor 1983). Gough et al. (1979) report that in soil, 1,370-2,740 ppm chromium caused chlorosis in citrus; 10 to 15 ppm chromium (as $K_2Cr_2O_7$) in the nutrient solution was toxic to barley; 150 ppm in soil was toxic to orange seedlings; 16 ppm (chromate) reduced growth in tomatoes, oats, kale, and potatoes whereas 10 ppm was toxic to corn and 5 ppm was toxic to tobacco. In corn leaves, 4 to 8 ppm (dry weight) chromium was toxic. Based on these agricultural data, there are indications that chromium may have adverse effects on plants at certain levels.

It is not known what effects volatile organic chemicals will have on a tulip poplar growing on this WAG.

7.6 TERRESTRIAL FAUNA

Three target species were selected to describe the potential impacts to terrestrial fauna on WAG 6: white-tailed deer, red-tailed hawk, and raccoon. In addition to these target species, likely effects upon the lower elements in their food chains are also discussed. Although there are three routes of exposure for each species, only one (ingestion) can be addressed because no data exist for the other exposure pathways (inhalation and direct contact). (See Figs. 7.2 through 7.4.) As discussed in Sect. 7.3.1, these target species were chosen to represent different trophic levels and exposure pathways.

7.6.1 White-Tailed Deer

7.6.1.1 Exposure pathways—white-tailed deer

The white-tailed deer (*Odocoileus virginianus*) is a common large mammal in Tennessee. Weighing an average of 50 kg, deer are excellent biological monitors according to Wren (1986), because they are widely distributed and abundant. Deer inhabit upland and bottomland forests and have a remarkably adaptable diet (Kitchings and Mann 1976). They are completely herbivorous, consuming approximately 3500 g of vegetation daily, and they eat according to seasonal availability (IT Corp. 1990). (See Table 7.1.)

In spring, white-tailed deer eat green leaves and stems from both woody and herbaceous species, fruit of the hawthorne, tulip poplar flowers, mushrooms, and acorns. Acorns dominate the deer's diet in autumn, followed by honeysuckle, mushrooms, grapes, apples, sumac, and blueberry. In winter, they eat acorns, grasses, honeysuckle, and mushrooms—all of which are subject to contamination from soil assimilation, gas exchange with the atmosphere, bioaccumulation from surface water, and surface deposition (IT Corp. 1990). Deer also typically consume 3000 mL of water and 5 g of soil, both of which are sources of additional uptake of contaminants. The white-tailed deer was selected for the WAG 6 environmental evaluation because it is a large herbivorous mammal and is common throughout the ORR and the WAG 6 environs.

As shown in Fig. 7.2, there are three routes of contamination to a white-tailed deer inhabiting WAG 6. They are inhalation, ingestion, and direct exposure (IT Corp. 1990). A deer is directly exposed to contamination when it brushes against vegetation, when it lies on the ground or paws the dirt to create beds, walks through water, or brushes against vegetation that holds water. A deer is also directly exposed to the air around it. Of the three routes of contamination, ingestion is the only route for which there are data available for characterization.

7.6.1.2 Exposure and toxicity assessments—white-tailed deer

Cesium-137 is not readily taken up by plants because it is strongly attracted by clay minerals and organic matter soils (Kabata-Pendias and Pendias 1984). Therefore, its toxic effects from ingested vegetation on white-tailed deer on WAG 6 will be negligible. Deer consume soil incidentally as they graze or browse, creating another source of contamination from cesium-137. However, because of the small amounts of soil consumed, cesium-137 likely does not pose a threat to the deer through the ingestion of soil.

According to Amano and Garten (1991), plants can absorb tritium from the soil and the atmosphere. If the source of tritium is atmospheric, then it is absorbed by plant leaves by diffusional processes. If the source of tritium to the plant is soil water, plants will absorb tritium and transpire it into the atmosphere. Tritium was found in high concentrations on WAG 6—4,300,000 pCi/L in groundwater and 14,000,000 pCi/L in surface water. (See Table 7.2.) Tritium concentrations in soil water at SWSA 5 range from 370 to 14,000 Bq/L during the growing season. Monthly changes in groundwater levels and soil water tritium concentrations indicate that tritium at this site migrates upward to the surface soil from the subsurface due to rise in groundwater levels during the wet season (Amano, Garten, and Lomax 1987). Tritium in the surface soil gives rise to tritium concentrations in the atmosphere by evaporation of surface soil water. Under conditions of chronic exposure to tritium as either liquid or vapor, an equilibrium concentration between body-water tritium and environmental tritium is rapidly attained (Elwood 1971). Deer killed at the Savannah River Plant site were found to have generally the same tritium concentrations in both their body water and tissue-bound components. Even with the high concentrations of tritium in WAG 6 groundwater and surface water, likely effects from tritium on a white-tailed deer on WAG 6 will come from ionizing radiation and not from bioaccumulation.

Strontium-90 is readily taken up by plants, and as early as the late 1950s, it was recognized as a potential accumulator in terrestrial ecosystem components of the WOL bed (DeSelm and Shanks 1963). Strontium-90 concentrations on the WOL bed were higher in leaves than in stems and higher in forbs and woody plants than in herbaceous plants. DeSelm and Shanks estimated that approximately 7% and 2% of the soil strontium was removed by lake bed willows and herbaceous vegetation, respectively. Much of this was recycled into the topsoil as accumulated litter.

Concentrations of strontium-90 were found on SWSA 5 at levels of 30 nCi/g n honeysuckle and 19 nCi/g in blackberry shoots. The maximum concentration in herbaceous vegetation was 90 nCi/g. According to Garten and Lomax (1987), the strontium-90 level in deer bone could easily exceed the TWRA confiscation limit of 30 pCi/g after browsing times of 1 week to 1 year. Although the standard was set to protect human consumers of deer taken from the ORR from the toxic effects of strontium-90, the argument can be made, based on effects data of strontium-90 on bone, that 30 pCi/g of strontium-90 in deer bone pose potential risk to deer.

Garten and Lomax (1987) report that there are few empirical data from which to derive representative plant:soil concentration ratios for strontium-90 in the WOC watershed. A plant:soil concentration ratio of approximately 1.5 is representative of strontium-90 uptake by herbaceous plants from the WOL bed and WOC floodplain. The ratio may be greater for forbs and woody browse vegetation growing on more acidic soils in the watershed.

Cadmium contamination of a deer will occur through the route of ingestion of vegetation or surface water. Cadmium is primarily associated with soil pH: the more acidic the soil, the more mobile the metal. It may accumulate in vegetation on WAG 6 and, therefore, may be ingested by deer. A deer could also consume cadmium found in surface water. Levels of cadmium were reported at 30 μg/L (5 μg/L is the MCL) in surface water on WAG 6. (See Table 7.2.) In a study of Japanese serows, base line data on cadmium accumulation in their organs and tissues showed that the mean cadmium concentration in muscle, liver, kidneys, and the whole body of fetuses was 1.0 ng/g; in fawns was 18, 29, 60, and 22 ng/g, respectively; in yearlings was 9, 51, 60, and 31 ng/g, respectively; in adults (2.5 years to 10 years) was 16, 130, 950, and 42 ng/g, respectively; and adults (10 years to 17.5 years) was 20, 131, 1080, and 52 ng/g. The body burdens of fetuses were low (<1%) compared with those of their mothers. The placental transfer of cadmium was negligible; after birth, cadmium levels increased each year. There was a significant difference in cadmium concentrations between collection locations, which was attributed to direct intake of cadmium in the soil (National Library of Medicine 1991).

Chromium will also be ingested by white-tailed deer eating vegetation and drinking surface water. It tends to accumulate in roots of plants, leaving the foliage largely unaffected. Chromium was found in surface water on WAG 6 at levels lower (58.3 μg/L) than the SDWA MCL standard of 100 μg/L. (See Table 7.2.)

7.6.1.3 Evaluation of potential impacts—white-tailed deer

Of the selected radionuclides of concern found on WAG 6, strontium-90 is the one most likely to impact a white-tailed deer. It bioaccumulates and is readily taken up by vegetation. Although there are no data on levels of strontium-90 in vegetation on WAG 6, levels of strontium-90 found in vegetation on nearby SWSA 5 are high—maximum concentration 90 nCi/g dry weight (Garten and Lomax 1987). On WAG 6, strontium-90 levels in both soil and surface water are high—a mean of 9.26 pCi/g for soils and a mean of 980.69 pCi/g for surface water compared to a standard of 8 pCi/L (SWDA MCL, see Table 7.2). This mean value was derived from the number of samples found to be contaminated. Given the fact that this contaminant is a bone-seeking radionuclide that is known to cause leukemia and bone tumors, it poses a likely threat to white-tailed deer consuming vegetation and surface water on WAG 6.

Of the remaining selected radionuclide contaminants, cesium-137 is bound to the clay fraction in soils and, therefore, does not pose a threat to deer through ingestion of vegetation. The small amount of soil incidentally consumed by deer also does not likely pose a threat of adverse effects on the animal. Tritium does not bioaccumulate, quickly reaches equilibrium in organisms, and has a short half-life. If it affects a deer on WAG 6 at all, it will be through radiation ionization.

Neither cadmium nor chromium is expected to adversely effect a deer inhabiting WAG 6. Cadmium, though taken up by vegetation, is greatly affected by soil properties, particularly pH. Given appropriate conditions, vegetation could provide a minor pathway for cadmium to the deer. Chromium is not accumulated to any great degree in foliage and is not expected to have adverse effects on deer on WAG 6.

The volatile organic chemicals selected for this evaluation are not expected to adversely effect deer on WAG 6 because of the properties that cause them to evaporate quickly in air and water.

7.6.2 Red-Tailed Hawk

7.6.2.1 Exposure pathways—red-tailed hawk

Red-tailed hawks are at the top of the food chain. Their preferred habitat is open country mixed with woodlands, nesting in the forests but hunting in more open terrain. Unlike some secondary consumers, such as raccoons, red-tailed hawks do not supplement their diet with any herbaceous material. Hawks have many pathways through which they may suffer adverse effects, beginning with lower trophic levels and culminating with the prey that make up the diet of this raptor (Fig. 7.3). The primary prey species selected for this environmental evaluation are rabbits and shrews (IT 1990). The estimated daily intake of food, water, and air for these species is presented in Table 7.1. In addition to ingestion, red-tailed hawks may be exposed to contaminants through inhalation and direct exposure.

Eastern cottontail rabbits, which are prey for hawks, are herbivorous, feeding on a great diversity of plant species. They consume primarily herbaceous species during summer and woody plants during winter. Their preferred habitat is heavy brush, strips of forest with open areas nearby, and weed patches (Chapman and Feldhamer 1982). Vegetation growing in the contaminated areas may bioaccumulate some of the constituents and be ingested by rabbits and other small mammals consumed by hawks. These rabbits have an average life span of 6 to 15 months, with a maximum of 3 years. Their average body weight is from 0.9 to 1.8 kg. Rabbits can be exposed to contaminants through inhalation, ingestion of vegetation and surface water, and direct exposure.

Shorttail shrews, the second major component of the diet for red-tailed hawks for this environmental evaluation, are small insectivorous animals that feed on earthworms, insects, snails, various invertebrates, and possibly young mice (Burt and Grossenheider 1964). They are active both day and night and make their own tunnels (or use those of other animals) in the earth. Their preferred habitat is not restrictive and includes forests, grasslands, marshes, and brushy areas. With a lifespan of 12 to 18 months, their average body weight is from 11 to 22 g. Because of their burrowing habits, shrews can ingest contaminated soil or wastes directly. Other avenues of exposure include ingestion of prey and surface water, inhalation of air, and direct exposure.

Earthworms ingest the soil and then are eaten by shrews, which consume from 7 to 36 of earthworms per day, a major portion of their diet. As cesium-137 becomes tightly bound to the clay fractions of soils, a large percentage of radioactivity accumulated by earthworms is ingested (Crossley, Reichle, and Edwards 1971), creating a potential source of contamination in the exposure pathway to a red-tailed hawk.

7.6.2.2 Exposure and toxicity assessments—red-tailed hawk

Cadmium is a naturally occurring element; however, at sufficient concentrations, it exhibits toxic effects. As a metallic analog of potassium, cadmium has a strong affinity for soft body tissues, especially the kidney and liver, and in general there appears to be a positive relationship between concentrations of cadmium in the soil and in the soft tissues of small mammals. The U.S. Fish and Wildlife Service estimates that adverse effects on fish and wildlife are either pronounced or probable when cadmium concentrations exceed 3 ppb in freshwater, 100 ppb in the diet, or 100 μg cadmium/m^3 in air (Eisler 1985). At low doses the concentration in the kidney is approximately 10 times that in the liver; however, at high exposures, the concentration in the liver may exceed that in the kidneys. The tissue concentration pattern is kidney > liver > muscle > heart > bone > brain. Cadmium residues in vertebrate kidney or liver that exceed 10 ppm fresh weight or 2 ppm whole body fresh weight should be viewed as evidence of cadmium contamination; residues of 200 ppm fresh weight kidney or more than 5 ppm whole animal fresh weight are probably life-threatening to the animal (Eisler 1985).

When compared to aquatic organisms, mammals and birds are relatively resistant to the biocidal properties of cadmium. The lowest oral doses producing death in rats and guinea pigs ranged from 150 to 250 mg cadmium/kg body weight (ppm). Sublethal effects of cadmium include growth retardation, anemia, and testicular damage (Eisler 1985). Teratogenic effects on animals appears to be higher for cadmium than for other metals. Rats subjected to >6 mg cadmium/kg body weight daily during pregnancy produced fetuses with jaw defects, cleft palates, club feet, and pulmonary hyperplasia (Ferm and Layton 1981). Cadmium has also been demonstrated to have mutagenic effects. Mice injected with 3 or 6 mg $CdCl_2$/kg body weight showed changes in chromosome number 12 h later. In assessing the carcinogenic effects of cadmium, laboratory studies have shown that the injection of cadmium metal or salts causes malignancies at the site of injection and testicular tumors.

There are species differences in accumulation related to the diet and trophic level. At a site that was heavily contaminated with cadmium, average whole body concentrations were lowest in a vole (*Microtus agrestis*), an exclusive herbivore, and highest in a shrew (*Sorex araneus*), which feeds primarily on insects, insect larvae, snails, and slugs (Talmage and Walton 1990). At a mine site where concentration of cadmium in the soil was 92 ppb, the whole body concentration in a shrew (*Sorex araneus*) was 40 ppb and in its invertebrate prey was 19 ppb. Total body concentration of cadmium:diet in *S. araneus* was 2.2. At other contaminated sites, concentrations of cadmium in liver tissue of shrews averaged 236 ppb and 280 ppb.

In the environment, cadmium is primarily associated with the soil and apparently does not accumulate in vegetation. It is present in invertebrates such as earthworms and insects, where the concentration factor of soil:invertebrates is approximately 1.

For the shrew, concentration factors for liver and kidney compared to soil or invertebrates ranged from 15 to 33 (Talmage and Walton 1990). Wren (1986) found that while cadmium levels in soft tissues were highly correlated with levels in soil and vegetation, cadmium levels in bone were not related to environmental concentrations. Furthermore, Andrews et al. (1984) postulated that cadmium in insects may be in a more biologically available form than that in vegetation.

Deficient intake of some essential nutrients may increase the absorption rate of cadmium. For example, some essential elements including zinc and copper do not bioaccumulate in mammal tissue relative to environmental levels as do some nonessential elements such as cadmium, lead, and mercury. Moreover, the relative influence of sex and age on metal accumulation and retention is not clear, but only cadmium levels appear to consistently increase with age (Eisler 1985).

The biological implications of metal pollution are not necessarily related directly to the abundance of contaminants but rather to their transfer potential and mobility within terrestrial ecosystems. Chromium, for instance, is most frequently found in the environment in the trivalent (+3) and hexavalent (+6) oxidation states. In general, the toxicity of trivalent chromium to mammals is low because its membrane permeability is poor and it is noncorrosive; furthermore, there is little tendency for trivalent chromium to bioaccumulate in food chains in the organic form (Eisler 1986). Organo-trivalent chromium compounds may have significantly different accumulation tendencies although little is known about these compounds (Steven et al. 1976). Hexavalent chromium is more toxic than the trivalent form, and it easily penetrates biological membranes (Steven et al. 1976, Taylor and Parr 1978, Langard and Norseth 1979, Ecological Analysts 1981). However, Taylor and Parr (1978) found a low assimilation rate and a rapid initial loss of hexavalent chromium by cotton rats, suggesting the nonessential nature of the element and indicating the lack of bioaccumulation.

Data on the environmental cycling of chromium are lacking, but it is generally agreed that suspended particulates are a major source of transport in aquatic systems, that most chromium in soil and sediment is unavailable to living organisms, and that hexavalent chromium in air and water is hazardous to fish and wildlife (Eisler 1986). Furthermore, available evidence suggests that organs and tissues of fish and wildlife that contain >4 mg total chromium/kg dry weight should be viewed as presumptive evidence of chromium contamination (Eisler 1986).

Acute and chronic adverse effects of chromium to warm-blooded animals are caused mainly by hexavalent chromium compounds. The LD-50s for mice were 260 mg/kg body weight for trivalent chromium, but only 5 mg/kg for hexavalent chromium (Steven et al. 1976). For most mammalian experimental animals, including mice, dogs, rabbits, cats, and guinea pigs, the minimum injected fatal dose of hexavalent chromium ranged from 1 to 5 mg/kg body weight, although doses of 0.2 to 0.5 mg/kg body weight produced marked kidney damage.

Hexavalent chromium compounds may cause skin ulceration, irritative dermatitis, ulcerations in mucous membranes, and perforations in the nasal septum. Under laboratory conditions, chromium is mutagenic, carcinogenic, and teratogenic to a wide variety of organisms (Eisler 1986). Teratogenic effects were documented in chicken embryos after eggs were injected with hexavalent chromium. Deformities included short and twisted limbs, microphthalmia, exencephaly, everted viscera, growth stunting, and parrot beaks (Ridgeway and Karnofsky 1952, Gilani and Marano 1979). Although both trivalent chromium and hexavalent chromium accumulated in brain, kidney, and myocardium of rabbits, the accumulation of hexavalent chromium was highest in brain and that of trivalent chromium in the kidney; for both valence states there was no correlation between dose and concentration of stored chromium or extent of tissue damage (Hatherill 1981). In rabbits, both trivalent chromium and hexavalent chromium, given at 1.7 mg/kg body weight daily for 6 weeks, adversely affected blood and serum chemistry and both produced significant morphological changes in liver of rabbits (Tandon et al. 1978). Similar effects were observed in rats (Laj et al. 1984). Kidney and liver lesions in rats were observed when the drinking water contained 134 ppm of hexavalent chromium for 2 to 3 months (Steven et al. 1976). Tissue accumulations were significant in dogs exposed to drinking water concentrations of 11.2 ppm chromium but were nil at 6 ppm.

Studies conducted in India showed that a concentration of 10 to 15 ppm of hexavalent chromium in irrigation water applied to soils was lethal to two species of earthworms in 58 to 60 days (Soni and Abbasi 1981, Abbasi and Soni 1983).

As with metals, radionuclides are transferred through components of ecosystems, often collecting in target tissues. The most significant radionuclide released by the water pathway is strontium-90 because of its radiotoxicity, environmental availability, and the quantities released at ORNL (Ohnesorge 1986). Other significant radionuclides are tritium, cesium-137, and the transuranics. Strontium-90 can be taken up by both plants and animals, accumulating in various plant tissues and in bone tissue in animals. It is a beta-emitting fission product with a relatively long half-life (28 years). As a metallic analog of calcium, it is readily absorbed from the gastrointestinal tract and lungs into the bloodstream and deposited in bone. In experimental animals, chronic ingestion of strontium-90 resulted in leukemia and bone tumors. Kaye and Dunaway (1962) found that concentrations of strontium-90 were higher in teeth and bone of muskrats than in stomach contents, indicating that this radionuclide, as an analog of calcium, is accumulated in tissues to concentrations exceeding those in the diet.

Although tritium also has the potential for marked uptake by terrestrial vegetation, it does not bioaccumulate in the environment or in food chains (Rowher and Wilcox 1976). The transfer coefficient for tritium from soil to vegetation is 0 (Argonne National Laboratory 1989), indicating that this radionuclide is not taken up by the roots of plants. In a study of the kinetics of tritium accumulation in adult rats, Istomina and Moskalev (1972) found that with chronic intake of varying activities, an equilibrium was established for intake, accumulation, and elimination of ^{3}HOH within the organism by day 20. After administration ceased, 98.6 to 99% of the ^{3}HOH was eliminated with a half-life of 3 to 4.3 days.

As a chemical analog of potassium, cesium-137 is deposited in soft tissues such as the liver and kidney. However, in ecosystems it is less mobile than strontium-90 and tritium, becoming bound to illitic clays. Because cesium-137 becomes tightly bound to the clay fraction of the soil, it can be ingested by earthworms, whose daily turnover of soil is equal to 29% of their live weight. In addition, assimilation of radiocesium from litter by earthworms appears to be much lower than for arthropods (Crossley et al. 1971). These earthworms and arthropods may then be consumed by small mammals, adding to the exposure pathways for contaminant transport in the ecosystem.

Small mammals may be an important vector for radioactive transport because of their burrowing activity, mobility, and potential high population numbers. In addition to animals becoming contaminated and transporting radionuclides to the surface, other potential problems are increased water infiltration due to excavations, soil erosion processes, and increased water holding capacity. These could result in increased deterioration of disposed waste containers (Arthur et al. 1986, Arthur and Markham 1983). Ninety-six percent of mice in waste disposal areas in southeastern Idaho had higher radiation doses than did control animals in fall-winter; in spring-summer, 73% of the mice had higher dose rates than the controls. These seasonal differences appear to be related to increased subsurface time in colder weather. Similarly, the autumn-winter dose rates received by kangaroo rats were significantly greater than the spring-summer rate, and this seasonal difference was attributed to the species entering torpor beneath the soil surface during winter (Arthur et al. 1986).

Arthur et al. (1986) found that radiation dose rates received by animals from radioactive waste disposal areas varied from 0.4 to 42,000 mrad/day and were significantly higher than the dose rate of 0.4 mrad/day received by control animals. Doses resulted primarily from external radiation from buried activation and fission products and not fi n internal radionuclides in/on tissues. The highest doses received by small mammals were of sufficient magnitude to cause premature deaths. French et al. (1974) found chronic yearly doses of 2.11 to 3.60 Gy sufficient to increase the death rate. Previous studies on the physiological effects of radiation on mice suggested that acute (0.02 Gy) and chronic (0.001 Gy) exposures caused loss of body weight and a decrease in the weight and size of spleen and adrenal glands.

While higher trophic levels generally have higher contaminant levels, concentrations of radionuclides can also be diluted as trophic level increases. In a study at Idaho National Engineering Laboratory, concentrations of radionuclides were much lower in red-tailed hawks, who inhabit large home ranges, than in their prey items, indicating that while radionuclides were transferred to higher trophic levels, concentration was diluted by consumption of uncontaminated animals (Craig et al. 1979). In fact, Craig, Halford, and Markham (1979) found that the mean concentration of radionuclides in rodents was approximately 4 to 290 times greater than in the raptors that preyed upon them.

Xylene (total) is designated as a hazardous substance under Section 311(b)(2)(A) of the Federal Water Pollution Control Act and further regulated by the Clean Water Act Amendments of 1977 and 1978. Most xylenes are released into the atmosphere where they photochemically degrade by reacting with hydroxyl radicals. Their half-life ranges from 1 to 1.7 h in summer and 10 to 18 h in winter, or a typical loss of 67 to 86% per day. In water, the dominant removal process is volatilization. Xylenes are moderately mobile in the soil and may leach into groundwater where they may persist for years, despite some evidence that they biodegrade in both soil and groundwater (National Library of Medicine 1991).

Xylenes are absorbed primarily through the mucous membranes and pulmonary system, where they are metabolized to a toxic aldehyde, methylbenzaldehyde. However, almost total excretion via the urinary tract occurs within 24 h, and the potential for bioconcentration is expected to be insignificant. The plasma half-life is 4 h, and this rapid clearance from blood prevents adequate biological monitoring of serum. The LD_{50} for ingestion by rats is 4.3 g/kg. The LC_{50} for rat inhalation is 6350 ppm/4 h and for mouse inhalation is 3907 ppm/6 h. In human studies, known toxic effects include fetotoxicity, hepatomegaly, hyperactivity, decreased body weight, increased mortality at higher dosages, central nervous system effects, and nose and throat irritation (HEAST 1990).

Like xylenes, toluene is designated as a hazardous substance under Section 311(b)(2)(A) of the Federal Water Pollution Control Act and further regulated by the Clean Water Act (CWA) Amendments of 1977 and 1978. As a toxic pollutant under Section 307(a)(1) of the CWA, it is subject to effluent limitations. The recommended Ambient Water Quality Criterion to protect freshwater aquatic life is 2300 μg/L as a 24-h average, with a ceiling of 5200 μg/L (National Library of Medicine 1991).

Toluene volatilizes rapidly from water; the expected evaporation half-lives from a river and lake are 1 and 4 days, respectively. Because of its high vapor pressure, toluene is also expected to volatilize fairly rapidly from dry soils. In the atmosphere, toluene degraded by reaction with photochemically produced hydroxyl radicals with a half-life of slightly more than a day and a degradation rate of 41% per day (National Library of Medicine 1991).

Toluene vapor is readily absorbed by inhalation and the liquid by the gastrointestinal tract; however, it is absorbed poorly by the skin. No simple relationship exists between uptake rate and peripheral venous concentration, indicating that no calculation can be made for toxic doses to internal organs. Following a single intraperitoneal administration of C-14 labeled toluene (290 μg/kg) to mice, the highest level of radioactivity was found in adipose tissue, followed in descending order by kidney, liver, and lung. Toluene reached high concentrations immediately after inhalation in lipid-rich tissues (brain and fat) and well-perfused organs (liver and kidney) but was rapidly eliminated resulting in low concentrations at 1 h in all maternal tissues except fat (National Library of Medicine 1991).

Tetrachloroethene is designated as a toxic pollutant under Section 307(a)(1) of the CWA. When tetrachloroethene is released to soil, it is subject to evaporation into the atmosphere and leaching into the groundwater. Photooxidation occurs in the atmosphere when tetrachloroethene reacts with hydroxyl radicals that are produced by sunlight in the troposphere; it has an estimated half-life of about 2 months or a loss of 1.5% per sunlit day. In anaerobic soils, there is evidence that slow biodegradation of tetrachloroethene may occur when microorganisms have been acclimated, yielding trichloroethene as a product. Because of its high vapor pressure (18.47 mm Hg at 25°C) and low adsorption, volatilization of tetrachloroethene from dry soil should be rapid. If tetrachloroethene is released to water, it will rapidly volatilize, with estimated half-lives ranging from less than one day to several weeks. Tetrachloroethene is expected to exhibit low to medium mobility in soil, allowing it to leach slowly into groundwater. It is also slightly adsorbed on sand and clay minerals (National Library of Medicine 1991).

Tetrachloroethene is readily absorbed through the lungs and to a much slower degree through skin or mucous membranes or following ingestion. Metabolism is relatively slow with only a small amount of the dose being excreted as metabolites, the primary of which is trichloroacetic acid. Elimination through exhalation is slow (biological half-life of 65 h for exhaled tetrachloroethene) because of continuing release of tetrachloroethene from fat stores, where it tends to distribute once in the bloodstream. Male Sprague-Dawley rats exposed to tetrachloroethene by either gavage (1.0 mg/kg) or inhalation (10 ppm) excreted 70% of the dose unchanged in expired air. Approximately 3% was excreted as carbon dioxide and approximately 23% was excreted in the urine and feces as nonvolatile metabolites. The hepatotoxicity of both tetrachloroethene and trichloroethene in mice is directly related to the extent of their metabolism. Four parameters of hepatoxicity were assessed: liver weight, triglycerides, glucose-6-phosphatase activity, and serum glutamic-pyruvic transaminase activity; tetrachloroethene affected all four parameters. Intubation of rats with mixtures of toluene and tetrachloroethene resulted in LD_{50} values of less than predicted for simple additivity, indicating synergistic effects. In another study of adverse effects of tetrachloroethene, it was shown to release lysosomal enzymes from granular fractions prepared from nematodes. The gut of nematodes seems to be specialized for lysosomal intracellular digestion of nutrients, and tetrachloroethene may interfere with this activity (National Library of Medicine 1991).

Trichloroethene is designated as a hazardous substance under Section 311(b)(2)(A) of the Federal Water Pollution Control Act and Section 307(a)(1) of the CWA as well as the CWA Amendments of 1977 and 1978.

Once in the environment, trichloroethene is fairly persistent. It is not hydrolized by water under normal conditions. In laboratory studies, trichloroethene does not appear to biodegrade in groundwater. However, in field studies, cis- and trans-1,2,-dichloroethene were detected in monitoring wells near trichloroethene-contaminated sites, suggesting that biodegradation was occurring. Only a few studies have noted significant aerobic biodegradation of trichloroethene, but acclimation was slow. Biodegradation under anaerobic conditions has been noted and ranged from very little after 12 weeks to 40% after 8 weeks. Trichloroethene has a low adsorption coefficient (log Koc = 2.0) to a number of soil types, which indicates ready transport through soil and low potential adsorption to sediments. Because trichloroethene does not adsorb light less than 290 nm, it should not directly photodegrade. However, slow (half-life 10.7 months) photooxidation in water has been noted (National Library of Medicine 1991).

Trichloroethene is an uncharged, nonpolar, and highly lipophilic compound that can be expected to cross the gastrointestinal mucosa by passive diffusion. It is a probable human carcinogen and is known to significantly increase the incidence of liver tumors, malignant lymphomas, and pulmonary adenocarcinomas in mice. The metabolism of trichloroethene in rats involves oxidation by the liver, producing an epoxide intermediate, which binds covalently to proteins and causes centrilobular damage to the liver (National Library of Medicine 1991).

7.6.2.3 Evaluation of potential impacts—red-tailed hawk

Of the metal contaminants selected for the terrestrial environmental evaluation, the greatest potential for adverse effects to the components of the food chain culminating in the red-tailed hawk is from the ingestion of surface water or prey contaminated with cadmium. As shown in Table 7.2, the maximum concentration of cadmium in unfiltered surface water was 30 μg/L compared to the SDWA MCL of 5 μg/L. The U.S. Fish and Wildlife Service suggests even more stringent standards and estimates that adverse effects are either pronounced or probable when cadmium concentrations exceed 3 ppb in freshwater or 100 ppb in the diet (Eisler 1985). The maximum concentration found in groundwater was 86 μg/L (Table 7.2), indicating that surface water could exhibit even higher cadmium levels during periods of recharge or at seeps. Using these estimates, the red-tailed hawk could become adversely affected by drinking water from WAG 6, as could its prey species, shorttail shrews and rabbits.

Talmage and Walton (1990) reported that earthworms and insects, which are components of the shrew's diet, exhibit the same concentration of cadmium as exists in the soil and that the total body concentration of cadmium in shrews is 2.[illegible] times that found in their diet. Using the maximum soil concentration of 4 mg/kg in soil at WAG 6, insects and earthworms would exhibit cadmium concentrations of 4 mg/kg, and shrews would have concentrations of 8.8 mg/kg. Furthermore, shrews are known to concentrate cadmium in liver and kidney tissues by factors ranging from 15 to 33 times the levels found in soil or invertebrates, such as earthworms, upon which they feed (Talmage and Walton 1990). Whereas the maximum concentration of cadmium in soils/sediments (4 mg/kg) did not exceed the action level standard of 40 mg/kg in WAG 6, allowing for bioconcentration in the shrew would result in values of 60 to 132 mg/kg in the liver and kidney tissues of this species. In laboratory animals, teratogenic and mutagenic effects were observed when rats and mice were subjected to 3 to 6 mg of cadmium per kg body weight. Thus, the potential exists for adverse effects from cadmium to the soft tissues (primarily liver and kidney) of the shrew.

Whereas cadmium is apparently associated with the soil and does not accumulate in vegetation, herbivorous animals have been shown to accumulate cadmium in soft tissues when that contaminant is deposited directly on vegetation. However, direct deposition of cadmium on vegetation on WAG 6 where it could be ingested by the herbivorous rabbit is not considered a major pathway.

By using a conservative approach and allowing the red-tailed hawk and the lower trophic levels in this food chain to feed and drink water only from WAG 6, the potential exists for adverse effects to the soft tissues of these species from cadmium. Surface water levels of cadmium (30 μg/L) surpass the SDWA MCL and exceed the concentration that, according to the U.S. Fish and Wildlife Service, causes adverse effects. Hawks, rabbits, and shrews could all be adversely affected by cadmium contamination in surface water. Another pathway of contaminant transport is from the soil to insects and earthworms to shrews, where the whole body concentration is 2.2 times that in the diet, to the red-tailed hawk, where the cadmium contamination could surpass the toxic dietary level of 100 ppb.

While hexavalent chromium is known to be toxic to wildlife because it easily penetrates biological membranes, the significance of chromium residues is unclear. The U.S. Fish and Wildlife Service states that organs and tissues of wildlife containing more than 4 mg total chromium/kg dry weight, should be presumed to have chromium contamination (Eisler 1986). Only chromium levels in groundwater in WAG 6 exceeded the SDWA MCL, with unfiltered groundwater having a maximum value of 216 μg/L compared to the regulatory standard of 100 μg/L. However, groundwater would be accessible to wildlife only during times of recharge to surface water or at seeps. In laboratory experiments with a variety of animals, the minimum injected fatal dose of hexavalent chromium ranged from 1 to 5 mg/kg body weight, although doses of 0.2 to 0.5 mg/kg body weight produced kidney damage. However, these dosage levels are much higher than levels of chromium found at WAG 6. Therefore, the effects of chromium on the elements of the food chain culminating in the red-tailed hawk are likely to be less than those from cadmium.

Radionuclides can adversely affect wildlife because they can become sources of both internal and external ionizing radiation. Of the three radionuclides selected for this environmental evaluation, strontium-90 likely poses the greatest risk to wildlife at WAG 6. The SDWA MCL is 8 pCi/L, and the maximum concentration detected at WAG 6 was 10,222 pCi/L, several orders of magnitude above the standard. Strontium-90 is a beta emitter with a relatively long half-life of 28 years. As an analog of calcium, it is readily absorbed and deposits in bone tissue. Chronic ingestion results in leukemia and bone tumors.

Because of the high concentrations of strontium-90, shrews and rabbits are likely to suffer adverse effects, to bioconcentrate strontium-90 in their bone tissue, and to cause adverse effects to the red-tailed hawk that preys upon them.

Cesium becomes tightly bound to the clay fraction of soil and is not taken up by vegetation. Therefore, the herbivorous rabbit feeding on WAG 6 is not likely to present a major pathway for contaminant transport. Conversely, earthworms can ingest the contaminated soil; they can then be consumed by shrews, which subsequently can be consumed by red-tailed hawks. All of these species can potentially be exposed to adverse effects of cesium-137 by drinking contaminated water. The maximum level of cesium-137 detected at WAG 6 was 4.57 pCi/g in soil/sediment and 11.2 pCi/L in surface water. (See Table 7.2.) While information is available to determine that cesium-137 has the potential for being transferred through the elements of the food chain culminating in the red-tailed hawk, it is not possible to describe the specific potential risks of 4.57 pCi/g of cesium-137 in soil/sediment and 11.2 pCi/L in surface water to the red-tailed hawk or the lower trophic levels composing that food chain. Furthermore, there are no regulatory standards to which to compare these concentrations.

Tritium is present at WAG 6 at levels up to 14,000,000 pCi/L in surface water, greatly exceeding the regulatory standard of 20,000 pCi/L (Table 7.2). However, tritium does not bioaccumulate in the environment, quickly reaches equilibrium in organisms, and has a relatively short half-life. The effects of tritium on the inhabitant species of WAG 6 will result from ionizing radiation rather than from bioaccumulation or bioconcentration in target organs. The specific effects on the biota at WAG 6 will be a function of the radiation dose.

Of the VOCs selected for this environmental evaluation, tetrachloroethene and trichloroethene exceeded regulatory guidelines (Table 7.2), and xylene and toluene are known to have been disposed in large quantities there. In human studies of xylene, known toxic effects include fetotoxicity, hepatomegaly, hyperactivity, decreased body weight, increased mortality at higher doses, central nervous system effects, and nose and throat irritation. It is possible that release of large quantities of xylene might cause similar reactions in small mammals such as shrews and rabbits at WAG 6. Toluene is readily absorbed by inhalation and through the gastrointestinal tract and tends to reach highest concentrations in lipid-rich tissues and well-perfused organs. However, toluene is rapidly eliminated. Studies on the effects of tetrachloroethene on laboratory animals showed that metabolism is relatively slow, that most was exhaled unchanged, and that hepatoxic effects occurred. Trichloroethene is known to significantly increase the incidence of liver tumors, malignant lymphomas, and pulmonary adenocarcinomas in mice.

While these results of studies on humans and laboratory animals indicate that these VOCs could pose a potential risk if there were extensive or persistent releases, they should volatilize fairly rapidly from surface water or soils and should not have major, long-term impacts on wildlife.

7.6.3 Raccoon

7.6.3.1 Exposure pathways—raccoon

Unlike the totally predacious red-tailed hawk, raccoons are omnivorous, consuming both aquatic and terrestrial prey species as well as vegetation. Raccoons, which are chiefly nocturnal, prefer habitat along streams and lake borders with nearby wooded areas or rock cliffs (Burt and Grossenheider 1964). Vegetation generally comprises more than half of their diet, with acorns being a principal component. Crayfish are also preferred food (Ewer 1973). Raccoons consume approximately .5 kg of food per day, 40% of which is animal matter (Table 7.1). Of this amount, crustaceans comprise 15%, fish 5%, and voles 20% (Newell, Johnson, and Allen 1987; Ewer 1973). In addition to the food chain, raccoons can be exposed to contaminants through ingestion of water, incidental ingestion of leaf litter and soil, inhalation, and direct exposure (Fig. 7.4).

Voles, which are favorite prey of raccoons, are active during both day and night (Burt and Grossenheider 1964). They are usually found where there is good grass cover and feed on grasses, sedges, seeds, grain, bark, and other vegetation. Voles can be exposed to contaminants not only through the foods that they consume but also directly as they burrow through contaminated soil and vegetation.

Crayfish, which also comprise a major portion of the diet of raccoons, complete their life cycle within contaminated aquatic and floodplain environs and directly contact contaminated mud. They feed upon filamentous algae, vascular aquatics, animal matter, and detritus. Since organic residues frequently contain elevated levels of nutrients and contaminants, there is an increased potential for the presence of contaminants in the food web leading to the crayfish. Crayfish are important in the detrital trophic level because they ingest material that otherwise would be very slowly decomposed.

7.6.3.2 Exposure and toxicity assessment—raccoon

This section contains a brief synopsis of the documented or likely toxic effects of the selected contaminants on the same or similar species as those that comprise the elements of the food chain for the raccoon. A more detailed discussion of the toxic effects of these contaminants is provided in Section 7.6.2.2. This section primarily provides additional information related to the species involved in the food chain culminating in the raccoon.

Cadmium has a strong affinity for soft body tissues, especially kidney and liver, and there appears to be a positive relationship between concentrations of cadmium in the soil and in the soft tissues of small mammals. When compared to aquatic organisms, mammals and birds seem relatively resistant to the biocidal properties of cadmium, but a review of its teratogenic, mutagenic, and carcinogenic effects is presented in Sect. 7.6.2.2. Adverse effects on fish and wildlife are either pronounced or probable when cadmium concentrations exceed 3 ppb in freshwater or 100 ppb in diet (Eisler 1985).

There are species differences in accumulation related to diet and trophic level. Of particular relevance to this exposure scenario is that whole body concentrations in the vole (*Microtus agrestis*), an exclusive herbivore and representative of the voles used in this food chain, were less than those in the surrounding soil. Talmage and Walton (1990) reported that for the mouse and vole, concentration factors of kidney:soil or kidney:invertebrates were similar and ranged from 0.5 to 1.5. An exception occurred at a site where cadmium was deposited directly on the vegetation. Here concentration factors of kidney:soil and kidney:vegetation were 2.7 and 11.7, respectively, for the vole. Johnson, Schreiber, and Burgess (1979) stated that in voles from abandoned metalliferous mines, the metal levels in herbaceous vegetation samples were indicative of dietary metal intake of voles.

Freshwater biota are very sensitive to cadmium; concentrations of 0.8 to 9.9 μg Cd/L (ppb) were lethal to several species of aquatic insects, crustaceans, and teleosts, and concentrations of 0.7 to 5.0 ppb were associated with sublethal effects such as decreased growth, inhibited reproduction, and population alterations (Eisler 1985).

Chromium is most often found in the environment in trivalent and hexavalent states, with hexavalent chromium being more toxic because it easily penetrates biological membranes. Under laboratory conditions, chromium is mutagenic, carcinogenic, and teratogenic to a wide variety of organisms (Eisler 1986). The significance of chromium residues is unclear, but the U.S. Fish and Wildlife Service estimates that organs and tissues of fish and wildlife that contain >4 mg total Cr/kg dry weight should be presumed to have chromium contamination (Eisler 1986).

Taylor and Parr (1978) found that concentrations of chromium in pelt, hair, and bone of cotton rats trapped near the cooling towers at Oak Ridge Gaseous Diffusion Plant were significantly higher ($P<0.01$) than concentrations in control animals. Chromium in pelt and hair resulted from direct deposition of drift or removal from plant materials during feeding or during movements along pathways. Chromium is an essential trace element to some mammalian species, and irreversible metabolic damage may result from long-term deficiency. Increased chromium levels may produce deleterious effects. However, cotton rats had a low assimilation rate (0.8%) and rapid initial loss of hexavalent chromium, suggesting that it is nonessential in this species.

In aquatic systems, chromium seldom exceeds 0.3 mg/kg fresh weight in edible crustacean tissues (Eisler 1981). One of the principal effects of chromium contamination in aquatic systems is damage to shells (Eisler 1986). Chromium apparently interferes with the normal functions of subcuticular epithelium, particularly cuticle formation, and subsequently causes structural weaknesses or perforations to develop in the cuticle. These chromium-induced exoskeleton deficiencies provide an avenue for pathogenic bacteria and direct chromium influx that perpetuates the development of the lesion.

Radionuclides are transferred through components of ecosystems and often accumulate in target tissues. Strontium-90, as an analog of calcium, is readily absorbed by bone tissue, and cesium-137 accumulates in soft tissue, especially kidney and liver. Although tritium also has the potential for marked uptake by terrestrial vegetation, and subsequently being incorporated into other trophic levels, it does not bioaccumulate in the environment or in food chains (Rohwer and Wilcox 1976). Cesium-137 is not expected to reach appreciable concentrations in vegetation because of its low transfer coefficient from soil. However, it can be rapidly translocated through major plant components of a forest when injected into trees. In addition to uptake by vegetation, small mammals may be an important vector for radioactive transport because of their burrowing activity, mobility, and potentially high population numbers.

As part of the Biological Monitoring and Abatement Program, Loar et al. (1988) found that all field-collected mammals from SWSA 4 and the WOC floodplain had detectable levels of strontium-90 in bone tissue. Highest levels were found in Eastern harvest mice (*Reithrodontomys humilis*), which were trapped in the grass immediately adjacent to the seeps in SWSA 4 and feed primarily on seeds found in the fescue grass that they inhabit. Levels ranged from 10,000 Bq/kg dry weight of bone to 281,000 Bq/kg, with the mean being 71,500 Bq/kg. Two pine voles (*Microtus pinetorum*) trapped at SWSA 4 had strontium-90 levels of 1450 Bq/kg and 31,810 Bq/kg, illustrating that this species is also capable of bioaccumulating this contaminant. In a study of consumption of radioactively labeled pine seeds by small mammals, Mathies et al. (1972) reported body burdens of cesium-137 in pine voles ranging from 5.89×10^3 to 13.37×10^3 dpm/g, indicating a significant uptake of radioisotopes.

Crayfish are a major component of the diet of raccoons, which ingest the soft tissues and digestive tract. Tanner (1964) found that crayfish from White Oak Creek had cobalt-60, ruthenium-106, and cesium-137 associated with their exoskeletons. Furthermore, Schurr and Stamper (1962) found that 90% of the strontium-85 found in crayfish was in the exoskeleton. Not only was 55 to 80% of the available strontium-85 retained in the body tissues after molting in young-of-the-year (Momot, Gowing, and Jones 1978), but uptake of strontium-85 was enhanced in recently molted crayfish (Schurr and Stamper 1962).

The VOCs selected for the WAG 6 environmental evaluation are toluene, xylene, tetrachloroethene, and trichloroethene. Some characteristics of these compounds, their behavior in the environment, and known toxic effects based upon studies of humans and/or laboratory animals are presented in Sect. 7.6.2.2. Information concerning their effects on terrestrial species is essentially nonexistent; effects upon these organisms can only be assumed to be similar to those found in laboratory animals.

7.6.3.3 Evaluation of potential impacts—raccoon

This evaluation is based on a conservative approach, allowing raccoons and their prey to eat and drink only from WAG 6.

The omnivorous raccoon has the potential for becoming contaminated through both aquatic and terrestrial food chains (Fig. 7.4). Raccoons can attain body burdens of contaminants through the ingestion of contaminated vegetation and drinking contaminated water. Voles, which are a major component of their diet, are herbivorous and can also become contaminated by ingesting contaminated vegetation. Fish and crayfish are exposed to the contaminants in the surface water and sediments at WAG 6 and can bioaccumulate them, resulting in even higher body burdens in raccoons that prey upon them.

Of the metal contaminants selected for this terrestrial environmental evaluation, maximum cadmium levels in unfiltered surface water were 30 μg/L, compared to the SDWA MCL of 5 μg/L. (See Table 7.2.) This concentration also exceeds the Federal Ambient Water Quality Criteria for the protection of aquatic organisms, such as the crayfish, in this food chain. (See Table 4.4.) U.S. Fish and Wildlife Service guidelines suggest that adverse effects to wildlife are likely when cadmium concentrations exceed 3 ppb in freshwater and 100 ppb in the diet (Eisler 1985). In fact, Eisler (1985) reported that freshwater biota are very sensitive to cadmium. Concentrations of 0.8 to 9.9 μg/L were lethal to several aquatic species, and concentrations of 0.7 to 5.0 μg/L were associated with sublethal effects such as decreased growth, inhibited reproduction, and population alterations. Based upon this information, the raccoon could suffer adverse effects from cadmium by drinking water from WAG 6. In addition, the aquatic prey species have the potential to be impacted by cadmium in the surface water and sediments and serving as a pathway for contaminant transport to the raccoon.

Cadmium is apparently associated with the soil and is not taken up by vegetation, suggesting that ingesting of vegetation by either the raccoon or its herbivorous prey, the vole, is not likely to result in contaminant body burdens. In fact, whole body concentrations of voles have been shown to be less than those in the surrounding soil. The greatest potential for adverse effects from cadmium to the soft tissues (liver and kidney) of the raccoon are through ingestion of surface water by the raccoon itself and by voles upon which it preys and from ingestion of contaminated aquatic organisms.

Hexavalent chromium easily penetrates biological membranes and has been shown to cause mutagenic, carcinogenic, and teratogenic effects in a variety of laboratory animals. Doses of hexavalent chromium from 0.2 to 0.5 mg/kg body weight produced kidney damage, and the minimum injected fatal dose ranged from 1 to 5 mg/kg body weight (Eisler 1986). In a study of hexavalent chromium in dogs, Steven et al. (1976) found that 100 ppm in food for three months was fatal, that 11.2 ppm in drinking water was not lethal but was accumulated in significant amounts over a 4-year period, and that 6.0 ppm in drinking water for 4 years had no measurable effects. The maximum chromium level at WAG 6 found in groundwater was 216 μg/L, compared to the SDWA MCL of 100 μg/L. However, not only is the chromium level in groundwater at WAG 6 lower than that producing noticeable effects in dogs, but groundwater is not generally accessible to wildlife. Therefore, chromium in groundwater at WAG 6 should not pose a major threat to the raccoon.

In a study of cotton rats, Taylor and Parr (1978) found that this species had a low assimilation rate and rapid initial loss of hexavalent chromium, suggesting that it is nonessential. Whether chromium is an essential element to voles or raccoons (which would enhance its assimilation) is not known.

The Federal Ambient Water Quality Criteria for the protection of freshwater organisms is 11 μg/L for hexavalent chromium, and the maximum level detected in surface water at WAG 6 was 58.3 µg/L, a level exceeding that standard. Although analysis of the WAG 6 samples was for total chromium, the human health evaluation (maintaining a conservative approach) assumed that this figure represented only hexavalent chromium. If the same assumption is made for the environmental evaluation, aquatic species such as the crayfish in this food chain could be adversely affected, with shell damage as one likely effect.

The radionuclides at WAG 6 can adversely affect wildlife either through bioaccumulation in target tissues or through providing sources of ionizing radiation. Of the three radionuclides selected for evaluation, strontium likely poses the greatest risk. The maximum concentration detected (10,222 pCi/L) greatly exceeds the regulatory standard of 8 pCi/L (Table 7.2). Strontium-90 readily accumulates in bone tissue and can be expected to adversely affect the raccoon directly and through ingestion of voles with body burdens in bone tissue.

Cesium-137, as a chemical analog of potassium, is accumulated in soft tissues such as liver and kidney. However, it is less mobile than strontium and tritium and becomes bound to the illitic clays. Information is available to determine that cesium-137 has the potential for being transferred through the trophic levels to the raccoon, but it is not possible to determine the specific potential impacts from 4.57 pCi/g of cesium-137 in soil/sediment or 11.2 pCi/L in surface water. Both strontium-90 and cesium-137 are known to accumulate in the exoskeletons of crayfish; however, raccoons consume the soft tissues and digestive tract of this species. Normal ingestion of crayfish should not provide a major pathway. The exception is that molting crayfish retain the majority of strontium-90 in their body tissues (Stamper 1962) and, if ingested during molting, could contribute to increased radionuclide levels in raccoons. Another potential pathway that should be considered is the direct radiation exposure to the raccoon that encounters contaminated crayfish while consuming them and the direct exposure from discarded molten exoskeletons.

Maximum tritium concentrations at WAG 6 were 14,000,000 pCi/L in surface water, greatly exceeding the SDWA MCL of 20,000 pCi/L (Table 7.2). Its primary impact on wildlife at WAG 6 is from the effects of ionizing radiation, because tritium does not bioaccumulate in ecosystems. The specific effects to the raccoon or its prey would depend on the dose received.

Of the VOCs selected for this evaluation, trichloroethene and tetrachloroethene exceeded regulatory standards (Table 7.2) and xylene and toluene are known to have been disposed there in large amounts. As discussed in Sect. 7.6.1.3, studies on humans and laboratory animals illustrate that these contaminants have the potential to produce toxic effects if there were extensive or persistent releases at WAG 6. However, they should volatilize fairly rapidly from the surface water or soils and should not have major, long-term effects on WAG 6 species.

7.7 AQUATIC VERTEBRATES—BLUEGILL SUNFISH AND FATHEAD MINNOWS

7.7.1 Exposure Assessment—Aquatic Vertebrates

Bluegill sunfish and fathead minnows were selected as key species representing the fish communities because both are common in the WOC watershed and because there is substantial literature on the toxicity of contaminants for these two species. Although both species were introduced to WOC, they are native to other waters in Tennessee and have probably established reproducing populations in WOC. Bluegills feed on a wide variety of food sources, including algae and aquatic plants, but rely primarily on aquatic macroinvertebrates. Bluegills mature in their second year and spawn from spring through summer in nests on sand/gravel substrates. The male constructs the nest and guards the eggs and young. Fathead minnows feed on algae and aquatic insects, mature in their second year, and spawn throughout the spring and summer. The eggs are deposited on the underside of suspended objects or rocks, and males guard the nests. Bluegills and fathead minnows are common forage fish in southern waters, with bass species as the top predator.

7.7.1.1 Characteristics of the contaminants

Cadmium and copper are naturally occurring divalent elements that are relatively soluble in water and may be found in high concentrations in sediments. Copper is a required minor nutrient for both plants and animals. It is an essential element for enzyme activity and is required for electron transport during oxidative phosphorylation. However, at concentrations over a few parts per billion, it can be toxic to aquatic plants and animals. There is no evidence that cadmium is biologically essential or beneficial to living organisms. At sufficient concentrations, cadmium is toxic to all forms of life. Freshwater biota are generally more sensitive to the toxic effects of cadmium and copper than marine organisms and terrestrial life forms. The toxicity of copper and cadmium are related primarily to the activity of the divalent ion (Cu^{+2} and Cd^{+2}) and possibly some metal hydroxy complexes.

In unpolluted freshwater systems, cadmium concentrations in water are reported to range from 0.05 to 0.2 ppb while copper concentrations are commonly in the 1- to 10-ppb range. Sediments in lakes and streams may contain a few thousand ppb of cadmium and copper (Eisler 1985). In areas where cadmium and copper are comparatively bioavailable, these metals can be chronically toxic to some fish and benthic macroinvertebrate species at concentrations slightly higher than the normal background ranges.

Toluene and xylene are main components in the scintillation fluids used in the radiological laboratories at ORNL. A large quantity of scintillation fluid, containing radioactive material, was buried in trenches at WAG 6. Toluene and xylene are moderately toxic in aquatic ecosystems in comparison to the more highly substituted benzene-type compounds. Their impact on surface water systems is limited because of their high volatility. In natural unpolluted waters, xylene and toluene may be present at low parts per billion concentrations. They are acutely toxic to fish at concentrations of about 10 mg/L (ppm) and up.

The radionuclides cesium-137 and strontium-90 are long-lived fission products. Strontium-90 is chemically similar to calcium and is metabolized by plants and animals as if it were calcium. Strontium-90 is transported in fresh water systems as a free ion or exchangeable cation similar to many metals and is one of the more mobile radionuclides. In animals, strontium-90, like calcium, concentrates in bones and shells. Cesium-137 has a high affinity for clays, gravels, and organic material and migrates in surface water systems as particulates or bound to sediments. In contaminated environments they are a hazard because they accumulate in living organisms and become internal sources of ionizing radiation, as well as external sources of gamma radiation.

In uncontaminated freshwater ecosystems, stable strontium and cesium exist in low concentrations, generally less than 1 ppm for cesium and less than 0.25 ppm for strontium (Vanderploeg et al. 1975; Kolehmainen and Nelson 1969). Strontium in freshwater ecosystems is generally equally distributed among three components: water, biota, and sediments (Klechkovskii, Polikarpov, and Aleksakhin 1973). Cesium, on the other hand, is found primarily in bottom sediments and suspended particulates (Chittenden 1983; Klechkovskii, Polikarpov, and Aleksakhin 1973). Two or more groundwater seeps have been identified at SWSA 6 that are sources of strontium-90 to the tributaries draining the WAG 6 area (Cerling and Spalding 1981).

7.7.1.2 Mode of uptake

Fish will accumulate metals, radionuclides, and organic contaminants from their environment primarily by absorption of dissolved materials across gill membranes and by ingesting contaminated food and sediments. The relative importance of these two avenues of contaminant uptake is sometimes in question (Dallinger et al. 1987). However, experiments comparing uptake of metals, radionuclides, and monocyclic aromatics via food versus direct uptake from water indicate that direct uptake is usually the principal route (Berry and Fisher 1979; Polikarpov 1966; Hatakeyama and Yasuno 1987). Uptake of cesium-137 by fish and macroinvertebrates, however, is primarily via the food chain (Kolehmainen 1972; Preston, Jefferies, and Dutton 1967). Because of the importance of direct uptake from water, the toxicity of contaminants in surface water ecosystems is usually based on the concentration of the contaminant in the water column, rather than estimating the contaminant concentration in food and then estimating the rate of food uptake. Acute and chronic toxicity values, therefore, are expressed as the lethal or harmful concentrations of the contaminant in water.

7.7.1.3 Factors affecting transport and bioavailability

Field investigations and laboratory experiments have shown that numerous factors can affect the transport and eventual fate of metals, simple aromatic compounds, and radionuclides in freshwater ecosystems. For example, cesium-137 has a high affinity for sediments, especially clays (Klechkovskii, Polikarpov, and Aleksakhin 1973), and in the surface water systems at ORNL, this radionuclide is held irreversibly to the illitic minerals in the sediments (Cerling and Spalding 1981; Huff 1986). Its movement in the surface water environment, therefore, reflects sediment transport mechanisms and movement of suspended particulates in the water column. Copper, cadmium, and strontium-90, on the other hand, are relatively soluble in water and exist as free ions or exchangeable cations that are bioavailable. Adsorption to clays and humic materials, however, may remove some proportion of these elements from the water column. Generally, the higher the distribution coefficients are for these elements (between sediments and water), the more effective adsorption processes are in reducing their concentrations in the water column. Humic materials in bottom sediments are one of the main components responsible for adsorption of cadmium and possibly copper. Changes in physicochemical conditions such as pH and the redox potential can also effect the proportion of cadmium, copper, and strontium-90 in the water column versus in the sediments and suspended particulates (Eisler 1985; Klechkovskii, Polikarpov, and Aleksakhin 1973).

Water quality parameters were measured for grab samples collected on March 2, 1989, from the two tributaries in SWSA 6. Tributary FA on the western side had a pH of 7.98, alkalinity of 131.0, and hardness of 124. Tributary DB draining the eastern side of SWSA 6 had a pH of 7.58, alkalinity of 112.0, and hardness of 148. These measurements did not indicate any important differences between streams. However, measurements from a control sample showed a pH of 7.70, alkalinity of 30.0, and hardness of 38, indicating that both hardness and alkalinity in the SWSA 6 tributaries were much higher.

Water hardness has a major effect on the toxicity of cadmium and copper. This effect may be due to the complexing capacity of the natural waters (Davies and Woodling 1980) or actual competition between calcium and cadmium/copper for binding sites (Versteeg and Giesy 1986). An increase in hardness from 50 to 200 mg/L (as $CaCo_3$) can reduce the toxicity of cadmium and copper by an order of magnitude or more (Eisler 1985; EPA 1986e). Several investigations have stated that hardness may not be the only factor influencing the toxicity of metals. Winner and Gauss (1986) found that humic acid was also important, and EPA (1985a) stated that total organic carbon may be more important than hardness in some situations. Alkalinity, temperature, and pH may also influence the toxicity of cadmium and copper and the bioavailability of strontium-90.

Toluene and xylene are among the more toxic components in the water soluble fraction of refined petroleum products. These low molecular weight hydrocarbons, however, volatilize relatively quickly in surface water systems. For example, Jori et al. (1986) estimated that the mean residence time for xylene was 30 minutes, assuming a flowing water system 1 m deep. During bioassay tests with six monocyclic aromatics, Benville and Korn (1977) observed 30 to 50% losses within 48 h and 60 to 90% with 72 h. These monocyclic aromatics are also sorbed onto suspended particulates and sediments, especially by organic matter. Toluene and xylenes are degraded rather rapidly in the atmosphere by photo-induced oxidation (Jori et al. 1986), and this process also occurs in near-surface zones in clear surface waters. Biodegradation processes in soil and sediments may be more important than adsorption in removing toluene and xylene from natural ecosystems.

The uptake (i.e., bioavailability) of cesium-137 and strontium-90 from water by living organisms is closely linked to the concentrations of their nonisotopic carriers, potassium and calcium, respectively. Cesium-137 acts chemically very much like potassium, and the accumulation of cesium-137 by aquatic plants and animals is inversely related to the concentration of potassium in the water. Similarly, the uptake of strontium-90 is inversely related to calcium concentration. Therefore, in aquatic ecosystems, the bioconcentration factors of cesium-137 and strontium-90 will be lower in waters with high concentrations of calcium and potassium. The influence of potassium on cesium-137 uptake is most important for the primary producers (algae and macrophytes), while factors that influence uptake via food chains (such as feeding rates and assimilation/elimination) are more important in the uptake of cesium-137 by secondary and higher consumers.

7.7.1.4 Sensitive populations

Sexual maturity is attained within 1 to 2 years, and the adults live for a few years (fathead minnows) to several years (bluegills). The scientific literature indicates that for bluegills and fathead minnows, the life stage most sensitive to metal and organic contaminants is the early larvae. The embryo and adult fish are generally the most tolerant (Benoit 1975; McKim 1977; Eaton, McKim, and Holcombe 1978; Fisher, Dilego, and Paulos 1984; Devlin, Branner, and Puyear 1982). This holds true for cadmium and copper and the simple monocyclic aromatics. The early larval stage is not necessarily the most sensitive stage for all metals or other contaminants, and early life stage mortality is not necessarily the most sensitive indicator of toxicity.

Field and laboratory studies have indicated that some aquatic organisms may be more tolerant of metal contaminants if they have been residing in a system with elevated metal concentrations (Davies and Woodling 1980; EPA 1985a; Bishop and McIntosh 1981). Acclimation to copper and cadmium toxicity generally requires protracted exposure to sublethal concentrations for days or weeks, and fish may become unacclimated if transferred to clean water for several weeks (Mance 1987). Acclimation in some cases may be related to the induction of metal-binding proteins or storage of the contaminants in various nontoxic forms (EPA 1985a).

Fish are apparently capable of metabolizing toluene and xylene in the liver, storing metabolites in the gall bladder, and excreting these substances via the gut (Berry 1980; Berry and Fisher 1979). Aquatic organisms with the ability to metabolize would be less vulnerable to the toxic effects of these organic compounds.

There is a relatively wide variation in the sensitivity of different fish species to metals, and to a lesser extent, organics. For example, salmonid teleosts such as brook trout and Atlantic salmon are about 10 times more sensitive to cadmium than sunfish and minnows. There is little difference in the sensitivities of bluegills and fathead minnows to cadmium, copper, toluene, or xylene.

There is very little information in the scientific literature concerning the relative sensitivities of different biotic subpopulations (such as life stage or size) to the harmful effects of radiation. Generally, simpler forms of life such as bacteria and algae are more resistant than the more complex plants and animals. Bacteria are especially resistant to ionizing radiation (Klechkovskii, Polikarpov, and Aleksakhin 1973). King (1964) stated that humans are at least twice as sensitive as fish to roentgen dosages.

Several investigators have studied the effect of radionuclides on fish eggs. They presented the effects of radiation on fish embryos, but the data on sensitivities of fish eggs relative to other life stages or fish species are variable (Klechkovskii, Polikarpov, and Aleksakhin 1973). Kolehmainen (1972) found that the concentration of cesium-137 in bluegills in WOL increased with size up to 70 g, apparently because assimilation of the radionuclide from food increased and elimination decreased with size. Thus, older fish would be more at risk because of higher body burdens as well as longer exposure. Fish species with shorter life cycles and higher reproductive potential are less susceptible to the harmful effects of radiation because of shorter exposure and the ability to cope with reduced fertility and/or fecundity (Krumholtz 1954).

7.7.2 Toxicity Assessment—Aquatic Vertebrates

7.7.2.1 Mode of action

Elevated concentrations of cadmium, copper, toluene, and xylene can affect fish through various mechanisms. In some cases, the specific biochemical or physiological process is known, and in other cases, the observable reaction by the fish may be measured, such as a change in ventilatory activity, but the specific mechanism or cause is unknown.

Toluene and xylene, like many organic chemicals used in industry, produce a nonspecific narcosis effect on fish. Narcosis is a reversible condition up to a point, but prolonged exposure may cause death (Veith, Call, and Brooke 1983; Broderius and Kahl 1985). Several investigators also observed initial behavioral changes (swimming activity) early in the exposure period, and with benzene compounds the fish exhibited erratic swimming behavior just prior to death (Broderius and Kahl 1985; Fisher, Dilego, and Paulus 1984; Norton, Mattie, and Kearns 1985). Exposure to aromatic hydrocarbons such as toluene during the embryo stage was observed to cause developmental abnormalities and premature hatching.

Heavy metals such as cadmium and copper can affect behavior, may interfere with or inhibit biochemical processes, can cause structural damage to tissues, and have teratogenic effects. Cadmium at relatively high concentrations can coagulate the mucous covering on the gill membranes, thereby inhibiting oxygen transport into the blood (Doudoroff and Katz 1953). Sublethal concentrations of both copper and cadmium over long exposure periods are known to cause structural damage to gill tissue (Dallinger et al. 1987; Versteeg and Giesy 1985,1986). Many of the metals can interfere with biochemical processes, and inhibition of enzyme systems is the specific cause for some of these effects. Different metals, however, generally affect different enzyme systems. Copper, for example, can alter the production of high energy phosphate bonds by disrupting enzymes involved in oxidative phosphorylation and can also disrupt osmoregulation by inhibiting active transport of salts across the gill membrane. Cadmium disrupts normal lysosome membrane activity (Versteeg and Giesy 1985), and both cadmium and copper may interfere with metabolic processes in liver mitochondria.

Chronic exposure to cadmium and copper can produce changes in behavior and ventilatory activity at concentrations well below those causing death (Bishop and McIntosh 1981; Diamond, Parson, and Gruber 1990; Ellgaard and Guillot 1988; Ellgaard, Tusa, and Malizia 1978). Changes in the frequency and amplitude of breathing and frequency of coughing have been observed by several investigators, and these responses may be directly proportional to the metal concentration (Bishop and McIntosh 1981). While several metals, including cadmium, cause hyperactivity, copper causes hypoactivity. These results indicate that copper causes a decrease in metabolic rate while cadmium increases metabolic processes. This agrees with studies that show that copper causes a decrease in production of high energy phosphate bonds (Heath 1984).

The selected radionuclides at WAG 6, cesium-137 and strontium-90, are potentially harmful to fish because they bioconcentrate in internal tissues and may become significant internal sources of ionizing radiation. Therefore, the mode of action is the same as the effect of radiation on humans. At high dosages, usually from external sources, radionuclides can cause acute mortality. King (1964) and Foster et al. (1949, cited by Krumholtz 1954) observed fish mortalities at exposures of 1000 to 3000 R. However, the effect of radionuclides in most cases, including the impact of cesium-137 and strontium-90 at ORNL, is more chronic. Krumholtz (1954) noted that exposure to radionuclides resulted in malformed ova and reduced growth rates. Ionizing radiation can cause genetic mutations, chromosomal aberrations, abnormal development, and reduced reproductive success (Klechkovskii, Polikarpov, and Aleksakhin 1973; King 1964). The effect of specific radionuclides, however, is impossible to measure in natural environments because the radiation from multiple radionuclides, both external and internal, is cumulative to some extent.

7.7.2.2 Toxicity of contaminants

Several different toxicity values are presented to assess the toxicity of the selected contaminants to freshwater fishes. These include the EPA ambient water quality criteria for the protection of aquatic life, acute toxicity values from several studies, and various measures of chronic toxicity. Acute toxicity is most frequently expressed as the concentration of a contaminant that will be lethal to 50% of the test population within a given time period, such as 96 h (96-h LC_{50}). Some of the more standard expressions of chronic toxicity include the highest concentration of a contaminant that results in no observable effect (the NOAEL), and the lowest concentration of a contaminant that produces a significant difference between test and control populations (the LOAEL). These two values theoretically bracket the "safe" concentration for that contaminant. The term "chronic" usually implies long duration, low severity, and high sensitivity, and is measured in numerous ways. Therefore, chronic toxicity is usually expressed as the concentration that produces some measurable response (e.g., decreased growth) over a defined exposure period. The harmful effect of radionuclides is expressed as a radiation dose.

7.7.2.3 Ambient water quality criteria

EPA (1986e) has established acute and chronic ambient water quality criteria for cadmium and copper, and acute criteria for benzene and toluene. Freshwater organisms (both fish and macroinvertebrates) and their habitat should not be affected unacceptably if the acute (Criterion Maximum Concentration, a 1-h average) and chronic (Criterion Continuous Concentration, a 4-day average) criteria are not exceeded more than once every 3 years. These criteria are presented in Table 7.3.

7.7.2.4 Acute and chronic toxicity of cadmium and copper

Some acute toxicity values for cadmium to bluegills and fathead minnows are presented in Table 7.4. All values are for 96-h LC_{50}s. Eisler (1985), Mance (1987), Suter, Vaughan, and Gardner (1983), and other investigators present LC_{50}s and LC_{10}s for time periods of 24 h to a month or more. Note that an increase in hardness can change the LC_{50} by an order of magnitude or more. The lowest acute toxicity value was 13.2 μg/L, for the larval life stage of fathead minnow in soft water (Spehar and Fiandt 1986). Generally, concentrations acutely toxic to bluegills and fathead minnows ranged between 340 and 21,100 μg/L and varied with life stage, water hardness, and other variables.

The concentrations of cadmium that are chronically toxic to bluegills and fathead minnows are shown in Table 7B.1 in Appendix 7B. Fathead minnows may be slightly more sensitive to cadmium than bluegills, with chronic toxicity values between 57 and 110 μg/L. However, cadmium at 5 μg/L produced lethal effects when mixed with five other metals in experiments by Spehar and Fiandt (Table 7B.1 in Appendix 7B). Concentrations chronically toxic to bluegills generally ranged between 50 and 250 μg/L, but concentrations as low as 4 μg/L induced changes in ventilatory response (Diamond, Parson, and Gruber 1990). The larvae are probably the most sensitive life stage to metal contaminants. For larval fathead minnow, the NOEL and LOEL are 37 and 57 μg/L, respectively (Pickering and Gast 1972). For bluegills, the NOEL and LOEL were reported by Eaton (1974) as 31 and 80 μg/L.

Copper concentrations that are acutely toxic to bluegills and fathead minnows are shown in Table 7B.2 in Appendix 7B. As with cadmium, fathead minnows may be slightly more sensitive to copper than bluegills. Copper concentrations causing 50% mortality over a 96-h exposure period ranged between 75 and 1,000 μg/L for fathead minnows and between 740 to over 4,000 μg/L for bluegills. Copper was more toxic in soft water, and the larvae of both species are more sensitive than the adults. Copper concentrations as low as 18.4 μg/L were reported to be chronically toxic to fathead minnows larvae in soft water (Mount and Stephan 1969); concentrations as low as 40 μg/L reduced growth and survival in bluegills (Table 7B.3, Appendix 7B). The NOEL and LOEL for copper for juvenile bluegills were reported as 21 and 40 μg/L by B[illegible]t (1975), and for fathead minnow larvae as 31 and 80 μg/L by Pickering, Brungs, and Gast (1977). Mount and Stephan (1969) reported NOEL and LOEL values for fathead minnow fry of 10.6 and 18.4 μg/L.

7.7.2.5 Acute and chronic toxicity of toluene and xylene

Toxicity tests by several investigators using fathead minnows and bluegills indicate that concentrations of toluene between 13 and 42 mg/L are acutely toxic to larval and juvenile fish (Table 7B.4, Appendix 7B). Concentrations between 57 and 72 mg/L were acutely toxic to fathead minnow embryos. Devlin, Brammer, and Puyear (1982) and Moles, Rice, and Korn (1979) found that the embryo life stage was relatively tolerant to the monocyclic aromatics, while early larvae were more sensitive. The more highly substituted benzenes are more toxic than toluene and xylene. The potencies of the chlorinated toluenes correlated closely with log octanal/water partition coefficients, and the toxicities of substituted benzenes can be predicted based on Quantitative Structure Activity Relationships (QSAR) (Babich and Barenfreund 1987; Hall, Maynard, and Kier 1989).

Data on chronic toxicity of toluene and xylene are limited probably reflecting the high volatility of these low molecular weight aromatics. In shallow freshwater systems, toluene and xylene would be reduced significantly within 2 or 3 days as they evaporate into the atmosphere. Chronic toxicities for toluene and benzene between 5 and 45 mg/L (Appendix 7B) have been reported by Devlin, Brammer, and Puyear (1982), Berry (1980), and Fisher, Dilego, and Paulos (1984). Jori et al. (1986) proposed a provisional water quality criterion of 0.02 mg/L for xylenes as a "safe level" for aquatic organisms in light of the limited chronic toxicity data in the literature.

7.7.2.6 Harmful effects of cesium-137 and strontium-90

To estimate the effects of radiation on aquatic organisms at WAG 6, at least six pieces of information are needed. These include: (1) the concentration of each radionuclide in the water, (2) the capacity of the organisms to concentrate the respective radionuclides, (3) the types of radiation emitted by each radionuclide, (4) the tissues where the radionuclides will accumulate, (5) the body geometry of the receptors, and (6) the additive effect of the different radionuclides. The harmful effect of radiation results from all radiation sources present, both internal and external. Therefore, the estimation of dose can be very complex, and it is almost impossible to isolate the effect of a single radionuclide when several are present. Additionally, most studies on aquatic organisms have been concerned with the potential impacts on humans rather than on the fish or other aquatic organisms, so there are very few data on the radiation doses that are harmful to the fish themselves.

Radiation dosages in the range of 100 to several thousand roentgens are known to produce harmful effects in fish, ranging from reduced growth to abnormal development of ova to death at the highest dosages (King 1964; Krumholz 1954). Krumholz (1954) found that radiophosphorus along with cesium-137 and strontium-90 were main sources in the WOC system at ORNL and estimated that long-lived vertebrates such as fish and turtles in WOL may receive the equivalent of more than 50 roentgens of external radiation per year and more than 50 R/year as internal irradiation. Krumholz also noted that the overall fitness of the aquatic vertebrate population in the WOC system was probably seriously impaired. However, Krumholz and others have indicated that it is impossible to determine the significance of one or a few radionuclides in light of all the radiological and nonradiological contaminants in the WOC ecosystem (Krumholz 1954; Kolehmainen and Nelson 1969; Kolehmainen 1972).

The body burdens of cesium-137 in bluegills and carp from WOL (20 to 63 pCi/g fresh weight) reported by Kolehmainen and Nelson (1969) and Edwards (1986) are in the same range or higher than the body burdens reported in brown trout and perch in Scotland and Finland (Preston, Jefferies, and Dutton 1967). Preston, Jefferies, and Dutton (1967) found some of the highest body burdens occurred in Finnish lakes (26.8 pCi/g) because the low mineral content of these lakes (low calcium and potassium) was conducive to high cesium bioconcentration factors. Kolehmainen and Nelson (1969) reported bioconcentration factors for cesium-137 between 460 to 1080, depending upon the fish species. Although cesium-137 bioconcentration factors may be high (the ratio of cesium concentrations in fish tissue versus the concentrations in water), most investigations agree that the bioaccumulation of cesium-137 through the food chain (that is, between successive trophic levels) is not very significant.

Klechkovskii, Polikarpov, and Aleksakhin (1973) reported that strontium-90 levels of 10^{-4} Ci/L caused some increase in morphological deformities in pike eggs and disrupted development of mollusc embryos. Edwards (1986) reported strontium-90 concentrations of 0.001 to 0.11 pCi/g observed in brown trout in several rivers in England by Preston, Jefferies, and Dutton (1967). Strontium-90 can be especially harmful because it concentrates in bone tissue where replacement is slow.

7.7.3 Evaluation of Potential Impacts—Aquatic Vertebrates

The maximum concentration of cadmium in surface water at WAG 6 was 30 μg/L. The Federal Ambient Water Quality Criterion for cadmium for acute exposure is 1.8 μg/L and for chronic exposure is 0.66 μg/L, assuming a hardness of 50 mg/L $CaCO_3$; the criterion is 8.6 for acute exposure and 2.0 for chronic exposure, assuming a hardness of 200 mg/L CaCO All of these levels were surpassed by concentrations found in surface water on WAG Similarly, the maximum copper concentration of 88 μg/L exceeded the Federal Ambient Water Quality Criteria for acute exposure (9.2 μg/L) and for chronic exposure (6.5 μg/L), assuming a hardness of 50 mg/L $CaCO_3$, as well as for acute exposure (34.0 μg/L) and chronic exposure (21.0 μg/L) assuming a hardness of 200 mg/L $CaCO_3$. However, the toxicity of these contaminants is dependent upon physicochemical conditions such as water hardness, pH, and temperature. In fact, an increase in hardness from 50 to 200 mg/L $CaCO_3$ can reduce the toxicity of cadmium and copper by an order of magnitude or more (Eisler 1985, EPA 1986e). Loar et al. (1990) reported hardness values of 124 and 148 for the two tributaries in WAG 6, making the higher ambient water quality criteria more appropriate for this evaluation.

Based on information in the literature, fathead minnows exhibit chronic toxicity to cadmium values ranging from 57 to 110 μg/L, while chronic toxicity for bluegill was 50 to 250 μg/L. However, concentrations as low as 4 μg/L induced changes in ventilatory response. Lifestage also influences toxicity, with larvae being more sensitive. For larval fathead minnows, the NOAEL and LOAEL were reported to be 37 and 57 μg/L, respectively; those values for bluegills were reported to be 31 and 80 μg/L. While cadmium levels in surface water at WAG 6 surpassed the Federal Ambient Water Quality Criteria, the maximum concentration detected during the RFI field sampling approximated the NOEL for larval bluegill based on values found in the literature.

Copper is more toxic in soft water, and larval fathead minnows and bluegills are more sensitive than adults. Copper concentrations as low as 18.4 μg/L were reported to be chronically toxic to fathead minnow larvae, and concentrations of 40 μg/L reduced growth and survival in bluegills. The maximum concentration detected at WAG 6 (88 μg/L) exceeded those concentrations known to cause adverse effects.

Strontium-90 and cesium-137 are potentially harmful to fish because they bioconcentrate in internal tissues and may become significant sources of ionizing radiation. Strontium-90 is transported in fresh water systems as a free ion or exchangeable cation and is one of the more mobile radionuclides. It is a concern for WAG 6 because the maximum concentration in surface water was 10,222 pCi/L, which greatly exceeds the SDWA MCL. Fish from WOL have been shown to have elevated strontium-90 levels, and this radionuclide concentrates in bone tissue where the biological half-life may be long. Cesium-137 has a high affinity for clays, gravels, and organic material, and it migrates in surface water systems as particulates or bound to sediments. The maximum concentration found in surface water at WAG 6 was 11.2 pCi/L, and the maximum in soil/sediment concentration was 4.57 pCi/L. Information is not available to determine the specific potential effects of those levels on fish species within WAG 6; any bioconcentration would likely affect the soft tissues such as the liver and kidney. At sufficient concentrations, exposure to radionuclides has resulted in mortality, malformed ova, reduced growth rates, chromosomal aberrations, abnormal development, and reduced reproductive success.

Xylene and toluene have been reported to be acutely toxic to fish at 10 to 42 mg/L. The maximum levels reported in groundwater for WAG 6 were 2.0 mg/L for toluene and 3.2 mg/L for xylene, indicating that there is some contamination but at levels lower than concentrations causing acute toxic effects. These compounds readily volatilize from surface water; however, an extensive or persistent release from the large amounts known to have been disposed there could produce acute effects on fish at WAG 6.

7.8 AQUATIC INVERTEBRATES—INSECTS, CRUSTACEANS, WORMS

7.8.1 Exposure Assessment—Aquatic Invertebrates

Because of their abundance, relatively sedentary nature, close association with bottom substrates, and lifespans of a few weeks to a year or more, benthic macroinvertebrates are good integrators of contaminant releases into aquatic ecosystems. These benthic communities are also relatively easy to sample and quantify and represent a wide range of trophic relationships. Aquatic macroinvertebrates feed on plants, bacteria, organic detritus, and other invertebrates. They range from active predators to grazers to net spinners. (Benthic macroinvertebrates, in turn, are a major food component for many fish.) For assessing potential toxicity of contaminants at WAG 6, four orders of aquatic insects were emphasized:

mayflies (Ephemeroptera), caddisflies (Trichoptera), stoneflies (Plecoptera), and true flies (Diptera). These aquatic insects molt through several larval forms (instars) until the terrestrial adult form emerges. The aquatic stages of the life cycle range from a few weeks to a year or more, while the adult terrestrial stage usually lasts for a few days to a month. Many species of Diptera are more tolerant of organic pollution and toxic contaminants than the EPT complex. Two Crustacea, isopods and amphipods (Gammaridae), and aquatic worms (Oligochaeta) were also emphasized because of their relative abundance in tributaries at WAG 6. The crustaceans are relatively sensitive to metal and organic contaminants, whereas many oligochaetes are relatively resistant.

As part of the Biological Monitoring and Abatement Program, water samples from the FA and DB tributaries in SWSA 6 were tested for acute and chronic toxicity in 1988 and for chronic toxicity in 1989 (Loar et al. 1989, Loar et al. 1990). The tests used Ceriodaphnia, a species that has been shown to be more sensitive than fathead minnow larvae to various effluents and toxicants (Loar et al. 1990). The 1988 tests resulted in survival of 100% of the Ceriodaphnia in the controls and in both concentrations (either 50% or 100%) of water from each of the tributaries. Average fecundity in water from the SWSA 6 tributaries was high (greater than 16 offspring per female) relative to the controls. When only the seven most fecund animals in each sample were compared, the number of offspring per female in water from FA tributary was virtually the same as the controls, but the fecundity of Ceriodaphnia in water from DB tributary exceeded that in the controls by 18%. These results indicate that there is likely a difference in the biological quality of water in the two streams but no evidence for acute or chronic toxicity in either stream.

Results from the 1989 test showed that neither stream was acutely toxic and that Ceriodaphnia reared in full-strength water from each stream had moderately high fecundity. The mean fecundity of Ceriodaphnia in water from the FA tributary was somewhat lower (11.8 versus [illegible]5.8) than those from the DB tributary, suggesting a lower biological quality for FA.

7.8.1.1 Characteristics of the contaminants

Both cadmium and copper are toxic to aquatic plants and animals at parts-per-billion concentrations. Xylene and toluene are toxic in the part-per-million range, but concentrations in shallow surface waters are typically low due to their high volatilities. The two radionuclides cesium-137 and strontium-90 are chemically similar to potassium and calcium, respectively. They are hazardous to living organisms because they emit ionizing radiation. These radionuclides can bioaccumulate in aquatic organisms since they are metabolized like potassium and calcium and then become internal sources of ionizing radiation.

7.8.1.2 Mode of uptake

Like fish, aquatic invertebrates may be exposed to contaminants from their environment by several different routes including ingestion of contaminated sediments and food and absorption of dissolved contaminants across respiratory surfaces. Additionally, contaminants may be absorbed through the integument, especially during molting and emergence. Finally, contaminants, particularly metals and strontium-90, may be accumulated by adsorption to the body wall or exoskeleton. Adsorbed metals may not be toxic to the organism itself, but the route has important implications because of the potential for bioaccumulation via the food chain. Radionuclides adsorbed to the exoskeleton may represent a hazardous radiation source. The principal mode of uptake by macroinvertebrates for the selected contaminants is direct uptake from water across the gills and integument, with the exception of cesium-137, which is taken up by eating contaminated food and sediments (Kolehmainen 1972; Vanderploeg et al. 1975).

7.8.1.3 Factors affecting transport and bioavailability

The fate and transport of metals, aromatic hydrocarbons, and radionuclides is influenced by several controlling factors which ultimately affect their respective toxicities to aquatic macroinvertebrates. These factors include water hardness, pH, total organic or humic acid concentrations, calcium and potassium concentrations (for strontium-90 and cesium-137), adsorption/desorption, and biodegradation.

In addition to the physical and chemical factors of the transporting medium (e.g., water or sediment), biological parameters such as the presence of metal-binding proteins also influence bioavailability and toxicity. Metallothioneins are present in a wide variety of organisms, from yeasts to mammals, but were only recently confirmed in freshwater invertebrates by Everard and Swain (1983). Metallothioneins bind selectively to Group II metals, particularly cadmium and zinc, allowing organisms to store metals in a nontoxic form.

7.8.1.4 Sensitive populations

The relative intraspecific susceptibility of aquatic macroinvertebrates to the toxic effects of contaminants depends upon the organism's lifestage or body size. In general, smaller organisms are more sensitive to metals and monocyclic aromatics than larger ones, and early instars are more sensitive than older instars. The difference in sensitivity of early and late instars can be significant (Williams et al. 1986; Gauss et al. 1985; McCahon, Whiles, and Pascoe 1989). Scott and Guthrie (1972) presented data showing that the first instar of mosquito larvae accumulated more radiophosphorus than later instars, while the fourth instar concentrated more cesium-137, strontium-90, and tritium than the earlier instars. Thus, the total radiation dose may be related to the biological processes ongoing in different instars as well as the radionuclide concentration in the water. Also, exposure duration is a major factor with respect to all radiation sources, so the length of the life stage must be considered.

Microhabitat may also be an important factor influencing the sensitivity of benthic macroinvertebrates to contaminants. McCahon, Whiles, and Pascoe (1989) reported that cased caddisflies, *Agapetus fuscipes*, were more vulnerable to cadmium as they grew and left their cases to build bigger ones. Cased and uncased larvae would also differ in their vulnerability to external radiation sources.

Interspecific susceptibility of benthic macroinvertebrates to organic and metal contamination has been described by several investigators. In flowing streams and rivers the macroinvertebrate community structure frequently exhibits a predictable graded response to point source pollution. The transition of dominant groups from unimpacted to impacted to recovery zones corresponds closely with the relative tolerance of different taxonomic groups to the metal or organic contaminants. Woodward and Riley (1983) observed the near complete elimination of stoneflies and caddisflies (Plecoptera and Trichoptera) in a stream contaminated by oil field discharge, and a corresponding increase in Diptera. Crustaceans, Cladocerans, mayflies, and some species of Diptera (e.g., Tanytarsiini) are more sensitive to metal contamination than stoneflies, caddisflies, tubificid worms, and several Diptera species (e.g., Orthocladiini) (Clements, Cherry, and Cairns 1988; Winner et al. 1980; Anderson 1979). Within some invertebrate groups, such as mayflies and midges, there is a wide range of sensitivities between species. Therefore, the transition from tolerant to sensitive invertebrates along an impacted stream depends to some extent on which species of mayflies and midges dominate the populations.

There is very little information on susceptibility of different species of benthic macroinvertebrates to radionuclides. King (1964) reported that *Daphnia pulex* bioconcentrated more cesium-137 than *D. magna* apparently due to different capabilities to assimilate this radionuclide from the tagged food source, *Chlamydomonas*. However, both species may be equally sensitive to the effects of ionizing radiation itself In general, the less complex benthic organisms and those with short life cycles are more resistant to ionizing radiation.

As with fish, elevated concentrations of metals, organics, and radionuclides can cause both direct and indirect effects on aquatic macroinvertebrates. Exposure to these contaminants may adversely affect biochemical and physiological processes. Since numerous metabolic (biochemical) processes are the same in fish and invertebrates, the cellular level effects of toxicants are frequently the same. For example, the inhibition of oxidative phosphorylation by high copper levels is common to many types of living organisms. In many cases, the toxicological endpoint, such as behavior or decreased growth, has been described for invertebrates, but the actual cause of toxicity at the cellular level is unknown.

Copper is generally more toxic than cadmium to freshwater macroinvertebrates (Khangarot and Rat 1989). Acute and chronic effects of copper exposure to aquatic insect larvae include reproductive effects such as teratogenicity (Kosalwat and Knight 1987b) and reduced larval growth (Hatakeyama 1989). Clements et al. (1989a,b) describe the indirect toxic effects of copper exposure to aquatic insect communities. They reported that two species of net-spinning caddisflies, *Chimarra* sp. and *Hydropsyche morosa*, were increasingly vulnerable to predation by the stonefly, *Paragnetina media*. It was hypothesized that copper exposure caused behavioral changes which made the caddisflies more vulnerable to predation. Peterson and Peterson (1983, as cited in Clements et al. 1989) introduced evidenced showing that heavy metal exposure altered silk-spinning in Hydropsychidae caddisflies causing structural anomalies to the web. These alterations may have caused *Hydropsyche* to spend more time in the open repairing nets, thus exposing them to increased predation pressure.

The chronic or sublethal effects of cadmium on freshwater invertebrates have been studied by numerous investigators and include reductions in percent emergence, abnormal behavior, reduced ability to construct cases (by caddisflies), and various effects on reproduction (Hatakeyama 1987; Gerhardt 1990; Spehar, Anderson, and Fiandt 1978; McCahon, Whiles, and Pascoe 1989; Williams et al. 1987).

Extensive data concerning the mode of action and general toxicities for xylene and toluene to macroinvertebrates were not encountered; however, some information is available. The mode of action of xylene is not completely understood but increased cellular permeability has been noted by Berry et al. [1978, as cited in Jori et al. (1986)]. Simpson [1980, as cited in Woodward and Riley (1983)] reported that Plecoptera and Trichoptera exposed to oil (xylene and toluene are major constituents) resulted in abnormalities in tracheal gills, which probably resulted in respiratory interference.

Exposure of aquatic macroinvertebrates to radiation sources has been reported to cause genetic anomalies, chromosomal aberrations, development abnormalities, depressed reproduction, decreased swimming activity, and sterilization (Klechkovshkii, Polikarpov, and Aleksakhin 1973; King 1964; Scott and Guthrie 1972). The specific mode of action of ionizing radiation in aquatic macroinvertebrates is the same effect that radiation has on all living organisms.

7.8.2 Toxicity Assessment—Aquatic Invertebrates

7.8.2.1 Toxicity of contaminants

The toxicity of copper, cadmium, xylene, toluene, cesium-137, and strontium-90 to freshwater invertebrates is assessed based on the ambient water quality criteria and the acute and chronic toxicity values reported in the literature. Acute toxicity is most frequently expressed as the 96-h LC_{50}. Chronic toxicity values for freshwater invertebrates are not reported in terms of the NOEL and LOEL as much as for fish, probably because of the shorter life spans and more numerous lifestages. The lowest observable end points may be changes in growth, egg production, emergence success, behavior, mortality, or other measurable signs of stress.

7.8.2.2 Ambient water quality criteria

Ambient water quality criteria for cadmium, copper, and toluene are depicted in Table 7.3. Criteria for xylene, cesium-137, and strontium-90 are not available.

7.8.2.3 Acute and chronic toxicity of cadmium and copper

The acute toxicity values for cadmium are given in Table 7B.5 in Appendix 7B. Values are for 96-h LC_{50} where possible to avoid the confusion of comparing different duration tests. Eisler (1985), Mance (1987), Williams et al. (1986) and others present LC_{50} values for test durations between 24 h and a month or more. The stoneflies (Plecoptera) and caddisflies (Trichoptera) are among the more tolerant groups of invertebrates. The cladocerans, freshwater shrimp, and crustaceans such a *Asellus aquaticus* are the least tolerant, with LC_{50}s frequently less than 1 mg/L. Mayflies and midges (Chironomidae) both exhibit a wide range of sensitivities among the species within these groups (Table 7B.5, Appendix 7B). The sensitivities of the different life stages of the crustacean reflect the higher tolerance of the embryos and adults seen in many benthic invertebrates.

The chronic or sublethal effects of cadmium on freshwater invertebrates have been studied by numerous investigators and include reductions in percent emergence, abnormal behavior, reduced ability to construct cases (by caddisflies), and various effects on reproduction (Table 7B.6, Appendix 7B). Although the 96-h LC_{50} for the fourth instar of the midge larvae *Polypedilum nubifer* was 372 mg/L, reproductive impairment was observed at concentrations as low as 0.02 mg/L. Sublethal exposures in the range of 46 to 186 mg/L reduced case building activities in the caddisfly *Agapetus fuscipes*, and at the highest end of this range the strength of the cases deteriorated. The net spinning capabilities of another caddisfly, *Hydropsyche angustipennis*, were reduced by cadmium concentrations as low as 0.3 mg/L.

In summary, the Plecoptera and Trichoptera are generally tolerant of cadmium concentrations below 10 mg/L, and older larva of chironomids may be nearly as tolerant (Table 7B.6, Appendix 7B). Cadmium concentrations less than 1 mg/L may have chronic effects on other invertebrate groups such as mayflies, some species of chironomids, cladocerans, and snails. The emergence of some species of mayflies, *Leptophlebia marginata* and *Baetis rhodani*, and midge larvae, *Polypedilum nubifer* and *Chironomus yoshimatsui*, may be adversely affected by cadmium concentrations between 0.02 and 0.2 mg/L. Within the literature reviewed, the lowest concentration of cadmium reported to affect aquatic invertebrates was 3 μg/L, which represented an LC_{50} for a mayfly, *Ephemerella* sp. over a 28-day exposure period.

As with cadmium, the range of acute and chronic toxicities of copper is broad for aquatic macroinvertebrates as depicted in Tables 7B.7 and 7B.8 in Appendix 7B. Acute 96-h toxicities range from 16.7 μg/L for the first instar of the chironomid *Chironomus tentans* to 8300 μ/L for the Plecopteran *Acroneria lycorlas* under similar water hardness conditions. The range of acute toxicities within the midge family Chironomidae (16.7 to 1690 μg/L) is as broad as the range of toxicities among the different types of aquatic invertebrates. Similar to cadmium, the stoneflies and caddisflies are relatively tolerant of copper. The different sensitivities of the four instars of *Chironomus tentans* illustrates that the youngest larval stage is the most sensitive. This has also been observed in other insect larvae.

Copper is chronically toxic to mayflies and midges at concentrations between 10 and 25 μg/L. Midge larvae in the taxonomic tribe Tanytarsiini represent some of the more sensitive groups of midges. Caddisflies and stoneflies are very tolerant of copper, especially the older instars. Exposure to copper via ingestion of food is apparently less detrimental and may be insignificant in comparison to exposure to copper in water (Kosalwat and Knight 1987a).

7.8.2.4 Acute and chronic exposure to toluene and xylene

Data regarding the acute and chronic toxicities of organic compounds to aquatic invertebrates are limited (Table 7B.9, Appendix 7B). Benville and Korn (1977) reported acute 96-h LC_{50}s for shrimp as 20 and 4.3 μg/L for benzene and toluene, respectively. LC_{50}s for xylenes were broken down into their meta, ortho, and para isomers; these values were 3.7, 1.3, and 2.0 μg/L, respectively. Twenty-four h LC_{50} for *Daphnia magna* was between 100 and 1000 mg/L (Dowden and Bennett 1965, as cited in Jore 1986). Mosquito larvae, *Aedes aegipti*, appear to be more sensitive with a reported 24-h LC_{50} of 14 mg/L (Berry and Brammer 1977, as cited in Jori 1986). Woodward and Riley (1983) studied changes in macroinvertebrate populations downstream of oil field discharges and stated that oil-in-water concentrations above 24 μg/L were biologically unsuitable. They noted that Plecoptera and Trichoptera were nearly eliminated by oil field contaminants, while one species of mayfly and two species of Diptera became relatively dominant at stations downgradient of the discharge area. Other investigators have also noted the sensitivity of Trichoptera and Plecoptera to oil contaminants (Snow and Rosenberg 1975 and Simpson 1980, as cited by Woodward and Riley 1983).

7.8.2.5 Harmful effects of cesium-137 and strontium-90

These is very little information regarding the harmful effects of cesium-137 and strontium-90 on aquatic organisms. King (1964) reviewed available literature and found a few estimates of roentgen dosages harmful to aquatic organisms, and reported LD_{50} values for crustaceans and fish ranged from 500 to 90,000 R, respectively. A radiation dosage of 70 r/hr reduced swimming activity in the cladoceran, *Daphnia magna*, within 16 days, and an accumulated dose of 10,500 R caused permanent sterility (King 1964). Sterility in the adult mosquito was apparently caused by lower levels; Guthrie and Brust (1971, cited by Scott and Guthrie 1972) estimated that mosquitos were sterilized by a total beta radiation of about 90 rad from the media in which they were reared. Depressed reproduction was observed in *D. magna* populations exposed to 10^{-5} Ci/L radiation for 80 days, and slightly higher levels of strontium-90 + yttrium-90 (10^{-4} to 10^{-3} Ci/L) disrupted development in mollusks (Klechkovskii, Polikarpov, and Aleksakhin 1973). A search of available literature did not produce any radiological effect levels for stoneflies, caddisflies, mayflies, or chironomids.

Aquatic macroinvertebrates are less susceptible to cesium-137 and strontium-90 than fish because of their shorter life cycle and other aspects of their life history and morphology. The exoskeleton of most macroinvertebrates, except mollusks, has a low mineral content and, therefore, does not accumulate strontium-90 as much as fish bone and scales. Also, more than 50% of the radioactivity in macroinvertebrate larvae may be lost during molting and emergence, and considerable radioactivity may also be lost by elimination (Wilhm 1970; Harvey 1971). Since up to half of the body burden of radioactive material is lost at each molting, the total accumulation of both adsorbed and absorbed radionuclides over the entire life cycle is significantly reduced. Also, the harmful effects of radiation on a single generation can be offset by the high reproductive potential of many aquatic macroinvertebrates.

7.8.3 Evaluation of Potential Impacts—Aquatic Invertebrates

As discussed in Sect. 7.7.3, maximum concentrations of cadmium and copper in surface water at WAG 6 exceeded the Federal Ambient Water Quality Criteria for both acute and chronic exposures. However, several factors influence the potential toxicity to aquatic macroinvertebrates including water hardness, pH, total organic or humic acid concentrations, adsorption/desorption, and biodegradation. Biological parameters such as the presence of metal-binding proteins also influence bioavailability and toxicity. Intraspecific susceptibility to toxic effects depends on lifestage and body size, with smaller organisms and earlier instars generally being more sensitive.

The maximum concentration of cadmium in surface water at WAG 6 was 30 μg/L, which exceeds the level reported to have caused reproductive impairment in the midge larvae *Polypedilum nubifer*. The emergence of some species of mayflies and midge larvae may also be adversely affected by concentrations lower than those found in WAG 6. The lowest concentration of cadmium reported to affect aquatic invertebrates was 3 μg/L, which was the LC_{50} for a mayfly over a 28-day exposure period. Chronic effects of cadmium on macroinvertebrates include reductions in percent emergence, abnormal behavior, reduced ability to construct cases, and various effects on reproduction. In general, stoneflies (Plecoptera) and caddisflies (Trichoptera) are more tolerant of cadmium contamination, whereas the cladocerans, freshwater shrimp, and crustaceans are the least tolerant. (See Tables 7B.8 and 7B.10.)

As with cadmium, the range of acute and chronic toxicities for copper is broad for aquatic macroinvertebrates. The maximum concentration reported in surface water at WAG 6 was 88 μg/L, which is higher than the level known to cause adverse effects to some aquatic macroinvertebrates. Copper is chronically toxic to mayflies and midges at concentrations between 10 and 25 μg/L. Conversely, caddisflies and stoneflies are relatively tolerant of copper, especially the older instars. Copper is generally more toxic than cadmium to fresh water invertebrates. (See Tables 7B.8 and 7B.10.) Acute and chronic effects include teratogenicity, reduced larval growth, behavioral changes, and the inhibition of oxidative phosphorylation.

The presence of VOCs, such as toluene and xylene, is not expected in surface water samples because of their rapid volatilization from shallow surface water. However, concentrations of toluene in groundwater samples from WAG 6 exceeded the ambient water quality criteria for chronic exposure (2000 μg/L compared to 20 μg/L). The maximum concentration of xylene was 3200 μg/L. These levels, in conjunction with historic records showing the large quantities disposed in WAG 6, indicate that the potential exists for contamination of surface water during recharge. The acute 96-h LC_{50} for shrimp was 4.3 μg/L for toluene and 3.7, 1.3, and 2.0 μg/L for meta, ortho, and para xylenes, respectively. The levels found in groundwater at WAG 6 far exceed those values. However, because of the rapid volatilization of toluene and xylene from shallow surface waters and the high reproductive potential of macroinvertebrates, these VOCs should not pose a major threat to the aquatic invertebrates on WAG 6.

Effects of radiation on aquatic macroinvertebrates include genetic anomalies, chromosomal aberrations, development abnormalities, depressed reproduction, decreased swimming activity, and sterilization. However, aquatic macroinvertebrates are less susceptible to cesium-137 and strontium-90 than fish for several reasons: (1) they have a shorter life cycle; (2) the exoskeleton of most macroinvertebrates, except mollusks, has a low mineral content and does not accumulate strontium-90 as much as fish bones and scales; (3) more than 50% of the radioactivity in macroinvertebrate larvae may be lost during molting and emergence; and (4) the harmful effects of radiation on a single generation can be offset by their high reproductive potential (Krumholtz 1954, Harvey 1971). Nonetheless, the strontium-90 levels at WAG 6 are quite high and could have some adverse effects on the aquatic macroinvertebrates.

7.9 SUMMARY

The results of this environmental evaluation indicate that adverse effects on the flora and fauna at WAG 6 are likely in the absence of any remediation. Certain assumptions were made in performing this evaluation.

1) Potential effects on WAG 6 inhabitants were evaluated under the hypothetical situation where fences were removed and animals were free to move on-WAG at will.

2) WAG 6 was allowed to undergo natural succession.

3) Because it was necessary to compare contaminant levels to standards for the protection of human health, it was assumed that nonhuman species received the same time of exposure and dose as humans and that nonhuman species exhibit the same sensitivity to contaminants as humans.

4) The contaminant effects reported from other field and laboratory studies would be exhibited by similar species on WAG 6.

Target species were selected to represent different trophic levels and various exposure pathways. The tulip poplar is ubiquitous on the ORR and is commonly found in disturbed areas such as WAG 6. The white-tailed deer is a large herbivorous mammal that is actually hunted on portions of the reservation. The red-tailed hawk is a carnivorous raptor, and the raccoon is omnivorous, consuming vegetation and both terrestrial and aquatic prey. Bluegill sunfish and fathead minnows were selected as key species because both are common in the WOC watershed. Benthic macroinvertebrates are relatively good integrators of contaminant releases because of their abundance, relatively sedentary nature, and close association with the substrate. Another important factor in selecting these target species was the availability of relevant information regarding effects of the selected contaminants on the organisms.

Factors influencing the selection of contaminants for this environmental evaluation were: (1) their known or suspected potential to cause ecological harm, (2) their known or suspected levels at WAG 6, (3) regulatory standards and guidelines, and (4) availability of information on potential adverse effects on key species or similar species.

Of the three radionuclides selected for this environmental evaluation, strontium-90 likely poses the greatest threat to fauna at WAG 6. As an analog of calcium, it is readily absorbed and subsequently deposited in bone tissue, where it can result in tumors and leukemia. Deer can ingest strontium-90 from contaminated vegetation, as was evidenced at SWSA 5 where contaminant concentrations in honeysuckle and blackberry were sufficiently high that the strontium-90 levels in the bone of a 45-kg deer could easily exceed the confiscation limit of 30 pCi/g by browsing there for a period of 1 week to 1 year (Garten and Lomax 1987). Because of the large inventory of strontium-90 at WAG 6, species from the lower trophic levels such as voles, shrews, and rabbits are likely to suffer adverse effects themselves, to bioconcentrate strontium-90 in their bone tissue, and, subsequently, to cause adverse effects in the red-tailed hawk and raccoon that prey upon them.

Cesium-137 is deposited in soft tissues, particularly liver and kidney. The most probable pathway through which cesium-137 may enter the food chain is through ingestion of contaminated soil, where it is tightly bound to the illitic clay particles. White-tailed deer may incidentally ingest soil while grazing. Earthworms also may ingest contaminated soil. They can then be eaten by shrews, which are subsequently consumed by red-tailed hawks. Similarly, voles may ingest soil and then be consumed by raccoons. These small, burrowing animals may also be directly exposed to cesium-137 as they move through the wastes. Contaminant concentrations found in soil/sediment and surface water at WAG 6 were above the levels found in background; however, there are no regulatory standards for cesium-137. There is sufficient evidence from the scientific literature to substantiate that cesium-137 can move through different components of the ecosystem but not to state definitively what effects would be expected from those concentrations.

The maximum tritium level at WAG 6 was 14,000,000 pCi/L, compared to the SDWA MCL of 20,000 pCi/L. The primary effect from tritium will be from ionizing radiation because tritium does not bioaccumulate, quickly reaches equilibrium in an organism, and has a relatively short half-life. The specific effects from the ionizing radiation will be dependent upon the dose.

Cadmium levels exceeded the SDWA MCL and the suggested standard from the U.S. Fish and Wildlife Service. This metal is also known to bioconcentrate in liver and kidney tissues of shrews by factors ranging from 15 to 33 times the levels found in the soil (Talmage and Walton 1990). Not only can cadmium adversely affect the soft tissue of shrews but also those of the red-tailed hawks that prey upon them. The greatest potential for adverse effects to raccoons is through ingestion of contaminated surface water by the raccoons themselves and by voles upon which they prey and from ingestion of contaminated aquatic organisms. Although cadmium is taken up by plants when the soils are acidic, ingestion of vegetation is not expected to be a major source of contamination to the deer. Incidental ingestion of soil may provide an additional pathway.

Hexavalent chromium easily penetrates biological membranes and has mutagenic, carcinogenic, and teratogenic effects. Chromium concentrations were higher in groundwater than in other media sampled--216 μg/L compared to the SDWA MCL of 100 μg/L. However, groundwater is generally not available to terrestrial fauna, and this should not pose a major threat to WAG 6 species. Chromium did exceed the Federal Ambient Water Quality Criteria for the protection of freshwater organisms. If it is assumed that this was all hexavalent chromium, aquatic species in the food chain to the raccoon could be adversely affected.

7.10 LIMITATIONS OF ANALYSIS

The primary limitation of the environmental evaluation for WAG 6 is the lack of site-specific and contaminant-effects data. The rationale for the methodology and approach for this environmental evaluation were presented in Sect. 7.3. Given the present disturbed nature of WAG 6, obvious, visible effects would not be expected, and it is extremely difficult to describe an anomaly between a contaminated area and a reference area.

The lack of specific quantifiable data resulted in a reliance on data found in the literature and data derived from studies on laboratory animals or agricultural crop species. The problems associated with making inferences from laboratory studies to the natural world, (specifically WAG 6), are numerous:

- Toxicity in a natural setting is independent of a dosing schedule. An organism in nature is exposed continuously throughout the daily feeding period and not in a single oral dose, as is typical in a laboratory (IT Corp. 1990).

- Feed consumption by organisms in the natural world will vary with body weight, ambient temperature, disease conditions, type of feed, and a number of other natural conditions (i.e., hibernation, growth, location, and reproduction). In a laboratory setting, an average daily food intake, which assumes no variation, is used for calculations.

- Endemic fauna are assumed to have unlimited access to a constant quantity of water and, therefore, constant daily uptake. Water consumption is usually more variable than food consumption. Factors such as ambient temperature and access to watering holes play a major role in determining the quantity of consumption in wild species.

- The influence of water quality on toxicity and/or bioavailability of contaminants is important. Since the water quality of WAG 6 is different from other laboratory and field studies, and variable through time, there is uncertainty in determining real on-site toxicity.

- In nature, there are mixtures of metals, organics, and radionuclides that constantly change through space and time. Data from laboratory studies usually reflect a single contaminant.

- There are wide variations in toxicity among different species, even closely related species. Therefore, there are uncertainties when the toxicity of species A from the literature is used to estimate impacts on species B at WAG 6.

- There are very limited data available to translate toxicity effects on individuals or populations to higher ecological levels, such as communities and ecosystems. For example, a 30% mortality in a population of short-lived invertebrates may have no real effect on an ecosystem because of the high reproductive capacity of that species; however, a 30% mortality in a species of a higher trophic level could impact the ecosystem.

Section 7 Tables

Table 7.1. Estimated diets for selected target species

Species	Average body weight (kg)	Water (mL/d)	Soil (g/d)	Inhalation of air (L/d)	Arthropods Earthworms (g/d)	Vegetation (g/d)	Mammalian prey (g/d)	Aquatic prey (g/d)
Red-tailed hawk	0.75	45		360			112	
White-tailed deer	50	3000	5	22,454		3500		
Eastern cottontail	1.45	210	0.145	470		87		
Shorttail shrew	0.02	3.2	0.002	10	22			
Raccoon	10					300	100	100

Sources: IT Corporation/Midwest Research Institute (MRI). 1990. *Final Environmental Impact Statement, Occidental Chemical Corporation Liquid Waste Incinerator Permit Modification, Niagara Falls Plant, Niagara, New York,* Vol.II.

Newell, A. J., D. W. Johnson, and L. K. Allen. 1987. *Niagara River Biota Contamination Project: Fish Flesh Criteria for Piscivorous Wildlife*, TR 87-3, Division of Fish and Wildlife, Bureau of Environmental Protection, New York State Department of Environmental Conservation.

Ewer, R. F. 1973. *The Carnivores*, Cornell University Press, Ithaca, New York.

Table 7.2. Comparision of contaminant levels to regulatory standards and background

Contaminant	Units	Maximum	Minimum	Mean	No. of hits	Std./ref.[a]	Background
				Soil/Sediment			
Cs-137	pCi/g	4.57	0.24	1.05	22/30		0.04-0.04 - sediment 0.34-0.38 - soils
Sr-90	pCi/g	25.50	0.79	9.26	3/3		0.92-0.92 - sediment 0.70-0.80 - soils
H-3	pCi/g	840	66.7	300.7	13/30		
Cd	mg/kg	4.0	1.10	2.72	26/32	40/2	3.30 - sediment 2.50-4.4 - soils
Cr	mg/kg	42.4	13.1	25.29	32/32	400/2	5.0-87.0 - sediment 19.8-36.9 - soils
				Surface Water			
Cs-137	pCi/L	11.2	11.2	11.2	1/64		None
Sr-90	pCi/L	10,222.0	5.1	980.69	42/91	8/1	None
H-3	pCi/L	14,000,000.0	1080	1,185,279	104/104	20,000/1	749-877
PCE	μg/L	83	1.0	26.73	15/59	5/1	None
TCE	μg/L	2200	1.00	257.88	26/59	5/1	None
				Surface Water-Filtered			
Cd	μg/L	2.0	4.6	15/53	5/1		None
Cr	μg/L	10.3	20.11	334/53	100/1		None

Table 7.2. (continued)

Contaminant	Units	Maximum	Minimum	Mean	No. of Hits	Std./ref.[a]	Background
				Surface Water-Unfiltered			
Cd	µg/L	30	2.0	5.42	24/119	5/1	None
Cr	µg/L	58.3	10.2	22.17	60/119	100/1	None
				Groundwater			
Sr-90	pCi/L	5400	5.98	1836.16	9/63	8/1	None
H-3	pCi/L	4,300,000	688	417099.46	56/64	20,000/1	2.67
PCE	µg/L	6800	1	2071.5	14/63	5/1	1-5
TCE	µg/L	1300	6	560.27	11/62	5/1	None
				Groundwater-Filtered			
Cd	µg/L	86	2.3	9.07	32/86	5/1	
Cr	µg/L	103	10	25.68	42/86	100/1	
				Groundwater-Unfiltered			
Cd	µg/L	94.5	2.1	10.83	30/91	5/1	2-3
Cr	µg/L	216	10.1	55.83	56/90	100/1	None

[a] Reference Codes:

1. Safe Drinking Water Act Maximum Contaminant Level (MCLs) (in Table 4.3 as RCRA Subparts).
2. Federal Register - July 27, 1990 - Appendix A, Action Levels.

Table 7.3. Acute and chronic ambient water quality criteria for selected contaminants

Contaminant	Hardness (mg/L $CaCO_3$)	Acute Criteria (μg/L)	Chronic Criteria (μg/L)
Cadmium	50	1.8	0.66
	200	8.6	2.0
Copper	50	9.2	6.5
	200	34.0	21.0
Benzene	N/A	17,500	20.0[a]
Toluene	N/A	5,300	20.0[a]

[a] Criterion proposed by Jori et al. (1986) for long-term exposure to xylenes.

Table 7.4. Acute toxicity (LC_{50}) of cadmium to bluegill and fathead minnows

Fish species life stage	Water hardness (mg/L-$CaCO_3$)	96-hour LC_{50} (μg/L)	Reference
Bluegill			EPA
Yearling	207	21,100	Eaton 1974
Adult	18	2300	Bishop and McIntosh 1981
Adult	20	1940	Pickering and Henderson 1966
Fathead Minnow			
Immature	202	30,000-32,000	Pickering and Gast 1972 (static tests)
Immature	202	1400-19,000	Pickering and Gast 1972 (flow-through tests)
Larvae	44	13.2	Spehar and Fiandt 1986

Section 7 Figures

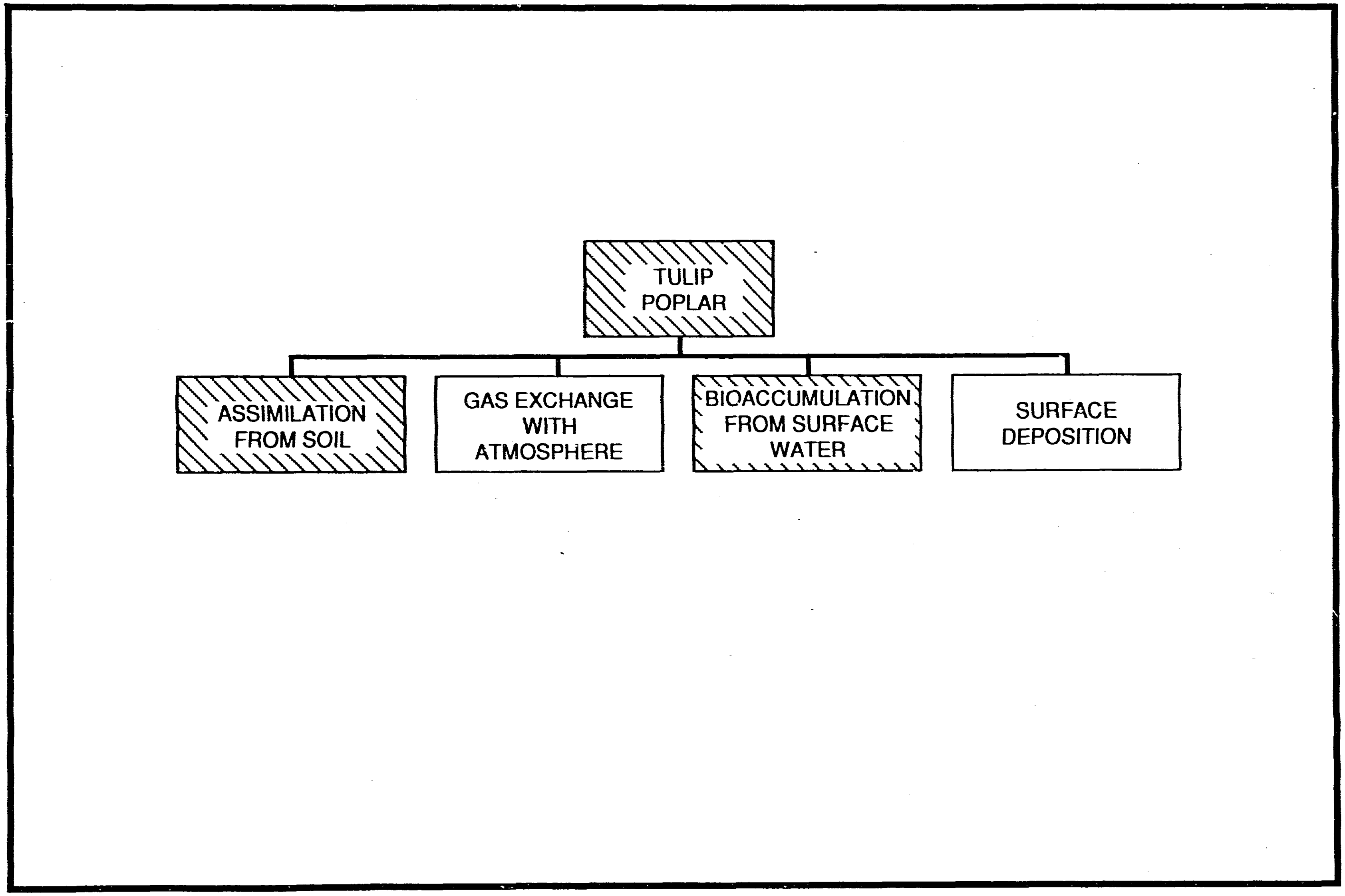

Fig. 7.1. Exposure scenario with the tulip poplar as the target species. (Highlighted boxes indicate primary exposure routes.)

5.18B 4104.78

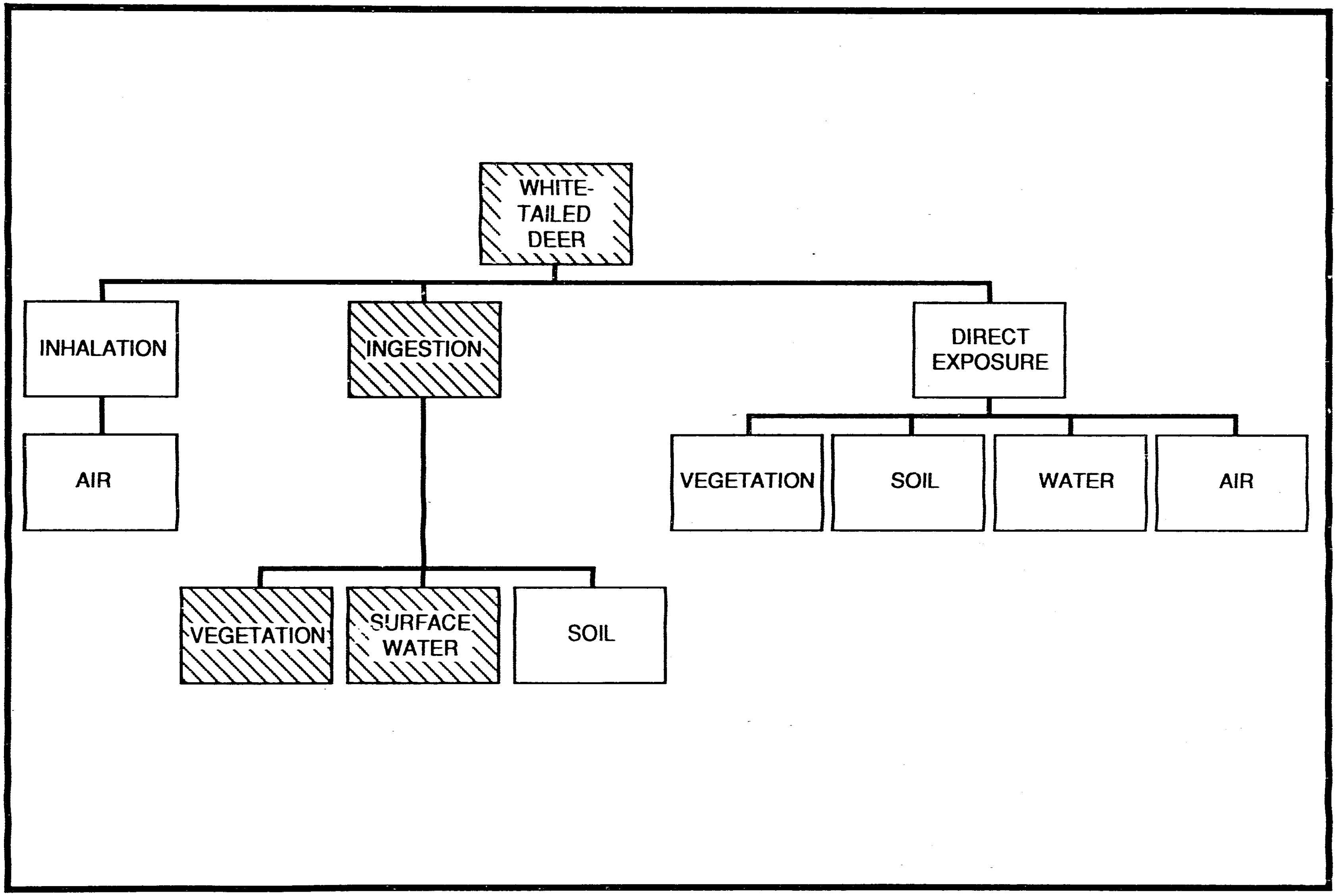

Fig. 7.2. Exposure scenario with the white-tailed deer as the target species. (Highlighted boxes indicate primary exposure routes.)

5.18B 4104.76

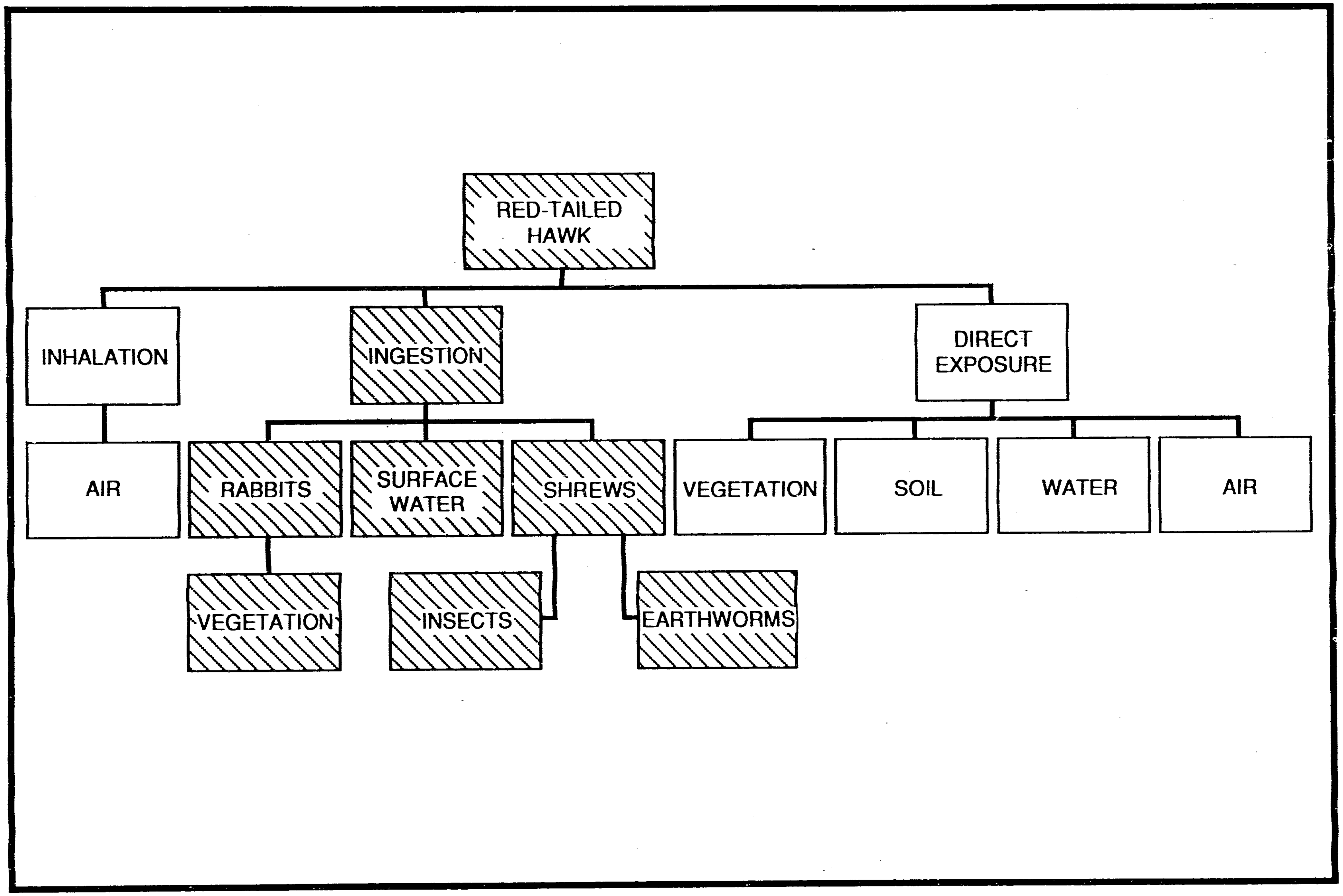

Fig. 7.3. Exposure scenario with the red-tailed hawk as the target species. (Highlighted boxes indicate primary exposure routes.)

5.18B 4104.77

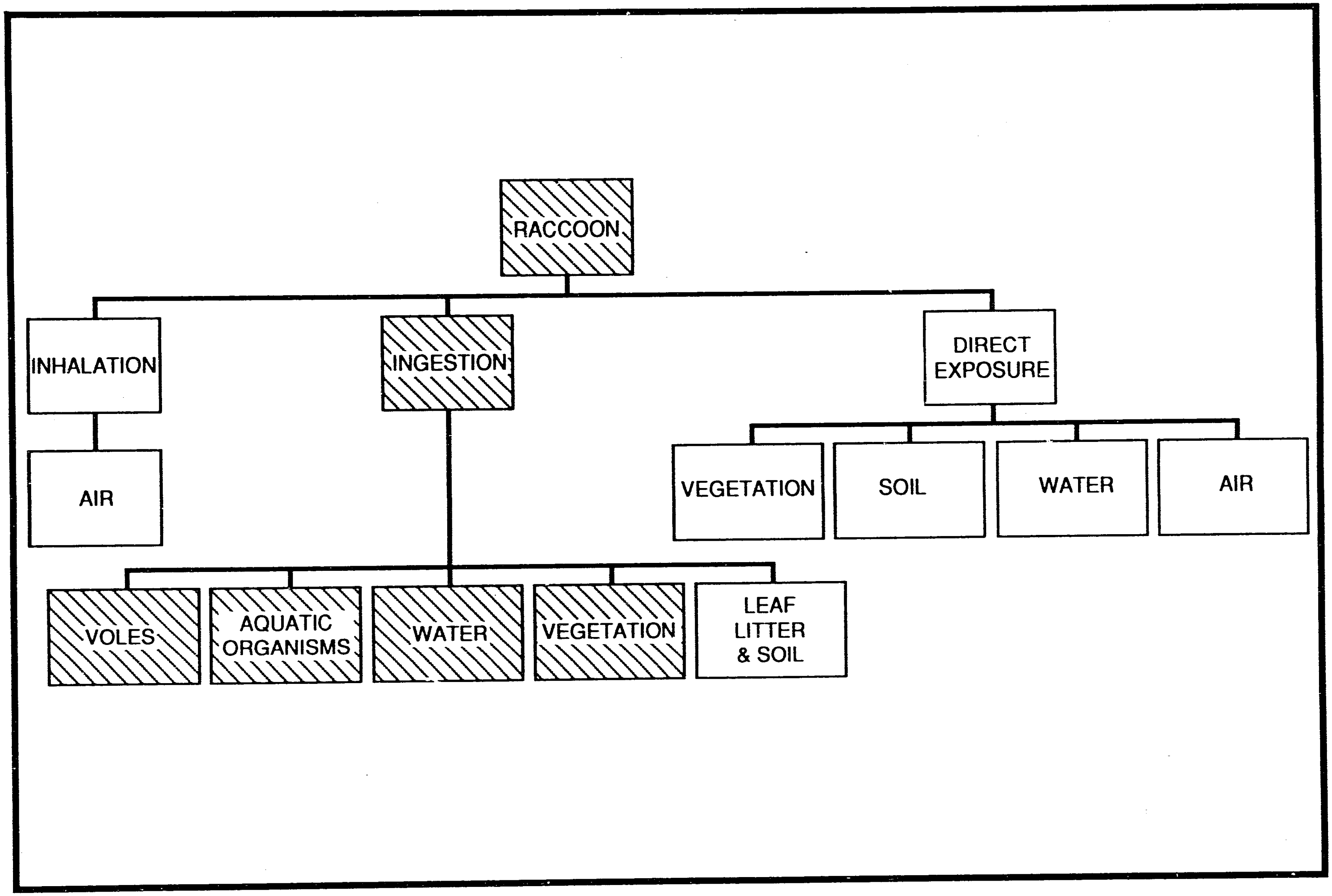

Fig. 7.4. Exposure scenario with the raccoon as the target species. (Highlighted boxes indicate primary exposure routes.)

5.18B 4104.75

8. PRELIMINARY DEVELOPMENT AND SCREENING OF ALTERNATIVES

8.1 INTRODUCTION

The purpose of this section is to develop alternatives that will meet the remedial action objectives for WAG 6. This section concludes with a preliminary list of site-wide alternatives. Each alternative will be defined and evaluated in detail in an integrated Feasibility Study/Corrective Measures Study/Environmental Assessment (FS/CMS/EA) (anticipated to be submitted to EPA and TDEC in February 1992. The EPA Superfund process was employed to develop the alternatives for RCRA closure and corrective action for WAG 6. The process is described in detail in the *Guidance for Conducting Remedial Investigations and Feasibility Studies Under CERCLA* (EPA 1988a). The guidance is substantially the same as the guidance for RCRA corrective actions as described in the *Interim Final RCRA Corrective Action Plan* (EPA 1988f). The process of developing alternatives is consistent with NEPA requirements.

8.1.1 Alternative Assessment Process

The following is a generalized discussion of the steps followed in performing the alternative development and assessment process for WAG 6, which is begun in this section and will be finalized in the FS/CMS/EA. The process is illustrated in Fig. 8.1.

Objectives for the remediation of the site have been formulated based on the information gathered during the RFI. The goal of this section of the report is to develop a range of alternatives that can meet those objectives so that a cost-effective remedy can be identified. To accomplish this, many remediation technologies and process options have been identified and evaluated for technical feasibility. The general relationship among the different elements is illustrated in Fig. 8.2. The technologies and process options were then screened to eliminate those obviously not suited to WAG 6 conditions or contaminants (see Sects. 8.4 and 8.5). The retained technology process options were then combined into alternatives for waste sources and groundwater. These alternatives were evaluated for implementability, effectiveness, and cost to reduce the number of alternatives (Sect. 8.6) and yet retain a range of reasonable alternatives. The last phase of the alternative assessment presented in this report is the assembly of waste source and groundwater alternatives into site-wide alternatives.

This alternative assessment for WAG 6 recognizes that unknowns exist in anticipated site conditions or in the expected performance of an alternative so that modifications or adjustments to the alternative may be required. Because there are numerous potential interpretations concerning existing and future site conditions and a variety of alternative responses for each, it is not feasible to specifically address all potential interpretations. The most credible interpretation and major variations on this interpretation have been considered while developing the alternatives. Monitoring of site conditions to identify changes or variations in site conditions will be part of every developed alternative.

8.1.2 Scope of Alternative Assessment

Based on the description of site contamination presented in Sect. 4 and the results of the base line risk assessment (Sects. 6 and 7), the alternative assessment has focused on the following.

- **Sources.** WAG 6 has SLB units, including high-activity trenches, low-activity trenches, biological trenches, asbestos trenches, silos, auger holes, and test areas. Although many different types of waste are buried in a variety of containers, it is felt that technologies considered will be similar because the sources are all buried. To simplify the alternatives assessment, all buried wastes and corresponding soil and leachate will be considered together, although they may not all be remediated by the same technology. The IWMF, which is still under construction, is not included under the scope of this RFI.

- **Groundwater.** Site characterization data presented in Sect. 4 show that the groundwater contains volatiles (notably vinyl chloride, 1,1-dichloroethene, 1,2-dichloroethene, and trichloroethene) at levels exceeding MCLs and health-based levels. Metals, especially barium, beryllium, and cadmium, were also detected above MCLs and site reference levels. Also detected were radionuclides, with tritium, strontium-90, and cobalt-60 being most frequently detected above levels of concern. Remedial actions for groundwater will not focus on off-WAG remediation since that is part of the cleanup of other WAGs (WAGs 2 and 7).

The EWB was determined to have no current risk and therefore is considered a potential future receiving water body and not a source of contamination. Therefore, no alternatives are developed for the EWB.

8.2 REMEDIAL ACTION OBJECTIVES

The general goal of WAG 6 closure and remedial action is to protect human health and the environment by providing a long-term closure of the disposal units to segregate the waste from the environment for as long as the waste remains a threat to human health and the environment and by remediating unacceptable levels of environmental contamination. This is consistent with the RCRA requirements [40 CFR 265 Subpart G (265.111)] presented in the RCRA Closure Plan that a facility be closed in a manner that controls, minimizes, or eliminates, to the extent required to protect human health and the environment, post-closure escape of hazardous waste, hazardous constituents, leachate, contaminated runoff, or hazardous waste decomposition products into the environment. Alternatives are to be developed for the SLB wastes and groundwater.

Objectives for waste sources and groundwater are as follows.

- SLB wastes: to provide protection of public health and the environment by limiting direct contact/emissions, surface runoff, and leachate generation from on-WAG sources and associated contaminated leachate and soil.

- Groundwater: to provide protection of public health and the environment by mitigating the effects of the migration of contaminants off WAG and prevent future use of on-WAG contaminated groundwater.

During the development of alternatives for site closure and remediation (including the definition of action levels and treatment options), a variety of regulations and standards must be considered. This requirement to consider all applicable or relevant and appropriate requirements (ARARs) is driven by §121 of CERCLA. The potential ARARs for WAG 6 are presented in Appendix 8A.

8.3 GENERAL RESPONSE ACTIONS

To protect human health and the environment from the contaminants at WAG 6, general response actions have been developed with respect to the site's source areas and groundwater. The actions are intended to address the environmental and public health impacts of the contaminants, possible migration pathways, and exposure routes for each source area and groundwater. Combinations of response actions may prove to be more effective in meeting the objectives.

The general response actions include the "No Action" category as mandated by the NCP. This No Action response will serve as a base line comparison against which the other developed actions can be compared. Table 8.1 summarizes the general response actions for each media/source and its risk type.

8.4 IDENTIFICATION AND SCREENING OF TECHNOLOGIES AND PROCESS OPTIONS

The specific technologies and process options that are potentially applicable at WAG 6 are a subset of the universe of remedial technologies. The technologies and process options that have been identified are those that are potentially applicable to the source areas and groundwater present and are not still in laboratory development. As shown in Fig. 8.2, the remedial technologies are broad categories, each of which addresses a similar problem. For example, the general response action, Containment, includes the remedial technologies capping, vertical barriers, etc. Each of these remedial technologies, in turn, comprises several process options that might be used. For example, capping comprises clay cap, multilayer cap, asphalt cap, etc.

The screening process eliminates technologies from further consideration on the basis of technical implementability. Two general screening criteria are used to eliminate process options or technologies that are not considered applicable for the site:

- Applicability to site physical conditions (and their ranges)

- Applicability to type and combinations of contaminants (and their ranges)

Table 8.2 identifies site and contaminant characteristics applied in screening potential technologies and process options. These criteria have been applied to the technologies and process options developed for WAG 6, and brief results of the screening are presented in Fig. 8.3.

8.5 DISCUSSION OF RETAINED TECHNOLOGIES

Each technology retained for further evaluation is described below in terms of relevant aspects of its implementability, effectiveness, and cost. In this discussion, a representative process encompassing several similar process options is often carried forward for further evaluation. This limits the number of options used in alternative development without losing the flexibility of reevaluating these options at a later phase in the FS or during the design. Options that are extremely difficult to implement or that do not offer significantly improved protection, yet are very expensive, have not been carried forward for further consideration (see Fig. 8.3).

8.5.1 Institutional Controls—Access and Use Restrictions

The objectives of access and use restrictions are to prevent prolonged exposure to contaminants, to control future development or disturbance of the site, and to prevent installation of water supply wells downgradient of or within the contaminated area. These objectives are accomplished by placing site usage and deed restrictions on property within potentially contaminated areas and fencing the site. DOE already maintains significant access restrictions for security. WAG 6 is fenced and has significant use restrictions, and the effectiveness of the existing access restrictions is high.

This technology would be implemented after DOE control of the facility is released, and its future effectiveness would depend upon its continued implementation. Access restrictions are subject to changes in political jurisdiction, legal interpretations, and regulatory enforcement. If implemented correctly, access restrictions provide moderate protection against direct contact with contaminants at low cost. Access restrictions will be carried forward for further evaluation.

8.5.2 Containment

8.5.2.1 Capping

Caps are intended to minimize any or all of the following:

- Infiltration of surface water and subsequent leachate generation caused by water percolating through the waste
- Release of contaminants by wind or water erosion
- Surface radiation levels
- Direct contact with surface and subsurface contaminants by site intruders

Capping technologies include single-layer and multilayer systems, both of which may involve clearing, grubbing, and grading before placement of the cap. In general, capping is performed when extensive subsurface contamination at a site precludes excavation and removal of wastes because of the potential hazards or costs, which may be the case at WAG 6.

The main disadvantages of capping are the need for long-term maintenance and uncertain design life. Caps must be periodically inspected for settlement, erosion, and invasion by animals and deep-rooted vegetation. If not stabilized before capping, trenches in WAG 6 might subside significantly as a result of degradation of trench wastes. Dynamic compaction and trench grouting are being demonstrated at ORNL as techniques for stabilizing trenches prior to capping and are discussed later in this section.

The presence of relatively high groundwater levels in the lower areas of WAG 6 might limit a cap's ability to significantly decrease groundwater contamination in these areas. In such cases, other technologies could be necessary to lower the groundwater levels below the trench bottoms (e.g., trench drains). Capping with no or minimal excavation limits worker risk and the difficulty in locating acceptable alternate permanent disposal locations over excavation and off-WAG disposal. Both a clay cap and a multilayer cap were retained after the process option screening based on technical implementability.

Clay cap. Although a single-layer clay cap is low in cost and easy to implement, it is unreliable and lacks durability. Local freeze-thaw and wet-dry cycles create fractures in the clay, reducing its effectiveness as a low permeability barrier and requiring frequent maintenance. Therefore, a multilayered cap will be considered as a representative process option.

Multilayer caps. Multilayer caps combine one or more layers of low-permeability material with other layers serving various design functions. Multilayer caps have three distinct types of layers that can be combined in a number of ways, (1) surface protection layer, (2) drainage layer, and (3) barrier layer.

The topsoil and fill layer separates the low-permeability or barrier layer from the environment, which protects the barrier layer from factors such as frost damage, maintenance vehicles, and burrowing animals. Topsoil also supports vegetation, which helps minimize erosion.

The drainage layer, which typically consists of clean sand and perforated drain pipes or a geonet, limits water infiltration from the surface and reduces the amount of water that could penetrate the barrier layer. The drainage layer is covered with a geotextile filter, which reduces clogging by fine-grained soil.

The barrier layer minimizes infiltration of water into the waste mass and the release of radioactivity to the atmosphere. Both clay and synthetic membranes are used as low-permeability barriers. There are many different kinds of clay, most of which can be compacted to provide a permeability of 10^{-7} cm/s. The performance life of a protected clay barrier is expected to exceed 30 years, unless large differential settlements occur. At WAG 6, settlement would be a concern with wastes that were containerized before burial or that are biodegradable.

When properly installed and protected, synthetic membranes are impermeable. The performance life is uncertain because of limited operational history.

A variety of multilayer caps will be considered for WAG 6 as a whole, with different caps used as appropriate for the different burial areas. The details of the cap(s) and their application will be developed as part of the alternatives. Any caps constructed over biodegradable material may include gas venting or collection to prevent gas buildup. (See Sect. 8.5.3.5, Vapor/Gas Collection.)

8.5.2.2 Vertical barriers

Vertical barriers control the horizontal flow of groundwater. If the barrier is hydraulically upgradient, groundwater is diverted around the waste. If downgradient or around the waste, the vertical barrier helps to contain or control the movement of contaminated groundwater. Potentially applicable vertical barriers include slurry walls and imposed hydraulic barriers.

Slurry walls. Two types of slurry walls—soil-bentonite or diaphragm—are under consideration. Either type would be constructed by excavating a trench that is filled with a bentonite slurry to provide temporary stabilization of the vertical trench walls, especially below the water table. For a soil-bentonite slurry wall, the trench is backfilled with a mixture of bentonite and soil. For a diaphragm wall, concrete is placed into a trench containing a bentonite slurry usually in conjunction with steel reinforcement. The concrete displaces the bentonite slurry and is allowed to harden. Typically the concrete is reinforced to provide additional strength.

Both soil-bentonite and diaphragm walls are effective in controlling the flow of groundwater. When used downgradient of a source area for the purpose of containing contaminated groundwater, they must be keyed into an impervious stratum or confining layer. They also need to be used in combination with an interior groundwater collection system to provide a negative hydraulic gradient to limit leaking through the wall. Because there is no known, confining layer at WAG 6, slurry walls in this application will not be considered.

Another application of slurry walls is the installation of shallow walls upgradient of buried sources to divert groundwater around the sources. The difficulty is that a very carefully designed system of walls is needed to prevent the diverted groundwater from entering other buried sources. Because the diversion of shallow groundwater would more effectively be accomplished if the groundwater were removed instead of just diverted, hydraulic barriers in combination with shallow slurry walls will be considered over slurry walls alone.

Hydraulic barriers. Another applicable process option under the vertical barrier technology is a hydraulic barrier. Well systems and drains may be used to reduce the flow of groundwater through waste and contaminated soil. The drains or wells are positioned upgradient of the sources and through the collection and removal of groundwater a barrier to flow is produced. However, because of hydrogeologic conditions an extraction well system may not be feasible at WAG 6. The site's subsurface is complex and the bedrock is fractured. Collection trenches are appropriate for such site conditions, because, regardless of subsurface conditions, they can intercept groundwater flowing through an area and prevent it from entering a waste disposal area. A French drain at WAG 6 has been successful in collecting groundwater. However, the system performance has diminished over less than 10 years, and this problem would need to be addressed in any trench design. Hydraulic barriers represented by collection drains will be carried forward for alternative development.

8.5.2.3 Surface controls

There are a number of surface controls potentially applicable for WAG 6. These are discussed below.

Grading and revegetation. Grading is used to modify the natural topography and surface characteristics of the site to control ponding, runoff, and infiltration. Revegetation stabilizes the site surfaces and decreases wind and water erosion, contributing to the development of a naturally fertile and stable surface environment. In addition, vegetative cover helps decrease infiltration of surface water by maximizing evapotranspiration. The option involves seeding, fertilizing, and watering the area until a stand of vegetation has been established. Grading and revegetation are easily implemented and are relatively inexpensive, effective options, although maintenance costs (e.g., mowing) will be incurred over the life of the alternative.

Diversion and collection. Surface water diversion and collection methods are effective and relatively inexpensive. Dikes and berms (well-compacted earthen ridges or ledges) isolate areas from runoff. Ditches and trenches collect and/or convey water around, through, or from an area. This option could be used as a secondary technology for control of surface runoff during construction or from a cap.

Dynamic compaction. Dynamic compaction involves dropping a weight on buried waste to compact it and reduce voids in the SLB waste. Filling in the voids diminishes the likelihood of subsidence, which could result in failure of a cap. This provides control of the surface condition of the buried waste. Dynamic compaction has been tested on WAG 6 trenches and has been shown to be effective in reducing estimated voids. Monitoring of the surface of the trench showed no release of volatiles or radioactivity. However, some dust was produced, which might require control in any site-wide application. This process option is applicable for buried waste materials that, if excavated, might cause unacceptable worker health risks. However, use on containerized waste may result in the rupturing of the containers.

Surface sealing. The purpose of surface sealing is similar to that of capping; i.e., to prevent surface water infiltration, isolate waste, and control erosion. However, compared to capping, this process option is temporary and has limited effectiveness. Surface sealing will not be retained for further evaluation.

In situ grouting. This process involves injecting grout into voids in SLB waste. If the grouting is successful, the material is solidified with sufficient structural integrity to support a cap, thereby controlling surface conditions of the buried waste. The solidified material may also reduce mobility of contaminants. This process option is applicable for buried waste materials that, if excavated, might cause unacceptable worker health risks.

For fine-grained soil with small pores, chemical grouts are better suited. For coarse-grained materials, particulate grouts are suitable. Although this process option is experimental, in situ grouting has been conducted at WAG 6 on existing trenches. The process appears to be relatively inexpensive compared to excavation and disposal and presents fewer health risks.

Grading, revegetation, and diversion/collection systems will be carried forward for use either with capping (to control SLB wastes/soil), or as part of an alternative for control of surface water. Dynamic compaction and in situ grouting will also be retained as methods for structurally stabilizing buried waste before capping. They would be used only with alternatives that involve capping of SLB buried waste/soil.

8.5.2.4 Dust and vapor suppression

The control of dust and vapors is a secondary activity that requires consideration with every technology that would disturb the waste. Various agents (water, polymer, foam, or hygroscopic salts) can be spread over disturbed areas to reduce the dust and vapor migration. Membranes and tarps can also be used. The process option used would be a design decision, but methods would be selected that would minimize environmental impacts yet be cost-effective.

8.5.3 Removal

8.5.3.1 Drum and debris removal

Among the buried wastes are casks, pieces of large equipment, old laboratory furniture, and a variety of other contaminated debris. To complete a remedial action, it may be necessary to use specialized equipment to remove this debris or miscellaneous site debris such as trees. Specific alternatives for debris removal or excavation techniques will not be developed, but the various process options under drum and debris removal will be reevaluated when designs for the final alternative require the movement of debris.

8.5.3.2 Excavation

Excavation of SLB waste and associated contaminated soil or sediments, combined with disposal and/or treatment, is an effective remedial technology that reduces potential risks associated with release of contaminants to groundwater and surface water. Soil and buried waste are removed with ordinary construction equipment such as backhoes. However, the actual excavation and subsequent handling techniques will vary due to the variety of waste materials and site conditions. After removal, wastes would need to be segregated for different handling, treatment, or disposal requirements. Excavated areas would be backfilled to grade consistent with the site area and capped or covered as appropriate.

In some areas of WAG 6, excavation of materials will pose high risk of worker exposure to hazardous or radioactive materials. These risks include accidental rupture of buried containers, release of contaminants, and the potential for elevated radiation exposure. Also, dust, surface water, and groundwater would have to be controlled and monitored during the excavation to avoid or minimize release of contaminants and off-site exposure. Ambient air monitoring and radiological surveys during excavation activities would be required. Because the risks to workers and the environment may be unacceptable, excavation would be limited to buried waste/soil with low radiation levels.

Excavation is an effective and implementable technology. With adequate health and safety controls, and combined with treatment and/or disposal technologies, it can provide a reliable means of protecting human health and the environment. Excavation will be carried forward.

8.5.3.3 Bulk liquid removal

With the proper equipment, liquid can be moved from one place to another with the use of hoses and pumps. There are significant amounts of leachate in some of the trenches, and there is the possibility that waste liquids may be formed in the process of treating the SLB waste or groundwater. Liquid removal minimizes the potential for leachate migration and improves the efficiency of in situ waste treatment processes. This technology also provides for the transfer of process wastewater to further treatment or disposal. The removal of liquids presents the possibility of spills from leaks, hose failures, or splashing; therefore, provisions for spill containment must be made.

Pumping is a reliable, inexpensive way to remove or collect bulk liquid, especially effective for removing water contained in the waste through the surface and for transferring waste water. It will be retained as an option for consideration but will not be developed specifically as an alternative because the need to remove bulk liquid can only be determined during design or construction. Bulk liquids are probably present in some of the SLB waste and may result from other remedial actions, but based on current data, the removal of bulk liquid is not expected to comprise a large part of the remediation. Continuous leachate collection is considered an element of groundwater collection. Any removed bulk liquid would need to be treated and/or disposed as discussed under Groundwater Treatment and Disposal.

8.5.3.4 Groundwater collection

The groundwater collection process options are well systems and drains. Collecting groundwater with wells involves installing and operating extraction wells or well points for removing contaminated groundwater (followed by treatment and/or disposal). If properly designed and constructed, the wells are generally reliable, effective, and durable. However, at WAG 6 an extraction well system may not be feasible because of hydrogeologic conditions (See Sect. 3). It would be difficult to site a well system that could prevent contaminated groundwater from escaping the collection system. Therefore, extraction wells will not be retained as an option for groundwater collection.

A drain collection system would consist of a series of perforated pipes laid in trenches excavated to the bottom of the active flow zone of the upper aquifer. The trenches would be backfilled with a permeable medium such as gravel. Drain systems without perforated pipes can also be used. The groundwater collected in the trenches would flow by gravity to a wet well, from which it would be pumped to a holding tank prior to treatment or discharge. Collection trenches are appropriate for WAG 6 conditions because of the relatively low volume of groundwater to be collected, and because they can intercept groundwater flowing through an area regardless of subsurface conditions. Also, by using the natural contours of the burial areas, trenches can be constructed to intercept and prevent clean groundwater from entering a waste disposal area. As cited earlier, a French drain has already been used successfully at WAG 6. Drains will be carried forward as alternatives associated with groundwater collection.

8.5.3.5 Vapor/gas collection

Collection and venting of gases is an important consideration when capping biodegradable material, such as the wastes in the biological trenches. Without a venting system, gas pressures produced could build up and cause the cap to crack. Gases could also lead to lateral subsurface migration of contaminants. Gases within covered areas can be collected and vented to the atmosphere, with or without treatment, depending on the composition of the gas. Collection systems include passive vents, which are free-venting structures, and active vents, which mechanically draw gases from the capped material.

Passive vent systems require that highly permeable material, typically gravel and a perforated pipe in a trench or a well, be placed in the flow path of gas near the upper part of the capped material. These gas collection trenches are often lined on one side to help channel gas into the collection pipes. Gas collected in the pipes may be monitored, then either discharged to the atmosphere or treated. An active gas system consists of a series of wells or trenches containing perforated pipe. A mechanical blower or compressor applies suction to draw the gas to a central point, where it is monitored and either discharged or treated.

Both process options for collection will be retained for incorporation into capping options. The active system is more costly and requires regular maintenance; however, it is more effective. Also, at WAG 6 the gas would need to be collected and monitored for radioactivity and hazardous constituents before release. Gas collection will be carried forward for further evaluation as part of a capping process option. Gas collection would be used for sources that are still biodegradable. It would not be used in areas that are stabilized/solidified as part of the alternative.

Vapor and/or gas collection may be a consideration if groundwater is collected for treatment and radon gas or organic volatiles are present in sufficient concentration to pose a hazard to treatment plant operators. Because of the potential for gases or vapors, any treatment system would include air monitoring, and, if needed, a gas/vapor collection system would be incorporated.

8.5.3.6 Decontamination

Decontamination, by washing or mechanical operations, is potentially viable at WAG 6. Although only routine health and safety decontamination has been identified as a definite requirement, decontamination will be included as part of all developed alternatives.

8.5.4 Treatment

8.5.4.1 Solids processing

Solids processing either separates material or changes the size of the material. Solids processing usually is required before treatment to create a more homogeneous material. Two processes—crushing/grinding and shredding/chopping—change a material's size. Crushing and grinding are used on brittle waste; shredding and chopping are used on nonbrittle waste. Both process options would be used as part of alternatives that involve excavation and treatment of SLB waste material that contains large solid objects such as drums or equipment.

The other process options, including magnetic processes, screening, and hydraulic classification, separate wastes by property or size. Magnetic processes separate magnetic from nonmagnetic waste, which is especially useful prior to resource recovery (not under consideration), or incineration, which is under consideration. However, because magnetic and nonmagnetic wastes have been mixed together, are equally contaminated, and equally require remediation, magnetic separation processes will not be useful and will not be retained.

Screening mechanically separates material particles on the basis of size using a uniformly perforated surface. Particles larger than the perforations are retained on the screen while smaller particles pass through. This process can be very effective on coarser particles such as pieces of glass or metal with the use of a slow feed and long screening period. Screening can be performed on very dry or very wet (suspended) material, but not damp material. At WAG 6 the material would be dry. Also, it will be effective only after larger pieces of waste (e.g., equipment, casks) have been removed. This is an inexpensive separation technique and will be retained.

Classification is the separation of particles according to their settling rate in a fluid. With the hydraulic classifier, water or air is introduced so that the direction of flow opposes that of settling particles. The simplest form of a hydraulic separator is the settling-cone. Solids-bearing liquid flows into the settling-cone as if into a funnel. The heavier, solid-laden flow exits the bottom while the liquid flows radially over the lip of the cone. Classifiers have a lengthy history of reliable, continuous operation for finer-sized particles. This could be used on contaminated soils at WAG 6. However, clayey soils are difficult to process.

Screening will be used to represent both separation processes, because it is more applicable to varying conditions. Classification could be reconsidered during the final design, if appropriate. This technology is considered a secondary technology.

8.5.4.2 Solidification and/or stabilization

Solidification produces a monolithic block of waste with high structural integrity and reduced leaching potential. The contaminants do not interact chemically with the solidification agents, but are mechanically bonded. Stabilization usually involves addition of reagents, which limit the solubility or mobility of the waste constituents. The intent of solidification and/or stabilization of contaminated materials would be to limit the spread of radioactive material via leaching, to trap and contain decay products in the densified mass, and to improve the waste's structural integrity.

Contaminated soil and/or sediments would be excavated and mixed with solidifying agents in either a continuous or a batch process. Portland cement, pozzolanic fly ash, or other chemical fixation, sorption, or encapsulation agents could be used. The varying waste types at WAG 6 would require different agents. In addition, much of the waste in the burial units at the site could not be stabilized because it includes materials (desks, casks, etc.), not amendable to the process. The material that could be stabilized would be placed in containers and disposed on-site.

As a treatment technology, solidification/stabilization, preceded by excavation and followed by disposal, would be very expensive. It would also have a much higher risk to workers and the environment than in situ solidification/stabilization. Because the variable waste at WAG 6 will contribute to reduced effectiveness, and because of the cost and significant materials-handling issues, stabilization/solidification will not be carried forward for further consideration as a primary treatment technology.

8.5.4.3 Solids dewatering

The solids dewatering process would be used to dewater excavated waste before treatment or used to dewater sludge produced from treating the groundwater. One such process, sludge drying beds, allows free water in a sludge or waste matrix to drain or evaporate. Because of the hazards associated with any water that has been in contact with the waste, any sludge drying bed would be constructed with a sophisticated leachate collection/detection system. The bed would also need to be covered to protect against rain/humidity. This process could be used for sediments resulting from treatment of groundwater; however, because of the small particle size of sediments, drying beds would not be as effective as dewatering equipment. They would be much more effective for larger-size waste. Sludge drying beds will be carried forward only for alternatives considering excavating and treatment or disposal; it is considered a secondary process option.

8.5.4.4 Physical/chemical water treatment

Any groundwater treatment must be coupled with its extraction, and the treatment system would need to be designed to produce water of suitable quality to meet discharge limits. Bulk liquid removed and/or collected from the disposal units would contain similar contaminants and may also be treated in this plant. Because groundwater is expected to be the most significant contributor to the plant influent waste stream, groundwater characteristics will be used to design the treatment system.

The following groups of contaminants are representative of those that have been detected in the groundwater during RFI activities as exceeding federal or state standards or health-based criteria. While these are the contaminants present, the treatment levels and ultimate treatment required will depend on the discharge point selected and its requirements. More than likely, a single process option will not be able to remove all constituents.

- Radionuclides, including strontium-90, tritium, and cobalt-60. Total radium levels were also elevated, but it is naturally occurring at WAG 6. Strontium also belongs to the same chemical series as radium and calcium. Tritium cannot be removed from groundwater and collected for disposal.

- Filtered inorganics such as antimony, barium, beryllium, cadmium, and silver were found at levels exceeding MCLs and above reference values. The extent of contamination appears to be limited.

- VOCs are present in the groundwater at levels exceeding federal standards and human health criteria. Fairly prevalent are 1,2-dichloroethene, 1,1-dichloroethane, and trichloroethene. BTX compounds were found, but the areal extent of contamination is limited.

When groundwater from different areas of the collection system is combined across the site, nonradioactive inorganics will most likely not be of significant concern due to dilution. Therefore, only those inorganic treatment process options that have proven to be effective on radionuclides or that may be necessary pretreatment or posttreatment processes for other options were carried forward.

Flow and strength equalization. By using temporary storage, influent water quantity and quality can be regulated to minimize process-disrupting changes. Initial temporary storage will be included as part of any wastewater treatment plant because of the anticipated variations in groundwater quantity and quality. Additionally, waste leachate or aqueous residual from soils treatment may require periodic treatment.

Coagulation/flocculation. If contaminants of concern are associated with suspended particles, this process accelerates the settling of suspended solids through the addition of floc-forming chemicals. While this process option is an effective method for significant removal, it is not effective on dissolved contaminants nor for very low effluent concentration requirements. Because it is anticipated that most of the contaminants of concern are dissolved, coagulation/flocculation will not be retained. If appropriate, it may be reconsidered during the design phase.

Media filtration. Filtration separates suspended matter from water. Water is passed through a filter medium, and particulates either accumulate on the surface of the medium or are collected throughout its depth. Filters have been found to be effective for removing a wide range of sizes and types of particulates, including algae, colloidal clay compounds, viruses, asbestos fibers, and colloidal clay particles. Examples of filter media include screens, with openings of 1 to 100 μm, and granular materials, usually sand, anthracite, or coal with grain sizes varying from 0.1 to 10 mm.

Filters are frequently used as a polishing step in the removal of suspended solids. They are more effective than coagulation/flocculation. However, when used with water containing too many suspended solids, filters may bind on the top and matter cannot be removed deeper in the filter. Filtration will be carried forward.

Adsorption. In an adsorption process, contaminated water would flow through a bed of granular activated carbon, using either pressurized or gravity tanks. This existing, conventional treatment process will remove VOCs, BNAEs, and some inorganics from dilute aqueous solutions. The process is effective and applicable for a wide range of contaminants. Carbon adsorption can achieve a high level of organic removal when regularly maintained to monitor for breakthrough and to replace spent carbon. Testing would be required to determine its effectiveness on radionuclides. In addition, spent carbon may be considered a radioactivity source. This process option will be carried forward.

Membrane processes. In a membrane process, contaminated groundwater would contact a membrane that separates contaminants from an aqueous stream. Three types of membrane processes are suitable for consideration: reverse osmosis, electrodialysis, and ultrafiltration. Reverse osmosis and ultrafiltration are similar; both remove water from a mixture by passing it through a device that retains the other components. There are significant differences: (1) osmotic pressure has an extremely important role in reverse osmosis and not in ultrafiltration; and (2) ultrafiltration separates mixtures based primarily on size. Many factors affect separation by reverse osmosis.

Electrodialysis uses an electric current to induce partial separation of components in an ionic solution. Separation is accomplished by alternately placing cation- and anion-selective membranes across the electric current path. When current is applied, the electrically attracted cations pass through the cation exchanger in one direction, and the anions pass through the anion exchanger membrane in the other direction.

All membrane processes are fairly developed and would be effective on some of the groups of contaminants at WAG 6. They require rather extensive pretreatment to control fouling and scaling. All processes also produce a contaminant-laden solution or brine requiring disposal. This brine may be a liquid mixed waste as a result of RCRA-metal and radionuclides removal. Further treatment such as solidification would be needed prior to disposal as a mixed solid waste. The processes' high energy demands, pretreatment needs, and brine disposal requirements make them fairly expensive compared to carbon adsorption and ion exchange. However, due to the varied WAG 6 wastes, several primary treatment processes may be necessary. Until more is known about the actual quality of the groundwater treated by the plant, ultrafiltration will be carried forward as representative of the membrane processes since organics and radionuclides (large ions) are of more concern than inorganics.

Air stripping. In the process of air stripping, a compound is transferred from a liquid stream to an air stream. The driving force for the transfer is related to the equilibrium partial pressure of the compound in air. The partition constant (Henry's law constant) indicates the volatility of the compound and is the primary parameter for determining air stripper removal efficiency for the compound. Removal efficiencies increase with increasing partition coefficients, temperature, and/or air-to-water ratios. Air stripping is usually conducted in counter-current packed towers. Air is forced through the bottom of the tower and exits at the top with the volatile compounds. If sufficiently low concentrations are present in the effluent, the gaseous compounds can be discharged directly to the air. Otherwise, air pollution control devices may be necessary.

Air stripping has been proven reliable for removing volatile compounds, including some organics and ammonia and will be carried forward. However, if it is used before removal of volatile radionuclides (radon), vapor phase carbon may be needed to treat the air emissions. This would present health and safety issues for the workers. It also results in disposal/regeneration issues for the carbon, as it would contain radionuclides.

Neutralization. The groundwater does not require neutralization prior to discharge; however, a base may be added to reduce the solubility of metals (discussed under Precipitation subsection). Although neutralization will not be considered as a primary treatment, it may be needed as post-treatment to readjust the pH of the water.

Precipitation. In a precipitation process, a chemical constituent (such as lime or sodium hydroxide) is added to the aqueous stream. The constituent alters the equilibrium and forms a precipitate with the contaminant. The precipitate is removed by clarification operations.

Precipitation has been used effectively to remove inorganic contaminants, including radionuclides. There are several methods of precipitation, each differing in the precipitation agent and the pH at which precipitation is performed. The precipitation method can be tailored to match the contaminants requiring removal. The process is relatively well developed, and equipment is available from a number of manufacturers and is readily implemented. Precipitation has generally been shown to be both cost- and performance-effective for removing heavy metal contaminants. One disadvantage is that it produces a significant volume of residuals that require additional treatment and disposal. Also, it has no effect on organics. However, it will be carried forward because of its effectiveness for radionuclides.

Ion exchange. In an ion exchange process, contaminated groundwater flows over a bed of special resin that removes ions from solution and replaces them with less harmful ions. Ion exchange is a physical-chemical process in which ions are transferred from a liquid to a solid phase or vice versa. Ions held by electrostatic forces to charged functional groups on the surfaces of a solid (an ion exchange resin) are exchanged for ions of similar charge from a solution that flows past the resin. These resins can tolerate a range of temperature and pH conditions, exhibit a high exchange capacity, and can be tailored to show selectivity towards specific ions. Synthetic resins are employed in many applications; however, some natural exchanges are available. For example, clinoptilolite, a sodium-calcium-aluminum silicate, has a very strong affinity for ammonium ions.

Due to its high removal efficiencies, ion exchange may be useful in a polishing operation following treatment by other processes. Ion exchange is relatively well developed and could be readily implemented. However, ion exchange units either must be regenerated, which produces a liquid waste stream requiring treatment, or the ion exchange resin itself must be disposed when exhausted. Its specific effectiveness would need to be tested because of the potential for competing ions. This option will be carried forward.

8.5.4.5 In situ treatment

Permeable treatment beds. Permeable treatment beds are trenches dug to intercept groundwater. Adsorbent chemicals are added so the groundwater is treated in the trench. The contaminants can be adsorbed, biodegraded, neutralized, oxidized, precipitated, or reduced. This process option is experimental, and permeable treatment beds cannot achieve the degree of removal that can be achieved by extraction and treatment. Also, effluent limits are likely to be lower than the bed can achieve, meaning the groundwater would require further treatment. Additionally, the contaminants removed remain in the trench and would most likely require excavation for ultimate disposal since the trench cannot be completely lined and still collect groundwater. The additional treatment requirements and the excavation and reburial of bed materials makes this process option expensive compared to traditional treatment with the potential for significantly less effectiveness. Therefore, this process option will not be carried forward.

Vitrification. In the vitrification process, the contaminated material is heated to its melting temperature then allowed to cool and solidify to a glassy mass. Testing to date has indicated that the glassy mass or monolith is structurally stable and resists leaching. Vitrification consumes high amounts of energy. In situ vitrification involves the application of electricity to electrodes placed in the ground over the waste mass. The ground and waste mass both heat and melt, with the melting zone growing downward. A hood is placed over the zone to catch gases, which are treated and removed to eliminate air emissions.

Vitrification destroys organic contaminants during processing via pyrolization in the melt followed by combustion in the offgas hood; however, it will also volatilize tritium and metals such as cesium, mercury, and lead. There is the potential for the production of incomplete combustion byproducts. These contaminants present a secondary (and possibly mixed) waste stream and potential worker exposure problems.

Although available commercially, this process option is still experimental for the type of complex waste matrices (e.g., buried steel and concrete containers) present at WAG 6.

However, tests performed by DOE on contaminated soils (at ORNL and Hanford) and on a small-scale (30-cm diameter) steel/concrete tank have been promising. Because in situ vitrification significantly reduces mobility of inorganics, including radionuclides, it will be carried forward.

Stabilization/solidification. The same general principals apply to in situ stabilization/ solidification as to excavation and stabilization/solidification, described alone. Because of the difficulties associated with stabilizing the types of waste at WAG 6, this option will not be carried forward.

8.5.4.6 Thermal treatment

The only process option remaining for the thermal treatment technology is incineration, which can be used both to minimize waste material volume and to destroy many compounds. Through incineration, materials are thermally destroyed in a controlled, oxygen-sufficient environment. The oxidation products are primarily carbon dioxide, water, and ash. The ash contains various end-products from the incineration of nitrogen- and halogen-containing materials, in addition to noncombustible materials.

The system is reliable and effective for destroying organic contaminants (consisting of combustible material) in the soil and the trenches. However, very little volume reduction is anticipated due to the large amounts of noncombustible material buried in or associated with the WAG 6 SLB units. In addition, other disadvantages include its high capital and operating costs and the fact that radionuclides are not destroyed. In fact, the radionuclides may be volatilized, requiring gas and ash to be collected and controlled. Incineration is expensive and may be useful only for a small portion of the waste. However, because of its high degree of effectiveness on organics and potential for some waste volume reduction, incineration will be carried forward.

8.5.5 Disposal

8.5.5.1 Temporary storage

Any temporary storage would be needed only during construction. There are no plans to remove waste materials to temporary storage to await technological advancements or development of a mixed waste disposal facility. Temporary waste piles and containers or tanks are the temporary storage options most likely to be used; however, the presence of radioactivity makes it more likely that construction will be scheduled to avoid any temporary storage. Therefore, this technology will no longer be considered. If storage becomes necessary as a secondary component in design, it will be reevaluated at that stage.

8.5.5.2 Nuclear waste repository

DOE Order 5820.2A, "Radioactive Waste Management," Chapter III, states "DOE low-level waste shall be disposed of on the site at which it is generated, if practical, or if on-site disposal capability is not available, at another DOE disposal facility consistent with the requirements detailed in reference 50." For this reason, and because limited on-site disposal capacity is available and a nuclear waste repository is not available, this option will not be considered for disposal of WAG 6 waste.

8.5.5.3 Wastewater discharge

After technology screening, discharge to surface water and discharge to an on-site existing treatment plant remain as process options. The only viable surface water discharge point is WOL, which is sufficiently large to handle additional flow from a groundwater treatment plant. Any treatment plant would have to be designed to protect the surface water environment, which will require meeting discharge standards specified in the plant's NPDES permit.

The other process option is using the existing on-site liquid waste treatment system prior to surface water discharge. The ORNL Process Wastewater Treatment Plant (PWTP) treats water that contains very low concentrations of radionuclides, mostly strontium-90 and cesium-137. About 70 percent of the strontium-90 is removed by softening at a pH of 11.5. Clarified water is then treated by strong acid ion exchange, which removes the rest of the strontium-90. To remove cesium-137, zeolite columns are used, either in series or parallel to the routine PWTP processes. The PWTP has an average flow of 120 gpm. The Nonradiological Wastewater Treatment Plant (NRWTP) usually treats only for organics by filtration, followed by air stripping, followed by carbon columns. The average flow is 250 gpm. This plant does have metals-removal capabilities. In addition, water from the PWTP can be piped to the nonradiological plant for treatment in series. Treatability tests using trench water from WAG 6 showed that the two plants have the processes necessary to remove the contaminants.

Depending on the flow predicted for collection at WAG 6 and the ORNL plants' capacities, discharge to the ORNL liquid waste treatment system could be a cost-effective process option. Most of the treatment process options carried forward during this evaluation are available in the ORNL plants. Therefore, the ORNL liquid waste treatment system will be used in alternatives instead of constructing a new treatment plant. A new plant or the option of treating wastewater at existing Y-12 and K-25 facilities may be reconsidered during the design if appropriate.

8.5.5.4 Solids disposal

After treatment, much of the excavated materials could be replaced in the locations from which they were removed. Material excavated from other areas of the site also could be consolidated within major areas of contamination or placed in the IWMF, providing they are classified as LLW. In addition, solidified brines, carbon, ion exchange resins, and other solid residues from liquid treatment could be disposed on-site if necessary. This solids disposal option could be effective and reliable when combined with a relatively impermeable cap. All treated RCRA waste would need to meet RCRA land ban requirements before disposal in its originating area. This option will be carried forward for further consideration.

8.5.6 Monitoring

Monitoring of groundwater and surface water contamination is used primarily to evaluate over time the effectiveness of the chosen remedial action and/or to determine whether existing and future receptors are threatened. This is a necessary component of a flexible remedial program. Samples would be taken regularly at various locations to determine required adjustments or additions to the implemented actions. Monitoring can be used in conjunction with other technologies or could be considered alone. Capital costs should be low since many monitoring wells are already in place. This technology will be carried forward.

8.6 PRELIMINARY DEVELOPMENT OF ALTERNATIVES

Specific remedial alternatives for waste sources and groundwater on the WAG 6 site are described and screened in this section. The alternatives represent a range of treatment and containment combinations specifically applicable to SLB disposal units and groundwater. The alternatives vary from the No Action alternative, in which no remedial efforts would be undertaken, to more aggressive and permanent alternatives.

The objective of the preliminary alternatives screening is to identify the alternatives that best meet the tests for effectiveness, implementability, and cost. The alternatives carried forward for incorporation into site-wide alternatives will provide a range of possible actions with respect to cost and degree of active remediation.

Effectiveness is primarily defined in terms of an alternative's ability to protect potential human and environmental receptors during and after remedial activities and in terms of its long-range reliability.

In assessing implementability, each alternative is evaluated in terms of both technical and administrative feasibility during construction, operation, and maintenance. Technical feasibility refers to the ability to construct, reliably operate, and meet technology-specific regulations for the remedial technology options. Administrative feasibility considers the procedure required to obtain approval or cooperation from federal, state, and/or local government agencies.

Relative costs, as used in this evaluation process, are for comparative purposes only and reflect judgment based on experience with the technologies in other situations. Cost of a remedial alternative is estimated as low, moderate, or high relative to other alternatives for each media. Those identified as being more costly without providing significant additional benefits may not be carried forward.

8.6.1 Development and Evaluation of Media-Specific Remedial Alternatives

This section begins with a brief description of the waste sources or groundwater at WAG 6. It also briefly examines the technology options retained from the technology screening and describes the rationale used to develop alternatives. Also in this section, waste source and media-specific alternatives developed from these technologies are presented and evaluated with respect to their effectiveness, implementability, and comparative cost. This results in a range of viable alternatives that can be combined into site-wide alternatives.

8.6.1.1 Shallow land burial waste alternatives

Source control alternatives are methods for protecting public health and the environment by limiting direct contact/emissions, surface runoff, and leachate generation from on-site sources and associated contaminated leachate and soil. For SLB wastes, the general response actions and process options carried forward to this phase include:

- No Action
- Institutional Controls
  - Deed Restrictions
  - Fences
- Containment
  - Multilayered Cap
  - Surface Controls
  - Dynamic Compaction
  - In Situ Grouting
- Removal
  - Solids Excavation
  - Liquid Collection (leachate)
- Treatment
  - In Situ Vitrification
  - Incineration
- Disposal
  - On-Site
  - Existing ORNL Liquid Waste Treatment System (leachate)

For the SLB wastes, alternatives have been developed to represent a range including no action, institutional controls, containment, and treatment. Not all possible combinations of technologies have been developed into alternatives, because many combinations essentially accomplish the same objective with virtually no difference in effectiveness, implementability, or cost. Also, alternatives that are not technically feasible have not been developed.

Not considered, for instance, is complete excavation of the sources. Much of the waste previously disposed of at WAG 6 is mixed waste. Currently, ORNL stores mixed waste as it is produced, awaiting the development of a mixed waste disposal facility. There is no existing DOE or commercial facility where the mixed waste at WAG 6 could be disposed of.

The other reason for not considering complete excavation is the potential for unacceptable worker exposure to radiation that could result while excavating many of the disposal units. Calculations (Appendix 8B) indicate that unacceptable radiation exposure to workers would be associated with excavation of many of the sources, even with 40-mm shielding (over the limit of reasonable shielding possible for heavy equipment). The calculated exposure levels exceed DOE orders for worker exposure and are counter to DOE orders requiring ALARA whenever possible. The use of remotely operated equipment is technically feasible but adds enormous costs to a site the size of WAG 6.

Specific containment or treatment technologies identified in each alternative will not necessarily be applied to all sources on the site. Instead, as part of defining the alternative, some sources will be assigned different technologies based on technical considerations. Thus, both dynamic compaction and in situ grouting are considered as waste structural stabilization technologies. Considering the wide variety of waste types and disposal methods at WAG 6, detailed alternatives would likely need to use more than one waste stabilization technology to address source characteristic differences. If an alternative containing stabilization is carried forward to the detailed evaluation, specific process options will be defined for the specific sources at that point.

The following alternatives have been developed and are evaluated in this section.

1. No Action
2. Institutional Controls
3. Surface Controls, Capping
4. Surface Controls, Cap, Consolidation, Structural Stabilization
5. Surface Controls, Cap, Consolidation, Structural Stabilization, In Situ Vitrification
6. Surface Controls, Cap, Consolidation, Incineration, Structural Stabilization

These are briefly described below and then evaluated and compared with respect to their effectiveness, implementability, and comparative cost. Except for No Action, all alternatives include performance monitoring to assess their effectiveness during and after implementation. Performance monitoring will also be included in each of the detailed alternatives, as an integral part of the total approach to remediation. Each detailed alternative also will define the appropriate monitoring and the appropriate contingent action to allow systematic, logical progress of remedial action.

SLB Alternative 1: No Action. In accordance with EPA guidance, No Action is evaluated under the assumption that WAG 6 has been abandoned, that there would be no institutional controls, and that the fence would no longer be effective in limiting access to the sources. This alternative provides no protection against existing and future hazardous material releases. As determined in the base line risk assessment, risk to human health (see Sect. 6) and the environment (see Sect. 7) would occur through leachate production and subsequent release to the groundwater or to the surface. Future risk would also occur if the existing soil cover were eroded.

The No Action alternative is the least expensive, as no costs would be incurred. The alternative will be carried forward for consideration to meet regulatory requirements of CERCLA, SARA, and NEPA to provide a base line comparison for other retained alternatives.

SLB Alternative 2: Institutional Controls. Two variations of institutional controls are considered. The first provides continued use of existing institutional controls through a 10-year operational period and a 100-year post-operational period. During the combined periods, the existing covers would be maintained, minimizing erosion. After 110 years, the institutional controls would be lifted and the facility abandoned. However, the base line risk assessment (Sect. 6) indicates that after 110 years, without active remediation the site would still pose a risk to human health. Therefore, such institutional controls alone will no longer be considered because they would not be effective in protecting public health.

The second variation of institutional controls requires controls for as long as the sources present a threat to public health (more than 110 years into the future). During the site's operational and post-operational periods, existing institutional controls (access controls) would be maintained. After the post-operational period, the access controls would be continued, with deed restrictions banning disturbance of the site and requiring continued maintenance of the existing covers. Although effectively implemented institutional controls would protect against exposure to humans, they would not protect the environment from future contaminant releases associated with leachate production. Also, they are not as reliable as more active measures.

Institutional controls would be fairly easy to implement at WAG 6, because most already exist. Also, institutional controls may have relatively low capital costs. However, the operational and maintenance costs may be comparable to those for a more active alternative. The existing covers need to be maintained during waste subsidence. In addition, the fences need to be maintained and patrolled.

Because of the high operation and maintenance costs and the possibility of failure of the controls before the risk from the site ends, and because they would not protect the environment from future contaminant releases, institutional controls will not be carried forward as an individual alternative. Instead, institutional controls will be considered as a part of the more active source control alternatives.

SLB Alternative 3: Surface Controls with Capping. Alternative 3 contains small waste-area specific caps with associated surface controls (Fig. 8.4). The purpose of these caps would be to limit infiltration through the waste, limit contaminant movement through surface runoff, and protect against direct exposure to the waste. No attempt is made to use the caps to lower the groundwater table below the bottom of the SLB waste units. As opposed to using any of the structural stabilization methods retained from the technology screening phase, this alternative would include a cap design to make repairs or replacement feasible as waste trenches subside over time. A grid mesh over the waste may help reduce the impact on the cap from waste subsidence by providing more support to the cap. Monitoring the condition of the cap and providing maintenance is an integral part of this alternative.

Surface controls also include grading and revegetation. The caps and surrounding areas would be graded to eliminate any surface ponding, and slopes would be graded to limit erosion. Upon completion of grading, disturbed areas would be revegetated.

Also in Alternative 3, French drains are included in selected areas of the site to intercept storm flow prior to its entering the disposal units. Since there will be no contact with the waste, it is expected that water collected during a storm event would be clean and would not require treatment. More detailed development of institutional controls, monitoring, and contingent actions will be provided in the FS/CMS/EA.

Under this alternative, some continued contamination of the groundwater may occur, because the smaller caps likely will not permanently lower the groundwater table significantly. However, intercepting storm flow prior to its contacting the wastes decreases continuing degradation of the groundwater quality. In addition, the caps will protect against direct contact with the waste and will minimize infiltration into the waste. The alternative can effectively meet the objectives of the remedial action. However, in the long term, the caps may need to be replaced because the waste would not have been stabilized and subsidence might undermine the caps.

Alternative 3 is one of the most easily implemented alternatives; none of the technologies require difficult construction techniques or unusually difficult worker protection. The alternative has relatively low capital costs compared to other alternatives, but maintenance costs may be relatively high if significant subsidence of the waste damages the caps. The consolidation component of the technology adds construction cost and complexity, but this is a component of all remaining alternatives. Alternative 3 will be carried forward to the detailed site-wide alternatives to allow a comparison of the merits of smaller caps versus larger caps (presented in other alternatives).

SLB Alternative 4: Surface Controls, Capping, Consolidation, and Structural Stabilization. Alternative 4 contains large caps (Fig. 8.5) with previous structural stabilization of the trenches and necessary surface water controls. Also, the southern trenches would be excavated and consolidated with other waste. The caps of this alternative would be designed to virtually eliminate recharge to the capped area, including recharge by storm flow, and consequentially would lower the water table. Caps sufficient in size to cover the recharge area also would reduce leachate generation caused by groundwater and/or infiltrating precipitation interacting with the waste, which would further reduce release to or transport via groundwater.

Studies conducted at WAG 6 indicate that interim caps 2 and 8, which were placed as part of the interim closure of the RCRA units, have been effective in reducing or preventing trench bathtubbing by stopping infiltration and by preventing recharge to the trenches via shallow storm flow. However, other caps have not prevented bathtubbing effects and some capped waste disposal areas continue to be inundated by groundwater (Ashwood and Spalding 1990). This indicates that even if direct infiltration is reduced locally, some existing caps are not large enough in area to prevent trench bathtubbing caused by storm flow seepage or interaction with the water table. Smaller caps would not significantly impact the water table level. Therefore, Alternative 4 will involve the larger caps. If further evaluation during the detailed alternative development shows that the caps alone do not lower the water table below the waste, interception drains upgradient of the disposal area may be included.

The degree of structural instability of the trenches and their contents has been documented in a report (Ashwood and Spalding 1990) that concluded that burial trenches in all areas of the site are inherently unstable and subject to substantial settlement under existing and newly imposed loads. Because the larger caps would be more difficult to replace in the event of failure then the smaller caps, all trenches would be stabilized by dynamic compaction or in situ grouting before the caps are constructed.

Dynamic compaction has been demonstrated to be effective in reducing the void volume in a trench as discussed in Sect. 1. Using a weight to compress the waste material does not in itself add much additional cost to capping. The negative aspect of this technology is that compaction could break containers, accelerating the release of contaminants to the subsurface. To date, monitoring wells have not detected any increase in contamination of the groundwater due to the dynamic compaction pilot studies. However, it is unclear if the demonstration trenches contained containerized wastes or whether sufficient time has elapsed for contamination to reach the monitoring wells.

Dynamic compaction would be used on trenches anticipated to have lower levels of contamination and a low likelihood of significant biodegradation. Even after compaction, biodegradation can result in significant subsidence. In this case, in situ grouting, which would limit biodegradation, would be the preferred technology. Future cap maintenance would be needed since any biodegradation would cause waste subsidence.

In situ grouting is also included in Alternative 4 as a process option for improving stability of trench waste. Prior to grouting, leachate present in the trenches may be removed through vacuum collection and transported to the ORNL liquid waste treatment system for treatment and disposal. ORNL performed a treatability study to test the capabilities for treating trench leachate using existing processes in the treatment system (Robinson 1990). This study indicated that the trench leachate could be effectively treated by the system.

As discussed in Sect. 1, testing at ORNL also has shown grouting technology to be implementable and effective in stabilizing waste (Tallent, McDaniel, and Spence 1987; Spence, Godsey, and McDaniel 1987). Preliminary unit cost estimates provided in the test report showed in situ grouting to be much more expensive than dynamic compaction.

Grouting would be used on the more highly contaminated trenches. Because grouting could leave some voids unfilled and because the grout itself could physically deteriorate, its reliability in the long-term may be less than that of dynamic compaction.

Also under this alternative, disposal units inundated with groundwater as a result of direct contact with WOL, as well as the Hill Cut Test Facility and the ETF, would be excavated and consolidated elsewhere on-site with other wastes. Depending on the types of waste from each disposal unit, the waste would either be placed under the cap at a higher elevation or, for non-RCRA waste, would be disposed in the Tumulus. Figure 8.5 shows the material being consolidated in the area of the existing landfill. The RCRA land disposal restrictions are not invoked under this scenario because the entire WAG is considered an area of contamination; the movement of waste from one area of the site to another area is not considered "placement" under RCRA. However, rather than just dispersing the trench wastes into other disposal areas it may be necessary to construct a disposal unit that is more protective. Most space appropriate for disposal is already in use. The on-site disposal possibilities will be defined and evaluated in detail in the FS/CMS/EA report.

The soil adjacent to the trenches remaining after excavation of the inundated waste would be left in place and the trenches would be backfilled. Without complete remediation of WOL, any soil placed below the flood level of the lake is expected to be contaminated by WOL after the site is capped and groundwater flow gradients are changed. Therefore, no attempt will be made to remove all contaminated soil. Neither would the area be capped. This will permit flexibility for future actions at WOL, which may require access to the southern portion of WAG 6. (The consolidation described herein is also considered as part of Alternatives 4 and 5.)

Alternative 4 is anticipated to provide better long-term effectiveness than Alternative 3, primarily because the caps would be more reliable and would offer improvements in protection from further contamination of groundwater. Alternative 4 is anticipated to be more difficult to implement due to the use of grouting and the construction of a large cap that will require significant earth-moving activities. Even though Alternative 4 may have higher capital costs than Alternative 3, the operational and maintenance costs are expected to be lower. To determine the benefit of the increased effectiveness versus the additional implementation difficulties and possible costs, Alternative 4 will be retained for further consideration during definition of detailed site-wide alternatives.

SLB Alternative 5: Consolidation, Surface Controls, Capping, Structural Stabilization, and In Situ Vitrification. Alternative 5 incorporates many of the same process options or technologies as Alternative 4 (surface controls, capping, grouting and/or dynamic compaction, and consolidation). It also includes in situ vitrification, which would be used on some of the highest activity trenches in place of structural stabilization. Other disposed waste of high activity would also be vitrified. (The actual disposal units selected for vitrification will be described during definition of the site-wide alternatives in the FS/CMS/EA report.)

In theory, and based on the demonstration tests described in Sect. 1, full-scale in situ vitrification appears to reduce the mobility of many of the radionuclides in addition to destroying the organic chemicals. The resulting vitrified matrix is structurally and biologically stable and should be reliable for a long time into the future. However, the long-term effectiveness of this technology has not been demonstrated.

In situ treatment has inherent implementation difficulties, especially with a varied waste matrix. Significant operational modifications may be necessary for different disposal areas. Significant field testing would be required, adding significant time (several years) required to the implementation of this alternative. However, the technology does not require excavation of the buried material; therefore, the hazards associated with materials handling are considerably less than with incineration (discussed below in this section). Proper worker protection against volatilized cesium would be required.

The costs of full-scale in situ vitrification are not well understood. There are reports that energy costs are very high, yet the process probably has costs between that of in situ grouting and incineration. Alternative 5 is anticipated to be in the same order of magnitude in cost as Alternative 4.

Alternative 5 will be carried forward to evaluate the added benefit of treatment versus the added implementation difficulties.

SLB Alternative 6: Consolidation, Surface Controls, Capping, Structural Stabilization, and Incineration. The surface control, capping, and structural stabilization aspects of Alternative 6 were discussed in Alternative 4. Under Alternative 6 waste contaminated with high levels of organics or containing combustible material (biological trenches) would be excavated and incinerated to reduce the volume of contamination and to destroy organic contaminants. A sufficient amount of waste must be considered for incineration to justify the extremely high mobilization costs.

Contaminated soil immediately surrounding the waste would be excavated. The volume of soil accumulated will depend on the excavation technique and on the contaminant concentrations, but volumes would be minimized as much as possible. Soil in the southern portion of the site would be left to be remediated along with WAG 2, because cleaned soil in the area would be recontaminated by the lake. The soil around the solvent auger holes would also be excavated as needed to remove the waste. Remaining contaminated areas would be backfilled and covered by the site cap.

Under this alternative, only an on-site mobile incinerator is considered. Although permanent off-site incinerators exist in the area [Scientific Ecology Group (SEG) incinerator and K-25 incinerator], they either are not permitted for mixed waste or do not have the capacity for the varied wastes found at WAG 6. (The only wastes under consideration for incineration from WAG 6 contain both radionuclides and RCRA organic wastes.) In addition, on-site incineration eliminates troublesome transportation considerations involved with off-site incinerators. Mobile incinerators currently are not present at ORNL, and one would need to be procured. Any existing incinerator would require considerable retrofitting since its design would not have been based on WAG 6 waste characteristics. It would also require up to 2 years lead time to set up, conduct trial burns, and obtain permits. This is a significant schedule delay over Alternatives 1 through 4.

The effectiveness of Alternative 6 in protecting public health and the environment is not significantly improved over other alternatives. Although some of the organic contaminants may be destroyed, most of the risk from the site is from radionuclides, which are not affected by incineration. Waste sources selected for incineration include the biological trenches and solvent auger holes. This waste is assumed to be combustible and would not present too great a hazard if excavated. However, extreme costs and mobilization efforts would be spent, only to achieve some volume reduction or to only destroy some of the organics on WAG 6. The majority of the waste at the facility would remain unaltered and hazardous (inorganics and radionuclides).

This alternative is considerably more difficult to implement than previous alternatives. Considerable materials handling and solids processing would be required before the excavated material could be incinerated. Large objects, including concrete pipes, would need to be crushed or segregated. Material excavated from beneath the existing groundwater table would need to be drained on drying beds to minimize moisture in the incinerator. Material might need to be separated and stored until there is sufficient volume of like material to warrant modifying the incinerator operating conditions, as not all waste could be incinerated using the same conditions.

After incineration, the ash and solids from emission controls would require disposal. Under the best conditions, the solids would be contaminated with radionuclides and could be disposed on-site in an above-ground Tumulus. Due to the fairly low volume reduction anticipated (no more than 50%) a considerable amount of LLW would require disposal. The on-site disposal units have very little future capacity, and there is no room on-site for new low-level disposal units. To dispose of the ash and noncombusted material under the cap would result in up to 4 to 5 years' delay in cap installation. Off-WAG disposal at a low-level disposal facility might be required. Under the most likely conditions, the residue would contain radionuclides and RCRA inorganics, making it a mixed waste. Because there are no existing on-site or off-site final disposal facilities for mixed waste, the material would have to be stored until an off-site facility becomes available. Any disposal of treated material also involves the land ban restrictions of RCRA. The extensive materials handling make this alternative very expensive to implement, both in terms of capital and operation and maintenance (O&M) costs, and the issues surrounding disposal of the incinerated material and the air pollution control residue make this alternative almost impossible to implement.

Alternative 6 will not be carried forward for incorporation into site-wide alternatives because of the high cost, difficult implementation, and minimal additional effectiveness.

8.6.1.2 Results of shallow land burial wastes alternatives

As a result of this alternative screening, the remedial alternatives for source control that will be carried forward include:

Alternative 1.	No Action
Alternative 3.	Surface Controls, Small Caps
Alternative 4.	Surface Controls, Large Caps, Consolidation, Structural Stabilization
Alternative 5.	Surface Controls, Large Caps, In Situ Vitrification, Structural Stabilization

A summary of this evaluation is presented in Table 8.3.

8.6.1.3 Groundwater alternatives

As defined in Sect. 6 of this report, radionuclides and volatile organics with respect to groundwater are the contaminants of primary concern at WAG 6. These contaminants have exceeded various federal standards, are migrating off-site, and are of risk to future potential users of the groundwater. The hydrogeologic setting at WAG 6 creates complex groundwater flow paths. Groundwater flow in the saturated soil and saprolite above competent bedrock is generally controlled by a combination of primary matrix porosity of the weathered saprolite and secondary porosity caused by relict bedrock structural features such as fractures and solution cavities. Groundwater flow in the soil and saprolite is somewhat predictable and follows the hydraulic gradient indicated by potentiometric mapping. Conversely, there is little or no primary porosity associated with the competent bedrock. Flow in this zone occurs along fractures, following a tortuous path related to intersecting fractures. As there are up to five distinct fracture orientations recognized at any one location within WAG 6, prediction of groundwater flow in the bedrock is not possible. Although groundwater contamination has been detected in many of the WAG 6 interior and perimeter monitoring wells, distinct on-site groundwater plumes related to specific sources are not recognizable. Alternatives for remediation of groundwater must recognize the unknowns associated with the subsurface. As part of the alternative evaluation process performed in the FS/CMS/EA, the alternatives will be compared with respect to how effectively they can respond to varying conditions.

Because of the site's topographic relief, groundwater flow of most concern within WAG 6 originates primarily on-site. It flows from points of recharge (generally topographically higher areas) within WAG 6 to points of discharge also within or adjacent to WAG 6 (typically in low lying areas and surface water drainages), which ultimately flow into WOL. Adjacent to WOL, groundwater appears to discharge directly to the lake. A principal point of groundwater discharge appears to occur along the stream in the low area outside the eastern boundary of WAG 6. Site characterization data suggest that in the northern half of the site, the water table is below competent bedrock surface for most of the year such that groundwater flow does occur in the bedrock in this area.

General response actions and process options selected thus far for carrying forward to this phase include:

- No Action
- Institutional Controls
  - Deed Restrictions
  - Fences
- Collection
  - Drains
- Disposal
  - ORNL Liquid Waste Treatment System
  - Discharge to Surface Water

Remedial actions for groundwater will not focus on restoration of on-site groundwater to specified levels because of the difficulty of characterizing on-site contaminant plumes and groundwater flow patterns and because all source alternatives include the permanent disposal of wastes on-site. Alternatives developed here will address the mitigation of the impacts of off-site migration of contaminated groundwater and preventing the public from using on-site groundwater. Except for No Action, all of the developed alternatives include monitoring and the necessary institutional controls to prevent access to on-site contaminated water.

The following alternatives are under consideration for groundwater:

1. No Action
2. Institutional Controls, Monitoring
3. Institutional Controls, Monitoring, Future Decision
4. Boundary Collection, ORNL Liquid Waste Treatment System, Surface Water Disposal, Monitoring

These alternatives are evaluated below with respect to their effectiveness, implementability, and cost.

Groundwater Alternative 1: No Action. Under Alternative 1, the contamination moving off-site will not be controlled. The base line risk assessment and environmental assessment (Sect. 6 and 7, respectively) have illustrated the effects on public health and the environment of no action on the groundwater. Although no action is easy to implement technically, this alternative will not protect public health or the environment. Alternative 1 has no associated costs. The No Action alternative will be carried forward for consideration to meet CERCLA, SARA, and NEPA requirements and to provide a base line comparison for other alternatives.

Groundwater Alternative 2: Institutional Controls and Monitoring. Institutional controls and monitoring, as with the institutional control alternative for sources, contains two variations. The first variation assumes that the ORNL facility will remain operational for 10 years. That period will be followed by a post-operational period of 100 years, during which existing institutional controls will remain in effect. After that time, all institutional controls on WAG 6 will be removed. Institutional controls for this variation are primarily access controls such as fences and patrols. The risk assessment has indicated that the groundwater will remain a threat to public health beyond 110 years. Therefore, this first variation on institutional controls would not be effective in protecting public health and will not be carried forward for further evaluation.

The second variation uses the existing institutional controls through the post-operational period but extends the controls to as long as the groundwater remains a threat to public health. After the post-operational period, deed restrictions on the WAG 6 land would be added to the controls, limiting use of the land and of the groundwater beneath the site. In addition, monitoring would continue to assess when the controls can be lifted.

As with no action, there would be no mitigation of continued off-site migration of contaminated groundwater, but the effect on public health would be minimized by not allowing access to the on-site contaminated groundwater. Institutional controls may not be effective over a long period; typical controls resulting from hazardous waste remediation have not been in place long enough to assess long-term effectiveness, and it is anticipated that institutional controls may fail over time as the reason for the controls is forgotten.

However, this may be more true for controls on private land than for controls on federal facilities.

Institutional controls on DOE property are easy to implement, and most of the controls necessary for WAG 6 are already in place. They need only to be extended in time, and they would have a relatively low capital cost. However, the operational and maintenance costs may be higher than typically anticipated because of the need to continue patrolling the area.

Because of the low cost as compared to a more active alternative and the ease of implementation, institutional controls will be carried forward for further consideration.

Groundwater Alternative 3: Institutional Controls, Monitoring, and Future Decision. Alternative 3 includes institutional controls, monitoring, and a future decision concerning groundwater remediation. The groundwater underlying individual WAGs, including WAG 6, is not isolated and is part of the overall hydrologic system prevalent in Melton Valley. Consequently, groundwater at WAG 6 interacts with groundwater and surface water from adjacent WAGs, and decisions to address each independently may result in redundancy in collection and treatment systems or may even result in lowering the effectiveness of individual actions through interference. Cleanup levels or action levels need to be consistent across adjoining WAGs, especially with WOL cleanup since the groundwater from WAG 6 ultimately discharges there. Because of the interdependence of groundwater in and around WAG 6 with groundwater and surface water associated with other WAGs, and because of the timing of the work at each WAG, it is not possible to define, evaluate, implement, or monitor separate groundwater remediation actions for WAG 6 and adjacent WAGs. At this stage, it is only possible to control migration between WAGs. This is the objective for Alternative 4. However, Alternative 3 allows the groundwater at WAG 6 to be addressed as part of a larger area-wide groundwater action. In the meantime, institutional controls would be implemented over WAG 6 to prevent use of on-site groundwater until sufficient information is known about the adjacent WAGs to select an area-wide groundwater remedial action.

Alternative 3 provides the same degree of immediate protection of public health and the environment as Alternative 2; yet it does not preclude the use of active restoration as a future action. Where there is a significant risk from the use of groundwater, most active remediation alternatives are more effective in protecting public health and the environment in the long-term than institutional controls. The implementation of this alternative may best be accomplished by addressing WAG 6 groundwater and other connected groundwater as another operable unit.

It is not possible to assign costs to WAG 6 from a future decision. However, greater protection at a lower cost would be possible across the WAGs under this alternative than with active restoration or migration control alternatives at each WAG.

Alternative 3 will be carried forward for evaluation with respect to other alternatives.

Groundwater Alternative 4: Collection and Treatment. Alternative 4, the final groundwater alternative developed, includes collection of contaminated groundwater and disposal to the existing ORNL liquid waste treatment system for treatment and ultimate surface water discharge through the system's permitted discharge.

A drain was installed in the 49-Trench area in bedrock in 1983. Initially the 826 foot drain resulted in depressing the water table 10 feet (13 feet maximum). The drawdown area extended almost 200 feet from the centerline (Davis and Marshall 1988). Until the drain started clogging, it clearly demonstrated the effectiveness of trench drains at WAG 6.

ORNL has conducted treatability studies on the technologies contained in their process waste treatment and non-rad waste treatment plants. One of the conclusions of the report was that the plants can successfully treat the leachate found at WAG 6 to meet NPDES effluent requirements. The existing treatment capabilities will be explored more thoroughly during the detailed definition of alternatives.

Because the future areas of groundwater contamination passing the boundary are unknown, the collection system would be designed to address the areas at the facility boundary that are contaminated now, as illustrated in Sect. 4. Monitoring would be used to identify future boundary locations that may become incorporated into the boundary collection system. Any groundwater collection system would need to be integrated with any groundwater diversion drain system used as part of the source control alternative.

The components of this alternative have been tested at ORNL and found to be effective in achieving their respective purposes. Alternative 4 is more effective in the long-term for protecting public health and the environment than the no action alternative and than institutional controls. This is primarily due to the removal of contaminants from the groundwater. As risk from exposure of groundwater is lessened, so are the consequences of a collection system failure in the long-term. Alternative 4 probably has the same degree of protection and reliability as Alternative 3, which would include area-wide remediation.

The necessity to excavate collection trenches into the bedrock to collect contaminated groundwater at the northern and eastern boundary of the site make Alternative 4 difficult to implement. The depth of bedrock excavation will be determined in the FS/CMS/EA, but the objective is to collect groundwater contaminated above unacceptable levels from the aquifer. However, the treatment component would be fairly easy to implement, even compared to Institutional Controls, as it would only be necessary to transfer groundwater to the existing treatment systems. However, if the existing plants need to be expanded, the difficulty of implementing the treatment component would be higher. This will be discussed further in the FS/CMS/EA.

The cost of this alternative is the highest of the groundwater alternatives. Drains in the northern portion of the site may need to be installed deep in bedrock since the water table is below the bedrock surface contributing significantly to the capital costs. In addition, operation and maintenance costs of collecting the groundwater and monitoring the system would continue for a very long time. It may be necessary to add drains to the system in the future, causing future significant cost impacts.

Because of its treatment component, its implementability, its protectiveness of the environment and public health from the further release of contaminated groundwater off-site, and its long-term reliability compared to Institutional Controls, Alternative 4 will be carried forward for development into site-wide alternatives.

8.6.1.4 Results of groundwater alternatives screening

As a result of the above screening process, all of the groundwater remedial alternatives developed (listed below) will be carried forward. A summary of the evaluation is presented in Table 8.4.

1. No Action
2. Institutional Controls, Monitoring
3. Institutional Controls, Delayed Decision, Monitoring
4. Collection, ORNL Liquid Waste Treatment System, Surface Water Discharge, Monitoring

8.6.2 Assembly of Site-Wide Alternatives

The SLB waste alternatives and the groundwater alternatives have been combined to satisfy the remedial response objectives for the site. Table 8.5 illustrates the major components of the assembled alternatives. These alternatives will be defined and evaluated in detail in the FS/CMS/EA report. During the technical evaluations, some of these alternatives may be modified.

Section 8 Tables

Table 8.1. General response actions, ORNL WAG 6

Media/source	Risk type	General response action
SLB solid waste	Environmental	No action Containment Removal/disposal Treatment
	Human health	No action Institutional control Containment Removal/disposal Treatment
Groundwater	Environmental	No action Containment Removal/disposal Treatment
	Human health	No action Institutional control Monitoring Containment Removal/disposal Treatment

Table 8.2. Site and contaminant characteristics considered in remedial technology screening, ORNL WAG 6

Physical Site Characteristics

Climate: Humid, minor freeze/thaw
Soil Properties: Clayey, disturbed, varying depth
Aquifer Properties: Shallow
Bedrock Properties: Fractured, areas of shallow bedrock
Terrain: Varied, steep
Access: Easy, yet controlled
Depth to Aquitard: Unknown

Contaminant and Waste Characteristics

Type: Mixed, organic, radioactive
Waste Form: Varied; mostly solid
Previous Actions: Varied; containment, cap, fence
Quantity: High volumes of waste
Persistence: Varied; most is high
Biodegradability: Varied; most is low
Volatility: Varied; most is low
Explosivity: Varied; most is low
Solubility: Varied
Radioactivity: Varied
Corrosivity: Varied

Table 8.3. Summary of evaluation of SLB waste alternatives

Alternative	Major components	Effectiveness	Implementability	Cost	Retained
1	No Action	No protectiveness.		Low	Yes
2	Institutional Controls	Protects public health, but limited reliability for long-term. Little protection of environment.	Easy. Most controls already in existence and in place.	Medium	No
3	Small caps	Protects public health with improved reliability. Improved effectiveness for environment.	Technically easy.	Medium	Yes
4	Large caps, structural stabilization, consolidation	Protects public health and environment. Long-term reliability still a concern.	Somewhat difficult technically especially due to consolidation, but not excessive.	High	Yes
5	Large caps, structural stabilization, consolidation, in situ vitrification	Protects public health and environment. Long-term reliability still a concern.	More difficult than Alts. 3 and 4. Not too difficult.	High	Yes
6	Large caps, structural stab., consolidation, incineration	Worker exposure a concern. Not much effectiveness improvement.	Technically difficult if not impossible due to disposal issues. Difficult administratively.	Very high	No

Table 8.4. Summary of evaluation of groundwater alternatives

Alternative	Major components	Effectiveness	Implementability	Cost	Retained
1	No Action	Not effective in protecting public health or environment.	Nothing to implement.	Low	Yes
2	Institutional Controls	Protects public health, but limited reliability. No improvement in environmental quality.	Easy. Most controls already in existence and in place.	Low	Yes
3	Future decision	Most effective as objectives of remediation are for entire area and consistent.	No interference between WAG remedial actions. Should be technically straightforward. Administratively, best to address as area-wide operable unit.	Low, but not definable	Yes
4	Boundary collection, treatment at ORNL system	Protects public health and improves environmental conditions. Fairly reliable.	Construction of trenches in bedrock is difficult.	Medium	Yes

Table 8.5. WAG 6 alternative assessment preliminary site-wide alternatives

	Alternatives						
Components	1	2	3	4	5	6	7
SLB Waste							
Small Caps	NO ACTION	X	X				
Large Caps				X	X	X	X
Structural Stabilization				X	X	X	X
In Situ Vitrification							X
Consolidation				X	X	X	X
Groundwater							
Monitoring		X	X	X	X	X	X
Institutional Controls		X	X	X	X	X	X
Future Decision		X			X		
Boundary Drains			X			X	X
ORNL Liquid Waste Treatment Systems			X			X	X

Section 8 Figures

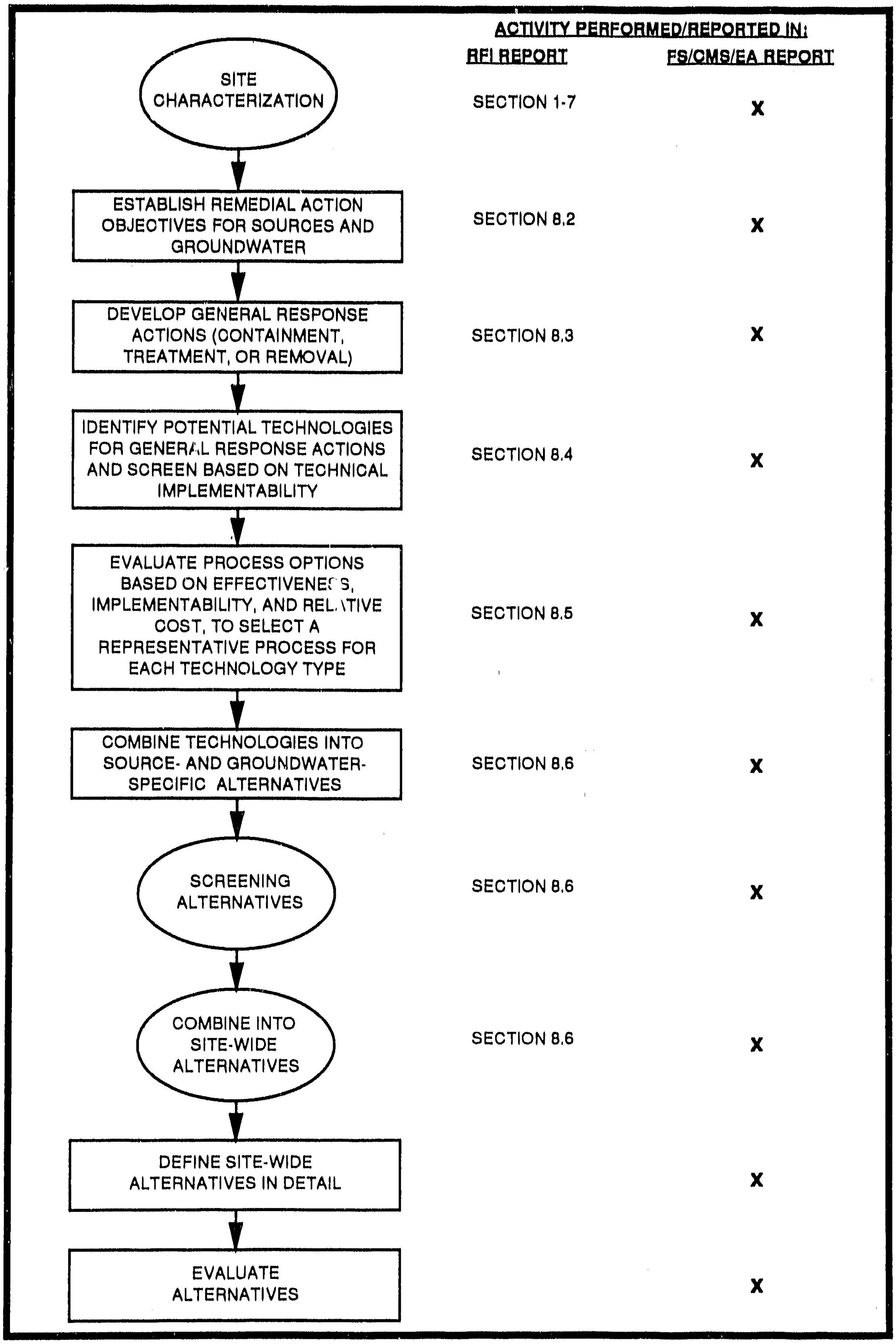

Fig. 8.1. Alternative development and evaluation process.

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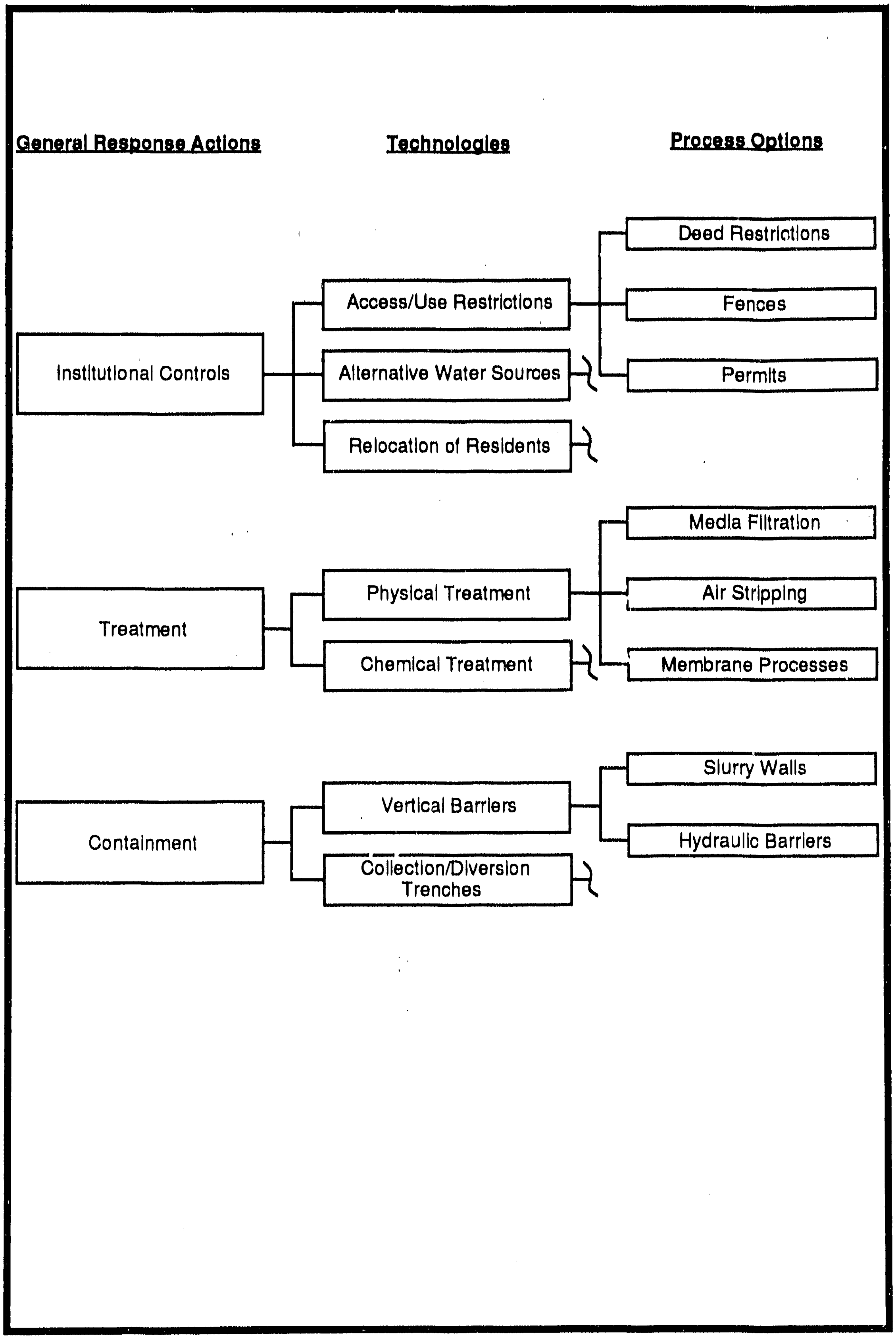

Fig. 8.2. Example of relationship among general response actions, technologies, and process options.

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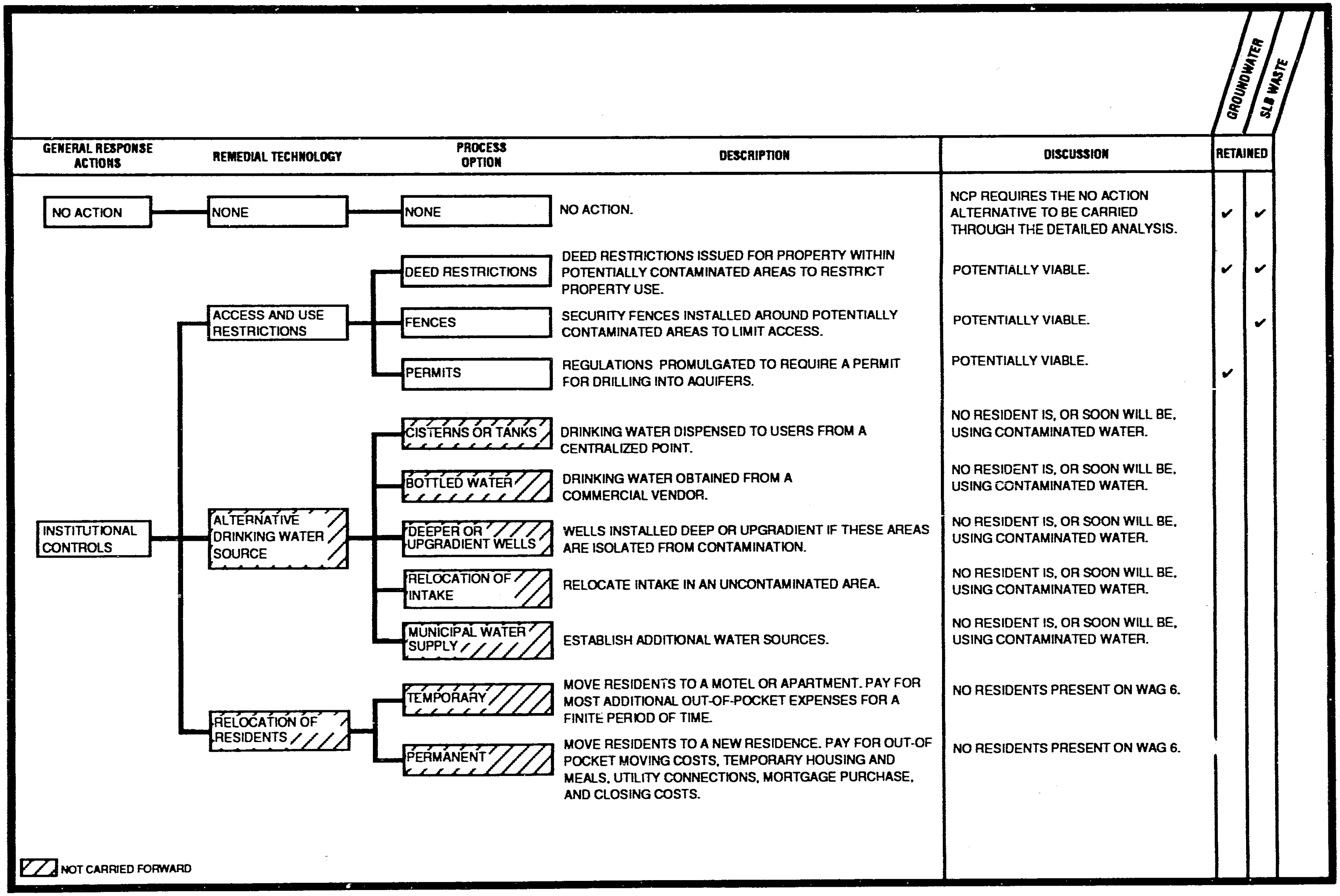

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED: GROUNDWATER	RETAINED: SLB WASTE
NO ACTION	NONE	NONE	NO ACTION.	NCP REQUIRES THE NO ACTION ALTERNATIVE TO BE CARRIED THROUGH THE DETAILED ANALYSIS.	✓	✓
INSTITUTIONAL CONTROLS	ACCESS AND USE RESTRICTIONS	DEED RESTRICTIONS	DEED RESTRICTIONS ISSUED FOR PROPERTY WITHIN POTENTIALLY CONTAMINATED AREAS TO RESTRICT PROPERTY USE.	POTENTIALLY VIABLE.	✓	✓
		FENCES	SECURITY FENCES INSTALLED AROUND POTENTIALLY CONTAMINATED AREAS TO LIMIT ACCESS.	POTENTIALLY VIABLE.		✓
		PERMITS	REGULATIONS PROMULGATED TO REQUIRE A PERMIT FOR DRILLING INTO AQUIFERS.	POTENTIALLY VIABLE.	✓	
	▨ ALTERNATIVE DRINKING WATER SOURCE	▨ CISTERNS OR TANKS	DRINKING WATER DISPENSED TO USERS FROM A CENTRALIZED POINT.	NO RESIDENT IS, OR SOON WILL BE, USING CONTAMINATED WATER.		
		▨ BOTTLED WATER	DRINKING WATER OBTAINED FROM A COMMERCIAL VENDOR.	NO RESIDENT IS, OR SOON WILL BE, USING CONTAMINATED WATER.		
		▨ DEEPER OR UPGRADIENT WELLS	WELLS INSTALLED DEEP OR UPGRADIENT IF THESE AREAS ARE ISOLATED FROM CONTAMINATION.	NO RESIDENT IS, OR SOON WILL BE, USING CONTAMINATED WATER.		
		▨ RELOCATION OF INTAKE	RELOCATE INTAKE IN AN UNCONTAMINATED AREA.	NO RESIDENT IS, OR SOON WILL BE, USING CONTAMINATED WATER.		
		▨ MUNICIPAL WATER SUPPLY	ESTABLISH ADDITIONAL WATER SOURCES.	NO RESIDENT IS, OR SOON WILL BE, USING CONTAMINATED WATER.		
	▨ RELOCATION OF RESIDENTS	▨ TEMPORARY	MOVE RESIDENTS TO A MOTEL OR APARTMENT. PAY FOR MOST ADDITIONAL OUT-OF-POCKET EXPENSES FOR A FINITE PERIOD OF TIME.	NO RESIDENTS PRESENT ON WAG 6.		
		▨ PERMANENT	MOVE RESIDENTS TO A NEW RESIDENCE. PAY FOR OUT-OF POCKET MOVING COSTS, TEMPORARY HOUSING AND MEALS, UTILITY CONNECTIONS, MORTGAGE PURCHASE, AND CLOSING COSTS.	NO RESIDENTS PRESENT ON WAG 6.		

▨ NOT CARRIED FORWARD

Fig. 8.3. Identification of applicable remedial technologies and process options.

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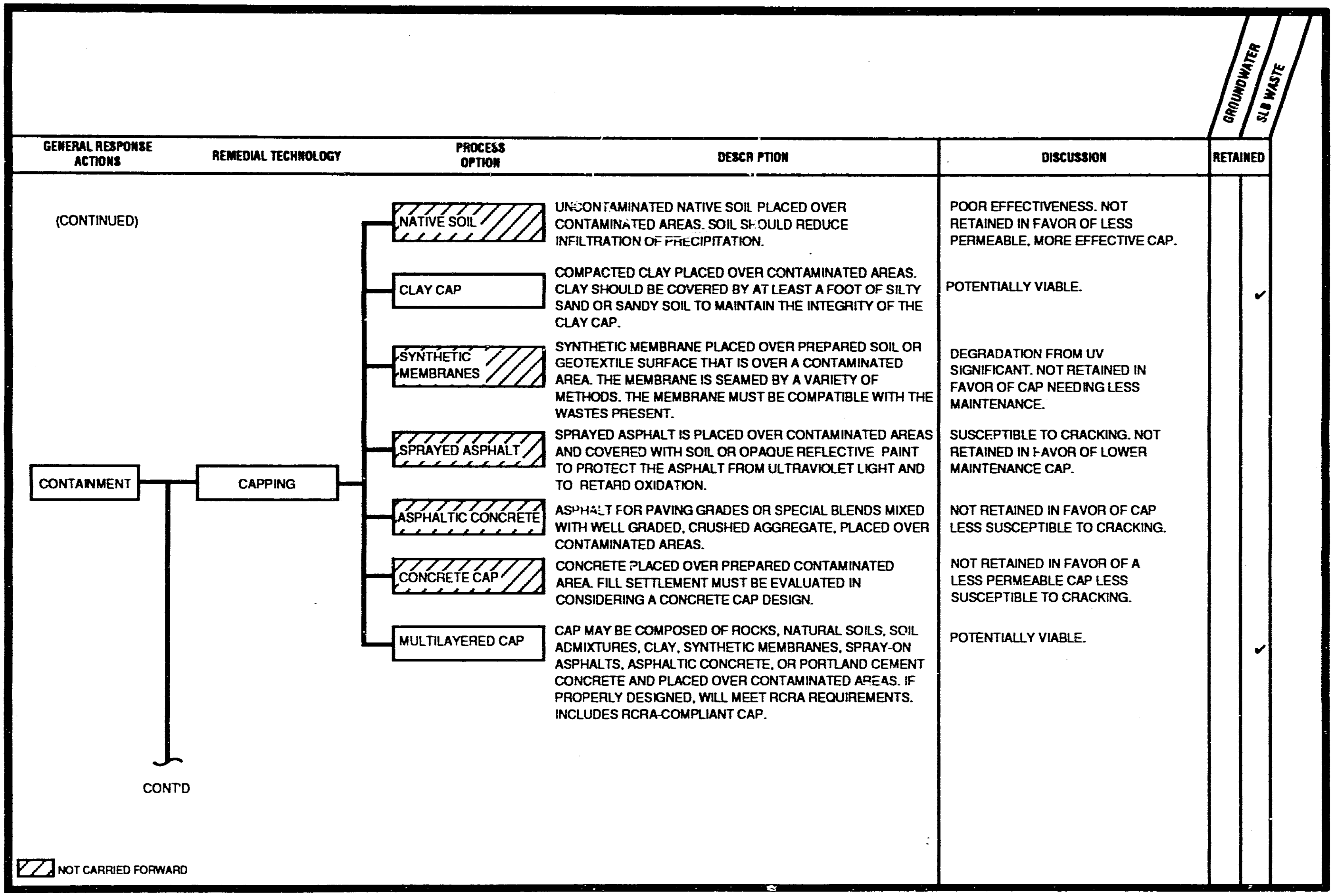

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED: GROUNDWATER	RETAINED: SLB WASTE
(CONTINUED) CONTAINMENT	CAPPING	▨ NATIVE SOIL	UNCONTAMINATED NATIVE SOIL PLACED OVER CONTAMINATED AREAS. SOIL SHOULD REDUCE INFILTRATION OF PRECIPITATION.	POOR EFFECTIVENESS. NOT RETAINED IN FAVOR OF LESS PERMEABLE, MORE EFFECTIVE CAP.		
		CLAY CAP	COMPACTED CLAY PLACED OVER CONTAMINATED AREAS. CLAY SHOULD BE COVERED BY AT LEAST A FOOT OF SILTY SAND OR SANDY SOIL TO MAINTAIN THE INTEGRITY OF THE CLAY CAP.	POTENTIALLY VIABLE.		✓
		▨ SYNTHETIC MEMBRANES	SYNTHETIC MEMBRANE PLACED OVER PREPARED SOIL OR GEOTEXTILE SURFACE THAT IS OVER A CONTAMINATED AREA. THE MEMBRANE IS SEAMED BY A VARIETY OF METHODS. THE MEMBRANE MUST BE COMPATIBLE WITH THE WASTES PRESENT.	DEGRADATION FROM UV SIGNIFICANT. NOT RETAINED IN FAVOR OF CAP NEEDING LESS MAINTENANCE.		
		▨ SPRAYED ASPHALT	SPRAYED ASPHALT IS PLACED OVER CONTAMINATED AREAS AND COVERED WITH SOIL OR OPAQUE REFLECTIVE PAINT TO PROTECT THE ASPHALT FROM ULTRAVIOLET LIGHT AND TO RETARD OXIDATION.	SUSCEPTIBLE TO CRACKING. NOT RETAINED IN FAVOR OF LOWER MAINTENANCE CAP.		
		▨ ASPHALTIC CONCRETE	ASPHALT FOR PAVING GRADES OR SPECIAL BLENDS MIXED WITH WELL GRADED, CRUSHED AGGREGATE, PLACED OVER CONTAMINATED AREAS.	NOT RETAINED IN FAVOR OF CAP LESS SUSCEPTIBLE TO CRACKING.		
		▨ CONCRETE CAP	CONCRETE PLACED OVER PREPARED CONTAMINATED AREA. FILL SETTLEMENT MUST BE EVALUATED IN CONSIDERING A CONCRETE CAP DESIGN.	NOT RETAINED IN FAVOR OF A LESS PERMEABLE CAP LESS SUSCEPTIBLE TO CRACKING.		
		MULTILAYERED CAP	CAP MAY BE COMPOSED OF ROCKS, NATURAL SOILS, SOIL ADMIXTURES, CLAY, SYNTHETIC MEMBRANES, SPRAY-ON ASPHALTS, ASPHALTIC CONCRETE, OR PORTLAND CEMENT CONCRETE AND PLACED OVER CONTAMINATED AREAS. IF PROPERLY DESIGNED, WILL MEET RCRA REQUIREMENTS. INCLUDES RCRA-COMPLIANT CAP.	POTENTIALLY VIABLE.		✓

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

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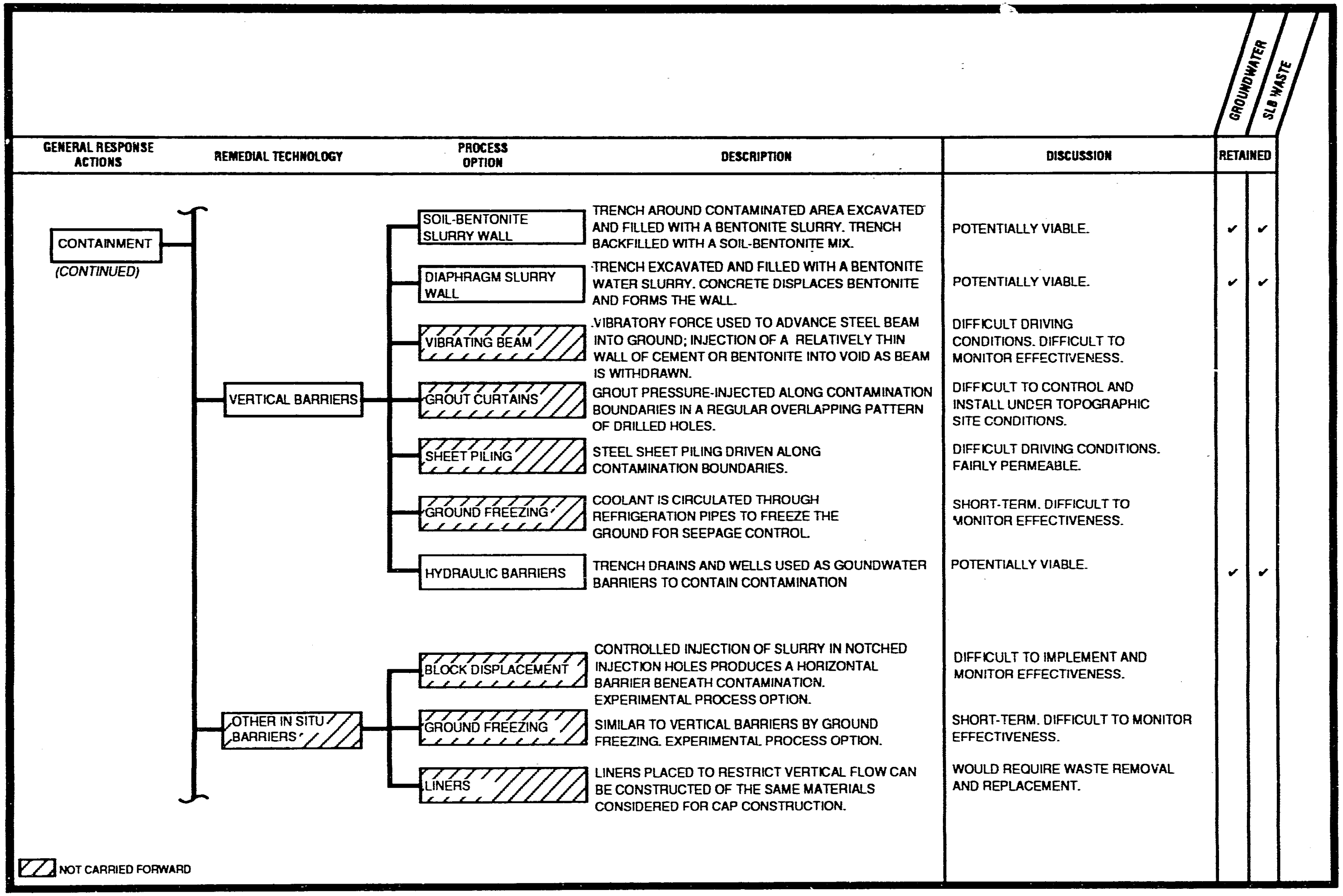

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED GROUNDWATER	RETAINED SLB WASTE
CONTAINMENT (CONTINUED)	VERTICAL BARRIERS	SOIL-BENTONITE SLURRY WALL	TRENCH AROUND CONTAMINATED AREA EXCAVATED AND FILLED WITH A BENTONITE SLURRY. TRENCH BACKFILLED WITH A SOIL-BENTONITE MIX.	POTENTIALLY VIABLE.	✓	✓
		DIAPHRAGM SLURRY WALL	TRENCH EXCAVATED AND FILLED WITH A BENTONITE WATER SLURRY. CONCRETE DISPLACES BENTONITE AND FORMS THE WALL.	POTENTIALLY VIABLE.	✓	✓
		▨ VIBRATING BEAM	VIBRATORY FORCE USED TO ADVANCE STEEL BEAM INTO GROUND; INJECTION OF A RELATIVELY THIN WALL OF CEMENT OR BENTONITE INTO VOID AS BEAM IS WITHDRAWN.	DIFFICULT DRIVING CONDITIONS. DIFFICULT TO MONITOR EFFECTIVENESS.		
		▨ GROUT CURTAINS	GROUT PRESSURE-INJECTED ALONG CONTAMINATION BOUNDARIES IN A REGULAR OVERLAPPING PATTERN OF DRILLED HOLES.	DIFFICULT TO CONTROL AND INSTALL UNDER TOPOGRAPHIC SITE CONDITIONS.		
		▨ SHEET PILING	STEEL SHEET PILING DRIVEN ALONG CONTAMINATION BOUNDARIES.	DIFFICULT DRIVING CONDITIONS. FAIRLY PERMEABLE.		
		▨ GROUND FREEZING	COOLANT IS CIRCULATED THROUGH REFRIGERATION PIPES TO FREEZE THE GROUND FOR SEEPAGE CONTROL.	SHORT-TERM. DIFFICULT TO MONITOR EFFECTIVENESS.		
		HYDRAULIC BARRIERS	TRENCH DRAINS AND WELLS USED AS GOUNDWATER BARRIERS TO CONTAIN CONTAMINATION	POTENTIALLY VIABLE.	✓	✓
	▨ OTHER IN SITU BARRIERS	▨ BLOCK DISPLACEMENT	CONTROLLED INJECTION OF SLURRY IN NOTCHED INJECTION HOLES PRODUCES A HORIZONTAL BARRIER BENEATH CONTAMINATION. EXPERIMENTAL PROCESS OPTION.	DIFFICULT TO IMPLEMENT AND MONITOR EFFECTIVENESS.		
		▨ GROUND FREEZING	SIMILAR TO VERTICAL BARRIERS BY GROUND FREEZING. EXPERIMENTAL PROCESS OPTION.	SHORT-TERM. DIFFICULT TO MONITOR EFFECTIVENESS.		
		▨ LINERS	LINERS PLACED TO RESTRICT VERTICAL FLOW CAN BE CONSTRUCTED OF THE SAME MATERIALS CONSIDERED FOR CAP CONSTRUCTION.	WOULD REQUIRE WASTE REMOVAL AND REPLACEMENT.		

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED: GROUNDWATER	RETAINED: SLB WASTE
CONTAINMENT (CONTINUED)	SURFACE CONTROLS	SURFACE SEALING	COVER MATERIALS AND SEAL TECHNIQUES IMPLEMENTED TO STABILIZE WASTE DEPOSITS AND TO PREVENT SURFACE WATER INFILTRATION, CONTROL EROSION, AND ISOLATE AND CONTAIN THE WASTES. SIMILAR TO CAPPING.	POTENTIALLY VIABLE.		✓
		GRADING	RESHAPING OF TOPOGRAPHY TO MANAGE SURFACE WATER, INFILTRATION AND RUNOFF TO CONTROL EROSION.	POTENTIALLY VIABLE.		✓
		IN SITU GROUTING	INJECTION OF GROUT INTO BURIED SOURCES TO FILL VOIDS AND PROVIDE STRUCTURAL STABILIZATION.	POTENTIALLY VIABLE.		✓
		REVEGETATION	A SYSTEMATIC REVEGETATION PLAN INCLUDES SELECTION OF A SUITABLE PLANT SPECIES, SEEDBED PREPARATION, SEEDING/PLANTING, MULCHING AND/OR CHEMICAL STABILIZATION, FERTILIZATION, AND MAINTENANCE.	POTENTIALLY VIABLE.		✓
		DIVERSION AND COLLECTION SYSTEMS	DIVERSION AND COLLECTION STRUCTURES INSTALLED UPSLOPE OR AT PERIMETER OF THE SITE TO CONTROL DRAINAGE OF STORMWATER RUNOFF. SYSTEM CAN ALSO BE IMPLEMENTED TO COLLECT CONTAMINATED SURFACE WATER FOR REMEDIATION.	POTENTIALLY VIABLE.		✓
		DYNAMIC COMPACTION	LARGE WEIGHT DROPPED ON WASTE TO COMPACT WASTE TO MINIMIZE FUTURE SUBSIDENCE.	POTENTIALLY VIABLE.		✓

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

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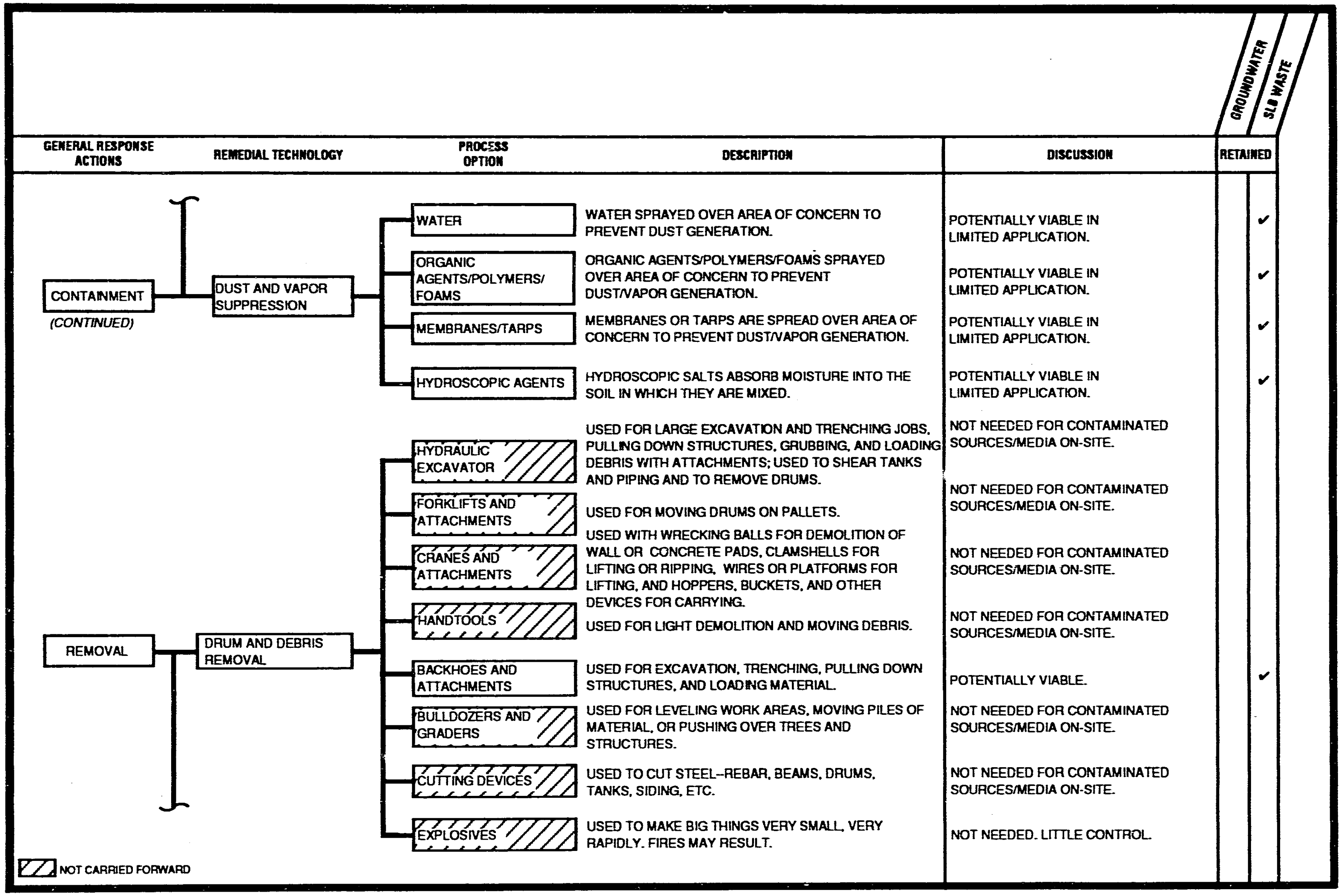

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED: GROUNDWATER	RETAINED: SLB WASTE
CONTAINMENT (CONTINUED)	DUST AND VAPOR SUPPRESSION	WATER	WATER SPRAYED OVER AREA OF CONCERN TO PREVENT DUST GENERATION.	POTENTIALLY VIABLE IN LIMITED APPLICATION.		✓
		ORGANIC AGENTS/POLYMERS/ FOAMS	ORGANIC AGENTS/POLYMERS/FOAMS SPRAYED OVER AREA OF CONCERN TO PREVENT DUST/VAPOR GENERATION.	POTENTIALLY VIABLE IN LIMITED APPLICATION.		✓
		MEMBRANES/TARPS	MEMBRANES OR TARPS ARE SPREAD OVER AREA OF CONCERN TO PREVENT DUST/VAPOR GENERATION.	POTENTIALLY VIABLE IN LIMITED APPLICATION.		✓
		HYDROSCOPIC AGENTS	HYDROSCOPIC SALTS ABSORB MOISTURE INTO THE SOIL IN WHICH THEY ARE MIXED.	POTENTIALLY VIABLE IN LIMITED APPLICATION.		✓
REMOVAL	DRUM AND DEBRIS REMOVAL	▨ HYDRAULIC EXCAVATOR	USED FOR LARGE EXCAVATION AND TRENCHING JOBS, PULLING DOWN STRUCTURES, GRUBBING, AND LOADING DEBRIS WITH ATTACHMENTS; USED TO SHEAR TANKS AND PIPING AND TO REMOVE DRUMS.	NOT NEEDED FOR CONTAMINATED SOURCES/MEDIA ON-SITE.		
		▨ FORKLIFTS AND ATTACHMENTS	USED FOR MOVING DRUMS ON PALLETS.	NOT NEEDED FOR CONTAMINATED SOURCES/MEDIA ON-SITE.		
		▨ CRANES AND ATTACHMENTS	USED WITH WRECKING BALLS FOR DEMOLITION OF WALL OR CONCRETE PADS, CLAMSHELLS FOR LIFTING OR RIPPING, WIRES OR PLATFORMS FOR LIFTING, AND HOPPERS, BUCKETS, AND OTHER DEVICES FOR CARRYING.	NOT NEEDED FOR CONTAMINATED SOURCES/MEDIA ON-SITE.		
		▨ HANDTOOLS	USED FOR LIGHT DEMOLITION AND MOVING DEBRIS.	NOT NEEDED FOR CONTAMINATED SOURCES/MEDIA ON-SITE.		
		BACKHOES AND ATTACHMENTS	USED FOR EXCAVATION, TRENCHING, PULLING DOWN STRUCTURES, AND LOADING MATERIAL.	POTENTIALLY VIABLE.		✓
		▨ BULLDOZERS AND GRADERS	USED FOR LEVELING WORK AREAS, MOVING PILES OF MATERIAL, OR PUSHING OVER TREES AND STRUCTURES.	NOT NEEDED FOR CONTAMINATED SOURCES/MEDIA ON-SITE.		
		▨ CUTTING DEVICES	USED TO CUT STEEL--REBAR, BEAMS, DRUMS, TANKS, SIDING, ETC.	NOT NEEDED FOR CONTAMINATED SOURCES/MEDIA ON-SITE.		
		▨ EXPLOSIVES	USED TO MAKE BIG THINGS VERY SMALL, VERY RAPIDLY. FIRES MAY RESULT.	NOT NEEDED. LITTLE CONTROL.		

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED: GROUNDWATER	RETAINED: SLB WASTE
REMOVAL (CONTINUED)	EXCAVATION	SOLIDS	EXCAVATION OF CONTAMINATED SOLIDS CAN USE ORDINARY CONSTRUCTION EQUIPMENT (BACKHOES, BULLDOZERS, AND FRONT-END LOADERS).	POTENTIALLY VIABLE.		✓
		▨ SEMI-SOLIDS	EXCAVATION OF CONTAMINATED SEMI-SOLIDS REQUIRES WORKING ON MATS OR FLOATING EQUIPMENT. A TOOTHLESS BUCKET IS RECOMMENDED OVER DRAGLINES OR CLAMSHELLS. TRANSPORTATION AND DISPOSAL METHODS MUST CONSIDER THE WATER ASSOCIATED WITH THE SEMI-SOLIDS.	NO WET AREA LARGE ENOUGH TO REQUIRE FLOATING EQUIPMENT.		
		▨ SEDIMENTS	EXCAVATION OF CONTAMINATED SEDIMENTS REQUIRES FLOATING EQUIPMENT OR OPERATION OF EQUIPMENT ON THE BANK. HYDRAULIC DREDGES, BARGE-MOUNTED PUMPS, VACUUM TRUCKS, AND PNEUMATIC DREDGES ARE USED FOR EXCAVATION. GENERALLY BULK LIQUID REMOVAL WILL ALSO BE REQUIRED.	NOT NEEDED FOR CONTAMINATED SOURCES/MEDIA ON-SITE.		
	BULK LIQUID REMOVAL	PUMPS	USED TO REMOVE LARGE AMOUNTS OF LIQUID. LIQUID MUST BE FREE OF DEBRIS.	POTENTIALLY VIABLE IN LIMITED APPLICATION.		✓
		▨ DIVERSION	USE OF DIVERSION/COLLECTION SYSTEMS TO REMOVE LARGE SURFACE WATER BODIES.	NOT NEEDED FOR CONTAMINATED SOURCES/MEDIA ON-SITE.		
		▨ GRAVITY/SIPHON	USED TO REMOVE LIQUIDS FROM TANKS.	NO TANKS REQUIRE DEWATERING.		
		▨ DREDGES	USED TO REMOVE LARGE QUANTITIES OF LIQUID OR SLUDGE. A SYSTEM OF PIPELINES AND CONTAINMENT SYSTEMS IS REQUIRED.	NOT NEEDED FOR CONTAMINATED SOURCES/MEDIA ON-SITE.		
		▨ BAILING	USED TO REMOVE SMALL AMOUNTS OF LIQUID OR SLUDGE.	INEFFICIENT.		

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

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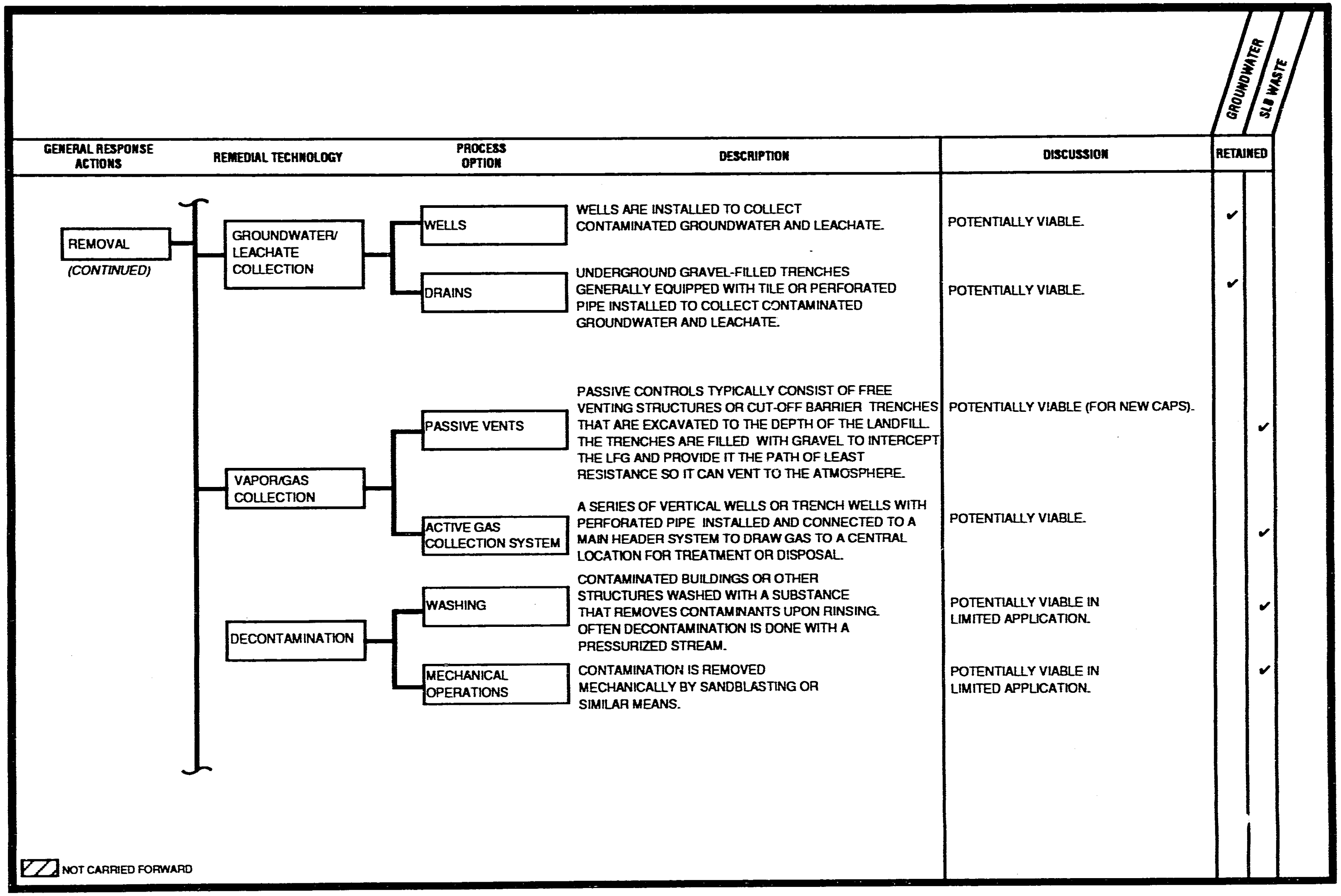
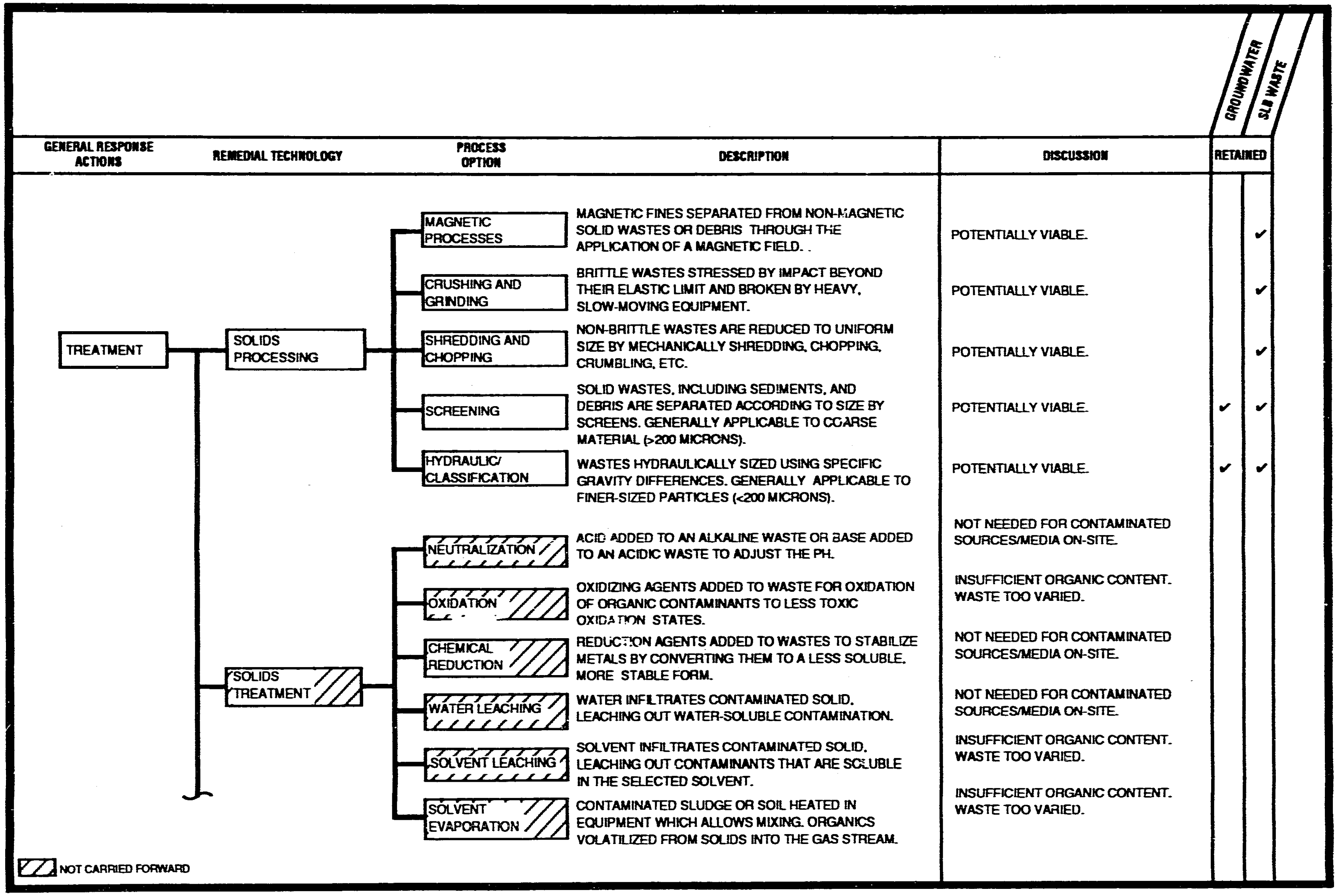


GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED: GROUNDWATER	RETAINED: SLB WASTE
REMOVAL *(CONTINUED)*	GROUNDWATER/ LEACHATE COLLECTION	WELLS	WELLS ARE INSTALLED TO COLLECT CONTAMINATED GROUNDWATER AND LEACHATE.	POTENTIALLY VIABLE.	✓	
		DRAINS	UNDERGROUND GRAVEL-FILLED TRENCHES GENERALLY EQUIPPED WITH TILE OR PERFORATED PIPE INSTALLED TO COLLECT CONTAMINATED GROUNDWATER AND LEACHATE.	POTENTIALLY VIABLE.	✓	
	VAPOR/GAS COLLECTION	PASSIVE VENTS	PASSIVE CONTROLS TYPICALLY CONSIST OF FREE VENTING STRUCTURES OR CUT-OFF BARRIER TRENCHES THAT ARE EXCAVATED TO THE DEPTH OF THE LANDFILL. THE TRENCHES ARE FILLED WITH GRAVEL TO INTERCEPT THE LFG AND PROVIDE IT THE PATH OF LEAST RESISTANCE SO IT CAN VENT TO THE ATMOSPHERE.	POTENTIALLY VIABLE (FOR NEW CAPS).		✓
		ACTIVE GAS COLLECTION SYSTEM	A SERIES OF VERTICAL WELLS OR TRENCH WELLS WITH PERFORATED PIPE INSTALLED AND CONNECTED TO A MAIN HEADER SYSTEM TO DRAW GAS TO A CENTRAL LOCATION FOR TREATMENT OR DISPOSAL.	POTENTIALLY VIABLE.		✓
	DECONTAMINATION	WASHING	CONTAMINATED BUILDINGS OR OTHER STRUCTURES WASHED WITH A SUBSTANCE THAT REMOVES CONTAMINANTS UPON RINSING. OFTEN DECONTAMINATION IS DONE WITH A PRESSURIZED STREAM.	POTENTIALLY VIABLE IN LIMITED APPLICATION.		✓
		MECHANICAL OPERATIONS	CONTAMINATION IS REMOVED MECHANICALLY BY SANDBLASTING OR SIMILAR MEANS.	POTENTIALLY VIABLE IN LIMITED APPLICATION.		✓

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)



GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED GROUNDWATER	RETAINED SLB WASTE
TREATMENT	SOLIDS PROCESSING	MAGNETIC PROCESSES	MAGNETIC FINES SEPARATED FROM NON-MAGNETIC SOLID WASTES OR DEBRIS THROUGH THE APPLICATION OF A MAGNETIC FIELD. .	POTENTIALLY VIABLE.		✓
		CRUSHING AND GRINDING	BRITTLE WASTES STRESSED BY IMPACT BEYOND THEIR ELASTIC LIMIT AND BROKEN BY HEAVY, SLOW-MOVING EQUIPMENT.	POTENTIALLY VIABLE.		✓
		SHREDDING AND CHOPPING	NON-BRITTLE WASTES ARE REDUCED TO UNIFORM SIZE BY MECHANICALLY SHREDDING, CHOPPING, CRUMBLING, ETC.	POTENTIALLY VIABLE.		✓
		SCREENING	SOLID WASTES, INCLUDING SEDIMENTS, AND DEBRIS ARE SEPARATED ACCORDING TO SIZE BY SCREENS. GENERALLY APPLICABLE TO COARSE MATERIAL (>200 MICRONS).	POTENTIALLY VIABLE.	✓	✓
		HYDRAULIC/ CLASSIFICATION	WASTES HYDRAULICALLY SIZED USING SPECIFIC GRAVITY DIFFERENCES. GENERALLY APPLICABLE TO FINER-SIZED PARTICLES (<200 MICRONS).	POTENTIALLY VIABLE.	✓	✓
	▨ SOLIDS TREATMENT	▨ NEUTRALIZATION	ACID ADDED TO AN ALKALINE WASTE OR BASE ADDED TO AN ACIDIC WASTE TO ADJUST THE PH.	NOT NEEDED FOR CONTAMINATED SOURCES/MEDIA ON-SITE.		
		▨ OXIDATION	OXIDIZING AGENTS ADDED TO WASTE FOR OXIDATION OF ORGANIC CONTAMINANTS TO LESS TOXIC OXIDATION STATES.	INSUFFICIENT ORGANIC CONTENT. WASTE TOO VARIED.		
		▨ CHEMICAL REDUCTION	REDUCTION AGENTS ADDED TO WASTES TO STABILIZE METALS BY CONVERTING THEM TO A LESS SOLUBLE, MORE STABLE FORM.	NOT NEEDED FOR CONTAMINATED SOURCES/MEDIA ON-SITE.		
		▨ WATER LEACHING	WATER INFILTRATES CONTAMINATED SOLID, LEACHING OUT WATER-SOLUBLE CONTAMINATION.	NOT NEEDED FOR CONTAMINATED SOURCES/MEDIA ON-SITE.		
		▨ SOLVENT LEACHING	SOLVENT INFILTRATES CONTAMINATED SOLID, LEACHING OUT CONTAMINANTS THAT ARE SOLUBLE IN THE SELECTED SOLVENT.	INSUFFICIENT ORGANIC CONTENT. WASTE TOO VARIED.		
		▨ SOLVENT EVAPORATION	CONTAMINATED SLUDGE OR SOIL HEATED IN EQUIPMENT WHICH ALLOWS MIXING. ORGANICS VOLATILIZED FROM SOLIDS INTO THE GAS STREAM.	INSUFFICIENT ORGANIC CONTENT. WASTE TOO VARIED.		

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

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GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED	
					GROUNDWATER	SLB WASTE
TREATMENT (CONTINUED)	SOLIDIFICATION, STABILIZATION	SORPTION	AN INERT, NONDEGRADABLE, NONREACTIVE SORBENT ADDED TO LIQUID OR SLUDGE TO SOAK UP MOISTURE AND PRODUCE A SOIL-LIKE MATERIAL.	POTENTIALLY VIABLE.		✓
		POZZOLANIC AGENTS	LIME AND FINE-GRAINED SILICEOUS MATERIAL ADDED TO AQUEOUS WASTE TO PRODUCE A CONCRETE-LIKE SOLID.	POTENTIALLY VIABLE.		✓
		ENCAPSULATION	MATERIAL IS ENCAPSULATED BY EITHER COATING THE INDIVIDUAL PARTICLES OR ENCLOSING ENTIRE MASS IN A JACKET OF INERT MATERIAL.	POTENTIALLY VIABLE.		✓
	SOLIDS DEWATERING	▨ SEDIMENTATION	SUSPENDED SOLIDS REMOVED FROM LIQUID BY GRAVITY IN A TANK OR LAGOON. SEDIMENTATION OFTEN PRECEDED BY CHEMICAL PRECIPITATION AND/OR BY COAGULATION AND FLOCCULATION.	NOT APPROPRIATE FOR LARGE SOLID MATERIALS.		
		▨ GRAVITY THICKENING	WATER REMOVED FROM SLUDGE BY GRAVITY IN THICKENING TANKS. CHEMICALS CAN AID IN SLUDGE DEWATERING.	NOT APPROPRIATE FOR LARGE SOLID MATERIALS.		
		▨ CENTRIFUGES	SLUDGE DEWATERED BY CENTRIFUGAL FORCE. CENTRIFUGES ARE USED WHEN SPACE IS LIMITED OR WHERE SLUDGE INCINERATION/DISPOSAL IS REQUIRED.	NOT APPROPRIATE FOR LARGE SOLID MATERIALS.		
		▨ FILTER PRESSES	SLUDGE PASSED INTO PRESS UNDER PRESSURE. WATER PASSES THROUGH FILTER CLOTH WHILE SOLIDS FORM A CAKE ON THE FILTER CLOTH SURFACE.	NOT APPROPRIATE FOR LARGE SOLID MATERIALS.		
		▨ VACUUM FILTRATION	A PARTIALLY SUBMERGED CYLINDRICAL DRUM ROTATES IN A VAT OR PAN OF CONDITIONED SLUDGE. A VACUUM IS PULLED INSIDE THE DRUM, SOLIDS COLLECT ON OUTSIDE AND ARE CONTINUOUSLY SCRAPED OFF.	NOT APPROPRIATE FOR LARGE SOLID MATERIALS.		
		DEWATERING AND DRYING BEDS	WASTE PLACED ON SAND LAYERED OVER GRAVEL. WATER EVAPORATES AND DRAINS.	POTENTIALLY VIABLE.		✓
		▨ SLUDGE DRYERS	CONTAMINATED SLUDGE OR SOIL HEATED IN EQUIPMENT THAT ALLOWS MIXING. AQUEOUS LIQUIDS EVAPORATED FROM SOLIDS INTO THE GAS STREAM.	NOT APPROPRIATE FOR LARGE SOLID MATERIALS.		

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

5.18A 4104.80

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED GROUNDWATER	RETAINED SLB WASTE
TREATMENT *(CONTINUED)*	PHYSICAL TREATMENT	FLOW AND STRENGTH EQUALIZATION	EQUALIZATION BASIN REGULATES FLOW THROUGH A TREATMENT FACILITY. MECHANICAL MIXING EQUALIZES CONCENTRATION.	POTENTIALLY VIABLE.	✓	
		COAGULATION/ FLOCCULATION	SUSPENDED PARTICLE AGGREGATION THAT RESULTS FROM CHEMICAL ADDITIONS TO THE WASTEWATER. ONCE PARTICLES HAVE AGGREGATED INTO LARGE PARTICLES, THEY CAN BE REMOVED BY SEDIMENTATION, FLOTATION, OR FILTRATION.	POTENTIALLY VIABLE.	✓	
		▨ OIL-WATER SEPARATION	OIL SEPARATED FROM WATER BY GRAVITY. TREATMENT MAY BE REQUIRED FOR DISSOLVED AND EMULSIFIED OIL PRIOR TO SIMPLE PHYSICAL SEPARATION.	NO OIL TO SEPARATE.		
		▨ FLOTATION	GAS BUBBLES RELEASED INTO WASTEWATER ATTACH TO SUSPENDED SOLIDS, FREE AND EMULSIFY OILS, AND GREASE. THESE FLOAT TO THE TOP AND ARE SKIMMED OFF.	INSUFFICIENT SOLIDS, OILS, AND GREASES.		
		MEDIA-FILTRATION	FINE SOLID PARTICLES REMOVED FROM LIQUID STREAM BY A FILTER MEDIUM. COMMON MEDIA ARE SAND, DIATOMITE, COAL, NATURAL OR SYNTHETIC FABRIC, AND WIRE CLOTH.	POTENTIALLY VIABLE.	✓	✓
		ADSORPTION	CONTAMINATED LIQUID OR GAS STREAM PASSED OVER ADSORBENT THAT REMOVES CONTAMINANTS.	POTENTIALLY VIABLE.	✓	✓
		MEMBRANE PROCESSES	BARRIER MEMBRANE PREFERENTIALLY PASSES SOME COMPONENTS OF A FLUID MIXTURE OR SOLUTION.	POTENTIALLY VIABLE.	✓	✓
		▨ CRYSTALLIZATION	DISSOLVED SOLIDS REMOVED FROM LOW-VISCOSITY LIQUIDS, SLURRIES, AND SLUDGES WHICH HAVE A COMPONENT THAT FREEZES.	WATER WOULD FREEZE WITH DISSOLVED SOLIDS. NOT EFFECTIVE ON GROUNDWATER.		
		AIR STRIPPING	LARGE VOLUMES OF AIR FORCED THROUGH LIQUID IN A PACKED COLUMN OR BY DIFFUSED AERATION TO PROMOTE TRANSFER OF VOCS TO AIR.	POTENTIALLY VIABLE.	✓	✓
		▨ STEAM STRIPPING	SIMILAR TO AIR STRIPPING EXCEPT STEAM PUMPED THROUGH STRIPPING COLUMN TO ADD HEAT IN THE PROMOTION OF VOC TRANSFER FROM LIQUID TO AIR.	WILL BE CONSIDERED A SUBSET OF AIR STRIPPING.		

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

5.18A 4104.81

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED: GROUNDWATER	RETAINED: SLB WASTE
TREATMENT (CONTINUED)	PHYSICAL TREATMENT (CONTINUED)	▨ EVAPORATION	WASTEWATER HEATED TO ITS BOILING POINT THEN CONDENSED TO RECOVER WATER AND A BRINE STREAM WHICH CONTAINS VIRTUALLY ALL THE DISSOLVED SOLIDS.	RADIOACTIVE CONTAMINANTS MAY BE VOLATILIZED. LOW VOC CONTENT.		
		▨ DISTILLATION	ORGANIC LIQUID WASTES ARE BOILED TO RELEASE VOLATILE COMPOUNDS INTO THE VAPOR PHASE. THE VAPOR PHASE IS CONDENSED, TREATED AND/OR DISPOSED.	USUALLY USED FOR MORE CONCENTRATED ORGANICS.		
		▨ LIQUID-LIQUID EXTRACTION	TWO IMMISCIBLE LIQUIDS BROUGHT TOGETHER BY FORCED MIXING OR COUNTERCURRENT FLOW TO PROMOTE THE TRANSFER OF A SOLUTE FROM ONE LIQUID TO THE OTHER. APPLICABLE FOR REMOVAL OF ORGANICS FROM AQUEOUS SOLUTIONS OR RADIONUCLIDES BY AN EXTRACTANT.	TYPES OF CONTAMINANTS TOO VARIED. PROCESS EXPERIMENTAL FOR RADIOACTIVITY.		
		▨ CRITICAL FLUID EXTRACTION	A GAS (TYPICALLY CO2) AT ITS CRITICAL POINT IS FED COUNTERCURRENT TO A CONTAMINATED WASTE, EXTRACTING ORGANICS THEN SEPARATE FROM THE GAS AT ATMOSPHERIC CONDITIONS.	USED FOR MORE CONCENTRATED ORGANICS.		
	CHEMICAL TREATMENT	NEUTRALIZATION	ACID ADDED TO AN ALKALINE WASTE OR BASE ADDED TO AN ACIDIC WASTE TO ADJUST THE STREAM TO A pH NEAR NEUTRALITY.	POTENTIALLY VIABLE.	✓	✓
		PRECIPITATION	CHEMICAL EQUILIBRIA OF WASTE ALTERED TO REDUCE THE SOLUBILITY OF HEAVY METALS THROUGH THE ADDITION OF A SUBSTANCE THAT REACTS WITH THE METALS OR CHANGES THE pH OR THROUGH A CHANGE IN TEMPERATURE.	POTENTIALLY VIABLE.	✓	✓
		ION EXCHANGE	CONTAMINATED WATER PASSED THROUGH A BED OF RESIN MATERIAL WHERE EXCHANGE OF IONS OCCURS BETWEEN THE BED AND THE WATER.	POTENTIALLY VIABLE.	✓	✓
		▨ OXIDATION	OXIDIZING AGENTS ADDED TO WASTES FOR OXIDATION OF CYANIDE, HEAVY METALS, UNSATURATED ORGANICS, SULFIDES, SULFITES, PHENOLICS, PESTICIDES, ALDEHYDES, AND AROMATIC HYDROCARBONS TO LESS TOXIC OXIDATION STATES.	CONTAMINANTS TREATED BY OXIDATION NOT OF CONCERN.		

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

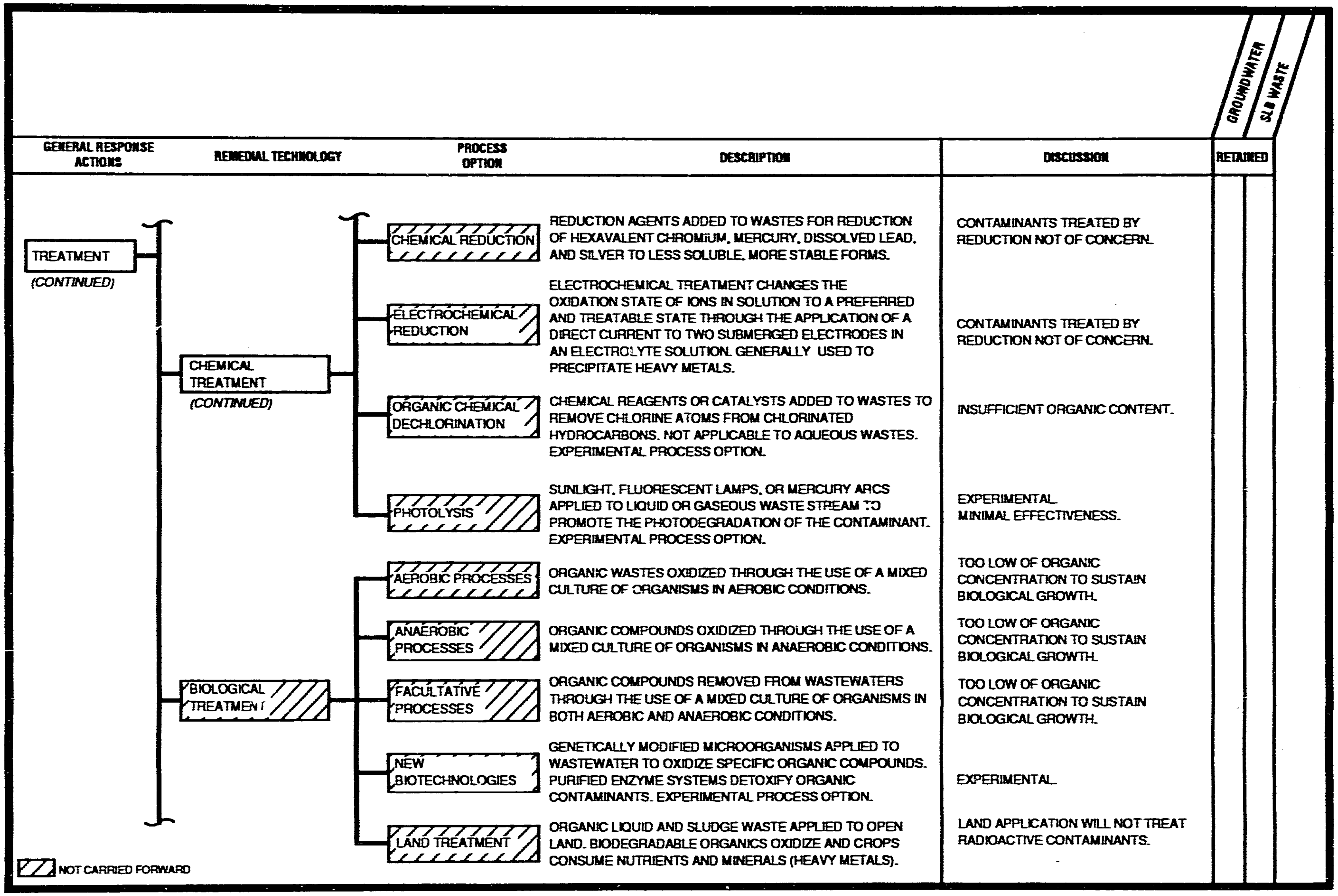

Fig. 8.3. (continued)

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED GROUNDWATER	RETAINED SLB WASTE
TREATMENT (CONTINUED)	CHEMICAL TREATMENT (CONTINUED)	CHEMICAL REDUCTION ▨	REDUCTION AGENTS ADDED TO WASTES FOR REDUCTION OF HEXAVALENT CHROMIUM, MERCURY, DISSOLVED LEAD, AND SILVER TO LESS SOLUBLE, MORE STABLE FORMS.	CONTAMINANTS TREATED BY REDUCTION NOT OF CONCERN.		
		ELECTROCHEMICAL REDUCTION ▨	ELECTROCHEMICAL TREATMENT CHANGES THE OXIDATION STATE OF IONS IN SOLUTION TO A PREFERRED AND TREATABLE STATE THROUGH THE APPLICATION OF A DIRECT CURRENT TO TWO SUBMERGED ELECTRODES IN AN ELECTROLYTE SOLUTION. GENERALLY USED TO PRECIPITATE HEAVY METALS.	CONTAMINANTS TREATED BY REDUCTION NOT OF CONCERN.		
		ORGANIC CHEMICAL DECHLORINATION ▨	CHEMICAL REAGENTS OR CATALYSTS ADDED TO WASTES TO REMOVE CHLORINE ATOMS FROM CHLORINATED HYDROCARBONS. NOT APPLICABLE TO AQUEOUS WASTES. EXPERIMENTAL PROCESS OPTION.	INSUFFICIENT ORGANIC CONTENT.		
		PHOTOLYSIS ▨	SUNLIGHT, FLUORESCENT LAMPS, OR MERCURY ARCS APPLIED TO LIQUID OR GASEOUS WASTE STREAM TO PROMOTE THE PHOTODEGRADATION OF THE CONTAMINANT. EXPERIMENTAL PROCESS OPTION.	EXPERIMENTAL. MINIMAL EFFECTIVENESS.		
	BIOLOGICAL TREATMENT ▨	AEROBIC PROCESSES ▨	ORGANIC WASTES OXIDIZED THROUGH THE USE OF A MIXED CULTURE OF ORGANISMS IN AEROBIC CONDITIONS.	TOO LOW OF ORGANIC CONCENTRATION TO SUSTAIN BIOLOGICAL GROWTH.		
		ANAEROBIC PROCESSES ▨	ORGANIC COMPOUNDS OXIDIZED THROUGH THE USE OF A MIXED CULTURE OF ORGANISMS IN ANAEROBIC CONDITIONS.	TOO LOW OF ORGANIC CONCENTRATION TO SUSTAIN BIOLOGICAL GROWTH.		
		FACULTATIVE PROCESSES ▨	ORGANIC COMPOUNDS REMOVED FROM WASTEWATERS THROUGH THE USE OF A MIXED CULTURE OF ORGANISMS IN BOTH AEROBIC AND ANAEROBIC CONDITIONS.	TOO LOW OF ORGANIC CONCENTRATION TO SUSTAIN BIOLOGICAL GROWTH.		
		NEW BIOTECHNOLOGIES ▨	GENETICALLY MODIFIED MICROORGANISMS APPLIED TO WASTEWATER TO OXIDIZE SPECIFIC ORGANIC COMPOUNDS. PURIFIED ENZYME SYSTEMS DETOXIFY ORGANIC CONTAMINANTS. EXPERIMENTAL PROCESS OPTION.	EXPERIMENTAL.		
		LAND TREATMENT ▨	ORGANIC LIQUID AND SLUDGE WASTE APPLIED TO OPEN LAND. BIODEGRADABLE ORGANICS OXIDIZE AND CROPS CONSUME NUTRIENTS AND MINERALS (HEAVY METALS).	LAND APPLICATION WILL NOT TREAT RADIOACTIVE CONTAMINANTS.		

▨ NOT CARRIED FORWARD

5.18A 4104.83

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED: GROUNDWATER	RETAINED: SLB WASTE
TREATMENT *(CONTINUED)*	IN SITU TREATMENT	▨ NEUTRALIZATION	SUBSTANCES APPLIED OR INJECTED INTO A CONTAMINATED SITE TO ADJUST THE PH OF THE WASTE. INNOVATIVE PROCESS OPTION FOR CONTAMINATED SOILS OR BURIED WASTES.	TOO DIFFICULT TO CONTROL INJECTION OF CHEMICALS AT SITE. WASTE TOO VARIED.		
		▨ OXIDATION	OZONE OR HYDROGEN PEROXIDE APPLIED IN WATER SOLUTIONS TO THE SOIL TO PROMOTE THE OXIDATION OF ORGANICS. CONCEPTUAL PROCESS OPTION.	TOO DIFFICULT TO CONTROL INJECTION OF CHEMICALS AT SITE. WASTE TOO VARIED.		
		▨ CHEMICAL REDUCTION	REDUCTION AGENTS APPLIED TO CONTAMINATED SOIL TO REDUCE CHLORINATED ORGANICS, UNSATURATED AROMATICS AND ALIPHATICS, AND HEAVY METALS TO LESS SOLUBLE, MORE STABLE FORMS. CONCEPTUAL PROCESS OPTION.	TOO DIFFICULT TO CONTROL INJECTION OF CHEMICALS AT SITE. WASTE TOO VARIED.		
		▨ PRECIPITATION	SUBSTANCES APPLIED TO SOIL OR STREAMS TO PROMOTE THE PRECIPITATION OF METALS AS SULFIDES, PHOSPHATES, AND HYDROXIDES. SUITABLE SOIL PH MUST BE MAINTAINED FOR MAXIMUM INSOLUBILITY OF METAL PRECIPITATES. EXPERIMENTAL PROCESS OPTION FOR SOILS.	TOO DIFFICULT TO CONTROL INJECTION OF CHEMICALS AT SITE. WASTE TOO VARIED.		
		▨ BIORECLAMATION	MICROBIOLOGICAL DEGRADATION, DETOXIFICATION, AND MINERALIZATION OF HAZARDOUS SUBSTANCES IS ENHANCED WITH RESPECT TO ELECTRON ACCEPTORS, NUTRIENT LEVELS, MOISTURE CONTENT, PH, AND TEMPERATURE.	TOO DIFFICULT TO CONTROL EFFECTIVENESS.		
		▨ THERMAL EVAPORATION	ELECTRODES PLACED OVER SURFACE OF CONTAMINATED SITE OR IN VERTICAL OR HORIZONTAL BORE-HOLES DRILLED THROUGH CONTAMINATED ZONE. ELECTROMAGNETIC ENERGY IN RADIOFREQUENCY BAND IS APPLIED TO ELECTRODES TO CAUSE RAPID HEATING OF THE SOIL BY MOLECULAR EXCITATION.	DIFFICULT TO CONTROL RELEASE OF RADIOACTIVE MATERIALS.		
		▨ STEAM STRIPPING	VOC REMOVAL FROM SOIL BY APPLICATION OF STEAM. INJECTION WELLS FORCE STEAM THROUGH CONTAMINATED SUBSURFACE. STEAM CARRYING STRIPPED VOLATILES IS COLLECTED AT SURFACE THROUGH EXTRACTION WELLS.	WILL BE CONSIDERED AS A SUBSET OF SOIL VAPOR EXTRACTION.		
		PERMEABLE TREATMENT BEDS	TRENCH EXCAVATED, FILLED WITH APPROPRIATE MATERIAL, AND CAPPED TO INTERCEPT CONTAMINATED GROUNDWATER. DEPENDING UPON BED MATERIAL, GROUNDWATER CAN BE NEUTRALIZED, METALS AND ORGANICS CAN BE REMOVED, AND NUTRIENTS CAN BE ADDED. CONCEPTUAL PROCESS OPTION.	POTENTIALLY VIABLE.	✓	

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

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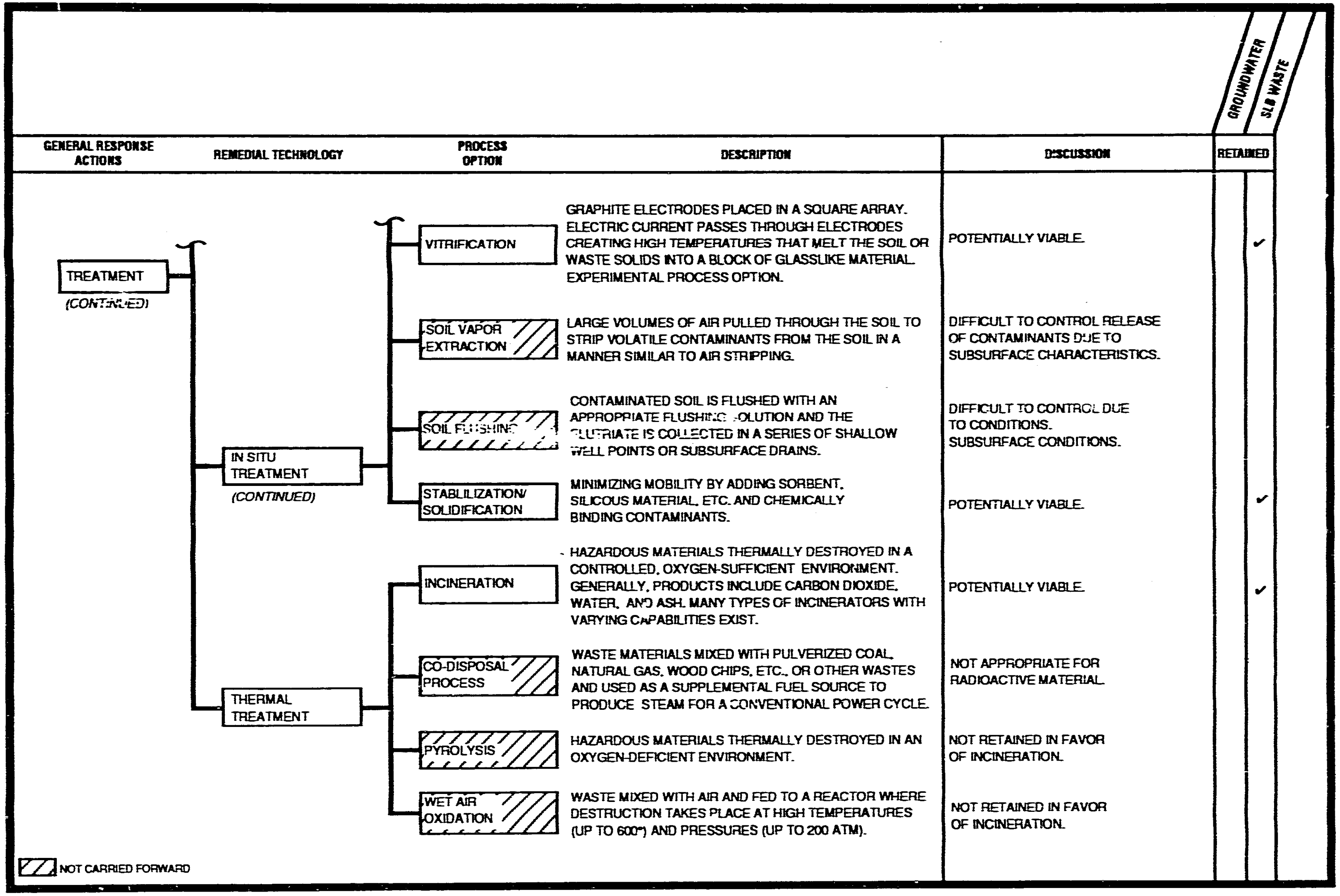

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED GROUNDWATER	RETAINED SLB WASTE
TREATMENT (CONTINUED)	IN SITU TREATMENT (CONTINUED)	VITRIFICATION	GRAPHITE ELECTRODES PLACED IN A SQUARE ARRAY. ELECTRIC CURRENT PASSES THROUGH ELECTRODES CREATING HIGH TEMPERATURES THAT MELT THE SOIL OR WASTE SOLIDS INTO A BLOCK OF GLASSLIKE MATERIAL. EXPERIMENTAL PROCESS OPTION.	POTENTIALLY VIABLE.		✓
		▨ SOIL VAPOR EXTRACTION	LARGE VOLUMES OF AIR PULLED THROUGH THE SOIL TO STRIP VOLATILE CONTAMINANTS FROM THE SOIL IN A MANNER SIMILAR TO AIR STRIPPING.	DIFFICULT TO CONTROL RELEASE OF CONTAMINANTS DUE TO SUBSURFACE CHARACTERISTICS.		
		▨ SOIL FLUSHING	CONTAMINATED SOIL IS FLUSHED WITH AN APPROPRIATE FLUSHING SOLUTION AND THE ELUTRIATE IS COLLECTED IN A SERIES OF SHALLOW WELL POINTS OR SUBSURFACE DRAINS.	DIFFICULT TO CONTROL DUE TO CONDITIONS. SUBSURFACE CONDITIONS.		
		STABILIZATION/ SOLIDIFICATION	MINIMIZING MOBILITY BY ADDING SORBENT, SILICOUS MATERIAL, ETC. AND CHEMICALLY BINDING CONTAMINANTS.	POTENTIALLY VIABLE.		✓
	THERMAL TREATMENT	INCINERATION	HAZARDOUS MATERIALS THERMALLY DESTROYED IN A CONTROLLED, OXYGEN-SUFFICIENT ENVIRONMENT. GENERALLY, PRODUCTS INCLUDE CARBON DIOXIDE, WATER, AND ASH. MANY TYPES OF INCINERATORS WITH VARYING CAPABILITIES EXIST.	POTENTIALLY VIABLE.		✓
		▨ CO-DISPOSAL PROCESS	WASTE MATERIALS MIXED WITH PULVERIZED COAL, NATURAL GAS, WOOD CHIPS, ETC., OR OTHER WASTES AND USED AS A SUPPLEMENTAL FUEL SOURCE TO PRODUCE STEAM FOR A CONVENTIONAL POWER CYCLE.	NOT APPROPRIATE FOR RADIOACTIVE MATERIAL.		
		▨ PYROLYSIS	HAZARDOUS MATERIALS THERMALLY DESTROYED IN AN OXYGEN-DEFICIENT ENVIRONMENT.	NOT RETAINED IN FAVOR OF INCINERATION.		
		▨ WET AIR OXIDATION	WASTE MIXED WITH AIR AND FED TO A REACTOR WHERE DESTRUCTION TAKES PLACE AT HIGH TEMPERATURES (UP TO 600°) AND PRESSURES (UP TO 200 ATM).	NOT RETAINED IN FAVOR OF INCINERATION.		

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

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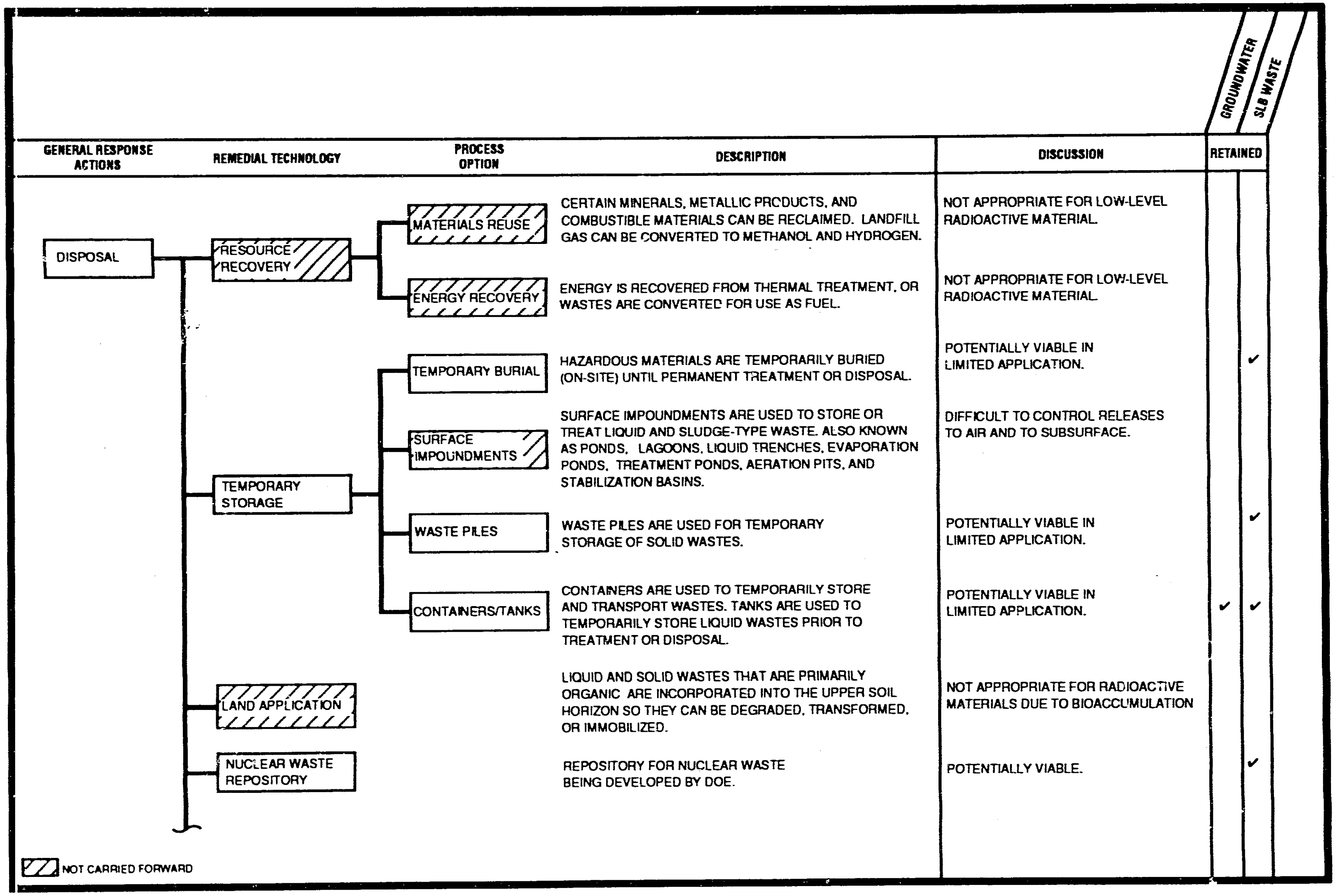

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED GROUNDWATER	RETAINED SLB WASTE
DISPOSAL	▨ RESOURCE RECOVERY	▨ MATERIALS REUSE	CERTAIN MINERALS, METALLIC PRODUCTS, AND COMBUSTIBLE MATERIALS CAN BE RECLAIMED. LANDFILL GAS CAN BE CONVERTED TO METHANOL AND HYDROGEN.	NOT APPROPRIATE FOR LOW-LEVEL RADIOACTIVE MATERIAL.		
		▨ ENERGY RECOVERY	ENERGY IS RECOVERED FROM THERMAL TREATMENT, OR WASTES ARE CONVERTED FOR USE AS FUEL.	NOT APPROPRIATE FOR LOW-LEVEL RADIOACTIVE MATERIAL.		
	TEMPORARY STORAGE	TEMPORARY BURIAL	HAZARDOUS MATERIALS ARE TEMPORARILY BURIED (ON-SITE) UNTIL PERMANENT TREATMENT OR DISPOSAL.	POTENTIALLY VIABLE IN LIMITED APPLICATION.		✓
		▨ SURFACE IMPOUNDMENTS	SURFACE IMPOUNDMENTS ARE USED TO STORE OR TREAT LIQUID AND SLUDGE-TYPE WASTE. ALSO KNOWN AS PONDS, LAGOONS, LIQUID TRENCHES, EVAPORATION PONDS, TREATMENT PONDS, AERATION PITS, AND STABILIZATION BASINS.	DIFFICULT TO CONTROL RELEASES TO AIR AND TO SUBSURFACE.		
		WASTE PILES	WASTE PILES ARE USED FOR TEMPORARY STORAGE OF SOLID WASTES.	POTENTIALLY VIABLE IN LIMITED APPLICATION.		✓
		CONTAINERS/TANKS	CONTAINERS ARE USED TO TEMPORARILY STORE AND TRANSPORT WASTES. TANKS ARE USED TO TEMPORARILY STORE LIQUID WASTES PRIOR TO TREATMENT OR DISPOSAL.	POTENTIALLY VIABLE IN LIMITED APPLICATION.	✓	✓
	▨ LAND APPLICATION		LIQUID AND SOLID WASTES THAT ARE PRIMARILY ORGANIC ARE INCORPORATED INTO THE UPPER SOIL HORIZON SO THEY CAN BE DEGRADED, TRANSFORMED, OR IMMOBILIZED.	NOT APPROPRIATE FOR RADIOACTIVE MATERIALS DUE TO BIOACCUMULATION		
	NUCLEAR WASTE REPOSITORY		REPOSITORY FOR NUCLEAR WASTE BEING DEVELOPED BY DOE.	POTENTIALLY VIABLE.		✓

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

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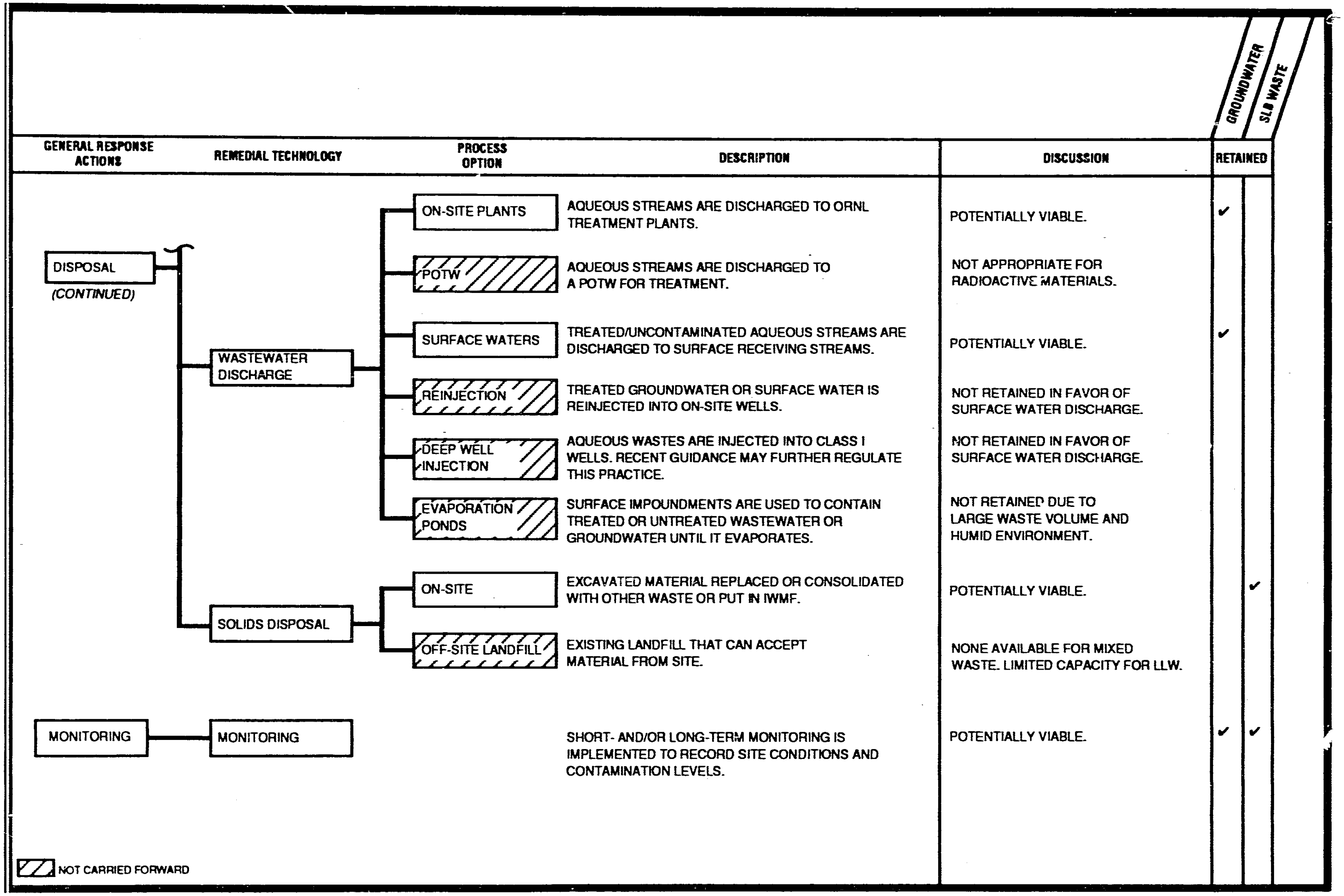

GENERAL RESPONSE ACTIONS	REMEDIAL TECHNOLOGY	PROCESS OPTION	DESCRIPTION	DISCUSSION	RETAINED GROUNDWATER	RETAINED SLB WASTE
DISPOSAL (CONTINUED)	WASTEWATER DISCHARGE	ON-SITE PLANTS	AQUEOUS STREAMS ARE DISCHARGED TO ORNL TREATMENT PLANTS.	POTENTIALLY VIABLE.	✓	
		▨ POTW	AQUEOUS STREAMS ARE DISCHARGED TO A POTW FOR TREATMENT.	NOT APPROPRIATE FOR RADIOACTIVE MATERIALS.		
		SURFACE WATERS	TREATED/UNCONTAMINATED AQUEOUS STREAMS ARE DISCHARGED TO SURFACE RECEIVING STREAMS.	POTENTIALLY VIABLE.	✓	
		▨ REINJECTION	TREATED GROUNDWATER OR SURFACE WATER IS REINJECTED INTO ON-SITE WELLS.	NOT RETAINED IN FAVOR OF SURFACE WATER DISCHARGE.		
		▨ DEEP WELL INJECTION	AQUEOUS WASTES ARE INJECTED INTO CLASS I WELLS. RECENT GUIDANCE MAY FURTHER REGULATE THIS PRACTICE.	NOT RETAINED IN FAVOR OF SURFACE WATER DISCHARGE.		
		▨ EVAPORATION PONDS	SURFACE IMPOUNDMENTS ARE USED TO CONTAIN TREATED OR UNTREATED WASTEWATER OR GROUNDWATER UNTIL IT EVAPORATES.	NOT RETAINED DUE TO LARGE WASTE VOLUME AND HUMID ENVIRONMENT.		
	SOLIDS DISPOSAL	ON-SITE	EXCAVATED MATERIAL REPLACED OR CONSOLIDATED WITH OTHER WASTE OR PUT IN IWMF.	POTENTIALLY VIABLE.		✓
		▨ OFF-SITE LANDFILL	EXISTING LANDFILL THAT CAN ACCEPT MATERIAL FROM SITE.	NONE AVAILABLE FOR MIXED WASTE. LIMITED CAPACITY FOR LLW.		
MONITORING	MONITORING		SHORT- AND/OR LONG-TERM MONITORING IS IMPLEMENTED TO RECORD SITE CONDITIONS AND CONTAMINATION LEVELS.	POTENTIALLY VIABLE.	✓	✓

▨ NOT CARRIED FORWARD

Fig. 8.3. (continued)

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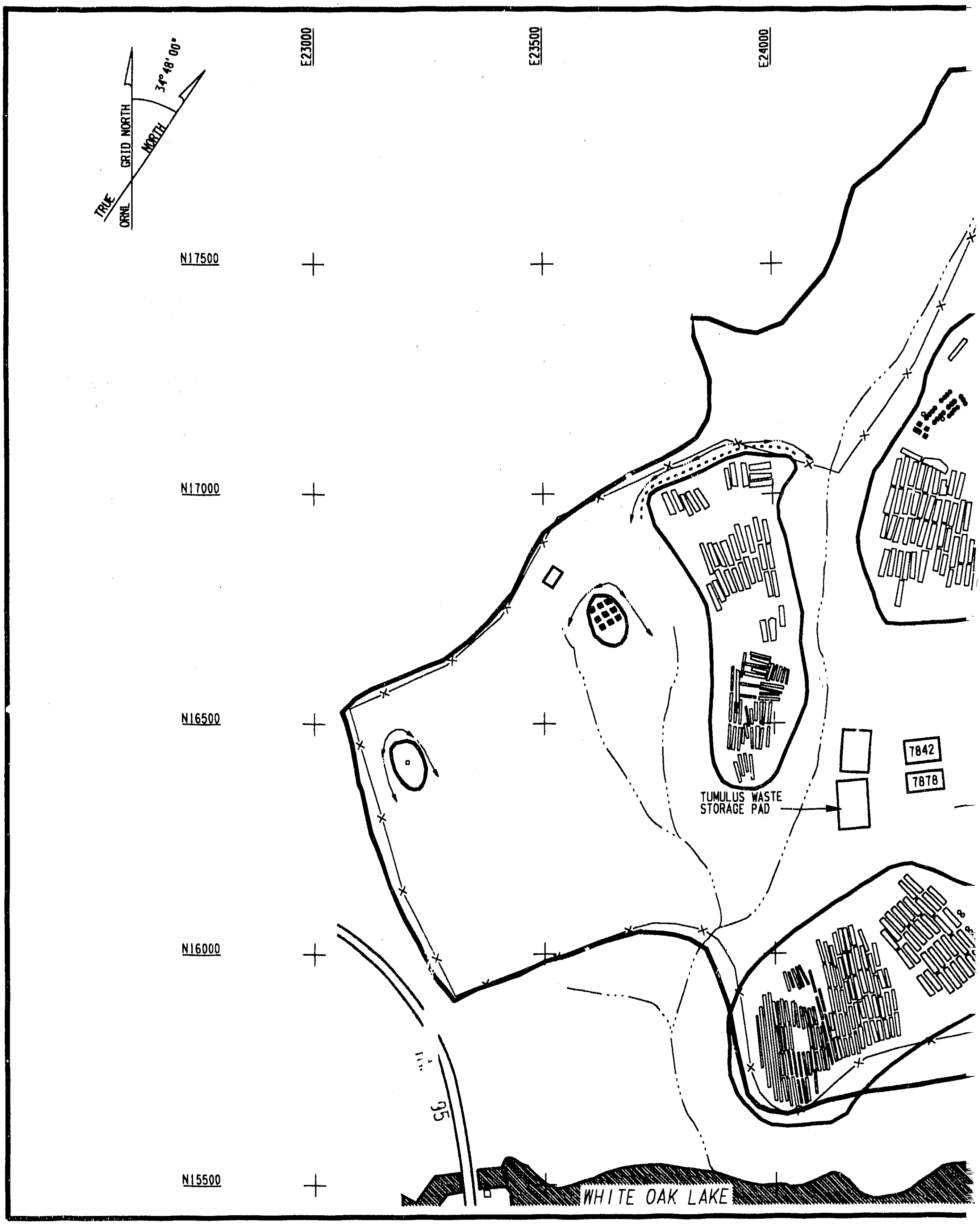

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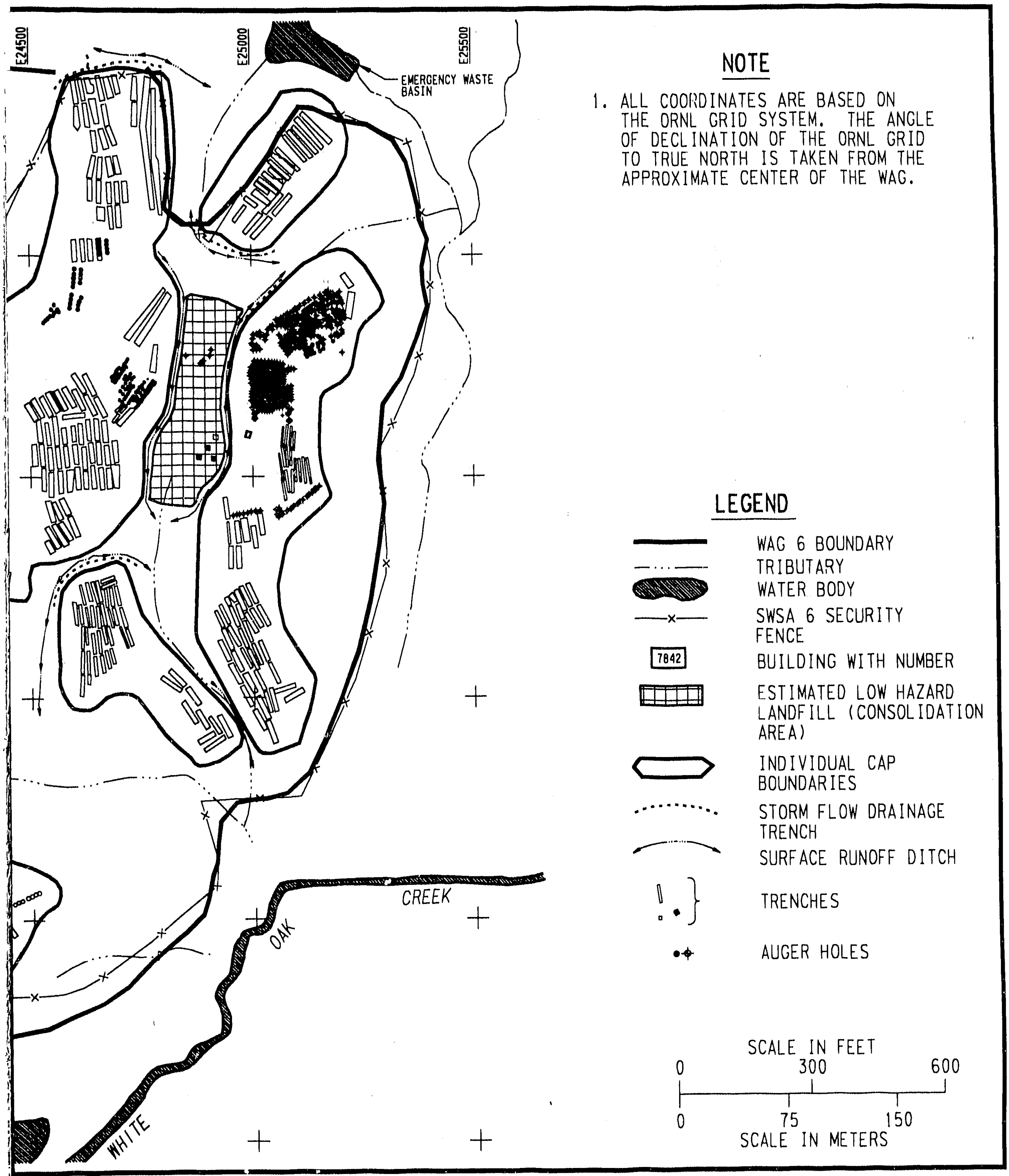
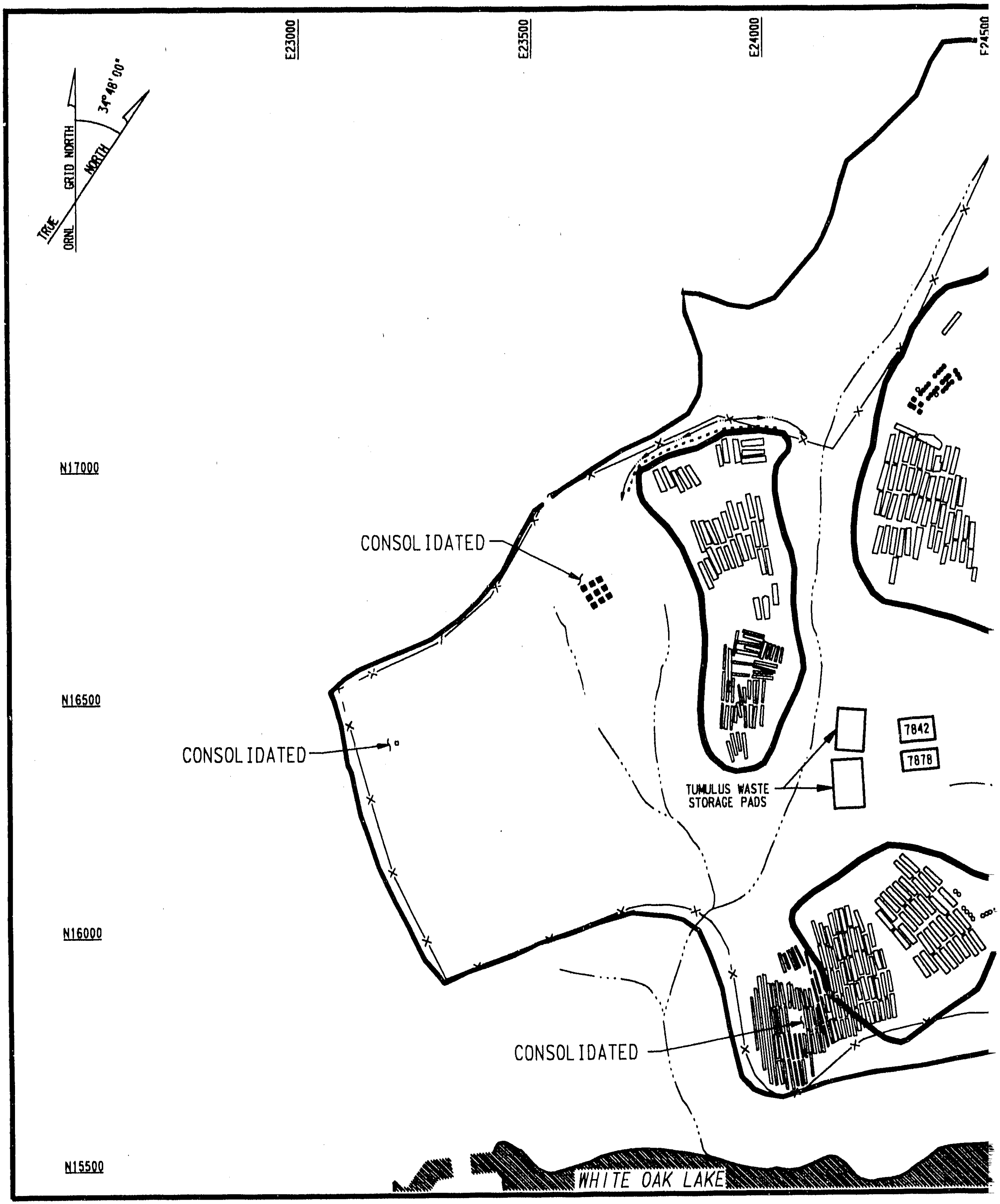


Fig. 8.4. Approximate location of small area caps, WAG 6.



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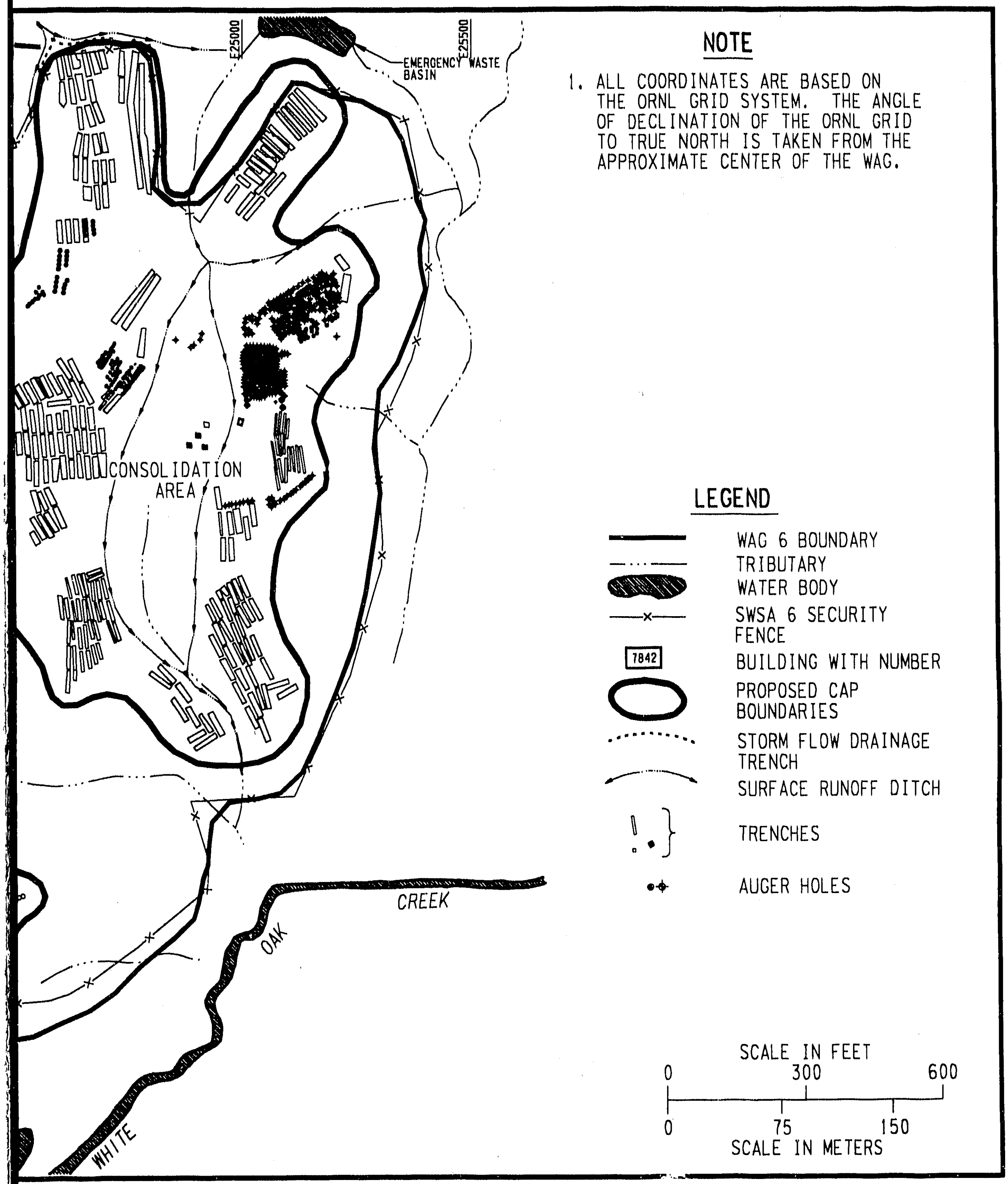

NOTE

1. ALL COORDINATES ARE BASED ON THE ORNL GRID SYSTEM. THE ANGLE OF DECLINATION OF THE ORNL GRID TO TRUE NORTH IS TAKEN FROM THE APPROXIMATE CENTER OF THE WAG.

Fig. 8.5. Approximate location of large caps, WAG 6.

9. SUMMARY AND CONCLUSIONS

9.1 NATURE AND EXTENT OF CONTAMINATION

Data collected during the WAG 6 RFI indicated that contaminants were widespread downgradient of source areas; however, specific contaminants generally were detected infrequently and/or at low concentrations. Within the site, there were areas where particular contaminants were detected at elevated concentrations.

9.1.1 Groundwater

Both man-made and naturally occurring radionuclides were detected in groundwater at WAG 6. Tritium was by far the most prevalent contaminant, and in locations adjacent to waste disposal areas, tritium concentrations frequently exceeded the 20,000 pCi/L MCL. Other man-made radionuclides detected included cobalt-60, strontium-90, and cesium-137; however, these were detected far less frequently and at relatively low concentrations. Cobalt-60 was detected predominantly in the northeastern corner of the site, and strontium-90 was primarily associated with the high-activity trench area. Transuranics were detected in groundwater infrequently and at low concentrations.

Like tritium, VOCs were detected in groundwater adjacent to most waste disposal areas. The highest concentrations, frequently exceeding MCLs, were associated with the biological trench areas and the solvent auger hole areas. Few SVOCs were detected in groundwater. Of metals detected that exceeded criteria identified in Sect. 4, lead and barium were detected most often. Metals contamination in groundwater appears to be restricted primarily to areas immediately adjacent to waste disposal units.

9.1.2 Surface Water

In WAG 6 surface water, as in groundwater, tritium was the most frequently detected contaminant. Tritium was detected in all drainages, and in the majority of samples, concentrations exceeded the MCL. Cobalt-60 was detected only once in surface water, near the same area where it was detected in groundwater and in soils. Strontium-90 was detected in several drainages at concentrations exceeding the MCL. The highest strontium-90 concentrations were detected in Drainage FB, which appears to be receiving strontium-90 from the high-activity trench area.

The primary VOCs detected in surface water at levels exceeding MCLs were 1,2-dichloroethene, tetrachloroethane, and trichloroethene. SVOCs generally were detected at low concentrations; frequently, these SVOCs were common laboratory contaminants. Barium and cadmium concentrations exceeded reference or MCL values for only two surface water samples.

9.1.3 Sediments

Cesium-137 was the man-made radionuclide most frequently detected in sediments. Apart from one cobalt-60 detection, concentrations of other man-made radionuclides did not exceed criteria identified in Sect. 4. Strontium-90 was detected in sediments in Drainage FB, the same drainage in which surface water appears to be receiving strontium-90 from the high-activity trench area.

VOCs were detected in most sediment sampling locations, but at concentrations less than the criteria identified in Sect. 4. SVOCs were detected infrequently and were typically common laboratory contaminants. Metals concentrations in sediments did not appear to be significantly elevated when compared to reference concentrations.

9.1.4 Soils

The WAG 6 radiological walkover survey and the soil sampling program indicated minimal surface soil contamination and the absence of gross soil contamination adjacent to source areas. Radiological contaminants detected in soils were limited primarily to strontium-90 and cobalt-60; each was detected in three soil borings. The maximum concentration of strontium-90 and cobalt-60 did not exceed criteria identified in Sect. 4.

Both VOCs and SVOCs were detected in soil samples. Compounds detected were frequently laboratory contaminants, and concentrations did not exceed criteria proposed in RCRA Subpart S. Metal contamination in soils at WAG 6 appears to be very localized. Arsenic, lead, mercury, and cobalt were detected in a few samples at concentrations significantly above background.

9.2 CONTAMINANT FATE AND TRANSPORT

Simplified contaminant fate and transport modeling was conducted to predict future on-WAG radionuclide concentrations in groundwater, surface water, sediments, and soils, and to predict the flux of contaminants off-WAG. Contaminant fluxes were used to calculate contaminant concentrations downstream in the Clinch River.

Modeling used essentially two models: an integrated surface/subsurface water contaminant fate and transport model, and an air model. For each contaminant modeled, time series predictions of concentrations were generated and employed in the base line risk assessment. Selected conclusions of the fate and transport analysis are summarized below.

- Water was the major transport mechanism for off-WAG migration of contaminants. Most of the radionuclide flux out of WAG 6 was expected to continue to occur via surface water.
- Tritium, cobalt-60, strontium-90, and cesium-137 were predicted to occur in most of the wells in WAG 6. Peak future groundwater concentrations were predicted to be within two orders of magnitude of present-day concentrations.

- The first appearances of europium and uranium in groundwater were predicted to occur in years 1998 and 2025, respectively.

- Air modeling, which was performed conservatively, indicated that the air pathway contributed negligible exposure-point concentrations of contaminants on-WAG and off-WAG.

9.3 BASE LINE RISK ASSESSMENT

A base line risk assessment was performed to assess the potential impacts WAG 6 contaminants would have on human health and the environment if no remedial actions were taken. The base line risk assessment included a human health evaluation and an environmental evaluation.

9.3.1 Human Health Evaluation

Two hypothetical scenarios were assumed for WAG 6—no action and institutional controls. The no action scenario assumed that DOE's current access restrictions for the site would become ineffective immediately. The institutional control scenario assumed that DOE would continue to use WAG 6 as an LLW disposal site for the next 10 years and that this operational period would be followed by a 100-year institutional control period. Under the toxicological and exposure assumptions used in the evaluation, the lower limit of EPA's target range (1×10^{-4} to 1×10^{-6}) was exceeded for the following receptors:

- No Action Scenario
  - On-WAG homesteader (1990–2020)

- Institutional Control Scenario
  - On-WAG ORNL employee (1990–2020)
  - Off-WAG (Clinch River) homesteader (1990–2020)
  - On-WAG homesteader (2100–2130)

Results for each scenario are briefly discussed in the following paragraphs.

9.3.1.1 Receptor risks for no action scenario

On-WAG homesteader (evaluated for 1990-2020). The estimated radionuclide and chemical carcinogen risks for the current day on-WAG receptor evaluated for site average concentrations were 1 (unity) and 3×10^{-4}, respectively. Ninety-nine percent of the radiological risk was associated with external exposure, with the majority of the dose resulting from radioisotopes of europium. The majority of the chemical risk was due to ingestion of groundwater and inhalation of water vapor while showering. Vinyl chloride, carbon tetrachloride, and trichloroethene were the predominant contributors to this risk.

Critical effect-specific hazard indexes computed for noncarcinogens were greater than 1 for the on-WAG homesteader (adult and child).

9.3.1.2 Receptor risks for institutional control scenario

Employee receptor (1990–2020). The estimated radionuclide risk for the maintenance receptor was 10^{-3}. The radionuclide risk was attributable almost exclusively to an estimated external radiation dose of 2 rem accumulated over 30 years, which is considerably below the 5 rem per day limit established in federal guidance for occupational exposure. Because the scenario assumed the worker spends 8 hours per day for 30 years working over SWSA 6 waste disposal areas, the risk estimate is conservative.

Off-WAG homesteader (1990–2020). The estimated radionuclide risk for the off-WAG homesteader was 6×10^{-5}. This risk was primarily a result of external radiation exposure from cobalt-60 and cesium-137 that have accumulated in soils as a result of crop irrigation over a period of decades. The contribution to radionuclide risk from assumed ingestion of surface water was 3×10^{-6}. Because heavy irrigation is highly unlikely given the abundant rainfall in the region, and because a homesteader is most likely to use groundwater as a drinking water source rather than untreated surface water, this scenario was conservative.

On-WAG homesteader (2100–2130). The estimated radionuclide risk for the future on-WAG receptor evaluated for site average concentrations was 3×10^{-1}. The majority of the radionuclide risk was associated with external exposure from isotopes of europium. Due to the assumption of steady state conditions for chemicals, chemical risks were the same as computed for the no action scenario on-WAG homesteader (1990–2020).

9.3.2 Environmental Evaluation

As an operating solid waste disposal facility, the majority of WAG 6 has been cleared of natural habitat. Regular site mowing and ongoing construction have discouraged reestablishment of natural habitats and wildlife communities. Aquatic and terrestrial surveys conducted at WAG 6 revealed no threatened and endangered species nor any developed wetland communities.

Potential impacts on target species were evaluated, including the tulip poplar—representing terrestrial flora; the white-tailed deer, red-tailed hawk, and raccoon—representing terrestrial fauna; bluegills and fathead minnows—representing aquatic vertebrates; and benthic macroinvertebrates—representing aquatic invertebrates. These species are found on the ORR, and might be expected on WAG 6 under a no action scenario.

The tulip poplar would likely uptake contaminants; the prediction of specific effects is not possible for most of the contaminants because of the lack of phytotoxicity data.

Strontium-90 likely would pose a significant threat to wildlife at WAG 6. It is readily absorbed and deposited in bone tissue, where it can result in bone tumors and leukemia. Species from lower trophic levels such as rabbits and shrews may bioconcentrate strontium-90 in their bone tissue, suffer adverse effects, and subsequently cause adverse effects in red-tailed hawks and raccoons that prey upon them.

Cadmium, copper, and strontium-90 could potentially affect fish populations. Cadmium and copper exceeded both chronic and acute ambient water quality criteria for protection of aquatic organisms. Strontium-90 may accumulate in bone tissues.

Macroinvertebrate communities could potentially be affected by cadmium and copper. These two metals produce chronic effects at concentrations above 10 ppb (see Tables 7B.8 and 7B.10). The maximum concentration of cadmium in surface water at WAG 6 was 30 μg/L, and the maximum concentration of copper was 88 μg/L. Both values exceed those reported to produce chronic effects in macroinvertebrates. Bioconcentration of strontium-90 by macroinvertebrates is expected to be low because of their short life cycle and frequent molting.

9.3.3 Conclusions of the Base Line Risk Assessment

Evaluation of hypothetical on-WAG homestead receptors demonstrated that the no action scenario could result in unacceptable risks to the public. Calculations further showed that if a receptor unknowingly excavated into one of the small number of high-activity auger holes in which reactor control plates have been disposed, radiation doses could cause acute effects. If the control plates were removed, residual risks associated with the remaining inventory were still predicted to exceed EPA's target range.

It is emphasized that the no action scenario is not a realistic scenario and was evaluated solely to define a base line against which to compare alternatives for site closure and remediation. The location of SWSA 6 on the U.S. government-owned ORR, its proximity to an operating facility (ORNL), and the existence of site fencing and security patrols make it highly unlikely that a member of the public could occupy the site and remain undetected.

Under the institutional control scenario, estimated risks for hypothetical public receptors did not exceed the upper limit of EPA's target risk range. For the off-WAG (Clinch River) receptor, risks fell within the upper end of the range; however, due to the conservative assumptions associated with this scenario, this result should not be the basis for implementing actions beyond those already inherent in the scenario (e.g., access restrictions, site maintenance).

The evaluation of an on-WAG receptor for the period 2100-2130 did indicate unacceptable risks. This demonstrates that, without appropriate site closure and remediation, access restrictions should continue beyond the year 2130 to prevent the possibility of on-WAG homesteading.

9.4 REMEDIAL ACTION OBJECTIVES

The results of the RFI base line risk assessment indicate that closure and remedial action are required for source areas and groundwater.

Specific remedial objectives for each of these waste media are as follows:

- **Sources**: protect public health and the environment by limiting direct contact, emissions, surface runoff, and leachate generation from on-WAG sources and associated contaminated leachate and soil.

- **Groundwater**: protect public health and the environment by mitigating the effects of migration of contaminants off-WAG and by protecting against future use of on-WAG contaminated groundwater.

9.5 CLOSURE AND CORRECTIVE ACTION ALTERNATIVES

Using the methodology recommended in EPA's *Guidance for Conducting Remedial Investigations and Feasibility Studies Under CERCLA* (EPA 1988a), a range of alternatives was developed for addressing source areas and groundwater. The alternatives, which were developed to span ranges of cost and permanence, are described in Sect. 8.

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