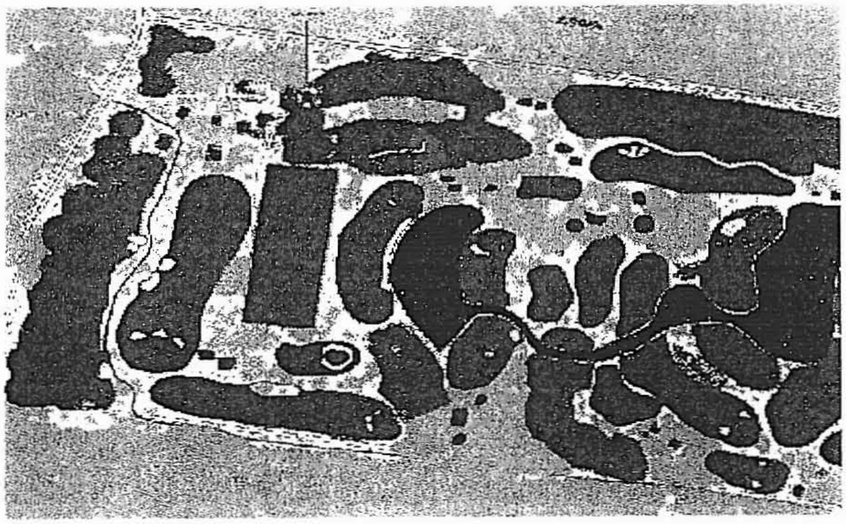
Dominion
Virginia Power

ASH STABILIZATION, GROUNDWATER MODELING & RISK EVALUATION UPDATED FINAL REPORT

CHESAPEAKE ENERGY CENTER

PROPOSED GOLF COURSE PROJECT



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1.1 PROJECT BACKGROUND AND OBJECTIVES

Dominion plans to provide approximately 1.5 million tons of stabilized fly ash to be used in the construction of a new 18-hole golf course in Chesapeake, Virginia. The Project Study Area is shown in **Figure 1-1**. The proposed golf course layout designed by Hassel & Folkes and CPM is presented in **Figure 1-2**. Dominion has contracted URS to evaluate the leachability of metals from the stabilized fly ash, and predict the maximum concentration of ash-related constituents in groundwater at the property boundary. Dominion is also interested in the effectiveness of cementicious reagents used to stabilize the Chesapeake ash, and how much stabilization is needed to protect human health and the environment in a responsible and cost-effective manner. These questions have been addressed with two studies designed by URS in cooperation with Dominion. They include:

- Laboratory bench scale leachability and geotechnical characterization testing using several candidate reagents in varied concentrations with Chesapeake fly ash; and
- Site hydro-geologic investigation, groundwater modeling, and associated risk assessment as described in our agreed to scope of services for this project.

The two studies cited were performed in tandem and recently completed. This approach provides a comprehensive approach to addressing stabilization and the potential environmental impacts that the stabilized ash placed at the golf course site is predicted to have on site groundwater and potential receptors. The hydrogeologic site investigation was completed in September 2001, and a report on that investigation has already been submitted to Dominion under separate cover.

URS also involved with Dominion to prepare a punch list of project-related items in an effort to identify any potential fatal flaws or schedule busters that would cause Dominion to have reservations about proceeding with the project. That task was completed and culminated in the submittal of a report including the project punch list and a pre-construction/construction project schedule submitted to Dominion by URS in September, 2001

1.2 SCOPE OF WORK AND VARIATIONS FROM ORIGINAL SCOPE

The baseline scope of work performed in support of this study is in conformance with the scope of services outlined in our agreement with Dominion. In addition, a more extensive modeling effort was performed resulting in the development of a project-specific *Integrated Pathway Model* as discussed in **Section 3.1** of this report. Some minor modifications and enhancements to the laboratory program were also implemented. These are discussed in more detail in **Section 2.2** of this report.

This section of the report addresses the stabilization, testing, and characterization of Chesapeake Energy Center fly ash to be used in the planned Golf Course Project. Included in this section is a discussion and review of the screening and evaluation of free-lime reagents considered for the project. This discussion is followed by a characterization of both the chemical and physical/geotechnical properties of both the amended and unamended fly ash from the Chesapeake Energy Center.

2.1 SAMPLING AND REPRESENTATIVENESS OF ASH SAMPLES USED IN TESTING

Representative samples of both the fresh and weathered fly ash were provided by the station and shipped to our lab for testing. The task of preparing and shipping representative samples was the sole responsibility of the station. We understand that through the use of best practices, Dominion collected all samples to represent "typical" operating conditions for the plant. If there is reason to believe that fly ash quality may vary significantly due to large swings in the plants' operating levels, or the installation of additional air quality control systems (e.g., SCR/SNCR, conversion to Low NOx burners, etc..), further consideration may be needed with regard to re-sampling and updated testing of the ash.

We understand that the fresh samples were collected from on-site silos. Weathered samples were collected from the on-site landfill by a Dominion subcontractor in conjunction with a recent sampling/investigation of fly ash stored in the onsite permitted landfill. Six five gallon pales of both the weathered and fresh ash were received by AEP labs, and thoroughly homogenized via physical mixing and recombining of each type of fly ash prior to use in any testing.

2.2 TESTING SCOPE OF WORK

The scope of work as defined in the approved agreements was adhered to with only minor variations from the original objectives and intents of the approved testing program. Some enhancements to the laboratory program were made including increasing the number of leach tests (SW 846 TCLP and SPLP methodology) on the unamended fly ash. This modification was made to better establish the baseline concentration of various constituents in leachate from the unamended fresh and weathered fly ash.

Dominion has authorized testing on an additional reagent, Blue Circle Cement which will be used in an ongoing highway project, and may under special conditions, be the reagent used to amend the ash in the golf course fill. Reporting on that additional reagent will be provided to Dominion under separate cover.

2.3 CHARACTERIZATION AND SELECTION OF CANDIDATE AND PREFERRED REAGENTS

Six (6) candidate cementicious reagents including cement kiln dust (CKD), lime kiln dust (LKD) and cement were pre-screened. The purpose of the screening was to ensure the reagents selected for use in the project are the most effective and cost-efficient available in the regional markets surrounding Chesapeake.

The candidate reagent list was compiled by Dominion and two ash marketing contractors, VFL Technologies and CPM. The short list of reagents included in the screening included the following vendors and products:

- Capitol Cement CKD
- Holnam Cement CKD
- Roanoke Cement CKD
- Chemical Lime LKD
- Global Stone LKD
- Roanoke Cement -Type 1 / 2 Cement.

Representative samples of the reagents were shipped to our lab as coordinated by the marketing contractors. Screening tests performed on the reagents and a blind duplicate (Roanoke CKD) included:

- Basic reagent chemistry - Calcium Carbonate Equivalency, Total CaO, and Available Free Lime,
- Physical testing - Grain-size distribution, and
- Qualitative reactivity - Immediate pH at various percentages of reagent batched with Chesapeake fly ash. Target pH's were set at pH=11 and 12.

Results of the screening are presented in **Figures 2-1** and **2-2**. The laboratory test results and certificates of analysis for the screening of reagents and all associated laboratory analysis is included in **Appendix A.**

Following the pre-screening activities, two (2) preferred reagents were selected based on the test results and generalized unit prices for each of the reagents as provided by the reagent vendors. The lowest relative priced and best performing reagents were:

- Global Stone LKD, and
- Roanoke Cement CKD.

The blind duplicate results (on the Roanoke CKD) indicated an acceptable level of comparability between the original and duplicate. The Available Lime Index – Estimated Free Lime determination was rerun for the Roanoke Cement because of the comparatively low percentage of free lime in the original analysis by the lab. The rerun of that analysis indicated that the original result was representative, confirming the previous test, and also providing a second point of quality assurance and reproducibility for the original lab analysis. Regarding fineness, the percent of the lime kiln dust and cement kiln dust samples passing the #200 sieve was 47.7% and 99.7%, respectively.

Trace metals analysis was then performed on the two preferred reagents to verify that these reagents would not adversely contribute to leachable metals in the stabilized ash. Results from those analysis indicated that the concentration of leachable metals (in particular arsenic, selenium and other potential risk drivers) are very low and will not likely contribute appreciable amounts of leachable metals in the amended ash leachates. Results from these analysis are included in **Appendix A**.

2.4 UNAMENDED AND AMENDED ASH CHARACTERIZATION

Contained in this section is a brief overview of the scope of testing and evaluation performed for characterization of both the unamended and amended ash from the Chesapeake Station. The testing program performed was intended to provide the necessary physical and chemical properties for both the weathered and fresh fly ash for input into the modeling exercises. A limited number of additional tests not directly utilized in this study were performed on the ash with the intent to provide Dominion and the golf course contractors with relevant engineering properties for use during construction on an as-needed basis.

2.4.1 Physical/Geotechnical Characterization

TEST PROCEDURES

Geotechnical testing was performed on weathered and fresh fly ash samples from the Chesapeake Station. The samples were either unamended, or amended with lime kiln dust (LKD) or cement kiln dust (CKD), at various concentration levels. Lime kiln dust and cement kiln dust used to prepare amended samples was obtained from the Global Stone Corporation and the Roanoke Cement Company, respectively as discussed in **Section 2.1**. Testing was performed by AEP PRO SERV, Inc of Groveport, Ohio.

Geotechnical tests included:

- ASTM D4222 - Particle-Size Analysis of Soils – 2 tests
- ASTM D698-91 - Laboratory Compaction Characteristics of Soil Using Std. Effort – 12 tests
- ASTM D1883 - CBR (California Bearing Ratio) of Laboratory-Compacted Soils – 4 tests
- ASTM D5102 - Unconfined Compressive Strength of Compacted Soil-Lime Mixtures – 14 tests
- ASTM D5084-90 - Hydraulic Conductivity of Soils – 2 tests

SUMMARY OF FINDINGS

Results from the geotechnical laboratory tests are tabulated in **Tables 2.1 and 2.2**, for the fresh fly ash and the weathered fly ash, with test result sheets from the lab provided in **Appendix B**. Results of Unconfined Compressive Testing are further portrayed in **Figure 2-1**. Raw data summaries provided by AEP Laboratories are included in **Appendix B**. The following sections present specific methods and results of the geotechnical laboratory testing.

TABLE 2-1
SUMMARY OF LABORATORY TEST RESULTS: GEOTECHNICAL AND PHYSICAL PROPERTIES – FRESH FLY ASH

LKD OR CKD ADDED (%)	PARTICLE SIZE (% Passing)			SPECIFIC GRAVITY	SHIPPING DRY UNIT WEIGHT		STANDARD PROCTOR		CALIFORNIA BEARING RATIO	HYDRAULIC CONDUCTIVITY	UNCONFINED COMPRESSIVE STRENGTH	
	4.75 mm (#4)	0.075 mm (#200)	0.045 mm (#325)		LOOSE (pcf)	DENSE (pcf)	MAX DRY DENSITY (pcf)	OMC (%)	CBR (%)	(cm/sec)	28-day Strength (psi)	63-day Strength (psi)
0.0	100.0	83.1	67.2	2.19	45	56.1	62.6	43.1	--	--	15	--
1.0 (CKD)	--	--	--	--	--	--	62.0	43.0	38.91	--	28	28
3.0 (CKD)	--	--	--	--	--	--	62.6	42.4	--	1.6 E-05	70	83
5.0 (CKD)	--	--	--	--	--	--	62.7	42.3	86.40	--	114	155
1.0 (LKD)	--	--	--	--	--	--	62.6	42.4	23.75	--	27	23
3.0 (LKD)	--	--	--	--	--	--	62.8	42.1	--	2.0 E-05	60	61
5.0 (LKD)	--	--	--	--	--	--	63.2	41.5	74.57	--	92	89

TABLE 2-2
SUMMARY OF LABORATORY TEST RESULTS: GEOTECHNICAL AND PHYSICAL PROPERTIES – WEATHERED FLY ASH

LKD OR CKD ADDED (%)	PARTICLE SIZE (% Passing)			SPECIFIC GRAVITY	SHIPPING DRY UNIT WEIGHT		STANDARD PROCTOR		CALIFORNIA BEARING RATIO	HYDRAULIC CONDUCTIVITY	UNCONFINED COMPRESSIVE STRENGTH	
	4.75 mm (#4)	0.075 mm (#200)	0.045 mm (#325)		LOOSE (pcf)	DENSE (pcf)	MAX DRY DENSITY (pcf)	OMC (%)	CBR (%)	(cm/sec)	28-day Strength (psi)	63-day Strength (psi)
0.0	98.0	88.1	66.3	2.25	58.5	70.6	75.1	31.5	--	--	16	--
3.0 (CKD)	--	--	--	--	--	--	74.4	32.6	--	--	--	--
5.0 (CKD)	--	--	--	--	--	--	74.7	31.4	--	--	--	--
3.0 (LKD)	--	--	--	--	--	--	74.6	31.9	--	--	--	--
5.0 (LKD)	--	--	--	--	--	--	75.8	31.1	--	--	--	--

Grain-Size Analysis, Specific Gravity, and Shipping Weight

Particle size, specific gravity, and shipping weight tests were performed on unamended samples of both weathered and fresh ash. Abbreviated particle size analyses were conducted on the cement kiln dust and lime kiln dust – i.e., the proportion of each passing through the #200 sieve was measured.

Results from the particle-size analysis indicate that both weathered and fresh unamended ashes are comprised of predominantly fine-grained particles. Percent of the samples passing the #4, #200, and #325 sieves, were, for the weathered and fresh ash respectively, 98% and 100%, 84.0% and 83.1%, and 66.3% and 67.2%. Results of the grain size analysis are presented in **Tables 2.1** and **2.2**, and the grain-size curves are presented in **Appendix B**.

Specific gravity was determined for unamended samples of both ash types. The specific gravity of the unamended weathered and fresh ashes was 2.25 and 2.19, respectively.

Shipping weight analyses were conducted for samples placed at two relative densities: a loosely placed condition (the loose shipping condition), and a denser, more compacted condition (the dense shipping condition). The latter state was achieved by vibrating samples at low frequency, and is thought to more closely simulate shipping conditions. The loose shipping weight of the samples was determined to be 58.5 pounds per cubic foot (pcf) for the weathered ash, and 45.0 pcf for the fresh ash. The dense shipping weight of the samples was determined to be 70.6 pcf for the weathered ash, and 56.1 pcf for the fresh ash. The results of this testing are presented in **Tables 2-1** and **2-2** and **Appendix B**.

Compaction Characteristics

Standard Proctor compaction curves were developed for unamended and amended weathered and fresh ash samples. For the amended samples, the proportion of lime or cement kiln dust was varied from 1 to 5%, to evaluate the effects reagent dosing on compaction characteristics. Results of the Proctor analyses are summarized in **Tables 2.1 and 2.2** and compaction curves are presented in **Appendix B**.

To summarize, the ranges of maximum dry density and optimum moisture content for weathered ash were 74.4 pcf to 75.8 pcf and 31.1% to 32.6%, respectively. For fresh ash the ranges were 62.0 pcf to 63.2 pcf and 41.5% to 43.1%. Generally, amended samples had slightly higher maximum dry densities and lower optimum moisture contents than the unamended samples, with increasing maximum dry density and decreasing optimum moisture content with increasing concentration of reagent. Changes in compaction characteristics (for a given reagent and type of fly ash) with the addition of reagents were nominal, however, and very little variation in the values of maximum dry density and optimum moisture content due to addition of reagents was observed. Similarly, the type of reagent (cement kiln dust or lime kiln dust) had little effect on compaction characteristics. From the results it is clear, however, that the weathered ash may be compacted to a significantly higher dry density and significantly lower moisture content, than the fresh ash.

California Bearing Ratio (CBR)

The California Bearing Ratio Test was performed on fresh fly ash samples amended with 1% and 5% of both lime and cement kiln dust, to evaluate the effects of increasing reagent concentration on the CBR. The amended samples were allowed to cure for 28 days prior to performing the CBR tests. Results of the CBR testing are portrayed in **Table 2.1,** with lab reports provided in **Appendix B**.

The CBR for samples at the 1% and 5% reagent concentrations are 38.9% and 86.4% for CKD, and 23.8% and 74.6% for LKD. Values for both reagent types are significantly higher at the 5% concentration level, and the CBR for samples amended with cement kiln dust is higher than for samples amended with lime kiln dust, at both reagent concentration levels. The higher CBR value achieved with the CKD may in part be associated with the greater fineness of the CKD when compared to LKD.

Generally, materials with CBR values greater than 7% are considered acceptable for use as subgrade; CBR values greater than 20% indicate an acceptable subbase; and CBR values above 50% indicate an acceptable base material

Unconfined Compressive Strength

Unconfined compressive strength (UC) was evaluated for unamended samples of weathered fly ash, and unamended and amended samples of fresh fly ash. The curing period for amended samples was either 28 or 63 days. Results of the UC analyses are presented in **Tables 2.1** and **2.2**, and the UC data are presented in **Appendix B**.

Unamended samples of both ash types displayed nearly identical unconfined compressive strengths: 16 psi for the weathered ash and 15 psi for the fresh ash. **Figure 2-1** presents a plot of the 28-day compressive strength of fresh fly ash as a function of the concentration of reagent added to the sample, for both the lime kiln dust and cement kiln dust reagents. From the figure, the relationship between strength and proportion of reagent is approximately linear for both reagent types, and it is apparent that significant increases in compressive strength may be realized with the addition of reagents. Compressive strength values for samples amended with cement kiln dust vary from 15 psi (unamended) to 114 psi (at 5% CKD), and for samples amended with lime kiln dust from 15 psi (unamended) to 92 psi (at 5% LKD). The curves portrayed on the figure indicate that cement kiln dust is more effective in increasing compressive strength than lime kiln dust, especially at higher reagent concentration levels. As with the CBR results, higher unconfined compressive strengths with CKD may in part be due to the greater fineness of the CKD.

The 63-day unconfined compressive strengths were determined for amended samples. Results of this testing are portrayed on **Table 2.1** and **Appendix B**. Generally, there is only nominal difference in the 28 and 63-day compressive strengths of samples amended with lime kiln dust. For example, for a sample prepared at 3% LKD, the 28 and 63-day compressive strengths are 60 and 61 psi, respectively. Extended curing periods have a more significant effect on samples amended with cement kiln dust, however, especially at higher reagent concentrations. For example, the 28 and 63-day compressive strengths at 5% CKD were 114 and 155 psi, respectively – an increase of 36% for the sample with the extended curing period.

Hydraulic Conductivity

Flexible-membrane permeability tests were run on fresh fly ash samples amended with 3% LKD and 3% CKD. Permeability coefficients were found to be 1.99E-05 cm/sec for the LKD sample and 1.57E-05 cm/sec for the CKD sample. The test data are presented in **Appendix B**.

2.4.2 Chemical Characterization – Totals and Leaching Test Results

Samples of fresh and weathered unamended ash were analyzed for total and leachable constituents by AEP Laboratory in August 2001, and again (in duplicate, for leachable metals only) in October 2001. All analyses were performed in accordance with SW-846 (solid waste) or EPA-600 (water and wastewater) methodologies. Quality control data included with the sample reports were reviewed and found to be acceptable, supporting the usability of the analytical results.

Based on these analyses, thirteen analytes were initially identified as potential 'indicator chemicals' for the groundwater modeling. The total metals results for the fresh and weathered ash were similar, with concentrations of the 'indicator' metals ranging from non-detected for cadmium and selenium (at 2 and 20 mg/Kg, respectively), to 224 mg/Kg for vanadium. The initial pH of a 1% slurry of the unamended ash was slightly alkaline, at 7.6 in the fresh and 7.9 in the weathered. The total analysis results are presented in **Table 2.3**.

TABLE 2.3
UNAMENDED FLY ASH DATA, TOTAL BASIS

	Ash Type	Fresh	Weathered
	Sample ID	CEC #1	SB-07
	Lab ID	011640-001	011640-002
	Date Collected	06/20/2001	04/17/2001
Parameter	Units	Total Basis	Total Basis
Antimony	mg/Kg	< 20	< 20
Arsenic	mg/Kg	98	87
Barium	mg/Kg	1050	1080
Beryllium	mg/Kg	16	15
Boron	mg/Kg	82	58
Cadmium	mg/Kg	< 2	< 2
Chromium	mg/Kg	113	128
Cobalt	mg/Kg	49.5	45.8
Copper	mg/Kg	125	136
Lead	mg/Kg	39.8	45.4
Manganese	mg/Kg	< 200	< 200
Mercury	mg/Kg	0.62	0.48
Molybdenum	mg/Kg	11.3	11.7
Nickel	mg/Kg	76.1	85.6
Selenium	mg/Kg	< 20	< 20
Silver	mg/Kg	< 1	< 1
Strontium	mg/Kg	796	975
Vanadium	mg/Kg	196	224
Zinc	mg/Kg	132	143

The leachate analyses demonstrated that approximately 1% to 10% of each metal analyzed was leachable from the fresh ash by the Toxicity Characteristic Leaching Procedure (TCLP, SW-846 Method 1311). Although not detected above the reporting limit in the total metals analyses, selenium was present in all the leachate samples (maximum concentration 0.33 mg/L), indicating that as much as 30% of the selenium in the fresh ash may be leachable. For most metals, a similar or somewhat lower amount was leachable by the Synthetic Precipitation Leaching Procedure (SPLP, SW-846 Method 1312). The leachate pH (i.e., the resulting pH in the leachate at the completion of the leaching test) ranged from slightly acidic (6.3 in the fresh ash SPLP) to slightly alkaline (7.7 in the weathered ash SPLP). The analytical results for the fly ash leachates are presented in **Table 2.4.**

TABLE 2.4
UNAMENDED FLY ASH LEACHATE DATA

Ash Type		Fresh Ash				Weathered Ash			
Result		Average		Maximum		Average		Maximum	
Parameter	Units	SPLP	TCLP	SPLP	TCLP	SPLP	TCLP	SPLP	TCLP
Aluminum	mg/L	0.03	3.10	0.05	3.38	0.26	3.02	0.29	3.52
Arsenic	mg/L	0.110	0.337	0.117	0.400	0.123	0.123	0.132	0.150
Barium	mg/L	0.166	0.643	0.31	0.74	0.140	0.927	0.217	1.2
Beryllium	mg/L	< 0.0002	0.003	< 0.0002	0.005	< 0.0002	< 0.002	< 0.0002	< 0.002
Boron	mg/L	0.50	0.39	0.52	1.01	0.34	0.22	0.35	0.53
Cadmium	mg/L	0.003	< 0.01	0.0039	< 0.01	0.002	< 0.01	0.0028	< 0.01
Calcium	mg/L	56.4	68.8	59	71.6	63.2	110	64	117
Chromium	mg/L	0.005	0.057	0.014	0.15	0.012	0.017	0.016	0.02
Chromium, 6^+	mg/L	< 0.003	< 0.003	< 0.003	< 0.003	0.009	0.008	0.01	0.01
Copper	mg/L	0.048	0.080	0.117	0.09	0.031	0.063	0.064	0.07
Iron	mg/L	< 0.01	0.115	< 0.01	0.13	< 0.01	0.075	< 0.01	0.11
Lead	mg/L	0.005	0.040	0.005	0.040	0.004	0.020	0.010	0.030
Magnesium	mg/L	10.3	11.7	10.8	12	5.6	8.4	5.7	8.8
Manganese	mg/L	0.015	0.10	0.02	0.1	< 0.01	0.04	< 0.01	0.06
Mercury	mg/L	< 0.0002	< 0.0002	< 0.0002	< 0.0002	< 0.0002	0.0002	< 0.0002	0.0003
Nickel	mg/L	0.005	0.063	0.006	0.08	< 0.003	0.030	< 0.003	0.06
Potassium	mg/L	12.2	13.3	12.8	14.3	9.7	11.6	10.6	12.4
Selenium	mg/L	0.076	0.223	0.084	0.33	0.090	0.143	0.133	0.150
Silver	mg/L	< 0.0002	< 0.02	< 0.0002	< 0.02	< 0.0002	< 0.02	< 0.0002	< 0.02
Sodium	mg/L	397	N/A	432	N/A	503	N/A	742	N/A
Thallium	mg/L	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	0.040	< 0.020	0.06
Vanadium	mg/L	0.077	0.167	0.078	0.3	0.083	0.070	0.088	0.07
Zinc	mg/L	0.077	0.247	0.112	0.34	0.044	0.293	0.062	0.39
Chloride	mg/L	107.0	138	107.0	138	44	57	44	57
Nitrate	mg/L	N/A	< 0.05	N/A	< 0.05	N/A	< 0.05	N/A	< 0.05
Orthophosphate	mg/L	0.06	0.11	0.06	0.11	0.12	0.07	0.12	0.07
Sulfate	mg/L	N/A	169	N/A	169	N/A	156	N/A	156
TOC	mg/L	1	N/A	1	N/A	1	N/A	1	N/A
Acidity	mg/L	13.0	1710	13.0	1710	< 2	1640	< 2	1640
Alkalinity, Total	mg/L	9.0	1820	9.0	1820	29	1890	29	1890
Alkalinity, Bicarb	mg/L	9.0	1820	9.0	1820	29	1890	29	1890
Conductivity	µmho	1890	4810	1890	4810	3290	4730	3290	4730
pH	S.U.	6.3	7	6.3	7	7.7	7.5	7.7	7.5
TDS	mg/L	1620	6450	1620	6450	3010	6360	3010	6360

In general, the TCLP and SPLP results for the weathered ash were lower than those for the fresh ash, indicating that a portion of the leachable metal content had been either naturally leached during the weathering process, or stabilized in the weathering process. Due to the higher concentrations in the fresh ash leachates (for most analytes), the fresh ash was selected to conservatively demonstrate the effects of the amending reagents on analyte concentrations.

After the addition of each of the selected reagents at three different levels – 1%, 3%, and 5% – samples of the amended fresh ash were analyzed once again for leachable constituents. At each level of reagent, the corresponding results for the two preferred reagents (i.e. Global Stone LKD and Roanoke Cement CKD) were similar, indicating that the effects of the reagents were, for the most part, indistinguishable. As with the unamended ash data, the majority of the SPLP results were similar to or lower than the corresponding TCLP results.

After an evaluation of the data (see **Section 4.0**), seven metals were determined to be Chemicals of Potential Concern (COPCs). The effects of increasing reagent amounts were not uniform across all the COPCs. Four analytes—specifically, arsenic, lead, selenium, and vanadium—show a progressive reduction in leachable concentrations (SPLP and/or TCLP) with increasing levels of reagent. Other analytes, specifically beryllium and chromium, exhibit initially increasing concentrations in the TCLP leachates (up to the 3% level), then appear to level off or decrease slightly at 5%. The last COPC, thallium, was non-detect in the fresh fly ash, and had two random detections near the detection limit at different levels of reagent. No clear pattern was discernable for thallium. The pH shows a significant shift from nearly neutral (6.3-7.0) in the unamended leachate to very alkaline (10.6-11.3) at the 5% reagent level.

The amended ash leachate results are summarized in **Table 2.5** and the results for the seven COPCs are presented graphically in **Figure 2-4A** through **2-4G**. For the purpose of graphing the results, the unamended ("0%" reagent) ash results presented in each chart are the average TCLP and SPLP results obtained from the original and replicate fresh fly ash analyses.

TABLE 2.5
AMENDED FLY ASH LEACHATE DATA

Sample ID		Unamended Fresh (Average)		1% Cement Kiln Dust		1% Lime Kiln Dust		3% Cement Kiln Dust		3% Lime Kiln Dust		5% Cement Kiln Dust		5% Lime Kiln Dust	
Lab ID Date Collected		011640-003 6/20/01		011640-005 4/17/01		011640-006 4/17/01		011640-007 4/17/01		011640-008 4/17/01		011640-009 4/17/01		011640-010 4/17/01	
Parameter	Units	SPLP	TCLP	SPLP	TCLP	SPLP	TCLP	SPLP	TCLP	SPLP	TCLP	SPLP	TCLP	SPLP	TCLP
Aluminum	mg/L	0.033	3.097	4.41	9.56	5.51	7.55	5.42	22.6	9.54	17.3	17.3	18.8	21.7	15.0
Arsenic	mg/L	0.110	0.337	0.325	0.29	0.411	0.33	0.203	0.15	0.183	0.16	0.088	0.11	0.085	0.09
Barium	mg/L	0.166	0.643	0.387	0.41	0.2	0.44	0.062	0.58	0.316	1.69	0.392	1.12	0.5	1.28
Beryllium	mg/L	< 0.0002	0.003	< 0.0002	0.007	< 0.0002	0.007	< 0.0002	0.017	< 0.0002	0.012	< 0.0002	0.017	< 0.0002	0.008
Boron	mg/L	0.497	0.387	0.38	0.71	0.42	0.7	0.2	0.84	0.16	0.9	0.1	0.96	0.07	0.94
Cadmium	mg/L	0.003	< 0.01	0.005	0.01	0.005	< 0.010	0.004	0.01	0.004	< 0.010	0.003	0.01	0.003	< 0.010
Calcium	mg/L	56.433	68.833	75.5	218	78.8	242	52.4	530	50.9	585	70.7	821	79.2	970
Chromium	mg/L	0.005	0.057	0.02	0.06	0.019	0.06	0.025	0.1	0.028	0.08	0.027	0.07	0.032	0.06
Chromium, Hexavalent	mg/L	< 0.003	< 0.003	0.01	< 0.003	0.012	< 0.003	0.018	< 0.003	0.02	< 0.003	0.018	< 0.003	0.024	< 0.003
Copper	mg/L	0.048	0.080	0.022	0.1	0.017	0.09	0.02	0.1	0.015	0.11	0.014	0.11	0.015	0.06
Iron	mg/L	< 0.01	0.115	0.02	0.46	< 0.01	0.29	0.03	1.39	< 0.01	0.91	< 0.01	1.29	0.01	0.96
Lead	mg/L	0.005	0.030	< 0.002	< 0.02	< 0.002	< 0.02	< 0.002	< 0.02	< 0.002	< 0.02	< 0.002	< 0.02	< 0.002	< 0.02
Magnesium	mg/L	10.333	11.733	0.3	13.7	0.4	12.2	0.9	24	< 0.1	17.1	< 0.1	31.5	< 0.1	20.2
Manganese	mg/L	0.015	0.100	< 0.01	0.19	< 0.01	0.11	0.01	0.4	< 0.01	0.17	< 0.01	0.56	< 0.01	0.22
Mercury	mg/L	< 0.0002	< 0.0002	< 0.0002	<0.0002	< 0.0002	< 0.0002	< 0.0002	< 0.0002	< 0.0002	< 0.0002	< 0.0002	< 0.0002	< 0.0002	< 0.0002
Nickel	mg/L	0.005	0.063	< 0.003	0.06	< 0.003	0.05	0.003	0.1	< 0.003	0.09	< 0.003	0.12	< 0.003	0.11
Potassium	mg/L	12.233	13.300	16.5	21.7	11.7	16.4	32.1	40.7	17.5	23.2	48.2	61.2	23.7	32.7
Selenium	mg/L	0.076	0.223	0.153	0.21	0.195	0.2	0.098	0.16	0.081	0.16	0.031	0.16	0.03	0.14
Silver	mg/L	< 0.0002	< 0.02	< 0.0002	< 0.020	< 0.0002	< 0.02	< 0.0002	< 0.02	< 0.0002	< 0.02	< 0.0002	< 0.02	< 0.0002	< 0.02
Sodium	mg/L	397	▨ N/A	378	▨ 3930	393	▨ 3800	450	▨ 3920	387	▨ 3770	381	▨ 5600	384	▨ 3740
Thallium	mg/L	< 0.020	< 0.020	< 0.020	0.04	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	< 0.020	0.02
Vanadium	mg/L	0.077	0.167	0.164	0.14	0.187	0.14	0.197	0.11	0.191	0.10	0.19	0.09	0.17	0.09
Zinc	mg/L	0.077	0.247	0.03	0.92	0.027	0.54	0.10	0.88	0.038	0.88	0.05	0.9	0.054	0.68
Chloride	mg/L	107	138	101	107	98	99	100	90	95	90	87	91	89	67
Nitrate	mg/L	▨ 160	< 0.05	▨ 15.8	< 0.05	▨ 15.6	< 0.05	▨ 15.8	< 0.05	▨ 16.7	< 0.05	▨ 15.4	< 0.05	▨ 15.7	0.16
Orthophosphate	mg/L	0.06	0.11	0.07	0.28	0.1	0.19	0.08	0.19	0.04	0.14	0.02	0.15	< 0.01	0.11
Sulfate	mg/L	▨ 270	169	▨ 172	176	▨ 697	154	▨ 605	245	▨ 588	166	▨ 673	308	▨ 669	179
TOC	mg/L	1	▨ 2000	1	▨ 6490	< 1	▨ 5150	4	▨ 6110	1	▨ 5470	1	▨ 6610	1	▨ 5130
Acidity	mg/L	13	1710	< 2	2380	< 2	2930	< 2	1970	< 2	1980	< 2	1590	< 2	1410
Alkalinity, Total	mg/L	9	1820	65	3590	66	3650	114	4140	131	4280	226	5880	294	6250
Alkalinity, Bicarbonate	mg/L	9	1820	19	3590	28	3650	16	4140	13	4280	11	5880	11	6250
Conductivity	umho	1890	4810	1950	7350	1840	7560	1830	8200	1800	7970	2000	8560	2000	8820
pH	S.U.	6.3	7	10.6	10.2	10.5	10.2	10.9	10.4	11	10.5	11.3	10.6	11.3	10.7
Residue, Filterable (TDS)	mg/L	1620	6450	1330	10800	1350	10800	1290	11800	1290	12000	1350	12900	1360	13300

▭	Highest of the results for this parameter.
▨	Result not valid due to contribution from constituents in leaching solution.
Bold	Chemicals of Potential Concern

In summary, the total metals content of the fresh and weathered fly ash appear to be similar, with a somewhat greater amount demonstrated to be leachable from the fresh. Of the two leaching procedures used, the TCLP yields higher leachate concentrations for nearly all analytes in both unamended and amended samples. The addition of either LKD or CKD in increasing amounts produces a progressive reduction in leachate concentrations for four of the seven COPCs, and an eventual drop after an initial rise in two additional metals. In addition, little or no apparent difference between the effects of the two reagents was demonstrated.

3.1 OVERVIEW OF MODELING APPROACH

To accomplish the key objective of this study, modeling (as envisioned in the original scope of work) consisting of an infiltration model (EPA's HELP Model) linked with a groundwater contaminant fate and transport model (MODFLOW and MT3D) was to be used to predict ash-related constituent concentrations in groundwater at the future golf course site. Analytical results of leachates derived from Chesapeake ash cured with various percentages of reagents along with site-specific baseline groundwater quality data were to be used as input data in the models. Steady state leaching from a non-depleting source of ash in the structural fill was used as a simplifying assumption in the initial modeling efforts, and a reasonable time duration for model run times was selected. The combined model was then calibrated and run.

After multiple runs, preliminary results indicated that assumptions intrinsic to the combined models were consistent with our original project assumptions, but likely too conservative and needed to be more site-specific. As a result of this initial conservatism, constituent concentrations from ash leachate resulted in predicted concentrations in groundwater above applicable drinking water standards. This included the recently reduced drinking water standard for arsenic (USEPA, 11/01, MCL=10 ug/L).

Several options were considered as a subsequent step to the initial modeling:

- Expand the testing program to include amending the ash with higher percentages of reagent (i.e., to levels greater than 5% and upward to 10% or even greater). This would be done in an attempt to lower the initial concentrations of constituents in the ash leachate;
- Introduce engineering controls such as moving the ash fills further from the south and eastern boundary of the golf course site. This would provide for a more expansive residential buffer zone along the south and east property lines; or
- Planning to provide the potentially impacted residential properties with an alternative water supply. This could take the form of replacement wells installed deeper into the next lower groundwater aquifer below the site, or installation of a city water main and individual home hook-ups.

Any of these options may have resulted in a potentially unacceptable project cost, requirements for project re-permitting with the City of Chesapeake, negative public perception, etc.. These developments could have lead to a decision to end the golf course project or development of a significantly restructured project plan.

As a result of internal discussions, is was determined that a fourth option was available. URS could develop a fully integrated model that incorporated additional site-specific conditions and more sophisticated modeling tools. The goal would be that by using this upgraded model, concentrations at the Site property boundaries could be modeled more accurately. To address that goal, it was determined that a new model should not only include infiltration and groundwater fate and transport components, but an integration of an additional component – an unsaturated zone flow model. This is justifiable in that the naturally occurring clays in the top 3 to 7 feet (in the unsaturated zone) will adsorb metals from the leachate coming from the ash fill. In this way, the integrated model could be used address adsorption of metals in the site clays. This has the indirect effect of reducing the concentration of those constituents prior to their entry into groundwater underlying the sight. To take advantage of this naturally occurring phenomenon, an unsaturated zone model (VLEACH) was integrated with the HELP, MODFLOW, and MT3D models to create a project-specific modeling package referred to as the *"Integrated Pathway Model"*.

The *Integrated Pathway Model* developed for this study satisfies the project goals and provides Dominion with a state of the art evaluation of the groundwater impacts associated with ash placement at the future golf course site. A schematic diagram of the *Integrated Pathway Model* is presented in **Figure 3-1**. This diagram illustrates how water introduced through rainfall infiltration into the new structural fill will become leachate, and flow through the unsaturated zone to the underlying groundwater and eventually to potential ground water receptors.

Each of the model components are discussed individually in the following sections along with a brief discussion of relevant issues and input parameter including:

- Site hydrogeologic conditions as determined from our recently completed hydrogeologic site investigation (URS 2001);
- Baseline groundwater quality, including the presence of risk-driving constituents like Arsenic which are naturally occurring in the regional groundwater regime;
- Residential well status in the area immediately downgradient of the proposed golf course site;
- The use of Partitioning Coefficients (Kd) which facilitate a more accurate prediction of how metals in the ash leachate are retarded from, and adsorbed during, transport through the unsaturated and saturated groundwater zones underlying the Site;
- Source depletion - more accurately modeling a stepped depletion of the leachable metals concentration in the ash leachate, and terminating that source at a defensible time into the future.

Following these introductory discussions, the model components are reviewed, and finally the resulting predicted exposure point concentrations (at the property boundary receptor) are presented as output from the integrated model. These results are then reviewed and compared with applicable risk-based drinking water standards in **Section 4.0** of this report.

3.1.1 Summary of Hydrogeologic Conditions and Site Environs

The Site is an approximate 200-acre parcel located at the southwest corner of the intersection of Centerville Turnpike and Whittamore Road in Chesapeake, Virginia (**Figure 1-1**). The Site is bounded on the north by Whittamore Road and agricultural fields, on the south by residential properties, on the west by Centerville Turnpike and residential properties, on the east by undeveloped, wooded property and residential properties. Water supply wells are not present on the Site; however, the surrounding properties utilize private water supply wells for domestic potable use. Public water supply is not currently available to the Site area.

The topography is relatively flat and slopes in an easterly direction from an elevation of 15 to 10 feet (ft) above mean sea level (msl). Site drainage is supported by east-west trending drainage swales that drain eastward eventually into the Pocaty River approximately 3.25 miles east of the Site.

Within the Site vicinity, one unconfined and six confined aquifers, and six confining units are documented (Meng and Harsh, 1988, McFarland, 1999). The total thickness of the sediment (aquifers and confining units) is estimated to exceed 3,000 feet in the Chesapeake area. The lithologic units are discussed in the Hydrogeologic Investigation (URS, 2001). The discussion of the upper two aquifers, the unconfined Columbia and the confined Yorktown-Eastover Aquifer is reiterated in this document since they are directly related to this investigation.

The uppermost aquifer is the Columbia aquifer. This aquifer is part of the Columbia Group and is the Lynnhaven Member of the Tabb Formation. The Tabb Formation consists of a fining-upward sequence of sediments composed of fine to coarse, gray sand with pebbles and cobbles grading upward into clayey and silty, fine sand and sandy silt. Locally, the aquifer is 20 to 50-feet thick and is unconfined throughout its extent. The Columbia aquifer is an important groundwater resource for rural and domestic users as well as a major source of recharge to the underlying aquifer system.

The Yorktown and Eastover formations of the Chesapeake group comprises the Yorktown-Eastover Aquifer and is the deepest aquifer used by residential wells near the Site and is approximately 350-feet thick. The aquifer is unconfined along a band parallel to the fall line to the west but confined in the vicinity of the Site. The aquifer consists of interfingered shelly, very fine- to coarse-grained sand, interbedded with silt, clay, shell beds, and gravel. This aquifer is used as a source of groundwater for domestic, commercial and light industrial uses and is considered an important recharge source for lower aquifers. The aquifer is confined by a bedded clay and silty clay unit formed by the fining-upwards of depositional sequences that formed the underlying sandy sediments of the Yorktown-Eastover Aquifer. This confining unit is highly dissected and may be absent less than 5 miles east and west of the Site resulting in a connection between the Yorktown-Eastover aquifer and the unconfined Columbia Aquifer.

3.1.2 Baseline Groundwater Quality

Groundwater samples were collected from three on-site wells in August 2001, and analyzed by Severn Trent Laboratories for 32 dissolved inorganic constituents. Additional samples were collected from the same wells in September 2001 for analysis of the same parameters on a total (unfiltered) basis, to further characterize the current quality of the groundwater. It is important to have accurate data on the baseline groundwater quality as it is used to determine the Total Predicted Exposure Point Concentrations in groundwater (see **Section 4.4** of this report for further discussion). All analyses were performed in accordance with SW-846 methodologies. Quality control data included with the sample reports were reviewed and found to be acceptable, supporting the usability of the analytical results.

Both sets of data include substantial positive detections for typical groundwater minerals such as calcium, iron, magnesium, potassium, and sodium, and inorganic anions such as chloride and sulfate. However, none of the Chemicals of Potential Concern were detected in either the filtered or unfiltered samples.

To achieve lower detection limits, arsenic and thallium were then reanalyzed in the unfiltered samples by ICP/MS (SW-846 Method 6020). Again, thallium was not detected in the samples (with a reporting limit of 1.0 µg/L), but arsenic was detected in all of the unfiltered samples at concentrations ranging from 0.81 to 1.2 µg/L. The baseline groundwater data and analytical method references are summarized in the table on the following page. Laboratory analytical reports for the baseline groundwater quality data are presented in the *Hydrogeologic Investigation Report* (URS, 2001). Analytical reports on the rerun at lower detection limits for arsenic and thallium are presented in **Appendix C**.

Baseline Groundwater Quality Data

		Sample ID	MW-1 Center	MW-2 NE	MW-2 NE Dup	MW-3 SE	MW-1	MW-2	MW-3	Duplicate (MW-2)	Maximum Concentration Detected (ug/L)
		Lab ID	A1H020245-001	A1H020245-002	A1H020245-003	A1H020245-004	A1J150117-001	A1J150117-002	A1J150117-003	A1J150117-004	
		Date Sampled	08/01/2001	08/01/2001	08/01/2001	08/01/2001	09/25/2001	09/25/2001	09/25/2001	09/25/2001	
Parameter	Units	Method	|---------- Filtered ----------|				|---------- Unfiltered ----------|				
Aluminum	ug/L	6010B	200 U	200 U	200 U	200 U	360	973	200 U	1440	1440
Arsenic	ug/L	6010B/6020	10.0 U	10.0 U	10.0 U	10.0 U	0.88 B	1.2 B	0.81 B	1.2 B	1.2 B
Barium	ug/L	6010B	200 U	200 U	200 U	200 U	200 U	200 U	200 U	200 U	200 U
Beryllium	ug/L	6010B	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U
Boron	ug/L	6010B	200 U	200 U	200 U	200 U	200 U	200 U	200 U	200 U	200 U
Cadmium	ug/L	6010B	2.0 U	2.0 U	2.0 U	2.0 U	2.0 U	2.0 U	2.0 U	2.0 U	2.0 U
Calcium	ug/L	6010B	92000	37800	38800	77300	115000	37000	87900	34100	115000
Chromium	ug/L	6010B	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U
Copper	ug/L	6010B	25.0 U	25.0 U	25.0 U	25.0 U	25.0 U	25.0 U	25.0 U	25.0 U	25.0 U
Iron	ug/L	6010B	10200	4860	4750	4790	12900	4860	4170	4760	12900
Lead	ug/L	6010B	3.0 U	3.0 U	3.0 U	3.0 U	3.0 U	3.0 U	3.0 U	3.0 U	3.0 U
Magnesium	ug/L	6010B	13200	18700	19100	15600	15000	17200	11900	15900	19100
Manganese	ug/L	6010B	339	237	242	160	429	241	153	225	429
Mercury	ug/L	7470A	0.20 U	0.20 U	0.20 U	0.20 U	0.20 U	0.20 U	0.20 U	0.20 U	0.20 U
Nickel	ug/L	6010B	40.0 U	40.0 U	40.0 U	40.0 U	40.0 U	40.0 U	40.0 U	40.0 U	40.0 U
Potassium	ug/L	6010B	5000 U	8190	8340	5000 U	5250	8450	5000 U	8040	8450
Selenium	ug/L	6010B	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U
Silver	ug/L	6010B	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U	5.0 U
Sodium	ug/L	6010B	32400	34000	34700	48200	38700	35100	48100	32900	48200
Thallium	ug/L	6010B/6020	10.0 U	10.0 U	10.0 U	10.0 U	1.0 U	1.0 U	1.0 U	1.0 U	10.0 U/ 1.0 U
Vanadium	ug/L	6010B	7.0 U	7.0 U	7.0 U	7.0 U	7.0 U	7.0 U	7.0 U	7.0 U	7.0 U
Zinc	ug/L	6010B	20.0 U	20.0 U	20.0 U	20.0 U	20.0 U	20.0 U	20.0 U	20.0 U	20.0 U
Alkalinity, Total	mg/L	310.1	130 J	75 J	73 J	160 J	160	77	180	79	180
Alkalinity, Bicarb.	mg/L	310.1	130 J	75 J	73 J	160 J	160	77	180	79	180
Bromide	ug/L	300.0A	500 U	500 U	500 U	500 U	500 U	500 U	500 U	500 U	500 U
Chloride	ug/L	300.0A	74300	54000	54000	53300	83600	49300	54800	50200	83600
Fluoride	ug/L	300.0A	1000 U	1000 U	1000 U	1000 U	1000 U	1000 U	1000 U	1000 U	1000 U
Nitrate	ug/L	300.0A	100 U	100 U	100 U	100 U	100 U	100 U	100 U	100 U	100 U
Sulfate	ug/L	300.0A	139000	103000	103000	112000	162000	97400	114000	96400	162000
Total Phosphorus	ug/L	365.2	210	170	190	390	200	170	220	160	390
TOC	mg/L	415.1	3	2	2	6	4	3	4	3	6
TDS	mg/L	160.1	510 J	390 J	380 J	460 J	550	330	460	320	550

XXX	Analysis by ICP/MS for lower detection limit
B	Result is below the lab's reporting limit, but above the MDL
J	Estimated result due to minor holding time exceedance

3.1.3 Residential Well Summary

Each of the surrounding residential properties likely uses a domestic water-supply well. A public record search provided information on three wells along Whittamore Road (northeast of the Site) and 14 wells along Murray Road (south of the Site) (URS, 2001). Of these 17 wells, ten are likely developed in the Columbia Aquifer and the remaining seven in the deeper Yorktown-Eastover Aquifer. The well logs for the remaining residential properties could not be located, but based on the reported distribution of well depths, 14:3 ratio, or approximately 82 percent of the residential wells surrounding the Site are developed in the shallow Columbia Aquifer. Wells developed in this aquifer are the potential receptors of constituents that may leach from the golf course structural fill comprised of amended ash.

3.1.4 Source Depletion

The Toxicity Characteristic Leaching Procedure (TCLP) and the Synthetic Precipitation Leaching Procedure (SPLP) are designed to simulate the leaching process undergone by a land-applied solid under natural conditions. In both procedures, a sample of the solid being tested is combined with a quantity of acidic leaching solution equivalent to 20 times its weight, then agitated in a sealed container for 16-20 hours. The mixture is then filtered under pressure, and the filtrate is analyzed to determine the leachable fraction of the chemicals of interest. The two procedures differ in the composition and pH of the leaching solution, and therefore may produce dissimilar results, but the concept and design are quite similar.

For the purposes of modeling, two assumptions are drawn from the results of the leachate analyses. The first is that the total mass of analyte found in the leachate represents the total mass of analyte available to be leached from the original sample (i.e., the amount of leachable analyte remaining in the solid is assumed to be negligible). The second assumption is that, given the same liquid-to-solid ratio, over time, that same amount of analyte will leach from the solid under natural conditions. On average, then, the concentration of the 'natural' leachate over that time period will equal the concentration found in the test leachate.

Based on these two assumptions, it can be demonstrated that the length of time necessary to deplete the leachable analyte in a specific mass of solid is equal to the number of years required to introduce a volume of water equivalent to 20 times the mass of the solid. Therefore, in order to recreate the 20:1 ratio of the TCLP/SPLP, given a specific mass of solid, the time period required depends solely on rate of water infiltration. Using the output from the HELP Model (see **Section 3.2**) and the estimated placement tonnage of approximately 1.5M tons, the time of depletion is calculated as follows:

$$\text{Mass of ash (Kg)} = 1{,}500{,}000 \text{ tons} \times \frac{2000 \text{ lbs}}{1 \text{ ton}} \times \frac{0.454 \text{ Kg}}{1 \text{ lb}} = 1.362 \times 10^{9} \text{ Kg}$$

$$\text{Infiltration Rate} = 420{,}000{,}000 \text{ L/year (from HELP Model)}$$

$$\text{Years to deplete} = \frac{1.362 \times 10^{9} \text{ Kg}}{4.2 \times 10^{8} \text{ L/year}} \times \frac{1 \text{ L}}{1 \text{ Kg}} \times 20 = 65 \text{ years}$$

The infiltration rate of 420,000,000 liters per year was determined from the HELP Model, however other models suggest a lower bound rate - possibly as low as 200,000,000 liters per year. Using that value (200,000,000 liters/year) in the above equation yields a time period for depletion of approximately 130 years. Therefore, the 130 years depletion period was conservatively used in both the VLEACH Model and MT3D Model to simulate the effects of the depletion of the leachable fraction of metals in the amended ash structural fill.

3.1.5 Selection of Soil/Water Partition Coefficients (Kd)

As summarized by the EPA (1996): "The soil-water partition (distribution) coefficient (Kd) for metals and other inorganic compounds is affected by numerous geochemical parameters and processes, including pH; sorption to clays, organic matter, iron oxides and other soil constituents; oxidation/reduction conditions; major ion chemistry; and the chemical form of the metal." As a result literature values for Kd vary by orders of magnitude. However, EPA conducted geochemical speciation analyses for metals for the following environmental conditions:

TABLE 3.1
COMPARISON OF ENVIRONMENTAL CONDITIONS ASSUMED IN EPA (1996) VERSUS SITE DATA

	pH (S.U.)	Iron Oxide (weight percent)	Fraction of Organic Carbon (g/g)	Pore-water Chemistry (mg/L)		Redox conditions
EPA (1996)	4.9-8.0	0.01-1.11	0.002	Aluminum	0.2	Viable redox conditions conducive to lowest Kd values used
				Bromine	0.3	
				Calcium	48	
				Chlorine	15	
				Iron	0.2	
				Magnesium	14	
				Manganese	0.04	
				Nitrate	1	
				Potassium	2.9	
				Sodium	22	
				Sulfate	25	
Site-specific data	5.8-7.0	-	0.002-0.003	Aluminum	<0.2	1.4 -1.88 mg/L DO
				Bromide	<0.5	
				Calcium	38-92	
				Chloride	53-74	
				Iron	5-10	
				Magnesium	13-19	
				Manganese	0.16-0.34	
				Nitrate as N	<0.1	
				Potassium	8	
				Sodium	32-48	
				Sulfate	103-139	

As can be seen, the conditions the EPA modeled cover a wider range of pH than are observed at the Site. Also, the most conservative and metal-mobilizing redox conditions were assumed in the EPA calculations whereas at the Site, oxidizing conditions are observed (DO of 1.4 to 1.88 mg/L). In addition, Kd values based on speciation analyses often under-estimate actual Kds because conservative assumptions are made due to the lack of complete reaction data. Therefore, using EPA (1996) Kd data will likely over-estimate the future metal concentrations in the groundwater and under-estimate concentrations on the soil (i.e., concentrations leaving the unsaturated zone and entering the groundwater will be higher which is conservative).

TABLE 3.2
SOIL/LIQUID PARTITION COEFFICIENTS USED IN MODELING ANALYSES (mL/g)

Constituent	Groundwater (sand) - MT3D [a]	Clay - VLEACH [b]
Aluminum	35,300 [c]	35,300 [c]
Arsenic	10 [d]	29
Beryllium	120	390
Cadmium	42	57
Chromium	4.2×10^5	1.2×10^6
Chromium, Hexavalent	22	20 [e]
Lead	5000 [f]	5000 [f]
Manganese	50 [g]	180 [g]
Mercury	7.5 [h]	30
Nickel	42	54
Selenium	7.5	5.7
Thallium	61	67
Vanadium	1,000	1,000

Notes:
a: Data for pH 6.2 (typical groundwater pH at the site) from EPA (1996), except where noted.
b: Data for pH 6.6 (typical pH of water in clay at the site) from EPA(1996), except where noted.
c: Texas Natural Resources Conservation Commission, chapter 350.)
d: Data from model-calibrated value for arsenic plume in sand aquifer (Chiyoda, Dames & Moore, 2000).
e: Hexavalent chromium was not a risk driver by the groundwater pathway.
f: No data available in EPA (1996). This is the median value recommended for lead in EPA (1999).
g: Sheppard and Thibault (1990).
h: Note that the median value recommended for mercury is 3,900 L/kg in EPA (1999).

As a reality check, these Kd values were compared to the data based on fly ash leachate and total metals analyses, as well as the Site soil and groundwater data depicted in **Table 3.3**.

TABLE 3.3
ESTIMATED SITE-SPECIFIC SOIL/LIQUID PARTITION COEFFICIENTS
ESTIMATED SITE-SPECIFIC Kd VALUES (mL/g)

	Kd for Unamended Fly Ash [a]	Equivalent Soil Kd [b]	Effective Kd for Sand Aquifer [c]	Effective Kd for Clay [c]
Arsenic	290	1,228	>170	>170
Beryllium	5,333	78,600	-	>108
	Kd for Unamended Fly Ash [a]	**Equivalent Soil Kd [b]**	**Effective Kd for Sand Aquifer [c]**	**Effective Kd for Clay [c]**
Cadmium	-	-	-	-
Chromium	1982	19,100	>320	>1,580
Lead	995	7,200	>230	>1,700
Mercury	>3,100	>36,200	-	-
Nickel	1,208	9,400	-	-
Selenium	<89	<227	-	>128
Thallium	-	-	-	-
Vanadium	1,174	9,050	-	>1,214

NOTES:

a: Kd values were calculated as the fresh ash metals content divided by unamended ash leachate concentrations.

b: Kd values were calculated using: log Kd (fly ash) = 0.7 log Kd (soil) + 0.3 (EPA, 1999).

c: Kd values were calculated as soil metal content divided by groundwater concentration.

The Kd data in **Table 3.3** are significantly higher than the Kd values used in the modeling analyses. From the results in **Table 3.3** it was concluded that:

- selenium and arsenic are still assessed to be the most mobile elements,
- the Kd for clay is greater than the Kd for sand, and
- the earlier conclusion that using EPA (1996) Kd data will likely over-estimate the future metal concentrations in the groundwater and under-estimate concentrations on the soil is further demonstrated.
- the selected Kd values used in the modeling (Table 3.2) although conservative, are defensible and appropriate for this type of modeling.

3.2 HELP INFILTRATION MODEL

The HELP model (Schroeder, 1997) provided the initial step in modeling the movement and dilution of leachate from the stablized fly ash fill of the proposed Etherridge Greens Golf Course. This model was developed by the U.S. Army Engineer Waterways Experiment Station (Vicksburg, Mississippi) for the U.S. Environmental Protection Agency (USEPA), Risk Reduction Engineering Laboratory (Cincinnati, Ohio). The primary purpose of the model is to assist in the comparison of waste structure design alternatives as judged by their water balances. The model accepts weather, soil and design data and uses solution techniques that account for the effects of surface storage, snowmelt, runoff, infiltration, evapotranspiration, vegetative growth, soil moisture storage, lateral subsurface drainage, leachate recirculation, unsaturated vertical drainage, and leakage through soil and liners. The version of HELP used in this investigation was release 3, which is the current version.

3.2.1 Data Input

The HELP model is programmed by assigning each material at the Site to specific layers and applying local climatic data. Data entry is conducted using HELP model's internal database of default values and user-defined values.

To meet the data requirements for the Site soils, the Site was defined using three soil layers. The top, vegetative cover (layer 1), the stabilized fly ash (layer 2), and the current ground surface (layer 3). Although the fill layer may vary in thickness, the cover and natural liner thickness were modeled as uniform. Below the natural ground surface (layer 3) is the aquifer. The aquifer is not modeled in HELP and is assumed to accept leachate draining through layer 3. The input parameters for each layer are summarized in **Table 3.4**.

TABLE 3.4
HELP MODEL SOIL AND DESIGN DATA ASSUMPTIONS
GROUNDWATER MODELING & RISK EVALUATION

	LAYER 1	LAYER 2	LAYER 3
Layer Type	Vertical Percolation	Vertical Percolation	Vertical Percolation
Texture	Silty Clay - Type 11	Fly ash - Type 30	Silty Clay – Type 12(1)
Thickness (feet)	0.5	5	5
Porosity (percent)	46.4	54.1	38.2
Field Capacity (volume per volume)	0.31	0.187	0.342
Wilting Point (volume per volume)	0.187	0.047	0.210
Initial Soil Water Content (percent)	22.29	31.24	38.20
Saturated Hydraulic Conductivity (cm/sec)	$6.40\text{x}10^{-5}$	$4.00\text{x}10^{-5}$	$8.20\text{x}10^{-7}$

NOTES:
(1) The default saturated hydraulic conductivity and porosity for soil texture type 12 was modified from the HELP model's default database to reflect site-specific data obtained from boring B1B.

Layer 1 was modeled as the fill cap with a 6-inch thick vegetated layer. The layer was defined as a silty clay, covered with well-maintained grass. This grass cover provides a high leaf area index and allows an evaporative zone depth through its full thickness. Since this upper layer permitted vertical water movement, it was designated as a vertical drainage layer in the model for calculating purposes. The slope of the ground surface was assumed to be an average 3 percent with a 200-foot drainage distance. The slope was evaluated using a sensitivity analysis to evaluate its influence on the total leachate production.

Layer 2 consisted of the stabilized fly ash. This layer was assigned an average thickness of 5 feet, although the preliminary planned thickness varies from nil to over 10 feet. This thickness was thought to provide a good estimate for the calculation of leachate production. Default values for the stabilized fly ash was obtained from the HELP model's internal database and then modified with data obtained from the testing of fly ash samples provided by Dominion. This layer was designated as a vertical drainage layer for calculating purposes.

Layer 3 was modeled as the base of the Site using data specific to the natural soil currently overlying the Site. This material is predominantly a sandy silt to silty clay and extends to an average depth of approximately 5 feet. The HELP model's default values for a silty clay were initially assigned to this layer. The inclusion of a site-specific permeability of 8.2 x 10^{-7} centimeters per second (cm/sec) and porosity of 38.2 percent was used to modify the default values. These values were determined from geotechnical analysis of a shelby tube sample collected from soil boring B1B (URS, 2001).

The climatic data were obtained from the default database with modification to make the data more site specific. These data are listed in **Table 3.5**. The climatic database for Norfolk and Richmond, Virginia were used as the defaults since these cities were geographically closest to the Site (about 10 and 90 miles away, respectively). The evapotranspiration parameters, relative humidity and mean wind speed for Richmond was manually entered using National Oceanic and Atmospheric Administration (NOAA) data (SC ODNR, 2001) to reflect Norfolk climatic conditions. The precipitation, temperature, and solar radiation were synthesized for Norfolk by the HELP model for a 5-year simulation. The recharge from irrigation was considered negligible since only the greens, which do not contain engineered fill, will be irrigated (per information provided by CPM, the future owner of the course 09/01).

TABLE 3.5
HELP MODEL CLIMATIC DATA ASSUMPTIONS
GROUNDWATER MODELING & RISK EVALUATION

Parameter	Source	Value	
Station Latitude (degrees)	Richmond DB	37.50	
Maximum Leaf Area Index	Richmond DB	4.50	
Start of Growing Season (Julian Date)	Richmond DB	91	
End of Growing Season (Julian Date)	Richmond DB	306	
Evaporative Zone Depth (inches)	Richmond DB	6.0	
Average Annual Wind Speed (mph)	SCDNR	10.60	
Average Percent Relative Humidity	SCDNR	1st Qtr	66
	SCDNR	2nd Qtr	68
	SCDNR	3rd Qtr	74
	SCDNR	4th Qtr	69
Mean Monthly Precipitation/Temperature	Norfolk S DB	January	3.72/39.9
		February	3.28/41.1
		March	3.86/48.5
		April	2.87/58.2
		May	3.75/66.4
		June	3.45/74/3
		July	5.15/78.4
		August	5.33/77.7
		September	4.35/72.2
		October	3.41/61.3
		November	2.88/51.9
		December	3.17/43.5

(1) Sources of information were as follows:
Norfolk S DB - HELP model's synthetically generated data using coefficients for Norfolk, VA

Richmond DB - HELP model's internal database for the Richmond, VA area;
SCDNR - Data from the South Carolina Department of Natural Resources for Richmond, VA. (SCDNR, 2001)

3.2.2 Analysis of HELP Model Results

The HELP model simulations were conducted based on the input parameters discussed earlier. The HELP model was operated using a 5-year simulation. The output from the simulation included monthly and annual precipitation, runoff, evapotranspiration, and leachate production rates for each of the years as well as the mean value for the 5-year simulation. The output units were inches of water per month and year and cubic feet per year under the criteria of a 216-acre site with 75 percent of that area allowing runoff and a slope length of 200 feet. The output from the HELP model is provided in **Appendix D**.

A sensitivity analysis was conducted to evaluate the slope of the fill piles since the golf course design had not been finalized prior to this investigation. The results of the sensitivity analysis (summarized in Table 3.6 shows that if the slope is varied between 1 and 10 percent, there is little change in the rate of leachate generation. Based on this analysis, a slope of 3 percent was selected for subsequent model simulations.

TABLE 3.6
HELP MODEL SLOPE SENSITIVITY ANALYSIS
GROUNDWATER MODELING & RISK EVALUATION

Slope (percent)	1	3	5	10
Precipitation (in/yr)	49.52	49.52	49.52	49.52
Runoff (in/yr)	1.176	1.271	1.32	1.373
Evapotranspiration (in/yr)	28.42	28.46	28.47	28.47
Leachate (in/yr)	18.97	18.87	18.83	18.79

The HELP model predicted that the leachate produced from the engineered fill material will be 18.9 inches per year and will range from 1.42 inches per month in February to 1.66 inches per month in October. These rates are based on the arithmetic means of a 5-year simulation. The mean annual recharge of 18.9 inches was used as an initial estimate of the rate of recharge entering the groundwater flow model. This value was later revised to 17 inches per year after model calibration indicated that the HELP model was over estimating recharge by approximately 10 percent (**Section 3.4.3**).

3.3 UNSATURATED ZONE GROUNDWATER MODELING – VLEACH

The purpose of this section is to present the results of unsaturated-zone groundwater modeling for potential constituents of concern including but not limited to arsenic and selenium at the proposed Chesapeake Golf Course Site. The unsaturated zone modeling was combined into the *Integrated Pathway Model* to answer the question: "What concentration of leachate from the fly ash backfill will migrate to the water table five feet below the backfill material?"

3.3.1 Modeling and Input Discussion

VLEACH was selected for this exercise because estimates of long-term seepage rates through the backfill were available from the HELP modeling results, and advective transport and chemical partitioning between the soil and groundwater were the main requirements from the unsaturated zone model. The model VLEACH, version 2.2, as developed for the U.S. EPA Office of Research and Development (Ravi and Johnson, 1997), was applied. VLEACH was used to predict the future migration of indicator chemicals arsenic, hexavalent chromium and selenium through the unsaturated zone and concentrations at the water table. These three constituents were run due to their lower Kd values (even though hexavalent chromium is not a driver in the risk evaluation - see Section 4 for more details). Results of a HELP model analysis, together with Site hydrogeologic and chemical property data were used as inputs to the VLEACH model.

VLEACH is a one-dimensional, finite difference model for making preliminary assessments of the effects on groundwater from the leaching of contaminants through the unsaturated zone. The program models four main processes: liquid-phase advection, solid-phase sorption, vapor-phase diffusion, and three-phase equilibration. In an individual run, VLEACH simulates leaching in a number of distinct polygons, which may differ in terms of soil properties, recharge rates, depth of water, and initial moisture or concentration conditions. VLEACH results are in the form of an overall, area-weighted assessment of groundwater impact.

The limitations of the VLEACH model are the following:

- Local or instantaneous equilibrium between liquid and solid phase is assumed within each polygon of the modeling grid. This is a reasonable assumption since the rate of leachate movement through the clay layer underlying the backfill is slow.
- Uniform (homogeneous) hydrogeologic conditions are assumed. No complex stratigraphy is considered. A uniform layer of average depth of 5 ft of clay is present at the Site and will be present underlying the backfill, so this assumption is also reasonable.
- A steady state moisture profile is assumed. This is reasonable since long-term leachate flow rates through the backfill are used in the modeling analysis.
- VLEACH assumes an irreversible partitioning between soil and pore water as described by a linear isotherm where the Kd is the soil-partitioning coefficient, i.e., desorption is not considered. Desorption will increase the time to flush soil clean, however desorption does not affect the maximum concentration in the groundwater. In addition, literature values suggest that a large fraction of metals are taken up by soil irreversibly (for example, arsenic is reported to be taken up 90% irreversibly (Kavanagh et al., 1997; Sandia National Labs, 1999)); therefore, this is a reasonable assumption.
- VLEACH does not consider adsorption limits; however, the adsorption of arsenic on soil as predicted by VLEACH plus that naturally occurring (1.7mg/kg, naturally, plus 0.96mg/kg, from VLEACH) is less than the average arsenic concentrations on western soils (7.2mg/kg) (Shaklette et al., 1984). This implies that the sorptive capacity will not be reached in the VLEACH model simulations, so this assumption is appropriate also.

The modeling approach consisted of running the VLEACH model for a range of leachate concentrations, depending on the fly ash amendment, at the HELP-predicted recharge rate, to predict the time varying concentration at the water table. The model was run for 130 years with a constant concentration at the top of the unsaturated clay, and zero concentration at the source thereafter. By the end of 130 years the leachable mass of arsenic is predicted to be leached from the backfill as discussed in Section 3.1 of this report. Sensitivity analyses were conducted for the most sensitive input parameter, the partition coefficient, Kd. Finally, runs were made for selenium and hexavalent chromium which have lower range Kd values.

Model input parameters and the values used in the model are shown in the **Table 3.7**. These values are based on a range of results from HELP modeling, field and literature values.

TABLE 3.7
INPUT DATA FOR VLEACH MODEL OF ARSENIC, SELENIUM AND HEXAVALENT CHROMIUM

Model Input	Values Used in Model	Comments/Source
Recharge/ Leachate Rate from Backfill (inches/year)	17 (8 was also evaluated)	Average leachate production rate based on a 5-year simulation using the HELP model; reasonable with respect to clay hydraulic conductivity of clay 8x10^{-7}cm/s
Porosity (dimensionless)	0.38	Field data from borehole B-1B,
Initial Volumetric Water Content (dimensionless)	0.38	Field data from borehole B-1B
Soil Bulk Density (kg/L)	1.65	Field data from borehole B-1B
Partition Coefficient (Kd) for Arsenic, Selenium and Hexavalent Chromium (mL/g)	Arsenic 10[a], 29[b], 270[a] Selenium 5.7[b] Hexavalent Chromium 20[b]	[a]Model-calibrated value for arsenic plume in sand aquifer and clay, respectively (Chiyoda Dames & Moore, 2000) [b]For pH 6.6 (typical pH in clay at the site) from EPA (1996)
Water Solubility (mg/L)	0.0 (1.0 was also evaluated)	EPA (1995); A solubility of 1 was evaluated in a sensitivity test and results found insensitive
Henry's Law Constant (atm-m^3/mole)	0.0	EPA (1995)
Modeled Area Width (feet)	4,000	Chosen as maximum width of fly ash backfill perpendicular to groundwater flow
Modeled Area Length in Unsaturated Zone (feet)	1	Nominal value.
Model Depth	20 cells of 0.25 ft depth each	Average depth between base of backfill and water table.
Source Concentration of Arsenic in Leachate (ug/L)	50, 100, 250, 500	Design parameter inputs
Initial Soil Concentration in Unsaturated Zone (ug/L)	0.0	Alternate soil concentrations were tested, but since the observed groundwater concentrations are very low (1 ug/L), the naturally-occurring arsenic content on the soil is irrelevant to future predicted groundwater concentrations unless the pH were to vary.
Initial Vapor Concentration (ug/L)	0.0	No volatilization for metals

Model Input	Values Used in Model	Comments/Source
Model Simulation Time (years)	500	130 years continuous source, plus time to evaluate long-term concentration trends

3.3.2 Summary of Results – Future Predictions

The following results show the predicted maximum fraction of original metal leachate concentration remaining at the water table for different adsorption distribution coefficients. Results show, for example, that an original arsenic leachate concentration of 100 ug/L, assuming an adsorption distribution coefficient of 29 mL/g (Table 3.7), would result in a maximum predicted arsenic concentration at the water table of 22 ug/L after 130 years of leaching (Table 3.8).

TABLE 3.8
PREDICTED MAXIMUM FRACTION OF ORIGINAL LEACHATE CONCENTRATION VERSUS TIME AT THE WATER TABLE FOR ARSENIC, SELENIUM AND HEXAVALENT CHROMIUM*

Time Since Emplacement of Backfill (Years)	Arsenic (used in integrated modeling) Kd = 29 mL/g	Arsenic (low-estimate Kd) Kd = 10 mL/g	Arsenic (high-estimate Kd) Kd = 270 mL/g	Selenium Kd = 5.7 mL/g	Hexavalent Chromium Kd = 20 mL/g
50	0.09	0.24	0.010	0.39	0.13
100	0.17	0.43	0.020	0.63	0.24
130	0.22	0.52	0.026	0.73	0.30
150	0.21	0.46	0.026	0.59	0.29
250	0.18	0.27	0.025	0.22	0.22
350	0.15	0.15	0.025	0.07	0.17
500	0.11	0.06	0.024	0.01	0.11

NOTES:
* For a leachate rate of 17 inches/year

These predicted results were then applied as source assumptions for the integrated groundwater model described in **Section 3.4** to estimate groundwater concentrations at the Site boundary.

3.4 GROUNDWATER MODELING

A numerical groundwater flow and transport model was constructed for the proposed Etheridge Greens Golf Course. The purpose of the model was to evaluate probable leachate migration from the proposed stabilized fill so to calculate the risk to potential receptors.

The groundwater modeling was conducted using the following public domain software:

- MODFLOW (McDonald and Harbaugh, 1988) was used as the groundwater flow code. This code is used to model the hydraulics of the aquifer.
- MT3D (Zheng, 1994) was used to simulate advective transport, adsorption, and dispersion of contaminants.

These models have been widely and successfully used in industry, research, and litigation-related projects and their capabilities are well-known by regulators. They were selected for these reasons and because they are capable of simulating the flow system and related transport at the Site.

3.4.1 MODFLOW and MT3D Models

Modeling was conducted using MODFLOW to simulate groundwater flow and calculate flow terms between model cells, and MT3D to simulate advective dispersion, sink/source mixing, and chemical reactions. These codes are modular so components may be added, deleted or changed independently of the other components. The versions of MODFLOW module components were Basic Model Package, Version 1, 9/1/87, Block-Centered Flow Package, Version 3, 7/9/92, Recharge Package, Version 1, 9/1/87, and Conjugate Gradient Solution Package, Version 2.1, 6/1/95. All modules of MT3D were Department of Defense, Version 1.5, July 1996. The data entry and initial post processing were facilitated with Groundwater Vistas Version 2.55 (Environmental Simulations, 1998).

3.4.2 Modeling Input and Discussion

The flow and transport model analysis that follows is an analysis of likely future groundwater conditions with the ash fill in place. Several conservative assumptions were made thus yielding estimated concentrations at receptors greater than expected. This choice was made to offset the fact that the modeling is being conducted for future conditions for which no site-specific transport information is available with which to calibrate the model. Conservative assumptions included:

- The backfill was assumed to be a finite source of contaminants leaching at the maximum water concentrations measured in TCLP tests.
- Adsorption distribution coefficients were selected lower than those recommended in the EPA soil screening guidance documentation. This assumption is prudent and conservative since the time of travel to receptors and loss to soil adsorption are reduced as a result. In addition, there is a wide body of opinions in the technical literature regarding the appropriate values for use as Kds.

3.4.3 Groundwater Flow Model Development

The groundwater flow model was implemented as a local, steady state groundwater flow model. The protocols for model development were the ASTM guidance documents (Designations D 5490-93, D 5447-93, D 5609-94, D 5610-94, D 5611-94, and D 5718-95). The groundwater flow model required input that included hydraulic characteristics such as hydraulic conductivity, recharge, water levels, and boundary conditions. These data were obtained from site-specific information, regional reports, and typical values reported in literature. Where possible, a sensitivity analysis was conducted to determine the need for additional accuracy in data input.

The groundwater model was constructed by overlaying a 110-column by 80-row grid over the Site and the surrounding area. The grid discretized the study area into 8,800 uniform, 50 by 50-foot cells. Each cell is provided with uniform values for each of the input parameters. The model domain that resulted from this discretization is shown in Figure 3-3. Figure 3-4 shows the grid overlain on the Site. The grid was rotated 25 degrees clockwise from north so that it is oriented with the rows parallel to the groundwater flow direction as determined from field measurements. This was done to avoid numerical dispersion in the transport calculations and to facilitate model design. The model was constructed as an unconfined, one-layer model since only the upper-most water-bearing zone, the Columbia Aquifer, was addressed in this study. Only the Columbia Aquifer was considered in this study as it is the first encountered groundwater below the Site. The saturated thickness of the model averages 50 ft.

The boundaries of the model were selected to establish an east-southeast groundwater gradient of 0.0016. This was done by setting the north and south boundaries as no-flow cells and the east and west boundaries as constant-head cells. The no-flow boundaries were set parallel to groundwater flow lines. The hydraulic head along the west and east constant head boundaries were set to elevations that would provide a west to east groundwater flow. The hydraulic heads along the constant head boundaries were a sensitive value during calibration.

Hydraulic conductivity values varied widely between sources of this data which are summarized in **Table 3.9**. Slug tests conducted at the Site at two monitoring wells provided a range of values from 0.00085 to 0.0026 cm/sec (arithmetic mean of 0.0018 cm/sec). This range is relatively low for the fine to medium sand encountered at the Site and it is suspected that the slug tests may have under estimated this parameter due to well loss that resulted from a poor connection with the aquifer. A calculation based on a two-hour specific capacity test at 25 gallons per minute (gpm) at a nearby residential well (1215 Murray Drive) provided an estimate of 0.027 cm/sec. Hamilton and Larson (1988) had used a value of 0.0184 cm/sec for the Columbia Aquifer in a regional model that included the Site. Based on this range of values, the model was calibrated within the hydraulic conductivity range of 0.0018 to 0.027 cm/sec. Hydraulic conductivity was a sensitive calibration parameter.

TABLE 3.9
ESTIMATES OF HYDRAULIC CONDUCTIVITY

Source	Tested	Hydraulic Conductivity (cm/sec)	Arithmetic Mean (cm/sec)	Arithmetic Mean (ft/day)
Slug Test	MW2-Falling Head	0.00085		
	MW2-Rising Head	0.0011		
	MW4-Falling Head	0.0026		
	MW4-Rising Head	0.0026		
			0.0018	5.1
Permeameter Tests(1)	B1, 20 to 22 feet bgs	0.0028		
	B2, 18 to 20 feet bgs	0.0030		
	B3, 18 to 20 feet bgs	0.0024		
			0.0027	7.7
Walton Specific Capacity(2)	1215 Murray Drive	0.027		
			0.027	77
Hamilton and Larson (1988)(3)			0.0184	52.2

(1) Permeameter tests represent vertical hydraulic conductivity from remolded samples.

(2) Walton (1962, 1970) specific capacity derived values are based on residential well, 1.5 to 2-hour specific capacity tests.

(3) Hamilton and Larson (1988) calculated horizontal conductivity based on the arithmetic mean of 9 tests in the Columbia aquifer.

The recharge to the model occurred from the western constant head boundary and infiltration from precipitation. The initial value of recharge was estimated at 4.4 inches per year (1×10^{-3} feet per day (ft/day)), for the case without backfill, using the HELP model. Since a range of values was not available for this parameter, it was allowed to vary by 10 percent during calibration. A value of 4 inches per year was selected in the calibrated model. Recharge was not a sensitive calibration parameter over this range.

The storage coefficient was assumed to be 0.15 based on the value used in Hamilton and Larson (1988) for the Columbia Aquifer. This value is not necessary for the creation of a steady state flow model, however, it a required value to calculate flow terms between cells which were used for the solute transport model (MT3D). This parameter was not included in the calibration process since storage coefficient cannot be calibrated unless transient data are available. A limited sensitivity analysis was conducted on this parameter to evaluate its relationship to chemical migration by comparing results for the assumed storage coefficient of 0.15 with a value of 0.30, which is the highest reasonable value for an unconfined aquifer. The high storage coefficient value did increase the mobility of chemicals in groundwater, however, the increase was relatively minor (see **Figure 3-5**).

The groundwater flow model calibration was conducted by comparing the hydraulic heads in monitoring wells MW-3 and MW-5 with those heads simulated by the flow model. These two monitoring wells are approximately along the same groundwater flow line, which therefore also provided a calibration of groundwater gradient. The remaining monitoring wells were not used in the calibration process since the groundwater surface was planar and these additional calibration targets would not add significantly to the calibration process. This comparison was both quantitative and qualitative. The quantitative technique was the calculation of residuals between simulated and measured hydraulic heads. The qualitative considerations included the ability of the model to produce the groundwater contours that are observed at the Site.

During the calibration process the results of each simulation was compared to the known site-specific information to ascertain the success of the calibration effort and to direct further calibration efforts. Each set of calibration attempts was documented on simulation logs.

The groundwater surface simulated by the calibrated model is shown in **Figure 3-6**. The residuals from the water levels observed at MW3 and MW5 were -0.46 and 0.30 feet respectively. The negative value indicates that the model over predicted the groundwater elevation while a positive value indicates that the model under predicted the water elevation relative to observed elevations. These residuals represent a slightly lower gradient than observed at the Site. These results are thought to be sufficiently accurate to use as a basis for a valid solute transport model.

The calibrated flow model was then used to simulate the flow at the Site under the assumption that fill was placed at the Site. The fill, which is more permeable than the natural surface soils, would result in an increase in recharge to groundwater as predicted by the HELP model. However, the recharge initially estimated from the HELP model was reduced by 10 percent from 18.9 to 17 inches per year. This reduction matched that applied during calibration to the natural recharge predicted by the HELP model prior to placement. Fill placement was considered under the scenario of the placement of fill in discrete mounds as indicated in the Hydrogeologic

Investigation (URS, 2001). The output from these model runs were used to create files containing the inter-cell flow terms needed to conduct the solute transport modeling.

3.4.4 Solute Transport Model Development

The solute transport model was constructed using MT3D. The MT3D software includes modules that simulate advection, dispersion, and chemical reactions of compounds.

The solute transport model was not calibrated with site-specific data. Specifically, calibration would require, for example, the matching of the concentration of contaminants or other tracer in the field with the simulation results. Since this investigation involved the effects of a hypothetical plume, no data was available that could be used for calibration of the solute transport portion of this study. Therefore conservative selections of transport parameters were made so that maximum groundwater concentrations would be predicted.

The solute transport model required as input flow terms from the groundwater flow model, concentrations of chemicals of concern in leachate and natural groundwater, and dispersion estimates. The flow terms define the rate of groundwater movement from a given model cell to adjacent cells and is the driving force in the advection module of the model. Advection is the dominant mode of transport for the contaminants in this investigation.

Adsorption retards the advective transport. This was modeled in MT3D by specifying a partition coefficient, Kd in the reaction module. When the Kd is set to zero, contaminants entering the aquifer from the fill area reach their maximum concentration at monitored locations within the model relatively quickly (i.e., less than 100 years). As the Kd is increased, the maximum concentration is reduced, and the length of time required to achieve that concentration increases (i.e., >500 years). A Kd value of 10 mL/g was selected for the simulation of arsenic while a Kd value of 7.5 mL/g was selected for the simulation of selenium. These were the only two Kd's ran in the groundwater model (with the exception of Kd=0), because they were the lowest values for any of the chemicals of concern (see **Table 3.3** of this report for a full list of Kd's used in the modeling evaluations). **Section 4.0** discusses in more detail how the results of arsenic drive the modeling if arsenic is shown to be below the drinking water standard via the *Integrated Pathway Model*, then predicted exposure point concentrations in groundwater for all other potential constituents of concern will be below their respective drinking water standards.

The concentrations of chemicals of concern were modeled as zero in the ambient aquifer and 100 units in the leachate as it enters the aquifer beneath the fill areas. The actual ambient concentrations were added to the model's predicted concentration at the completion of the simulation (see **Section 4.4** report regarding "Total" Predicted Exposure Point Concentrations). The convention of using generic units provided an evaluation of the contribution of contaminants as the percent of the initial leachate concentration at the water table. A concentration of 50 units, therefore, would represent 50 percent of the initial concentration of a particular chemical in the leachate, at the water table.

Dispersion is the spreading of contaminants over a greater region than would be predicted solely from the groundwater velocity vectors. This occurs because of heteorogeneities and tortuous flow paths in the subsurface. Dispersion is expressed as transverse and longitudinal dispersion. Because of the difficulty and expense of obtaining dispersion values in the field, most modeling studies rely on published data (Zheng and Bennett, 1995). In this investigation values of 0.1 and 100 feet were conservatively selected for transverse and longitudinal dispersion, respectively.

The solute transport model simulated the two scenarios of flyash placement previously described. Since the modeled duration of the simulation must be for a reasonable duration, the model was operated to simulate 750 years of migration and the peak concentrations were selected for risk evaluations (see **Section 4.0**). Concentration distribution maps were generated at 10, 50, 100, 250, 500, and 750- year intervals.

3.4.5 MODFLOW/MT3D Modeling Results

Since the recharge to groundwater under the flyash placement simulations differed slightly from the calibrated groundwater flow, the groundwater flow model was run to simulate the groundwater flow pattern that would result from ash placement. The resulting groundwater contours are shown in **Figure 3-7**.

The solute transport model indicated that the contaminants would migrate to the south-southeast at a decreasing rate over time under both scenarios. The solute transport model output is provided on **Figures 3-8** through **3-21** which show concentration as a percent of the initial leachate concentration. Thus, the 80 isoconcentration line on the figures represent a concentration that is 80 percent of the initial leachate concentration of a particular chemical. Chemical concentrations are highest at the ash placement areas where leachate enters groundwater, and decrease downgradient, cross gradient and upgradient of those areas. The cross-gradient and upgradient spread is due to advective transport resulting from mounding under the backfilled area and mechanical dispersion which is controlled by the dispersion terms described above. Since the dispersion terms were obtained from literature, the cross-gradient and upgradient migration results must be used prudently as the calculated dispersion may vary slightly from the actual dispersion. Downgradient movement is primarily the result of advective transport which is determined using the inter-cell flow terms from the calibrated groundwater flow model. Longitudinal dispersion is a component of the downgradient transport of the chemicals, however; it is relatively insignificant compared to the downgradient advective transport. Volatilization and degradation has been neglected in this phase of the investigation to be conservative.

The scenario of fly ash placement is shown on **Figures 3-8** through **3-13** for 50 through 750 years of the migration of arsenic (Kd = 10 mL/g). The first potential receptors to encounter chemicals from the flyash would be the residence along the southern property line of the Site (North side of Murray Road). Potential receptors east of the Site will encounter lower concentrations of these chemicals due to the effects of dilution by groundwater while those north and west of the Site are not likely to be impacted to any reportable degree. The resulting predicted percent of initial concentration is plotted on **Figure 3-14** at hypothetical wells located near potential receptors along the north side of Murray Road and at a point along the eastern Site boundary. This plot shows that at this location, the maximum concentration of 10.55% (of the original concentration) is reached after approximately 455 years. After this time, the effects of dilution by the regional groundwater flow, flush sufficient arsenic mass from the aquifer so that concentrations decrease.

A similar set of results are presented for selenium by using a Kd value of 7.5 mL/g on **Figures 3-15** through **3-20** for 50 through 750 years of migration. The breakthrough curve for selenium at the hypothetical receptor wells is provided on **Figure 3-21**. This plot is very similar to Figure 3-14, however, due to the numerically lower Kd value, the maximum concentration of

13.99% is reached after 365 years. For the purpose of illustrating the effect of Kd, the change in maximum concentration and elapsed time to reach that value is seen when the Kd is reduced to zero as shown on **Figure 3-22**. In this scenario (although not considered representative for the Site), a maximum concentration of 84.1% is reached after 130 years. At that time, the source will be depleted of its leachate constituents and the concentration immediately begins to decrease to zero. This relationship of Kd to time and maximum concentration is understandable since this variable considers the retardation potential of the media. The same mass is allowed to pass through the media regardless of the selection of this value, however, higher Kd values will tend to retard the chemical's movement so that a lower maximum concentration will occur at a later time.

These results were used in the risk evaluation to calculate the concentration of leachate components at potential receptors.

3.5 SUMMARY OF RESULTS DERIVED FROM THE *INTEGRATED PATHWAY MODEL*

The final step in the *Integrated Pathway Model* is to merge the results from the three previous models to determine the maximum predicted concentrations in groundwater – that is:

HELP + VLEACH + MODFLOW/ MT3D → **Maximum Predicted Concentrations in GW**

These maximum concentrations were evaluated along the south and east downgradient property boundaries of the proposed golf course site. The actual concentrations are determined as the product of the remaining percent of original concentrations from both the VLEACH and MT3D runs. They are calculated as chemical specific concentrations using the breakthrough curves for each chemical of concern (corresponding to chemical-specific Kds used in the VLEACH and MT3D models). As stated previously, only arsenic and selenium were run in the groundwater modeling as they represent the limiting cases. **Table 3.10** summarizes the predicted percent remaining results from the unsaturated soil and groundwater models and their integration to calculate maximum predicted concentration for arsenic in groundwater.

TABLE 3.10
PREDICTED MAXIMUM PERCENT OF ORIGINAL ARSENIC LEACHATE CONCENTRATION
INTEGRATED PATHWAY MODEL

Time Since Placement of Backfill (Years)	Arsenic VLEACH : Soils Kd = 29 mL/g	Arsenic MT3D : GW Kd = 10 mL/g	Integrated Pathway Model Result (VLEACH * MT3D)
	% of Orig. Conc. [max. VLEACH value used in analysis]	% of Orig. Conc.	% of Orig. Conc.
50	22.0	0.09	0.10
100	22.0	0.47	0.46
150	22.0	4.44	0.98
250	22.0	7.46	1.64
455[1]	**22.0**	**10.55**	**2.32**
500	22.0	10.44	2.30

(1) Maximum concentration in groundwater as predicted by *Integrated Pathway Model.*

The maximum predicted arsenic percentage of original concentration via the *Integrated Pathway Model* is 2.32%. This maximum concentration is expected at an estimated period of 455 years from the construction of the golf course (see **Section 3.4** for details). Therefore, for any initial leachate concentration ranging from about 50 ug/L through 500 ug/L (the limits of the model investigated for this study), the maximum predicted concentration at the downgradient property boundary of the Site can then be calculated via the *Integrated Pathway Model*. Performing the same analysis for selenium, results in a maximum selenium percentage of the original concentration of 10.21% at an estimated period of 365 years once the golf course construction is completed. Due to the consistently higher Kd values selected for all other constituents of concern (see **Table 3.2**), all constituents, with the exception of selenium, will have a predicted result equal to or lower than that seen for arsenic at the downgradient property boundary.

For a point of comparison, if one were to consider only the MT3D model (as envisioned in the original scope of work), the maximum arsenic percentage of original concentration is predicted at 10.55%, or about five times higher than the integrated model. Incorporating the maximum measured concentrations of arsenic in the leaching solutions, the following **Table 3.11** was developed.

TABLE 3.11
PREDICTED MAXIMUM ARSENIC CONCENTRATION AT DOWNGRADIENT PROPERTY BOUNDARY - COMPARISON OF MODELING RESULTS

	Predicted % of Original Concentration	1% Reagent [Max. Conc. – ug/L]	3% Reagent [Max. Conc. – ug/L]	5% Reagent [Max. Conc. – ug/L]
Max. Leachate Conc. [No Modeling]	100%	411	203	110
Max. Leachate Conc. MT3D Only	10.55%	44.6	22.4	12.8
Integrated Pathway Model	2.32%	9.5	4.7	2.6

Figure 3-23 presents a graphical presentation of the *Integrated Pathway Model* results for arsenic versus the maximum leachate concentration (no modeling) and MT3D groundwater model alone. These have also been plotted against the post-amended concentrations of Arsenic in leachate for 1%, 3% and 5% reagent for comparison. It is apparent that the integrated model predicts concentrations for less than the new MCL of 10 ug/L. Even at 5% reagent, the groundwater model alone predicts concentrations greater than the 10 ug/L standard. By inspection it is obvious that the amending alone (without the benefit of the modeling exercises performed in this study) would predict concentrations in groundwater the same as the maximum leach tests results for each of the three reagent percentages considered.

For simplicity in reporting and to maintain a reasonable level of conservatism, the highest concentration detected from any leach (either TCLP or SPLP) at each of the amended increments has been plotted. With the same reasoning, no distinction was made between the two preferred reagents used in the study - Roanoke CKD and Global LKD. It should be noted, that although this is conservative, it is not unreasonable as either reagent may be used at any time during the golf course construction project, and both reagents demonstrated similar strong performances during the reagent screening and chemical/physical testing performed for this study.

Additional discussion regarding comparison of output from the *Integrated Pathway Model*, with appropriate drinking water standards for all constituents of concern in the leachate is discussed in **Section 4.0** that follows. Discussion regarding the recommended minimum required percent reagent to meet target drinking water standards is evaluated and discussed in **Section 5.0** of this report.

Potential human health risks due to groundwater exposures were evaluated with respect to the potential of the ash fill to leach chemicals that are then transported to downgradient drinking water wells. Within approximately 400 to 500 feet of the proposed fill areas of the golf course, there are a number of residences whose sole source of drinking water is their property well. Since there is no readily available public water supply in the area, other drinking water wells could be installed in the area just beyond the property boundary in the foreseeable future. Property boundaries for the golf course proper are approximately 200 to 300 feet from the nearest proposed ash fill area.

The risk evaluation undertaken for this project consists of the following steps:

1) selection of chemicals of potential concern that could leach to the groundwater,
2) determination of the exposure point concentration for each chemical reaching the groundwater,
3) selection of chemical-specific drinking water standards protective of the health of nearby residents, and
4) comparison the estimated chemical concentrations to the drinking water standard.

4.1 SELECTION OF CHEMICALS OF POTENTIAL CONCERN

The first step in the risk evaluation is the selection of chemicals of potential concern. These are the chemicals leachable from the fill material and that could be transported through the vadose zone to the groundwater. These chemicals form the basis for the subsequent risk evaluation process. The leachate data on both unamended (either fresh or weathered) or amended fly ash were evaluated with regard to chemicals that could be present at levels that could represent a human health concern. If the maximum concentration from any leaching test results exceeded the chemical-specific drinking water standard (as developed in **Section 4.3**), that chemical was selected for the evaluation of potential risks.

Table 4.1 summarizes the selection process. Of the 26 chemicals reported from the leach test, seven chemicals were selected as chemicals of potential concern: arsenic, beryllium, chromium, lead, selenium, thallium, and vanadium. For consistency in comparisons, all concentration units have been converted to ug/L.

As shown on **Table 4.1**, drinking water standards are not available for five chemicals analyzed for and detected in the leachate tests (i.e., calcium, magnesium, potassium, sodium, and orthophosphate). These chemicals are considered nutrients and, therefore, are generally not evaluated with respect to adverse health effects. Therefore, they were not included in this risk evaluation.

TABLE 4.1
SELECTION OF CHEMICALS OF POTENTIAL CONCERN

Detected Chemical	Drinking Water Standard		Maximum Detected Leachate Concentration, ug/L [(1)]	Selected as Chemical of Potential Concern? [(7)]
	Value, ug/L	Source		
Aluminum	37,000	Region III RBC[(2)]	22,600	No
Arsenic	**10**	**Proposed MCL[(3)]**	**411**	**Yes**
Barium	2,000	MCL [(4)]	1,690	No
Beryllium	**4**	**MCL**	**17**	**Yes**
Boron	3,300	Region III RBC	1,010	No
Cadmium	5	MCL	3.9	No
Calcium	Not Available	--	--	No
Chromium	**100**	**MCL**	**150**	**Yes**
Chromium, Hexavalent	110	Region III RBC	24	No
Copper	1,300	TT- Action Level [(5)]	117	No
Iron	22,000	Region III RBC	1,390	No
Lead	**15**	**TT-Action Level**	**40**	**Yes**
Magnesium	Not Available	--	--	No
Manganese	5,100	Region III RBC	560	No
Mercury	2	MCL	0.3	No
Nickel	730	Region III RBC	120	No
Potassium	Not Available	--	--	No
Selenium	**50**	**MCL**	**330**	**Yes**
Sodium	Not Available	--	--	No
Thallium	**2**	**MCL**	**60**	**Yes**
Vanadium	**260**	**Region III RBC**	**300**	**Yes**
Zinc	11,000	Region III RBC	920	No
Chloride	250,000	Secondary	138,000	No
Nitrate	10,000	MCL	160	No
Orthophosphate	Not Available	--	--	No
Sulfate	500,000	Health Advisory [(6)]	308,000	No

NOTES:

[(1)] Maximum concentration detected in either fresh or weathered, amended or unamended ash regardless of leaching method (i.e., SPLP or TCLP).

[(2)] Risk-based concentration assuming conservative drinking water exposures for residential populations (USEPA, 2001b).

[(3)] Safe Drinking Water Act Proposed Maximum Contaminant Level, November 1, 2001 (USEPA, 2001c).

[(4)] Safe Drinking Water Act Primary Maximum Contaminant Level (USEPA, 2001a).

[(5)] Action Level based on Treatment Technique (corrosivity) (USEPA, 2001a)

[(6)] USEPA (1999)

[(7)]. Constituents selected as chemicals of potential concern, based on the evaluation represented in this table, are shown in bold.

4.2 DETERMINATION OF EXPOSURE POINT CONCENTRATIONS

Exposure to a chemical is dependent not only on the activities of nearby residents (i.e., drinking the water from their wells), but also on the concentration of each chemical in the groundwater at the drinking water well. This value is called the exposure point concentration. The "Total" predicted exposure point concentration for each chemical of potential concern is the sum of two elements:

- the baseline chemical concentration; and
- the amount of chemical attributable to the ash fill (i.e., the predicted concentration from the integrated pathway modeling).

The baseline chemical concentration data, the current levels of chemicals in the groundwater, were described in **Section 3.1.2.**

The predicted exposure point concentrations are the concentrations of chemicals released from the leachate, through the subsurface soil, to the groundwater and transported to the current or potentially future drinking water wells. For this evaluation, predicted exposure point concentrations (and total predicted exposure point concentrations) were estimated at a location downgradient of the golf course property and at the property boundary. This assumption is more conservative than predicting concentrations at current, existing wells that are further away. This assumption also conservatively includes any future wells that could be placed at the current property boundary.

From the initial leachate sampling results, concentrations in groundwater were estimated using the *Integrated Pathway Model.* The modeling effort focused on arsenic and selenium based on the variability among the chemical-specific Kd values and the fact that the Kd values for these two constituents were the limiting cases. The modeling effort, as described in **Section 3.0**, predicts that 2.32% of the arsenic and 10.21% of the selenium initial leachate concentrations in groundwater reach the downgradient property boundary. Exposure point concentrations for the other chemicals of potential concern were extrapolated from the arsenic results, understanding that due to higher Kd values for those constituents, no more than 2.32% of the chemical (with the exception of selenium) would reach the downgradient property boundary. **Table 4.2** summarizes the predicted exposure point concentrations; **Table 4.3** summarizes the "total" predicted exposure point concentrations for the seven chemicals of potential concern.

TABLE 4.2
COMPARISON OF PREDICTED EXPOSURE POINT CONCENTRATIONS AND DRINKING WATER STANDARDS FOR CHEMICALS OF POTENTIAL CONCERN

Chemical of Potential Concern	Drinking Water Standard, ug/L	Estimated Exposure Point Concentrations, ug/L [1]			
		Unamended	1% Amended	3% Amended	5% Amended
Arsenic	10	9.3	9.5	4.7	2.6
Beryllium	4	0.12	0.16	0.39	0.39
Chromium	100	3.5	1.4	2.3	1.6
Lead	15	0.93	ND(2)	ND	ND
Selenium	50	33.8	21.3	16.2	3.1
Thallium	2	1.4	0.93	ND	0.5
Vanadium	260	7.0	4.3	4.6	4.4

NOTES:

(1) For arsenic and selenium, concentrations are from integrated model results (Section 3.5). For all other chemicals, concentrations were extrapolated from the arsenic results by multiplying the maximum leachate concentration by 2.32%, the maximum percent of chemical remaining in the groundwater at the property boundary for all other constituents with the exception of selenium.

(2) "ND" – Chemical not detected in leachate test.

TABLE 4.3
COMPARISON OF "TOTAL" PREDICTED EXPOSURE POINT CONCENTRATIONS AND DRINKING WATER STANDARDS FOR CHEMICALS OF POTENTIAL CONCERN

Chemical of Potential Concern	Drinking Water Standard, ug/L	"Total" Predicted Exposure Point Concentrations, ug/L [1]			
		Unamended	1% Amended	3% Amended	5% Amended
Arsenic	**10**	**10.5**	**10.7**	**5.9**	**3.8**
Beryllium	4	0.12	0.16	0.39	0.39
Chromium	100	3.5	1.4	2.3	1.6
Lead	15	0.93	ND(2)	ND	ND
Selenium	50	33.8	21.3	16.2	3.1
Thallium	2	1.4	0.93	ND	0.5
Vanadium	260	7.0	4.3	4.6	4.4

NOTES:

(1) Total predicted exposure point concentration is the sum of the predicted (modeled) exposure point concentration and the baseline concentration of each chemical in groundwater.

(2) "ND" –Chemical not detected in both the amended leachate tests and in baseline groundwater sample.

4.3 SELECTION OF CHEMICAL-SPECIFIC DRINKING WATER STANDARDS

The Safe Drinking Water Act (SDWA) passed in 1974 gives the U.S. Environmental Protection Agency (USEPA) the authority to set drinking water standards that are designed to control the level of contamination in the nation's drinking water.

Primary standards, set by the USEPA under the SDWA, limit the levels of specific chemicals that can adversely affect public health. These standards are legally enforceable and apply to public water systems. There are two types of standards based on health effects: Maximum Contaminant Levels (MCLs) or Treatment Techniques (TTs). The USEPA has determined MCLs and TTs for many of the chemicals detected in the fly ash leachate (USEPA, 2001a, c). **Table 4.2** summarizes these values, which were used to select chemicals of potential concern (**Table 4.1**) and to evaluate potential risks.

There are a number of chemicals for which no primary standard (i.e., an MCL) has been derived. To evaluate potential risks a risk-based "standard" can be calculated by assuming default values for ingestion of groundwater as recommended by the USEPA and a fixed level of risk. This procedure is documented in the USEPA Region III Risk-Based Concentration (RBC) Tables (USEPA, 2001b). Development of these values assumes the ingestion of 2 liters of water for 350 days for 30 years. Since all these chemicals are noncarcinogens, the fixed risk level is assumed to be equal to a level of exposure that is likely to be without an appreciable risk for any noncancer toxic effect. The RBCs are listed in **Table 4.1** for those chemicals for which an MCL has not been derived.

For sulfate and chloride, neither an MCL nor an RBC is available. The USEPA has, however, conducted scientific investigations to determine what concentration of sulfate in drinking water would be protective of sensitive subpopulations. A 1998 Centers for Disease Control workshop evaluating the scientific evidence for sulfate health effects concluded that there is insufficient information upon which to base a regulation, but recommended a health advisory where sulfate levels exceed 500 mg/L (USEPA, 1999). The 500 mg/L value was adopted as a drinking water standard for sulfate in this risk evaluation.

Similar information for chloride was not found. Therefore, the secondary standard for chloride (250 mg/L, based on aesthetic reasons) was adopted for use in this risk evaluation.

4.4 COMPARISON OF DRINKING WATER STANDARDS TO CHEMICAL CONCENTRATIONS

The evaluation of risks integrates the results of the groundwater modeling and baseline groundwater concentrations (i.e., the estimation of total predicted exposure point concentrations) with the drinking water standards. The potential for adverse health effects is evaluated by comparing the chemical concentrations in groundwater with the drinking water standards, which are the levels considered to be protective of residential populations. If the groundwater concentrations are below the standard, then it is reasonable to conclude that there is no appreciable risk from chemicals in the drinking water either now or in the future.

4.4.1 Predicted Exposure Point Concentrations in Groundwater

Table 4.2 illustrates that for all chemicals of potential concern, the predicted concentrations at the property boundary from the modeling effort are all below the drinking water standards. Based on this, there is no concern for chemicals leaching from the unamended fly ash placed as fill material on the proposed golf course. Similarly, the various amendment regimes also result in acceptable levels of chemicals once they reach the property boundary. It should be noted that at the lowest percent reagent additions (i.e., 1% LKD/CKD), the results from the integrated pathway model predicts concentrations for arsenic just below the proposed MCL of 10 ug/L (i.e., maximum predicted concentration at 1% reagent of 9.5 ug/L).

4.4.2 "Total" Predicted Exposure Point Concentrations in Groundwater

In evaluating total predicted concentrations, however, the contribution of leaching arsenic (approximately 9.5 ug/L, considering 1% amendment) with the maximum baseline concentration of arsenic in the groundwater (approximately 1.2 ug/L), will result in levels of arsenic (i.e., 9.5 + 1.2 = 10.7 ug/L) above the drinking water standard of 10 ug/L. For the other chemicals of potential concern, the total predicted groundwater concentrations are well below the drinking water standards (**Table 4.3**).

Figure 4-1 presents a graphical comparison of the predicted exposure point concentration and the "total" predicted exposure point concentration for arsenic as a function of the percent reagent added to the ash. See **Section 5.0** for further evaluation of recommended minimum percent reagent to meet target drinking water standards at the Site property boundaries.

4.5 RISK EVALUATION AND INHERENT CONSERVATISM OF THE APPROACH

There are a number of factors that can introduce uncertainty into any estimate of risk. In this risk evaluation, there is some uncertainty in the modeling effort since assumptions must be made. For the most part, however, conservative assumptions were used in this effort so that the chemical concentrations predicted at the downgradient property and consequently at downgradient drinking water wells are likely to be overestimated. These conservative assumptions include how much infiltration will occur at the Site, how long it will take for the chemicals leaching from the ash to be depleted, and how much chemical retardation in the soil and groundwater will occur.

Similarly, the accuracy of the chemical-specific drinking water standards in the risk evaluation is dependent on the accuracy of the assumptions made about how much water is ingested and for how long. In addition, development of these standards requires information about the toxic effects of each chemical. Estimates of both exposure and toxicity are based on limited information and therefore some uncertainty exists. To be protective, the assumptions made in the calculation of risk-based standards were intentionally conservative. These included the amount of water a person drinks daily (1 to 2 liters), and the fact that they reside and use their well at this rate for long periods of time (30 years).

This risk evaluation addressed potential exposures to individual chemicals in groundwater. In reality, exposure would occur simultaneously to all chemicals. This results in some small portion of additive risk that is not accounted for. This underestimate of risk is likely, however, to be minor given that all these chemicals have different noncancer effects and are not truly additive.

In summary, this risk evaluation is conservative due to the assumptions contained in the various steps of the evaluation. This level of conservatism is appropriate for this type of risk evaluation considering receptor proximity, the inherent uncertainties in the modeling effort, and finally the accuracy of the chemical-specific drinking water standards used in the risk evaluation.

Utilizing the results of the stabilization testing in concert with the *Integrated Pathway Model* and risk evaluations performed for this study, the following summary, conclusions and recommendations are made.

5.1 SUMMARY AND CONCLUSIONS

- Two free-lime reagents, Roanoke Cement CKD and Global Stone LKD, were selected as preferred reagents from a portfolio of six potentially viable reagents suggested by Dominon's stabilization contractor, VFL Technologies. The preferred reagents were used throughout the stabilization program for this study. As requested by Dominion, a third preferred reagent, Blue Circle Cement, is being tested, and will be reported in a subsequent letter to Dominion.
- Two types of Chesapeake fly ash were considered – fresh (silo) and weathered (landfilled). Representative samples of both were provided by Dominion to our lab AEP. The total metals content of the fresh and weathered fly ash appear to be similar. Based on two industry standard leach tests used in this study (TCLP and SPLP - SW 846 Methods 1311 and 1312, respectively), it was determined that, in general, a greater amount of metals were leachable from the fresh ash. As a result, fresh ash was selected for use in all remaining lab testing as agreed to by Dominion. Of the two leaching procedures used, the TCLP generally yielded higher leachate concentrations for nearly all analytes in both unamended and amended samples of the Chesapeake fly ash.
- The addition of either the Global Stone LKD or Roanoke Cement CKD in increasing amounts generally produced a progressive reduction in leachate concentrations. In addition, little or no apparent difference was noted between the effects of the two preferred reagents on leachablity of metals. As expected, geotechnical test results indicated that the addition of the LKD and CKD resulted in improved material properties. Hallmarks of those improvements included increased compressive strength and California Bearing Ratio (CBR) commensurate with the increasing percentages of reagent added to the ash. CKD demonstrated greater benefits to strength and CBR than LKD.
- As part of this study, Site hydrogeologic conditions were determined from the recently completed hydrogeologic site investigation at the proposed golf course site (URS 2001). Results from that study including subsurface stratigraphy, physical properties of stratagraphic units, groundwater flow conditions, baseline groundwater quality, and well survey information were utilized in the modeling exercises performed for this project.
- The *Integrated Pathway Model* developed for this study satisfies the project goals and provides Dominion with a state of the art evaluation of the groundwater impacts associated with ash placement at the future golf course site. This model is an enhancement of the approach originally envisioned for this project in that it incorporates additional site-specific conditions including unsaturated flow modeling and utilizes more sophisticated modeling tools. In addition, less than realistic and/or overly conservative assumptions associated with the more simplistic model initially considered in this study were eliminated. The goal of the upgraded model was to predict the groundwater concentrations at the Site property boundaries more accurately while providing Dominion with a technically defensible approach. Four USEPA recognized models were successfully integrated in this effort

including HELP(infiltration), VLEACH (unsaturated flow modeling), MODFLOW (groundwater flow modeling) and MT3D (groundwater chemical transport modeling).

- Soil-water distribution coefficients (Kd) for metals and other inorganic compounds found in the Chesapeake ash leachates were examined and evaluated extensively as part of this study. The goal was to select realistic and defendable values for Kd incorporated into both the unsaturated and groundwater transport models - literature Kd values vary by orders of magnitude. Additional effort in this regard was warranted as Kd has more effect on the outcome of the modeling than any other single input assumption.
- The *Integrated Pathway Model* predicts that approximately 2% to 10% of the initial leachate concentrations, reach the groundwater at the Site property boundary. The modeling effort focused on arsenic and selenium based on the variability among the chemical-specific Kd values and the fact that the Kd values for these two constituents were the limiting cases. The modeling effort, as described in **Section 3.0**, predicts that 2.32% of the arsenic and 10.21% of selenium's initial leachate concentrations in groundwater reach the downgradient property boundary. Exposure point concentrations for the other chemicals of potential concern were extrapolated from the arsenic results, understanding that due to higher Kd values for those constituents, no more than 2.32% of the chemical (with the exception of selenium) would reach the downgradient property boundary.
- The risk evaluation undertaken for this study consisted of the following steps: 1) selection of chemicals of potential concern that could leach to the groundwater; 2) determination of the exposure point concentration for each chemical reaching the groundwater; 3) selection of chemical-specific drinking water standards protective of the health of nearby residents; and 4) comparison the estimated chemical concentrations to the drinking water standard.
- The downgradient property boundary of the Site was conservatively selected as the exposure point concentration of concern for risk evaluations performed in this study. The downgradient property lines are located approximately 200 to 300 ft from the nearest ash fill. Residential wells (nearest existing receptor) are located approximately 400 to 500 ft from the nearest ash fill.
- The results of a risk evaluation comparison between "Total" Predicted Exposure Point Concentration and drinking water standards are summarized below:

Comparison of "Total" Predicted Exposure Point Concentrations and Drinking Water Standards for Chemicals of Potential Concern

Chemical of Potential Concern	Drinking Water Standard, ug/L	"Total" Predicted Exposure Point Concentrations, ug/L (1)			
		Unamended	1% Amended	3% Amended	5% Amended
Arsenic	10	10.5	10.7	5.9	3.8
Beryllium	4	0.12	0.16	0.39	0.39
Chromium	100	3.5	1.4	2.3	1.6
Lead	15	0.93	ND	ND	ND
Selenium	50	33.8	21.3	16.2	3.1
Thallium	2	1.4	0.93	ND	0.5
Vanadium	260	7.0	4.3	4.6	4.4

(1) "Total" Predicted Exposure Point Concentration is equal to the sum of the predicted exposure point concentration from the *Integrated Pathway Model* and the baseline concentration of each chemical found in Site groundwater.

- Results indicate that for reagent dosages with the preferred LKD or CKD greater than approximately 1.3% by weight, the total predicted exposure point concentrations in groundwater for all potential chemicals of concern at the property boundary are below drinking water standards.
- The risk evaluation performed for this study is conservative due to the assumptions contained in the various steps of the evaluation. This level of conservatism is appropriate for this type of risk evaluation considering receptor proximity, the inherent uncertainties in the modeling effort, and finally the accuracy of the chemical-specific drinking water standards used in the risk evaluation.

5.2 RECOMMENDATIONS

5.2.1 Recommended Reagent Dosage - Stabilization of Fly Ash Fill for the Golf Course

It is recommended that a target reagent dose for the Chesapeake Station fly for this project be set between 1.5 to 2.0% (by weight) at Dominion's discretion. Either the Global Stone LKD or Roanoke Cement CKD can be used to achieve adequate stabilization. Using these reagents at the recommended dosages will result in leachate concentrations corresponding to total predicted exposure point concentrations in groundwater (at the property boundary) below applicable drinking water standards for any constituents leaching from the amended ash. This includes arsenic at the newly proposed level of 10 ug/L.

Figure 5-1 illustrates the effect of percent reagent versus predicted concentrations in groundwater. Addition of 1.5% reagent by weight results in a maximum total predicted concentration of arsenic in groundwater (at the property boundary) of approximately 9.5 ug/L. At 2.0% reagent by weight, the predicted concentration in groundwater for the same conditions is approximately 8.3 ug/L. Either dosage would be acceptable; however, Dominion will need to weigh the modeling results of the 2.0% addition in light of the project objectives, Dominion's perspectives on risk tolerance and impacts to overall project costs. Dominion's contractor preparing the amended ash fill (for use in the golf course construction) will need to verify their ability to dose the ash with reagents at 0.5% increments. If the tightest tolerance for dosing is a single percent, Dominion may be required to target 2.0% reagent by default.

5.2.2 Material Production and Fill Construction – Specifications and Quality Assurance Manuals

It is recommended that a Material Production and Quality Assurance Manual be developed for this project which outlines and specifies as a minimum:

- reagent grades, minimum dosing percentages and procedures to assure that the minimum reagent dosage is met,
- a program of both scheduled and unscheduled quality assurance (QA) testing on both the reagents and the amended ash materials,
- verification of proper moisture conditioning and completeness of mixing,

- loading protocols and maximum pre-use, on-site storage times, and
- control procedures associated with fugitive dust, run off and a checklist of regulatory issues or management controls necessary for proper and compliant operation of the on-site production operations.

In addition, Construction Specifications and a Quality Assurance Manual are recommended in advance of placement of Dominion's stabilized ash material on the golf course site. Considerations need to be made for controls to ensure compliance during off-site transport, and to ensure that the stabilized ash is being utilized in a manner that is compliant with all construction permits and industry standard best management practices. Considerations should include fugitive dust and surface run off, on-site storage and re-use/replacement limitations, compaction and moisture conditioning requirements, and any other controls associated with extent of placement, minimum property buffer requirements and the like. It is recommended that independent third party QA be employed by Dominion or mandated for Dominion's ash handling contractors.

5.2.3 Long-Term Management - Groundwater Quality Issues

The results of this study indicate that with proper amending of the Chesapeake fly ash, the maximum predicted concentrations of constituents leachable from the fly ash in groundwater will be below drinking water standards at the Site property boundary. Findings from these studies, although legitimate and defensible in their own right, do not guarantee the actual groundwater quality once construction begins. Groundwater conditions during and after completion of the ash fill can be predicted by models, but can only be authenticated by groundwater quality monitoring once the project is underway. In other words, the studies completed for this project and presented in this report are critical to evaluation of whether to proceed with the project (Dominion's go/no-go decision). However, they are not intended to replace the intrinsic benefit of real-time groundwater quality monitoring. Therefore, we recommend that Dominion consider developing a strategy for approaching long-term groundwater monitoring.

In our opinion, the long-term groundwater management strategy for this project should be considered in light of the following:

- The golf course applicant CPM (and indirectly Dominion who will provide 1.5M tons of amended ash for this project) is potentially liable for the financial burden (and community perception) of Stipulation No. 10 from the City of Chesapeake agreement. That stipulation covers a period for up to 7 years, and includes potentially 50+/- residential wells located within 2,000 feet of the site. According to Stipulation No. 10 of the City's agreement:

  > *"The applicant agrees that he will provide such replacement wells upon receiving a complaint of well damage unless a professional hydraulic/or water quality studies shows conclusively that the diminution of groundwater and/or contamination was not caused or contributed to by the construction or operation of the golf course or related facilities."*

  Although the findings from groundwater modeling studies and related on-going activities can be combined into a competent argument, should the time come when a resident is claiming that Stipulation No. 10 applies, statistically robust, long-term groundwater monitoring data could be the most conclusive approach to defending a case against such implied liability.

Long-term groundwater quality monitoring in conjunction with (and as a confirmation of) the pre-construction modeling is a defensible argument against such claims should they materialize. Predictive modeling before and on-going monitoring during and after construction could be the best available technology to support the requirements from within Stipulation No. 10.

- We recommend that Dominion consider the value of early warning tools (like groundwater monitoring) to avoid the potential for vulnerabilities associated with an uninformed position regarding on-going groundwater quality. In our opinion, an awareness of any statistically significant changes in groundwater quality trends (should they occur) would be best known by Dominion first. This would also allow Dominion to track any significant changes in groundwater concentrations in comparison to VDEQ's default groundwater quality criteria (Maximum Contaminant Levels, Alternative Contaminant Levels, etc.).
- It is important to point out that groundwater monitoring is not mandated by the VDEQ for this type of project, and (by virtue of the lack of requirements) is, in our opinion, left up to the discretion of the owner.

In the event that Dominion decides groundwater monitoring is necessary, there are several long-term groundwater monitoring strategies that we recommend be considered for the golf course site. These are briefly reviewed in the following bulletized list, not necessarily listed in order of preference:

- **Existing On-Site Well Network**: Utilizing the five shallow monitoring wells recently installed for the hydrogeologic site characterization – either alone or in conjunction with other perimeter sentry wells. Data from these interior wells would be helpful in that one could track the trends in groundwater quality immediately under the course over time. These wells are cost effective in that the cost for installation is associated with previous work activities. According to CPM, these wells would need to be converted into flush mounted wells once the golf course is operational.
- **Property Boundary Sentry Wells:** Installing nested or single sentry wells on down and side gradient boundaries of the Site. These could be used alone or in conjunction with the interior wells cited in the on-site well network option. The perimeter wells allow one to treat the Site like a "black box", avoiding the focus on individual areas of the Site and be more concerned with the compliance point of the Site boundary. This is a more global monitoring tool than the interior wells alone in that the potential diminimus impacts or changes in groundwater quality may be dampened (attenuated) by the time the groundwater moves to the perimeter location. At that point, the boundary would likely show no statistically significant trend changes.
- **Off-Site Residential Wells**: Monitoring of a statistically representative number (and location) of the off-site residential wells. Samples could be collected at the taps of off-site wells, as in the initial round of residential water supply sampling to be performed by CPM. The downside of this approach is that on-going access agreements need to be maintained; there may be difficulty keeping analytical results confidential; Dominion could find out late that there are elevated concentrations in certain wells; and Dominion could have a hard time

defending the noncontribution of the Site if the off-site water supply wells were all the data that had been collected.

Other issues that need to be considered in developing a long-term groundwater monitoring program could include, but would not be limited to:

- frequency of sampling (quarterly, semi-annual, annually, etc.),
- list of parameters, lab and field quality assurance/quality control samples (i.e., QA/dups/MS/DS, etc.),
- statistical methods and software tools to use in data evaluation,
- number of sampling rounds to establish background,
- work plans and field protocol, plus documentation standards, and
- field sampling techniques (bailer, low-flow pumps, filtered/nonfiltered, etc).

In addition, issues related to reporting, confidentiality (attorney confidential, etc.), joint funding (CPM/Dominion) or sole source funding for sampling activities, and other related issues need to be considered. Recommendations presented in this report related to long-term groundwater monitoring are not intended to be an exhaustive discussion, but representative of the type of issues and questions that will need to be addressed in the strategy and implementation of a long-term groundwater monitoring program. The cost for long-term monitoring will be directly related to the list of parameters, duration and frequency of sampling and number of wells monitored.

5.2.4 Recommendations for Establishing Background Conditions for Off-Site Potable Wells

We understand that CPM is planning[1] on collecting a round of groundwater quality samples from the taps at all 50+/- residents, potentially located within the 2,000 ft boundary limits set forth in the City's Stipulation No. 10. This is the most comprehensive and prudent approach to establishing a snap-shot background assessment. Although it is limited to a single event, a data set will be established that is comprehensive (i.e., covers every well and not just statistically valid sets of a smaller number of wells), and represents the pre-construction condition referenced in Stipulation No. 10. This will also give CPM/Dominion the opportunity to verify the absence or presence of residential wells and potable water use within the 2000 ft boundary condition. There are potentially less expensive approaches to the sampling and analysis for background, but this appears to be CPM's method of choice and cost burden.

It is recommended as discussed previously, that Dominion consider the following QA features to be built into CPM's background water quality assessment:

- There should be a documented work plan that CPM will be following. That work plan should be prepared before contacts are made and sampling schedules are established. Third party review of the plan would be important to verify that all relevant issues and protocols are established and agreed before the start of the fieldwork.

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[1] CPM may have already initiated or completed sampling at the issue date for this report.

- The work plan should include as a minimum protocols and procedures to address the following:
  - ✓ Sample collection (including tap priming, etc.),
  - ✓ Containerization and minimum volumes,
  - ✓ Sample container labeling and preparation of chain of custody,
  - ✓ Shipping and preservation,
  - ✓ Documentation requirements including visual observations and an simplified evaluation of tap pressure/delivery capabilities (as applicable),
  - ✓ Development of a master summary map and spreadsheet inventory of all potential potable wells prior to the start of the field activities,
  - ✓ List of constituents and methods (to assure a complete list of relevant parameters, to verify that proper laboratory reporting limits are being utilized, etc.),
  - ✓ Analytical testing QA and data results validation, and
  - ✓ Reporting requirements and data presentation formats.

In addition, the following items should be considered:

- ✓ A protocol and agreed to strategy for contacting residents and gaining permission for sampling,
- ✓ A discussion whether the results will be shared with residents, or kept confidential;
- ✓ Release of liability or conditional liability for any activities associated with the sampling, and
- ✓ Address the need for third-party oversight and documentation of sampling activities. Split sampling and independent confirmation of analytical results on a select number of representative samples. Third-party oversight plus proper and complete documentation testing QA would be important in the event that any of the data is used in a legal proceeding associated with Stipulation No. 10.

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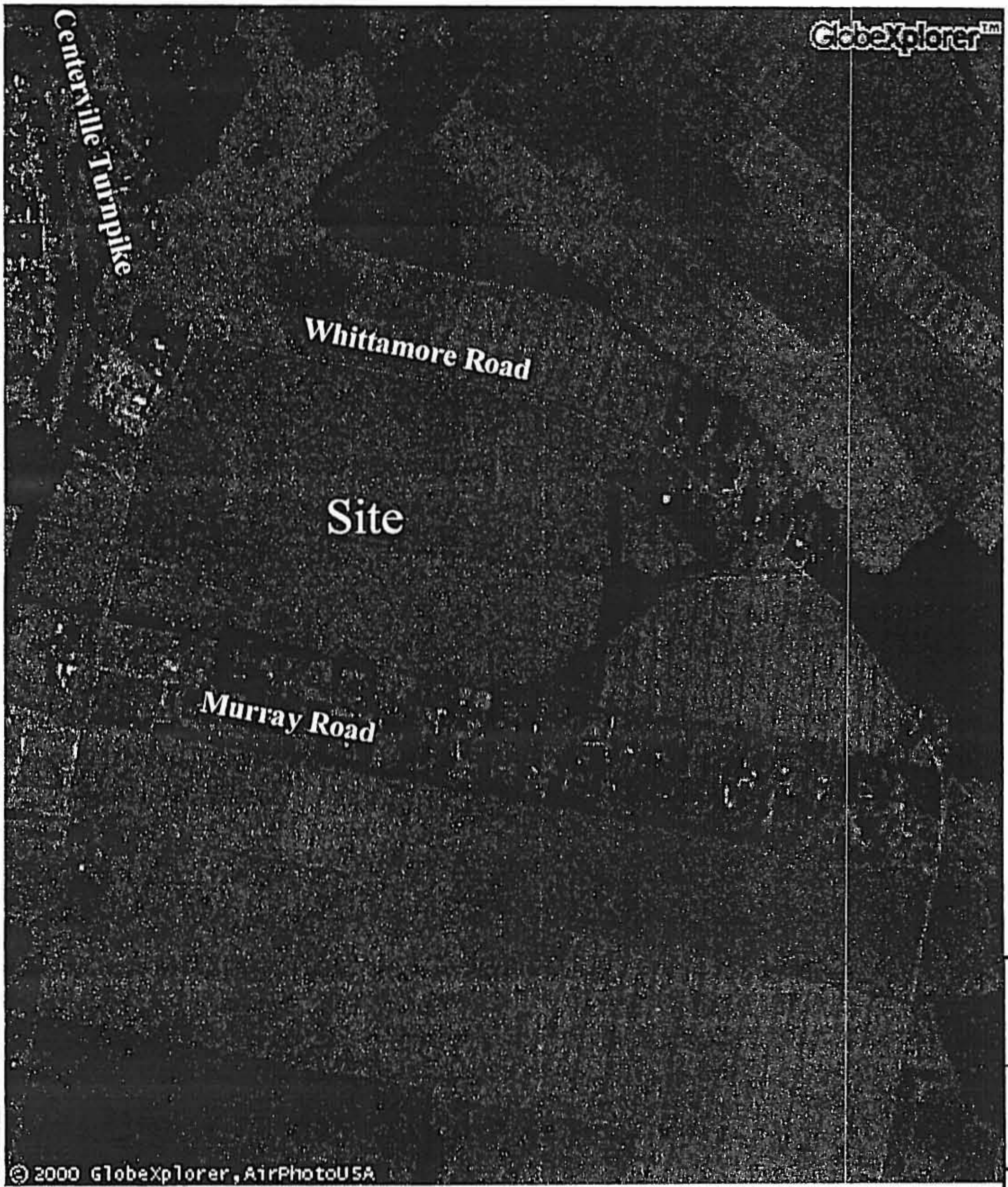
FIGURES



Groundwater Modeling & Risk Assessment
Chesapeake Energy Center
Proposed Golf Course Project

Figure 1-1
Study Area

URS

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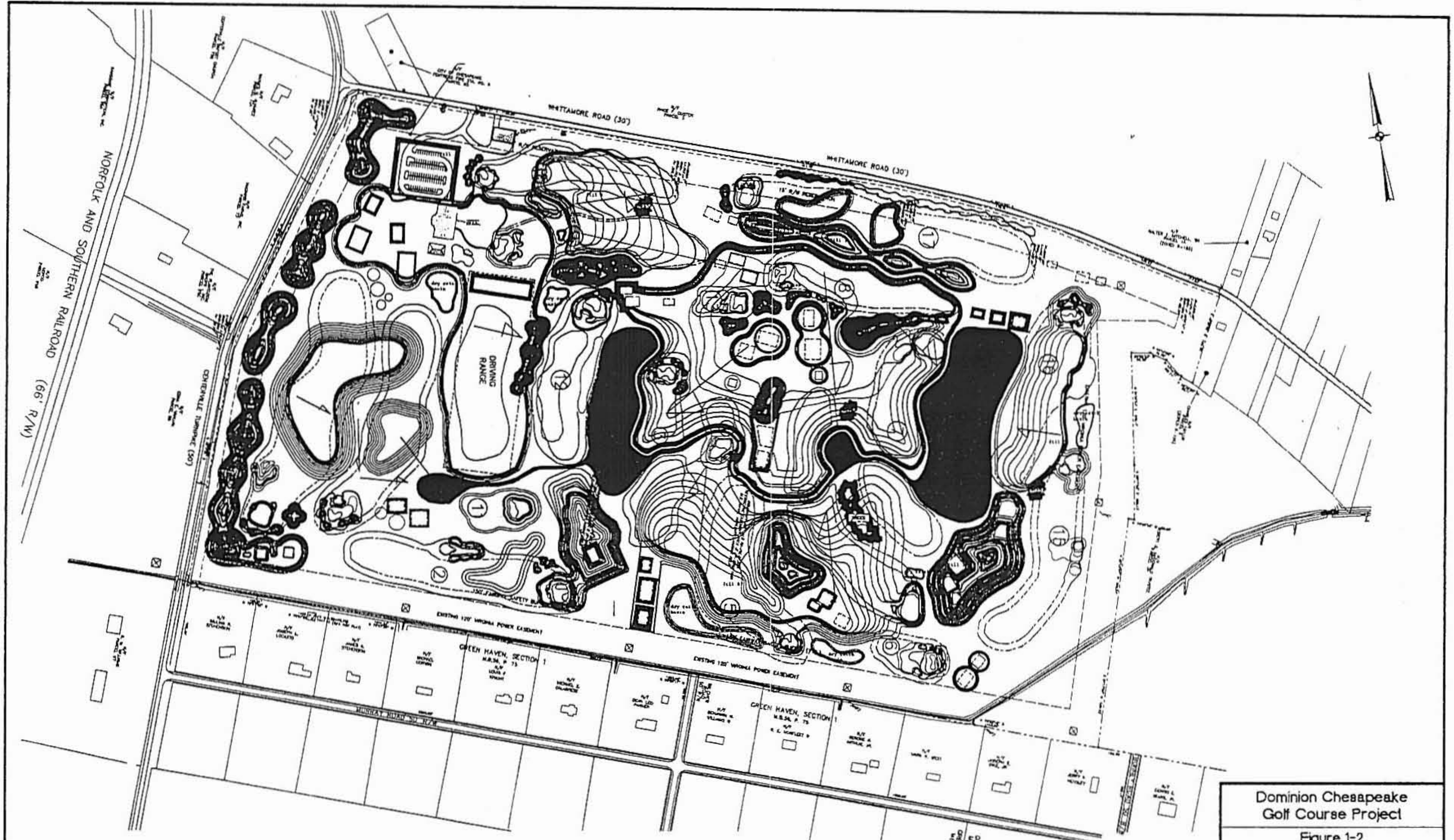


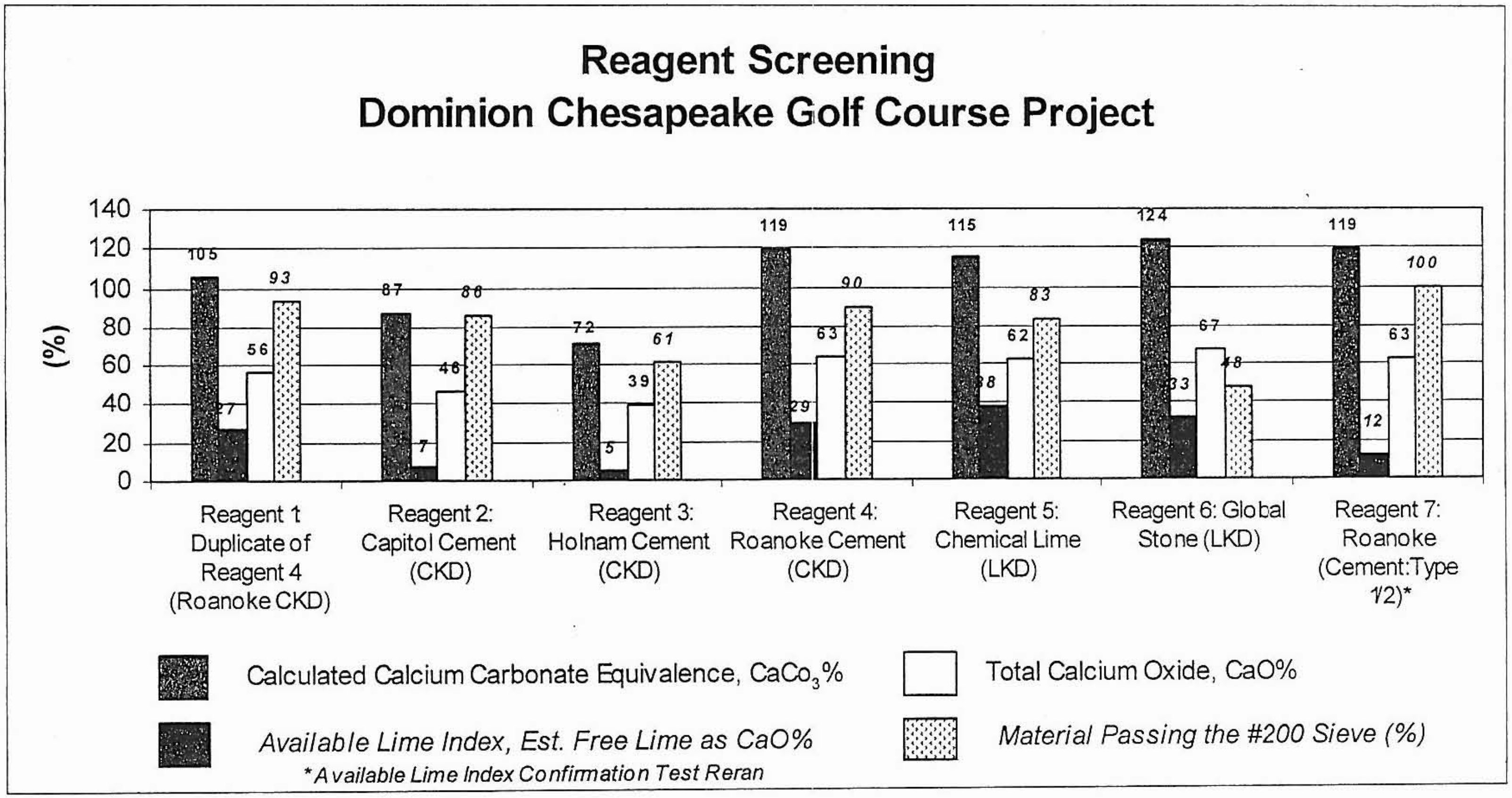
Dominion Chesapeake
Golf Course Project

Figure 1-2

GOLF COURSE
LAYOUT PLAN

URS

Base Map Source:
Etheridge Greens Site Plan, Drawings 1 & 2
Prepared by Hassell and Folkes, P. C.
Chesapeake, Virginia
Dated: April 2001



*Available Lime Index Confirmation Test Reran

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Chesapeake Energy Center
Proposed Golf Course Project

Figure 2-1
Reagent Screening

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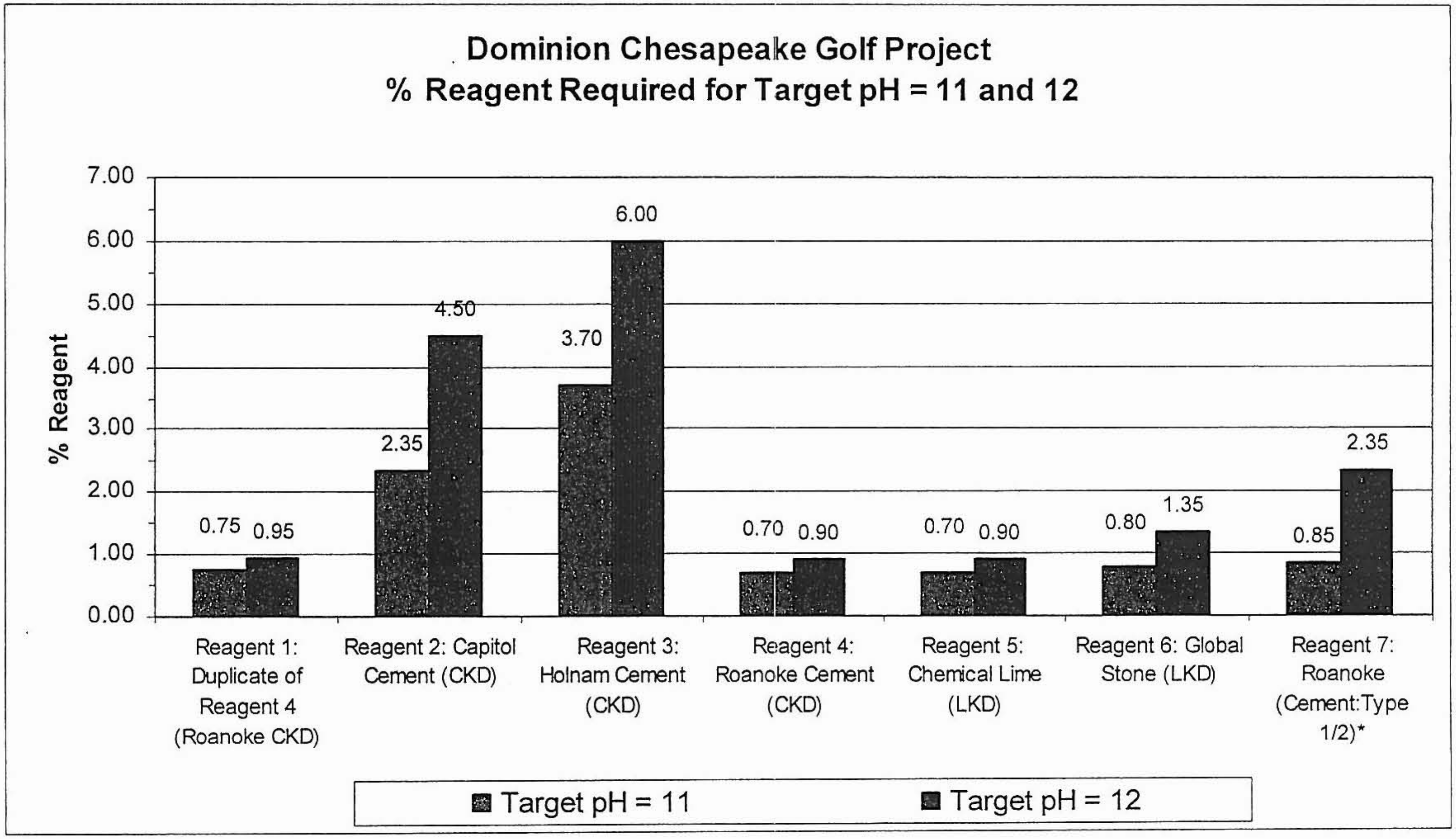

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Figure 2-2
% Reagent Required for Target pH = 11 & 12

URS

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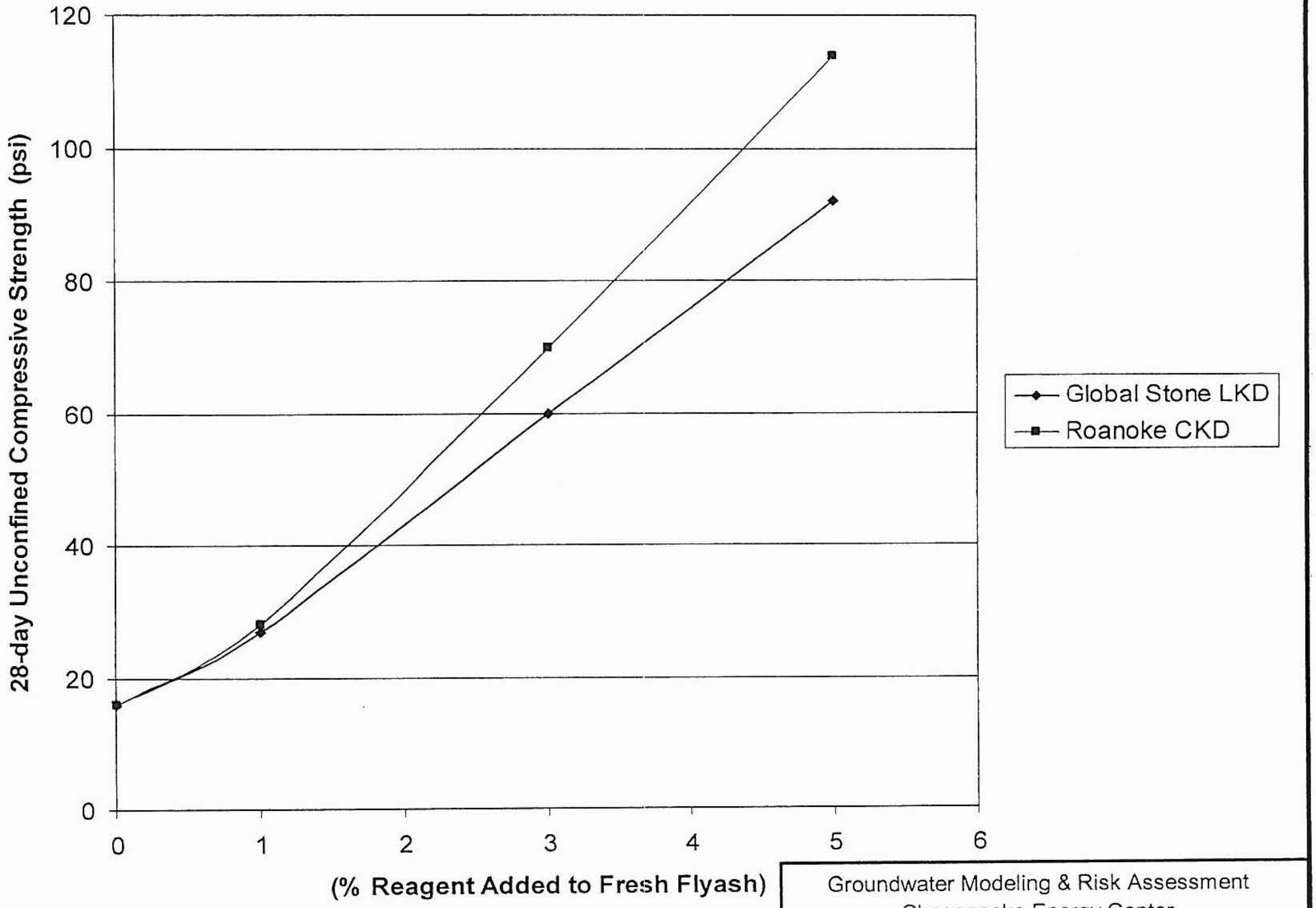

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Figure 2-3 Unconfined Compressive Strength vs. Reagent Concentration

URS

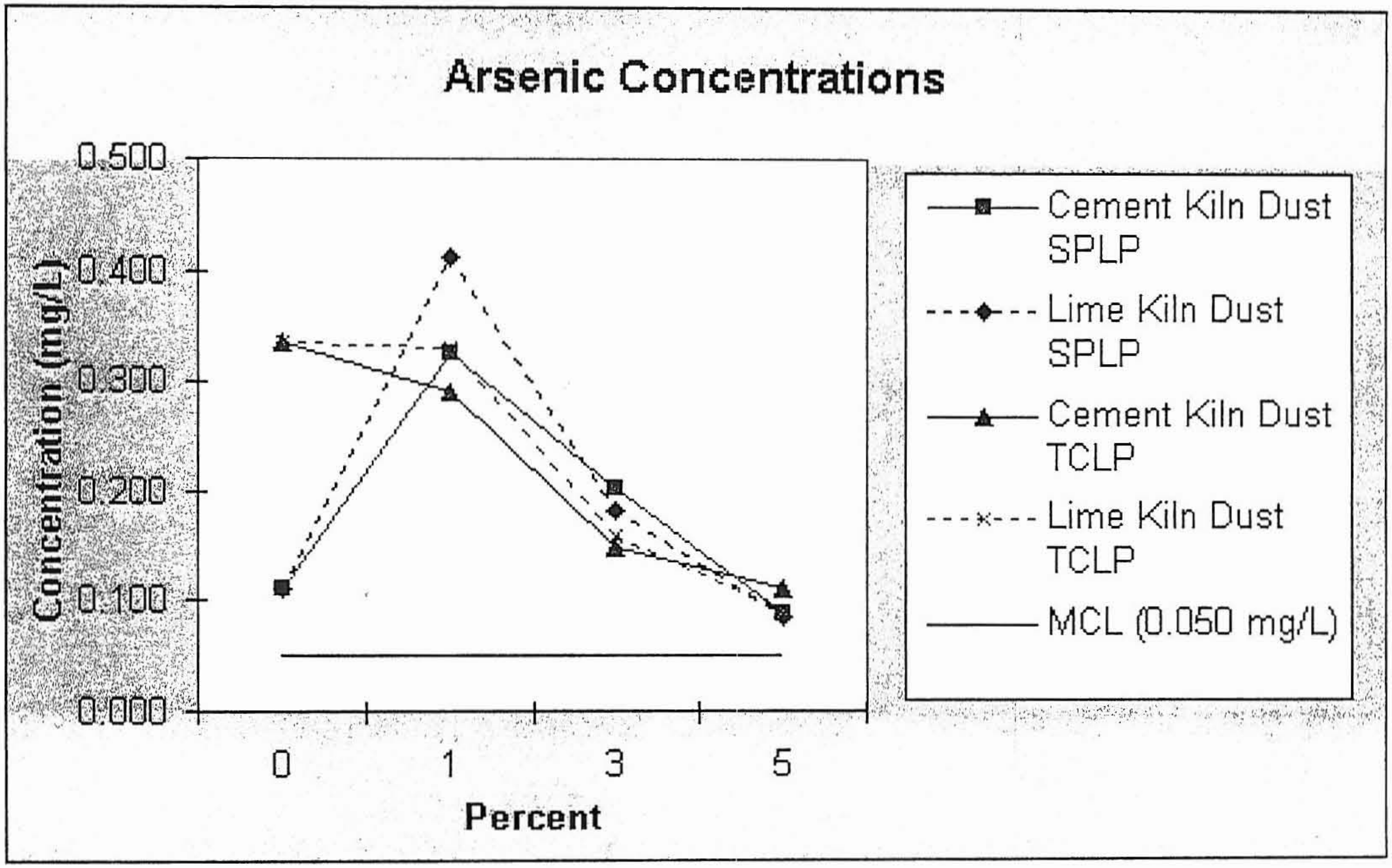

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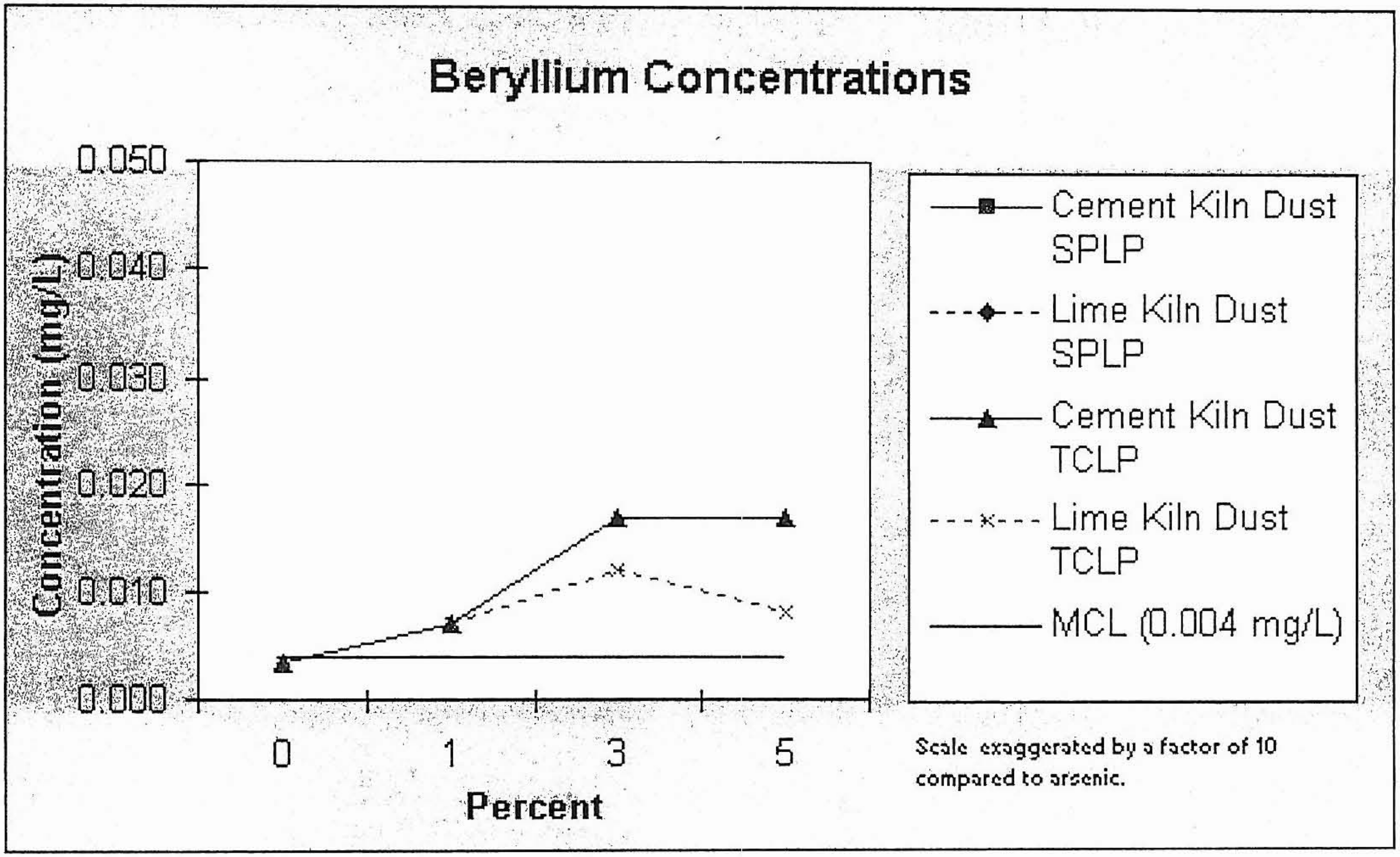
Figure 2-4 (A)
Amended Fly Ash Leachate Data (Arsenic)

URS



Note: SPLP results for beryllium were all non-detect at 0.0002 mg/L.

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Figure 2-4 (B)
Amended Fly Ash Leachate Data (Beryllium)

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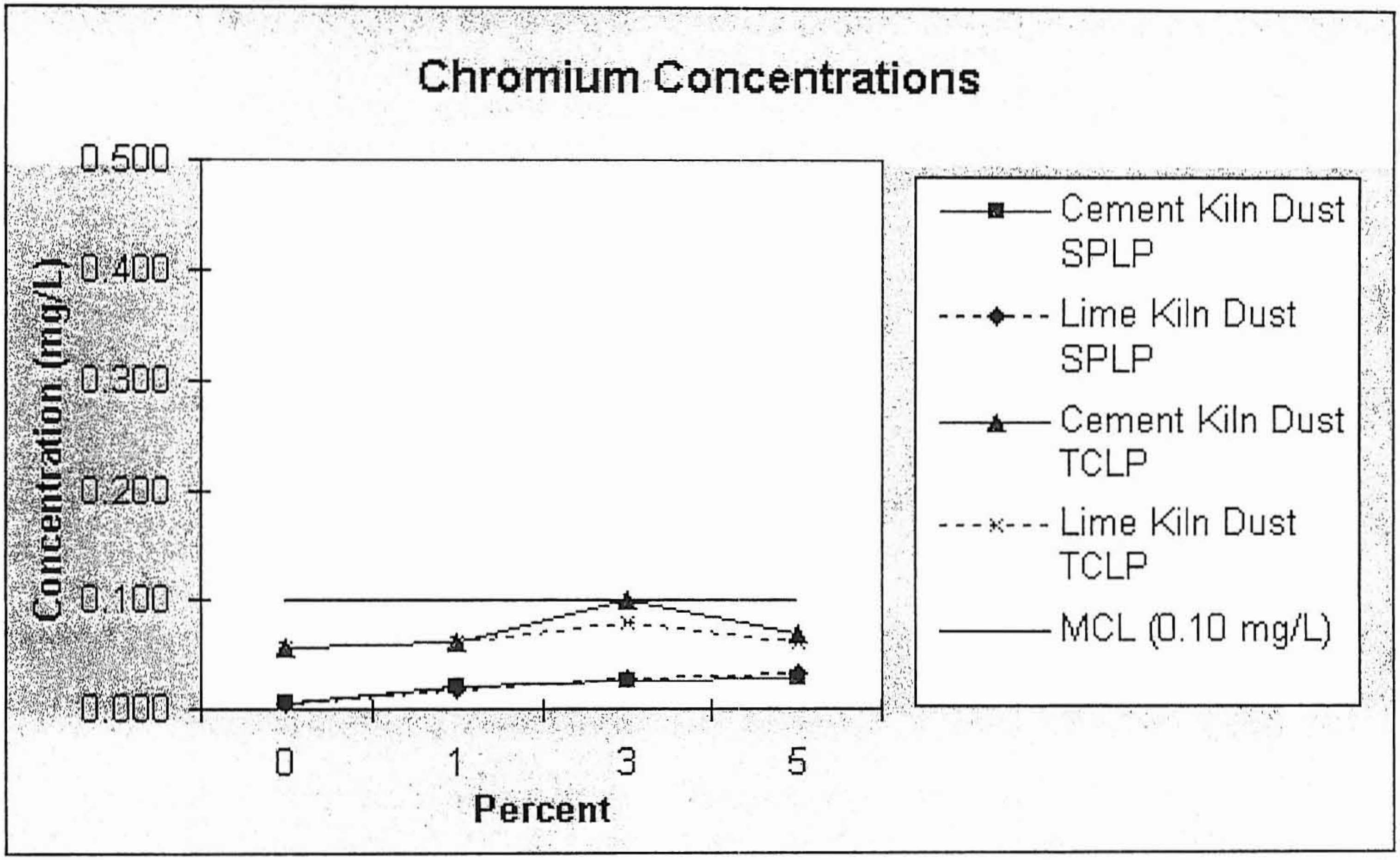

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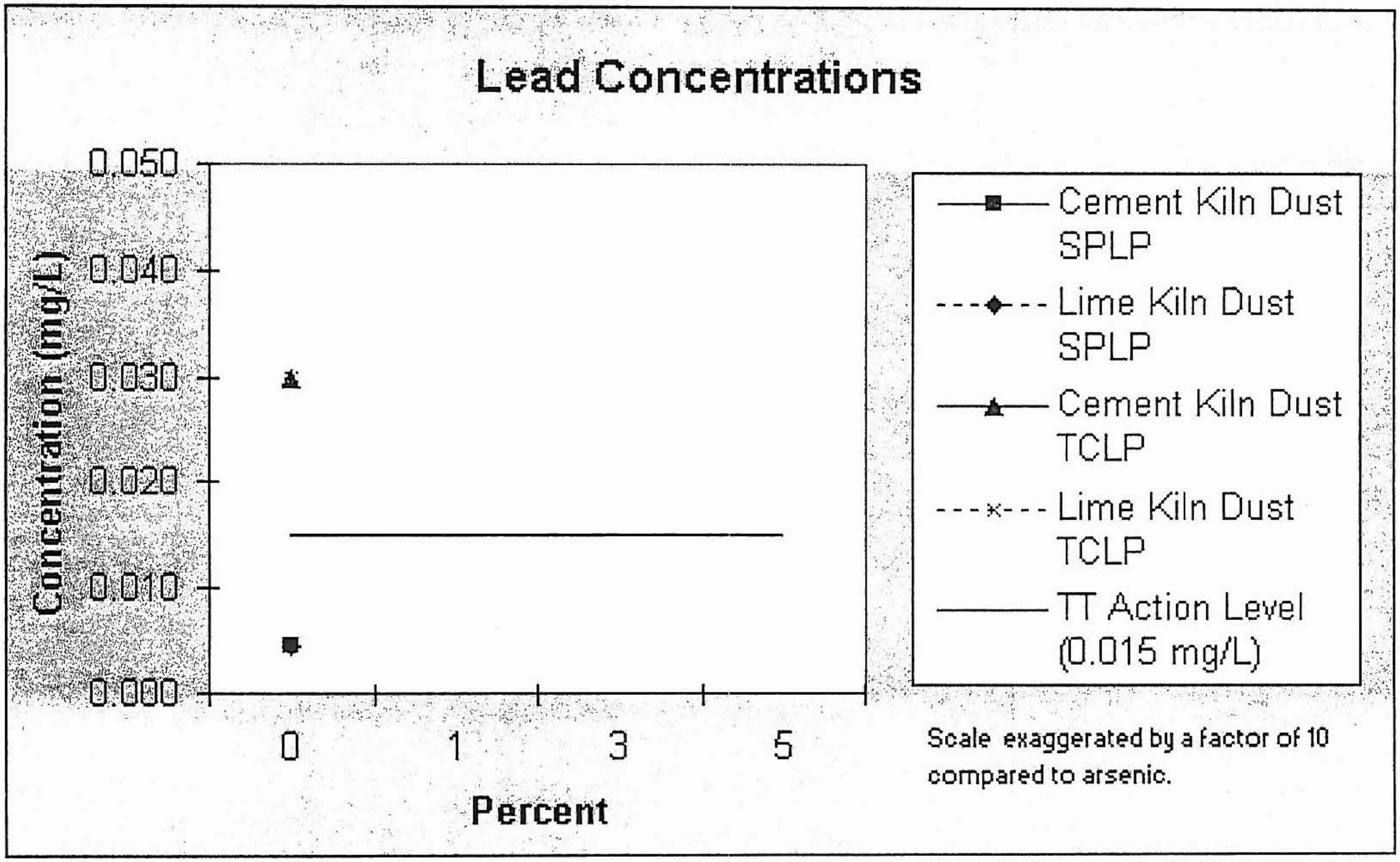
Figure 2-4 (C)
Amended Fly Ash Leachate Data (Chromium)

URS



Scale exaggerated by a factor of 10 compared to arsenic.

Note: All of the amended SPLP results were non-detect at 0.002 mg/L, and all amended TCLP results were non-detect at 0.020 mg/L.

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Figure 2-4 (D)
Amended Fly Ash Leachate Data (Lead)

URS

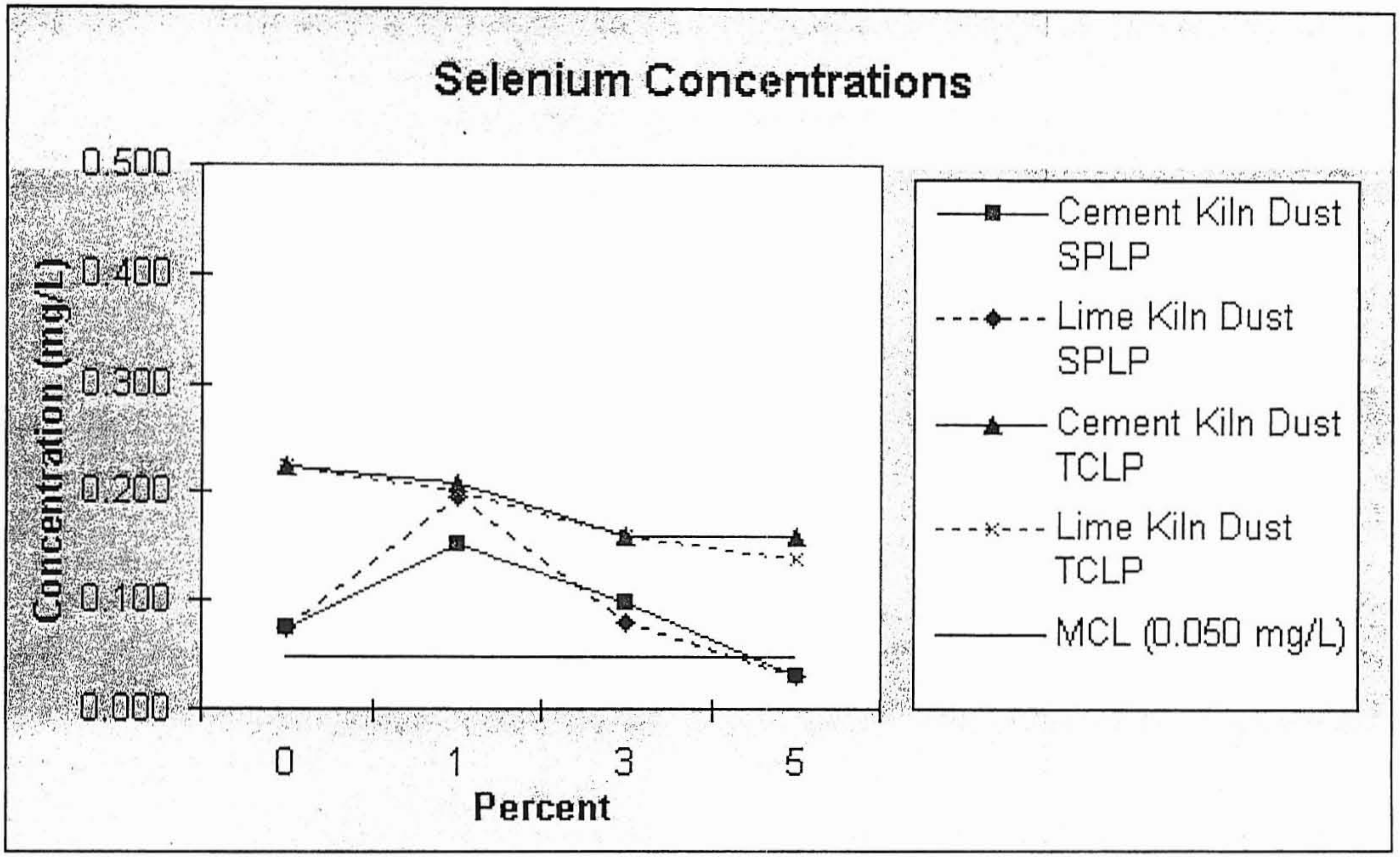

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Figure 2-4 (E)
Amended Fly Ash Leachate Data (Selemium)

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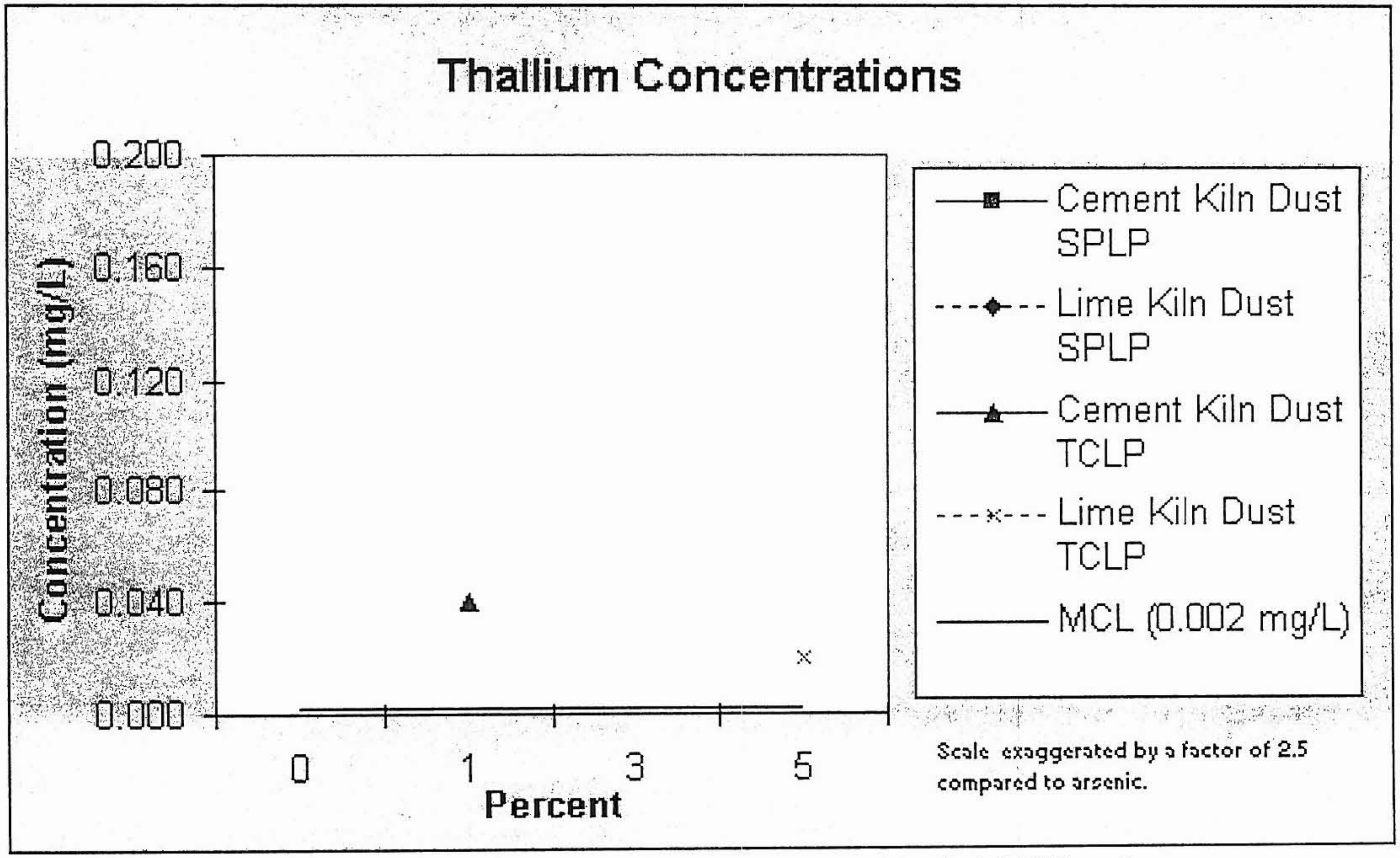

Note: All results except the two points shown above were non-detect at 0.020 mg/L.

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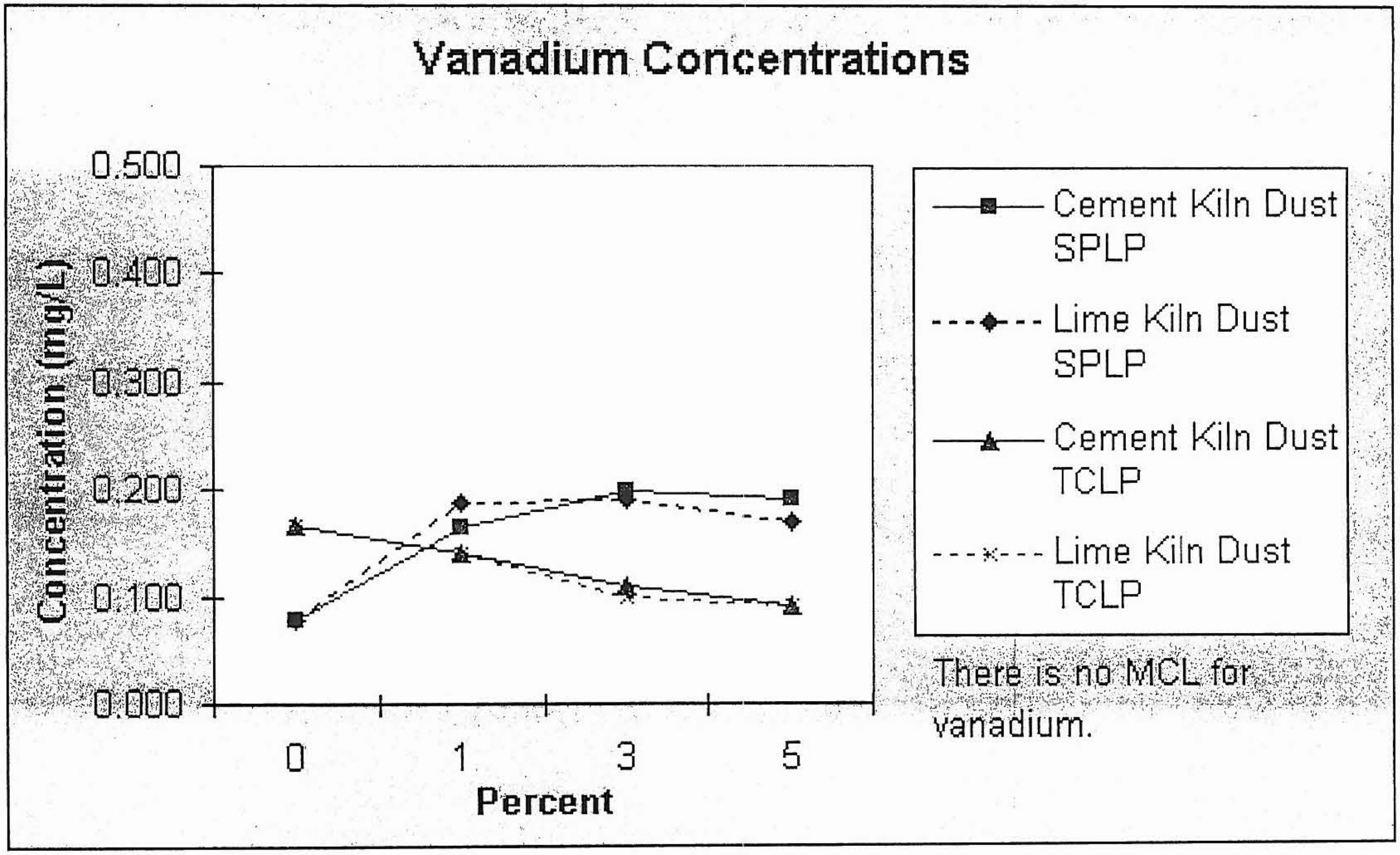
Figure 2-4 (F)
Amended Fly Ash Leachate Data (Thallium)

URS



There is no MCL for vanadium.

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Chesapeake Energy Center
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Figure 2-4 (G)
Amended Fly Ash Leachate Data (Vanadium)

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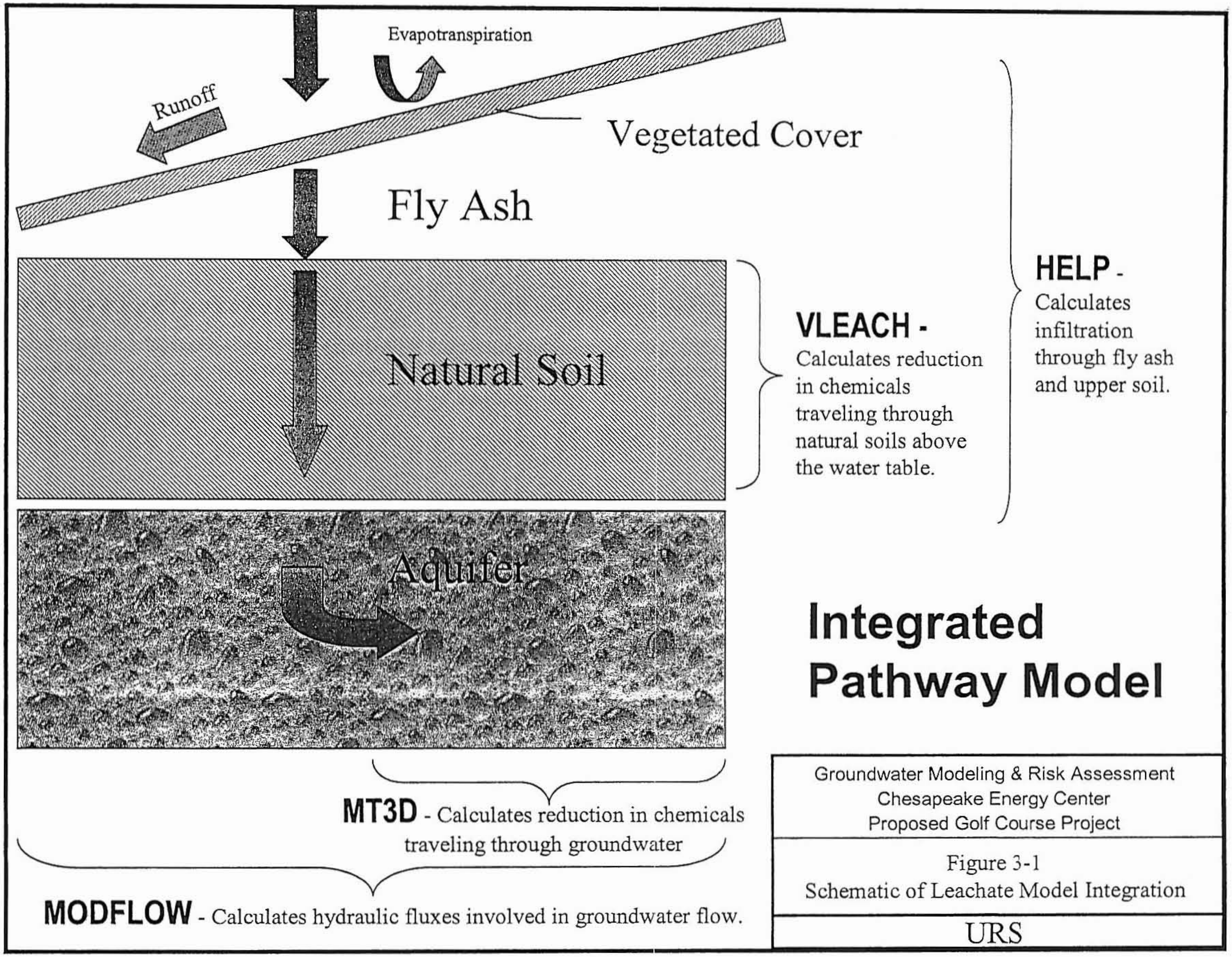

Integrated Pathway Model

Groundwater Modeling & Risk Assessment
Chesapeake Energy Center
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Figure 3-1
Schematic of Leachate Model Integration

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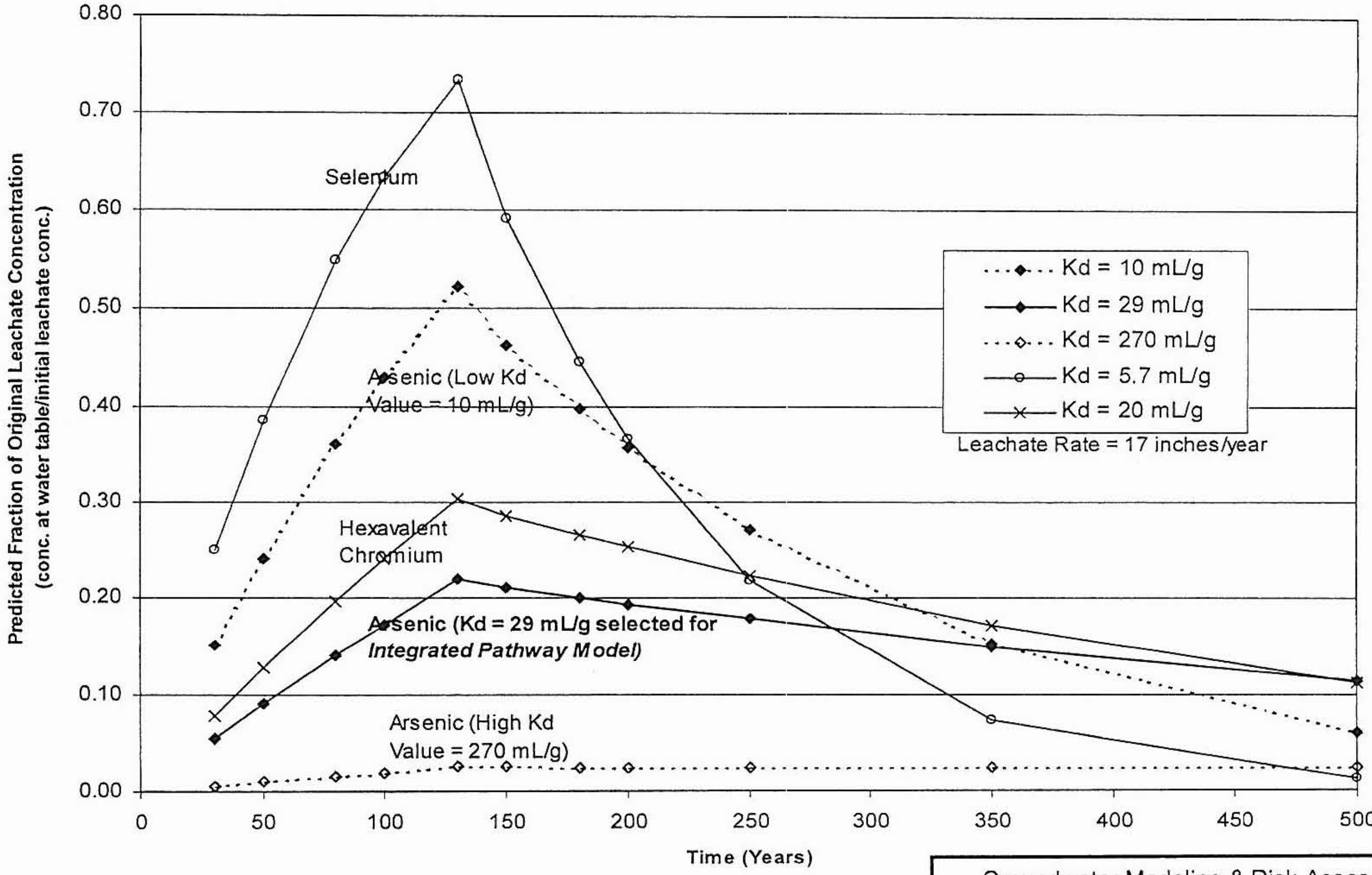

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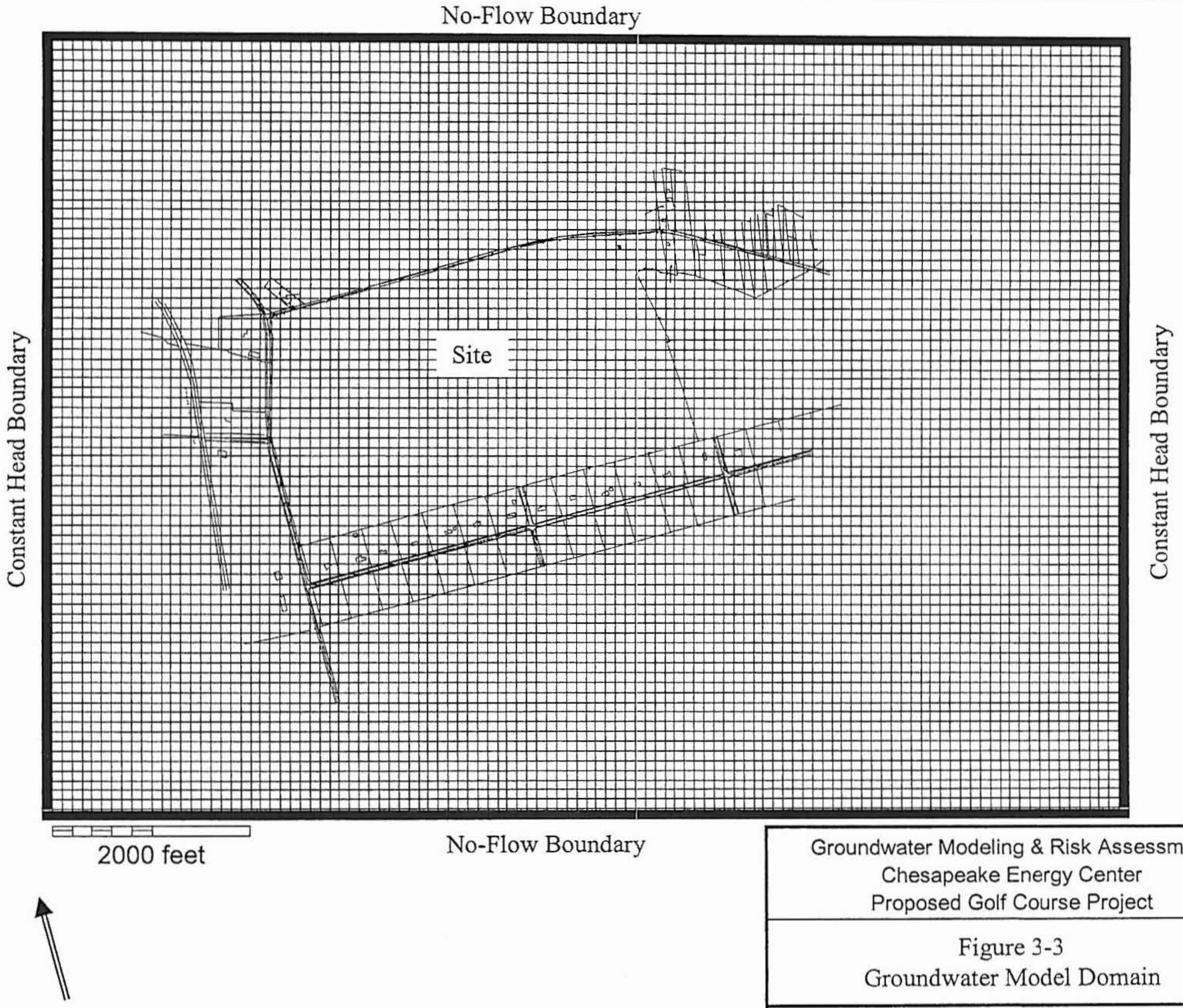
Figure 3-2 - VLEACH -Predicted Concentration at Water Table

URS



Groundwater Modeling & Risk Assessment
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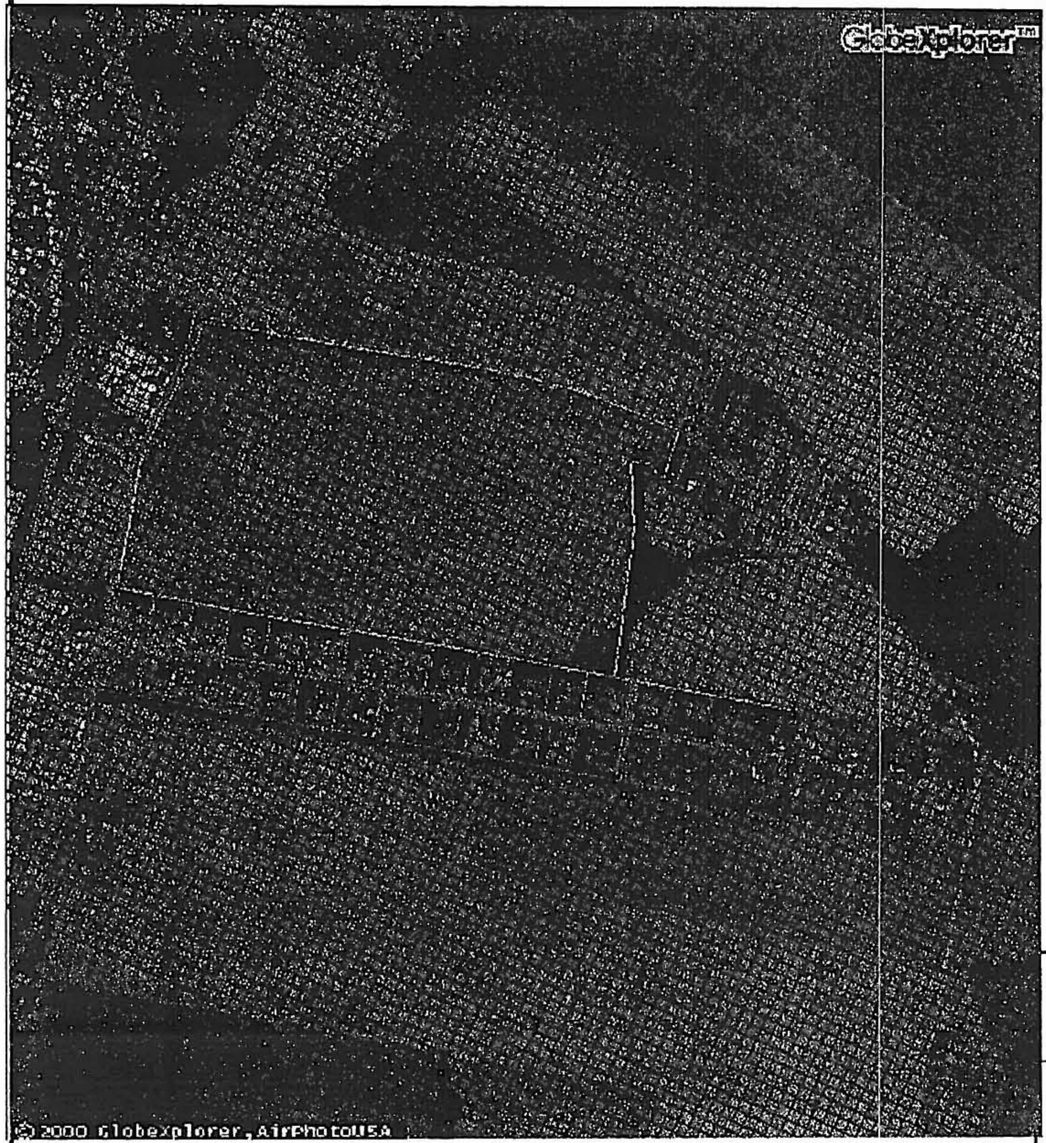
Figure 3-3
Groundwater Model Domain

URS



Groundwater Modeling & Risk Assessment
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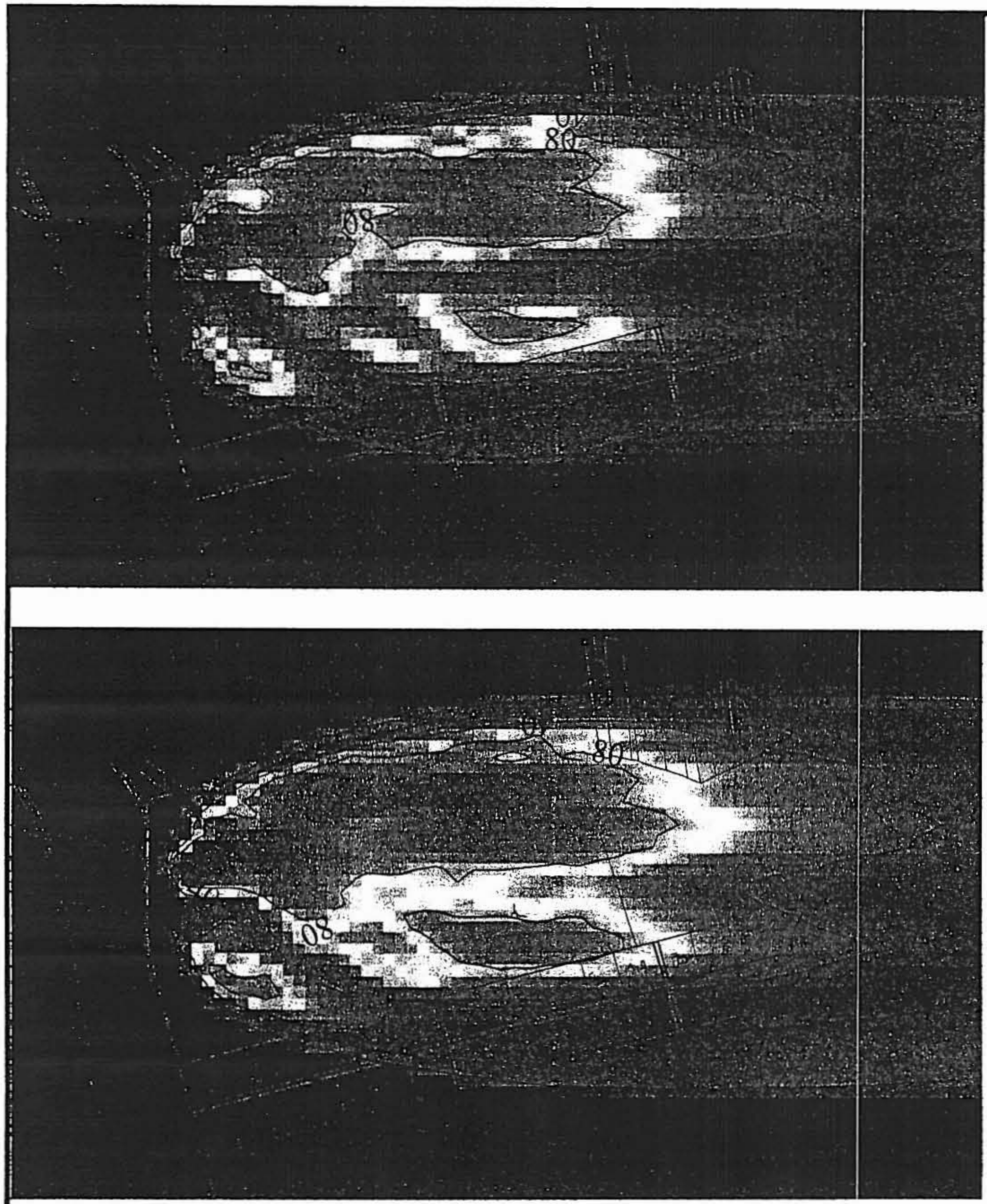
Figure 3-4
Aerial Photograph of Study Area with Finite Difference Grid

URS



Contours are of percent of the initial leachate concentration

Storage Coefficient of 0.15

Storage Coefficient of 0.30

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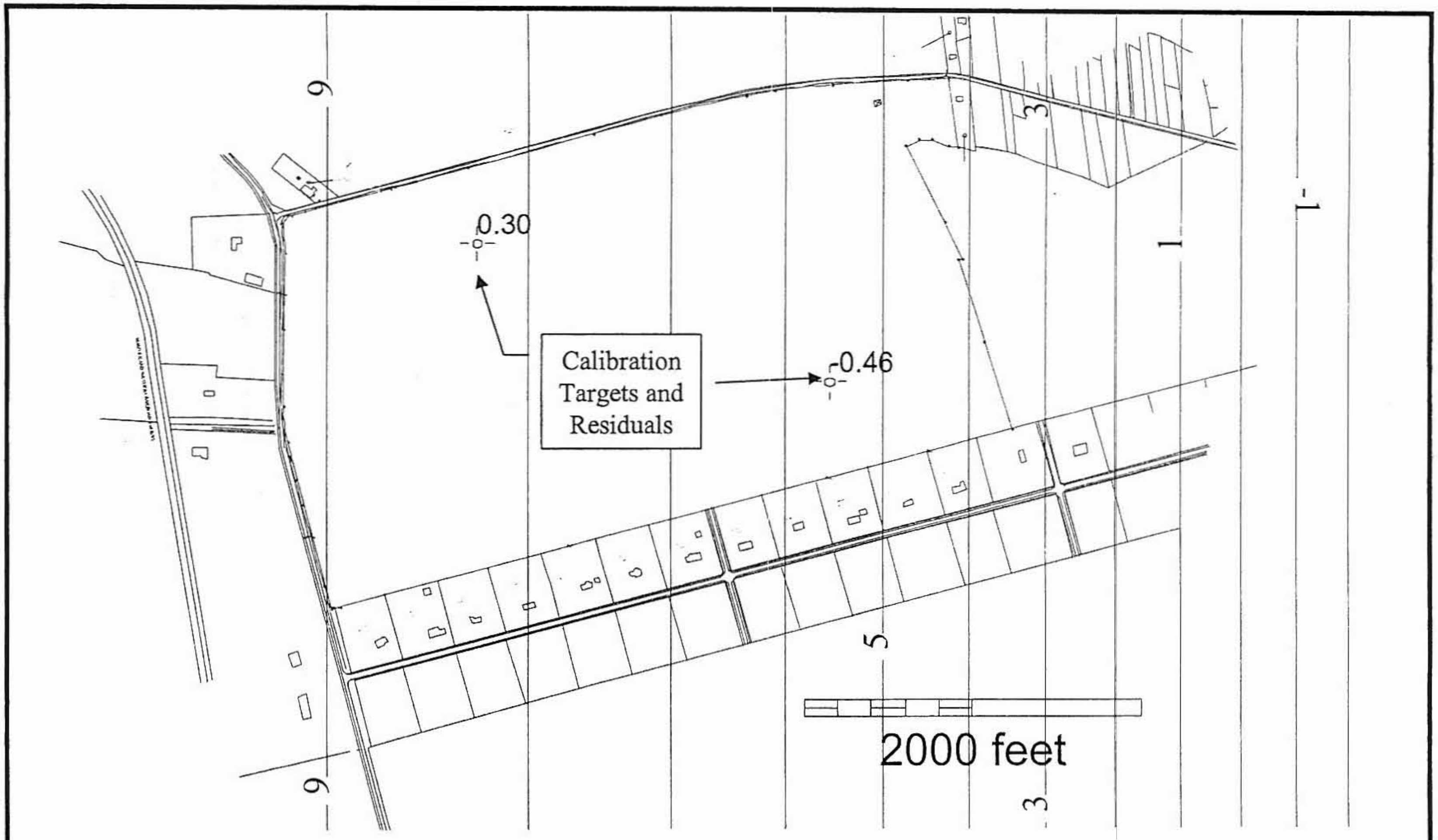
Figure 3-5

30-Year Preliminary Simulation
Sensitivity of Storage Coefficient Value

URS



Contours are of simulated groundwater surface in feet above an arbitrary elevation.

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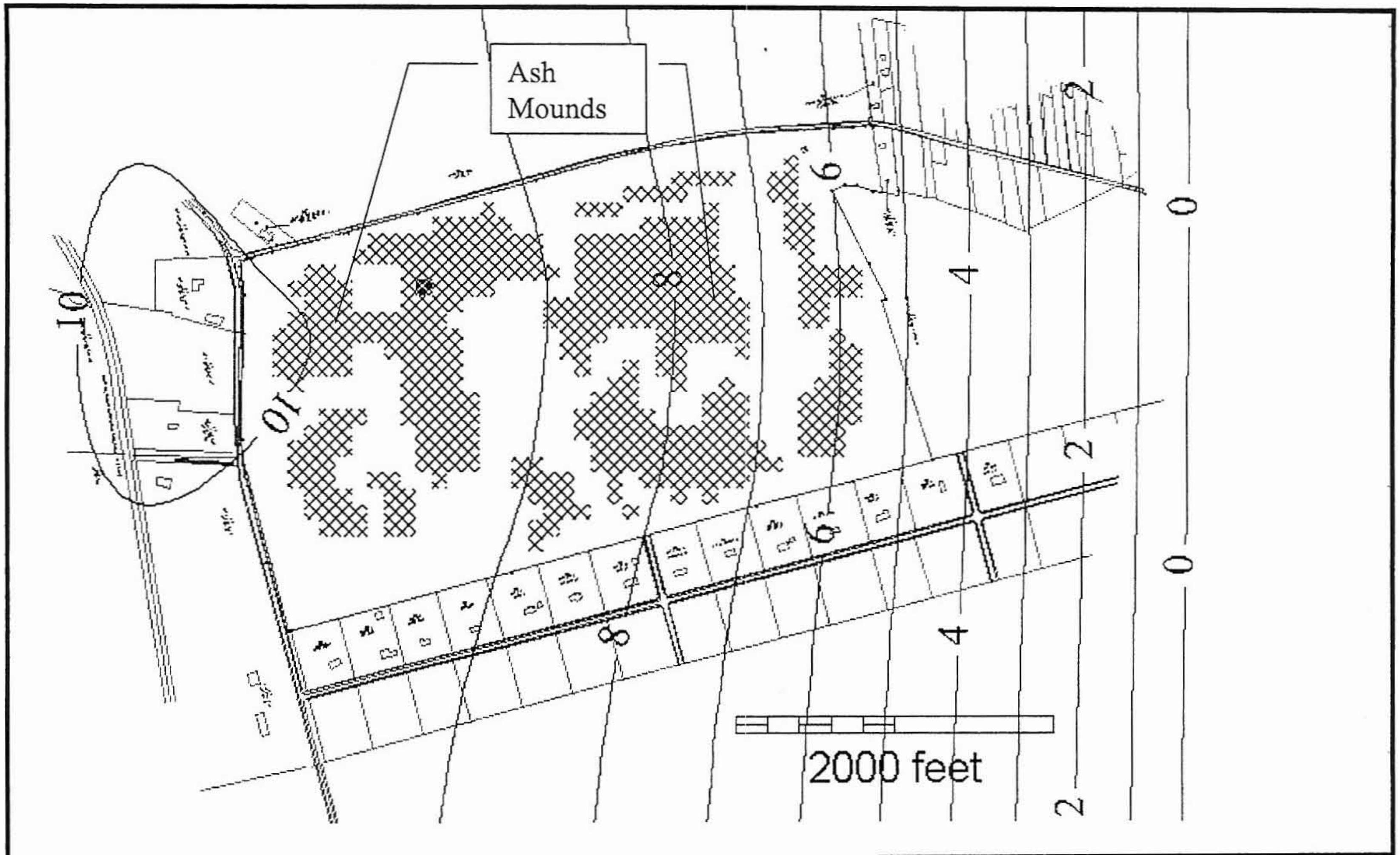
Figure 3-6
MODFLOW Simulated Groundwater Surface and Calibration Residual

URS



Contours are of simulated groundwater surface in feet above an arbitrary elevation.

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Figure 3-7
MODFLOW Simulated Ash Placement in Mounds - Groundwater Surface

URS

Contours are of percent initial leachate concentration

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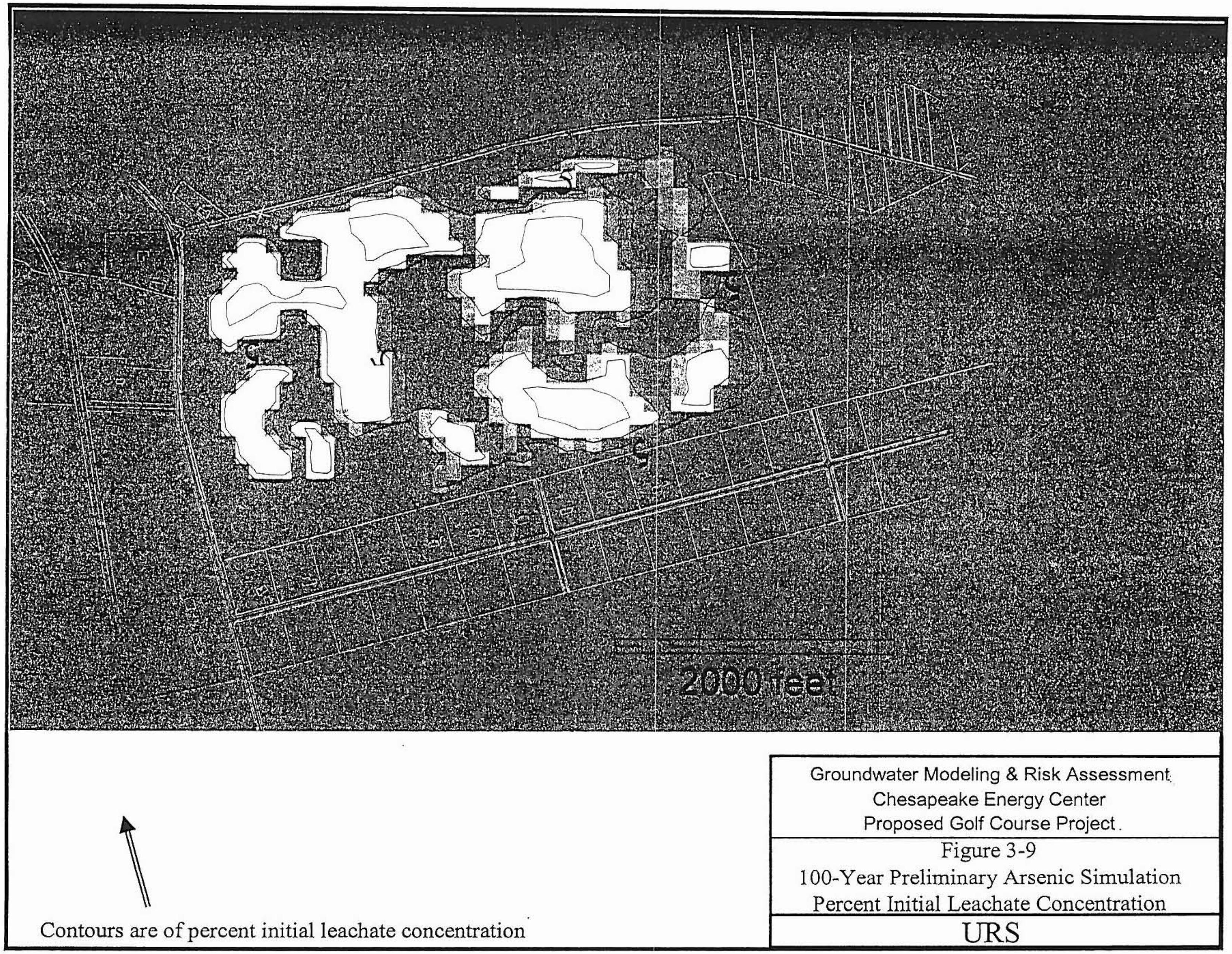
Figure 3-8
50-Year Preliminary Arsenic Simulation
Percent Initial Leachate Concentration

URS



Contours are of percent initial leachate concentration

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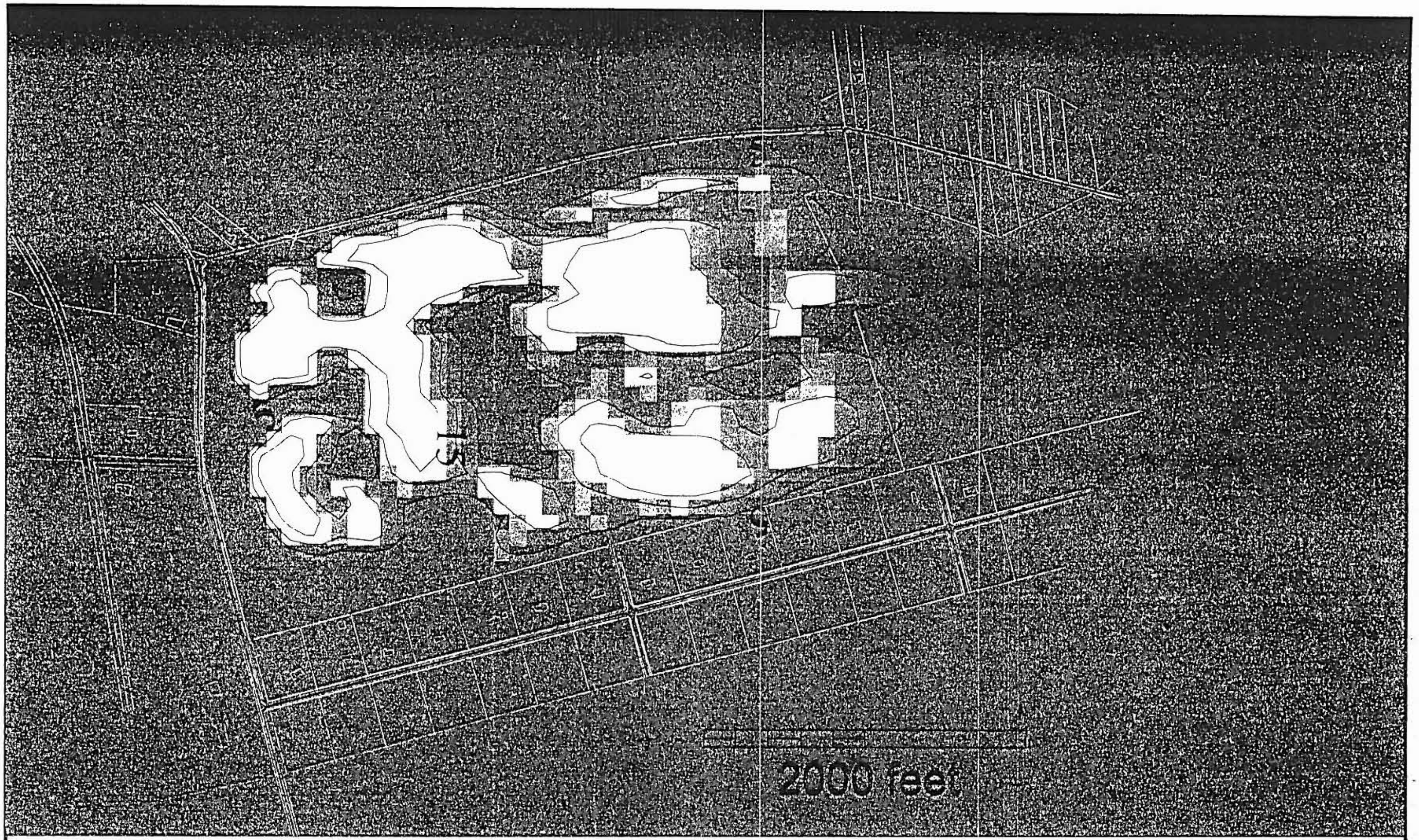
Figure 3-9
100-Year Preliminary Arsenic Simulation
Percent Initial Leachate Concentration

URS



Contours are of percent initial leachate concentration

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Figure 3-10
150-Year Preliminary Arsenic Simulation
Percent Initial Leachate Concentration

URS

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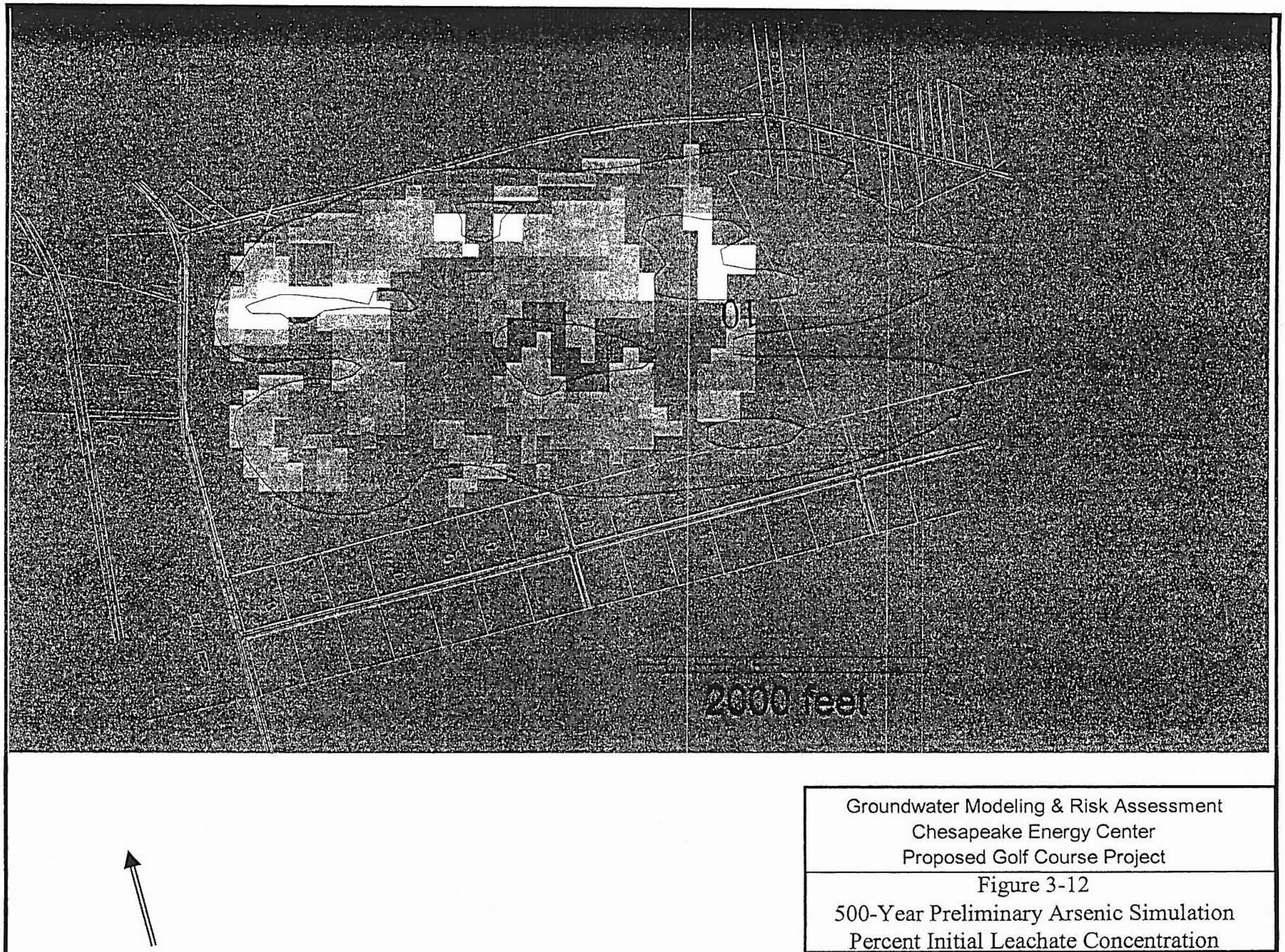
Figure 3-11
250-Year Preliminary Arsenic Simulation
Percent Initial Leachate Concentration

URS

Contours are of percent initial leachate concentration



Contours are of percent initial leachate concentration

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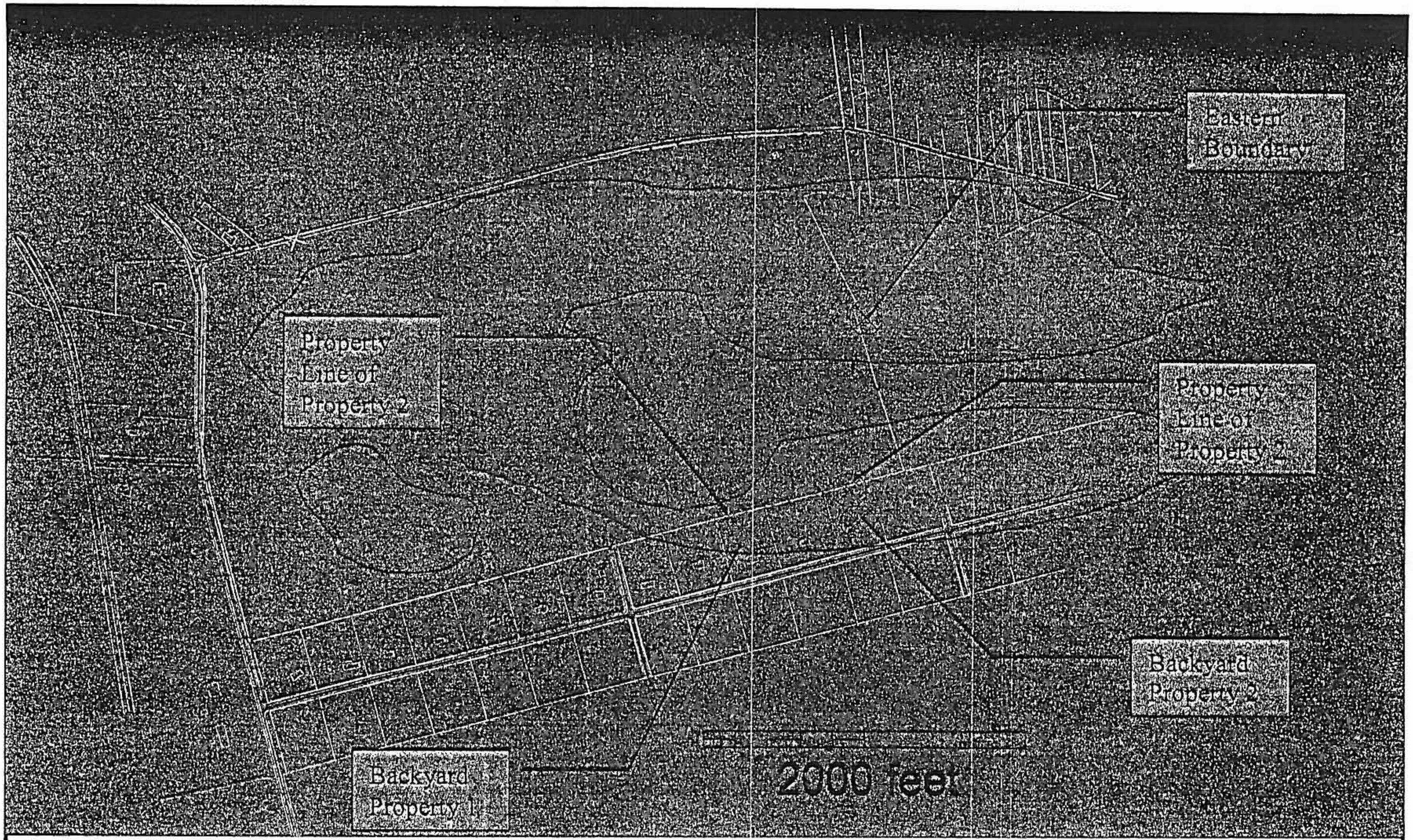
Figure 3-12
500-Year Preliminary Arsenic Simulation
Percent Initial Leachate Concentration

URS



Break-through curves for the above locations are shown on Figure 3-14.

Contours are of percent initial leachate concentration

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Figure 3-13
750-Year Preliminary Arsenic Simulation
Percent Initial Leachate Concentration

URS

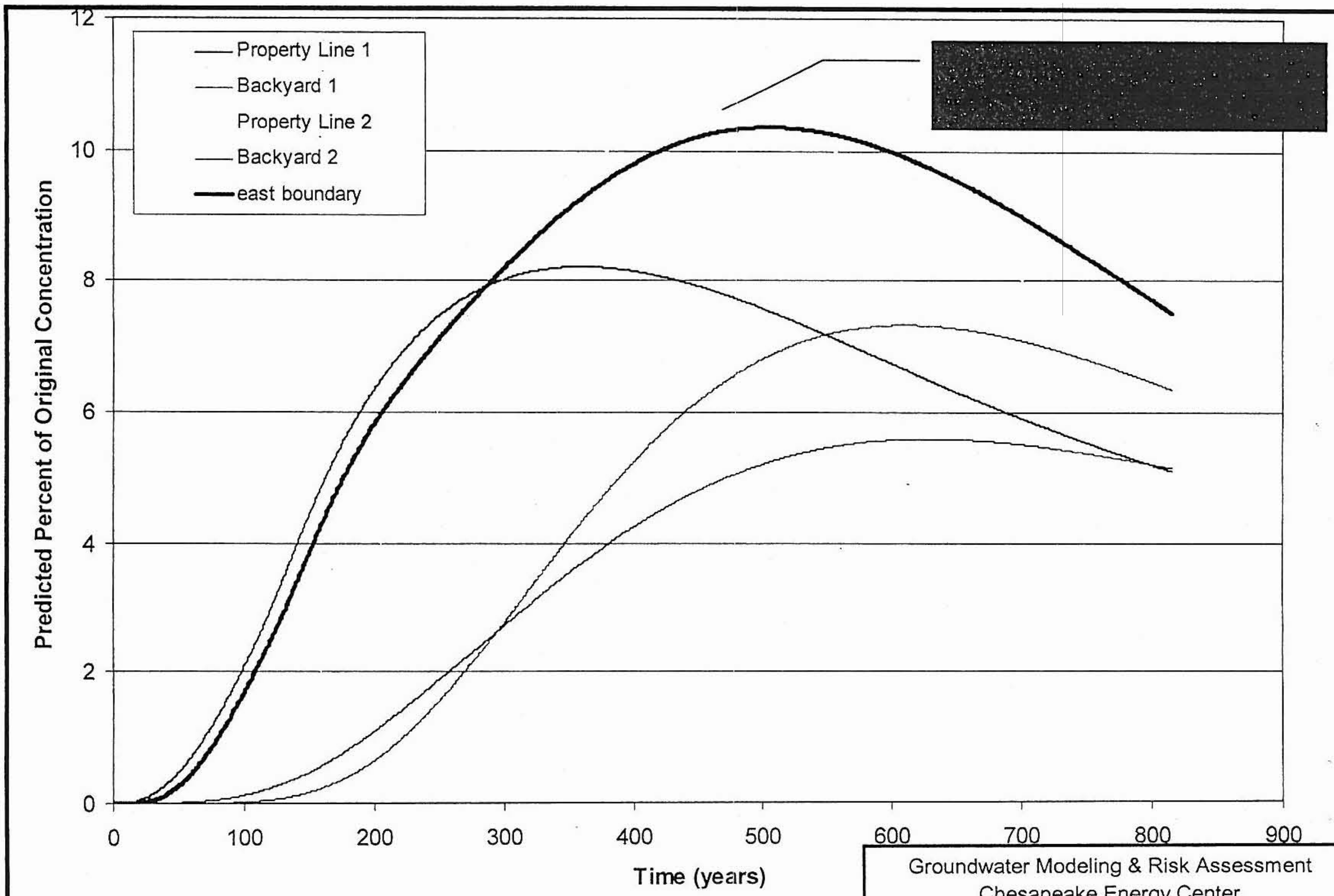

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Figure 3-14
Breakthrough Curve of Percent Initial
Leachate Concentration of Arsenic

URS

Contours are of percent initial leachate concentration

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Figure 3-15
50-Year Preliminary Selenium Simulation
Percent Initial Leachate Concentration

URS

Contours are of percent initial leachate concentration

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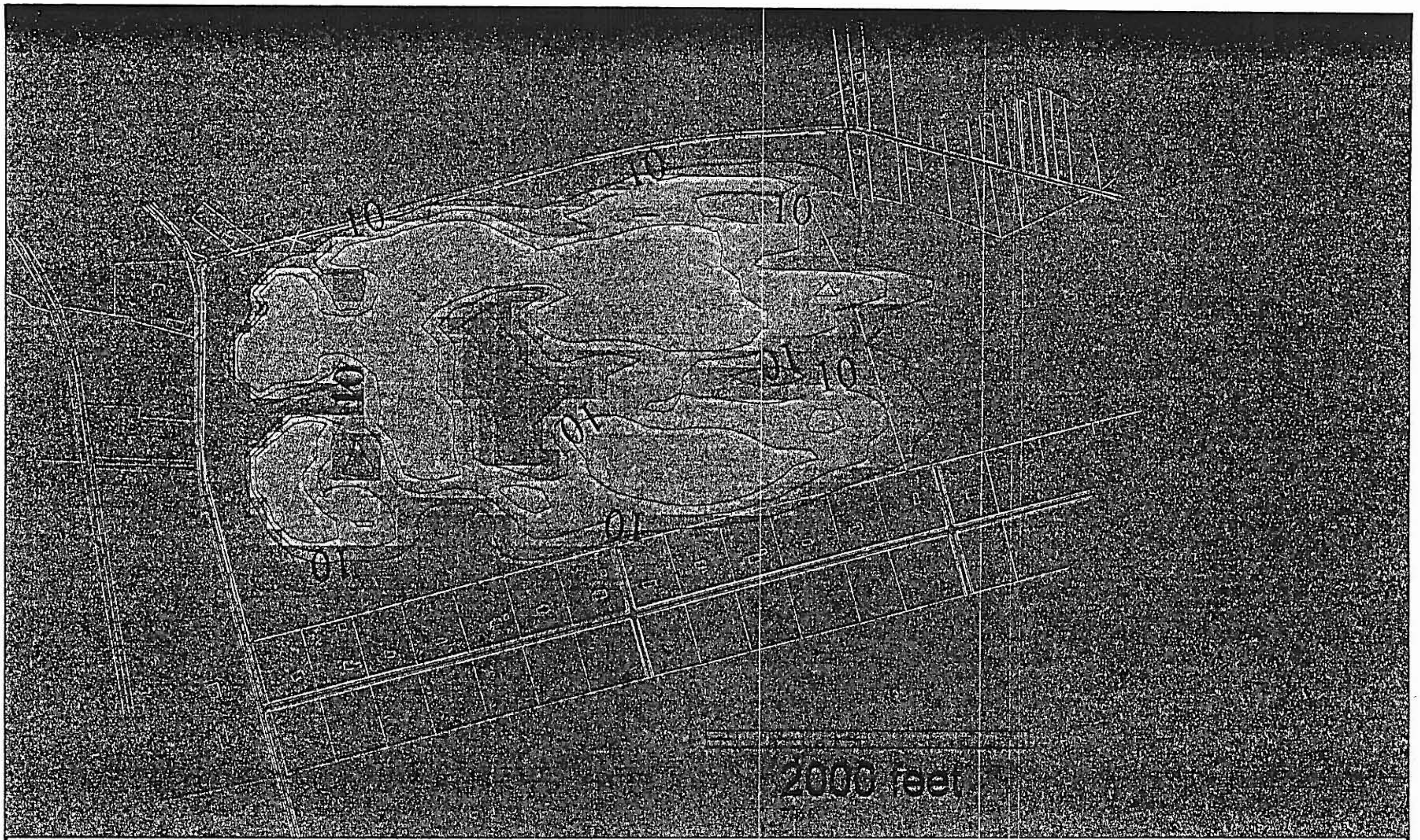
Figure 3-16
100-Year Preliminary Selenium Simulation
Percent Initial Leachate Concentration

URS



Contours are of percent initial leachate concentration

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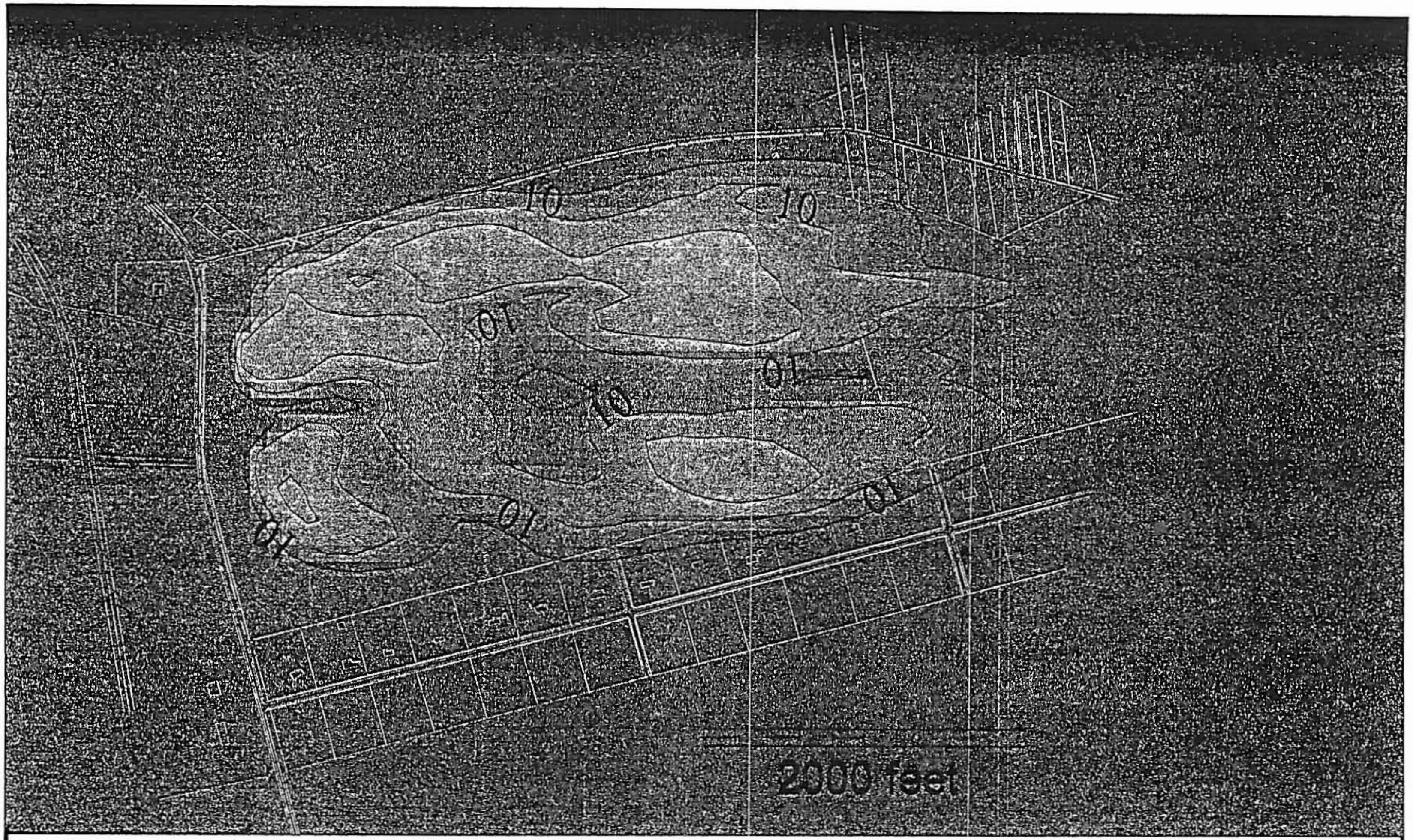
Figure 3-17
150-Year Preliminary Selenium Simulation
Percent Initial Leachate Concentration

URS



Contours are of percent initial leachate concentration

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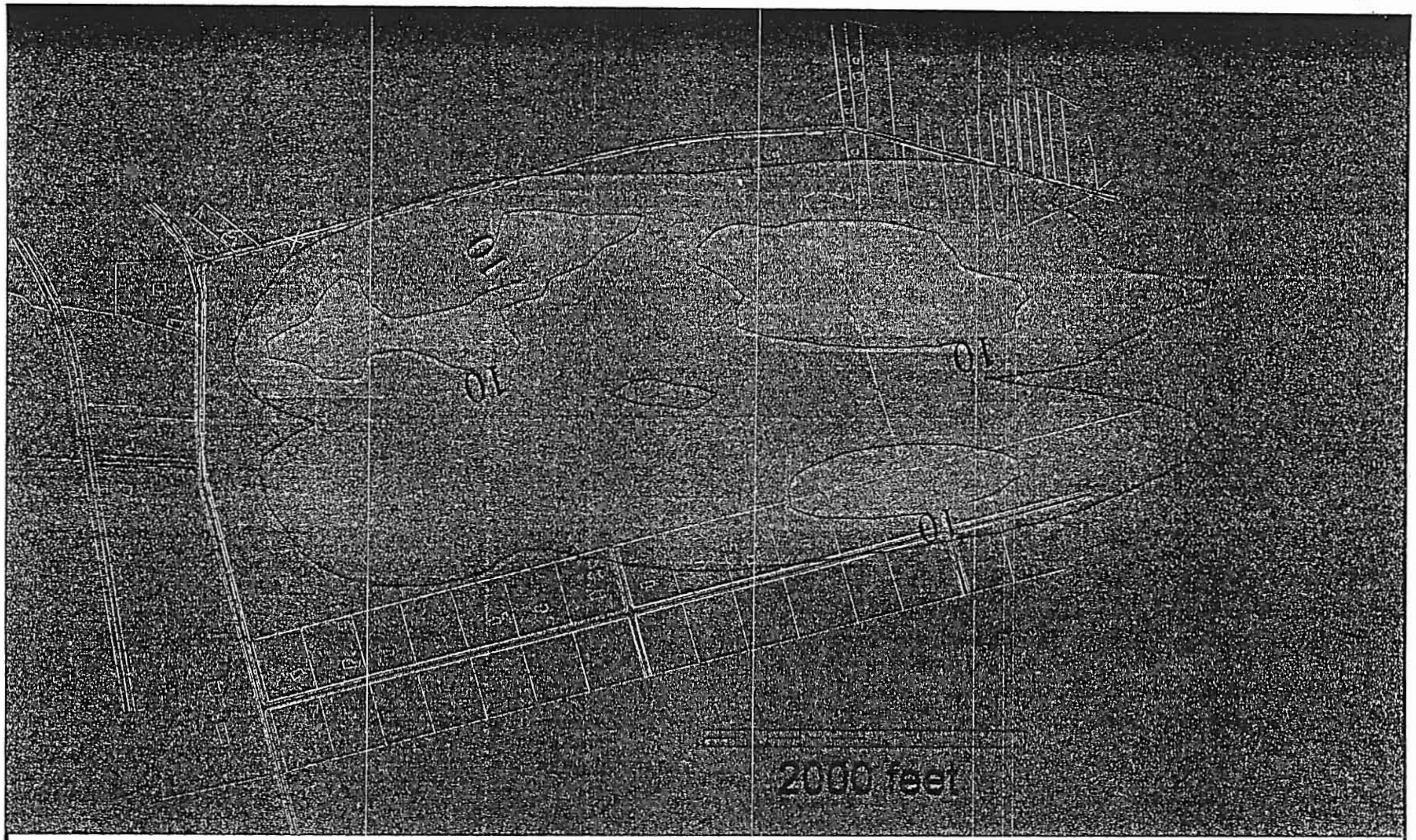
Figure 3-18
250-Year Preliminary Selenium Simulation
Percent Initial Leachate Concentration

URS



Contours are of percent initial leachate concentration

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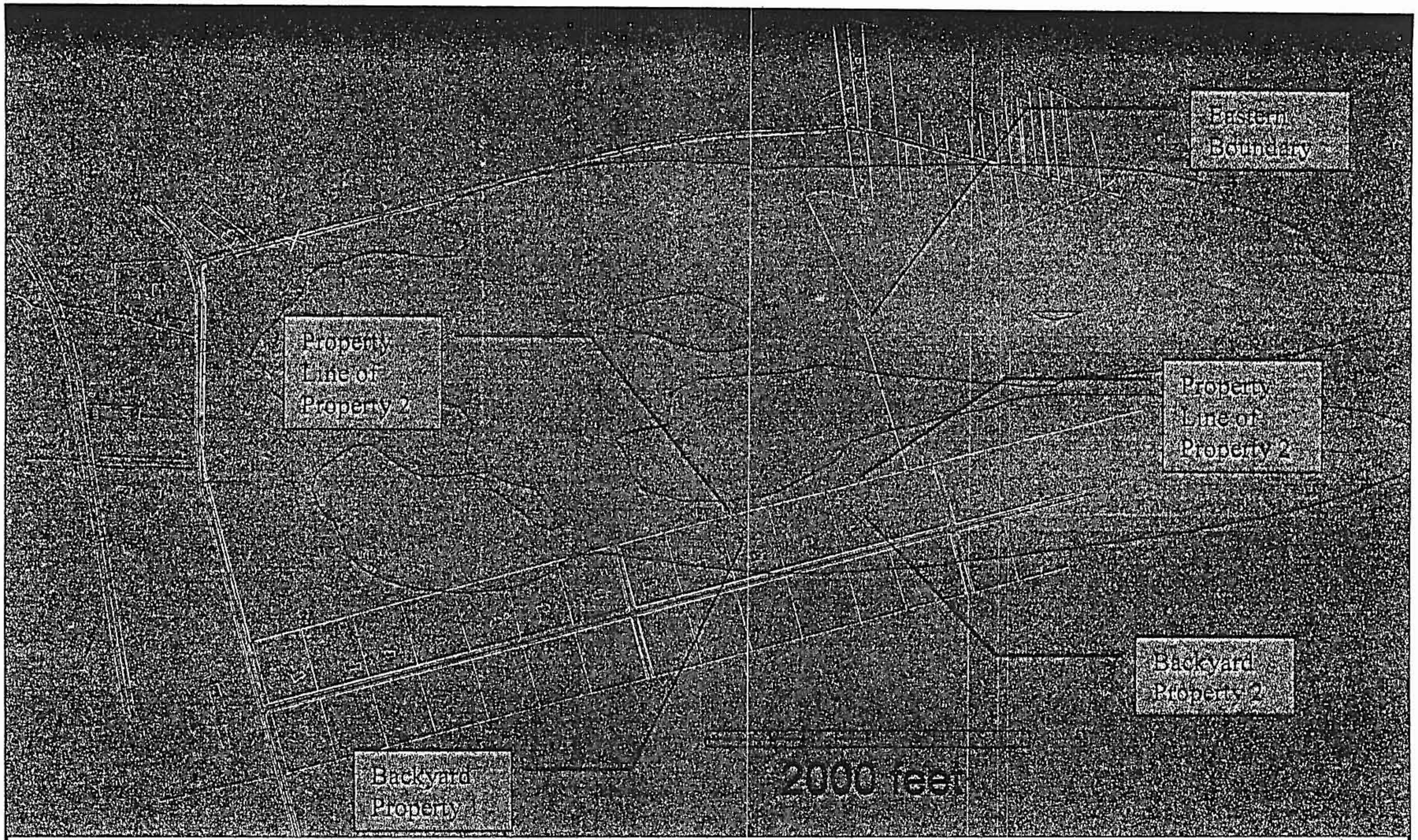
Figure 3-19
500-Year Preliminary Selenium Simulation
Percent Initial Leachate Concentration

URS



Break-through curves for the above locations are shown on the next figure.

Contours are of percent initial leachate concentration

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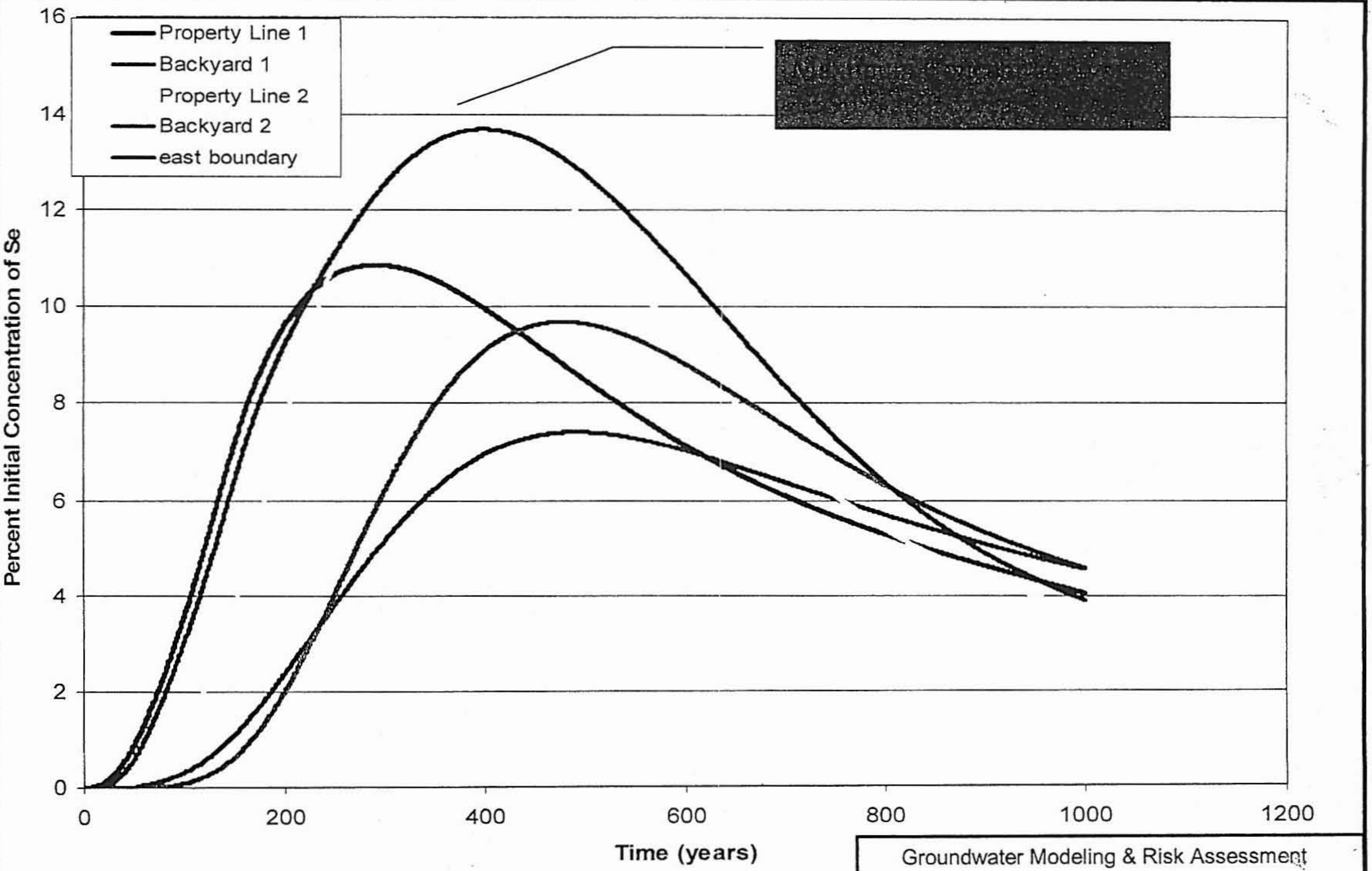
Figure 3-20
750-Year Preliminary Selenium Simulation
Percent Initial Leachate Concentration

URS



Contours are of percent initial leachate concentration

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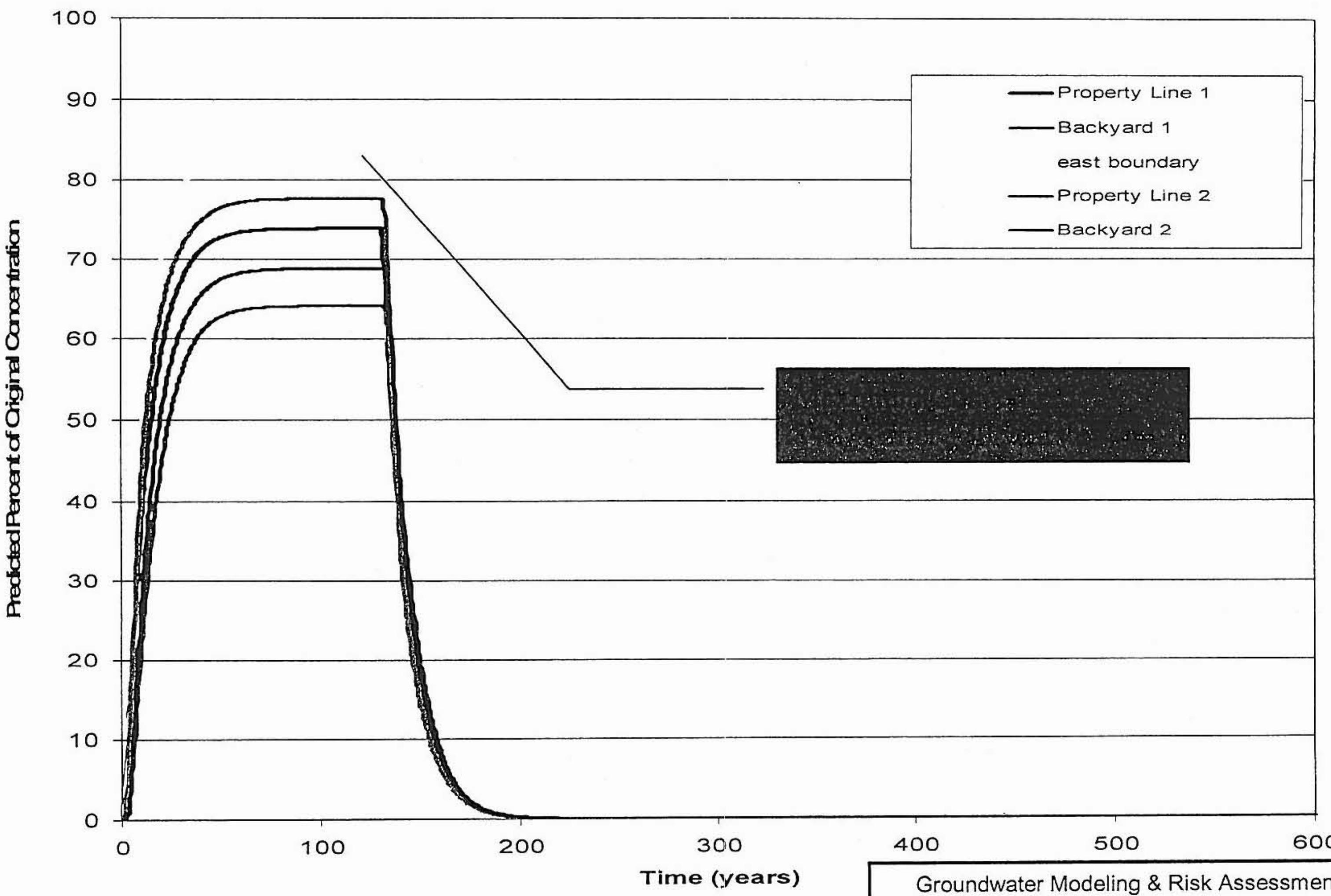
Figure 3-21
Breakthrough Curve of Percent Initial
Leachate Concentration of Selenium

URS



[redacted]

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Figure 3-22
Breakthrough Curve of Percent Initial
Leachate Concentration w/Kd=0, 130 yr Source

URS

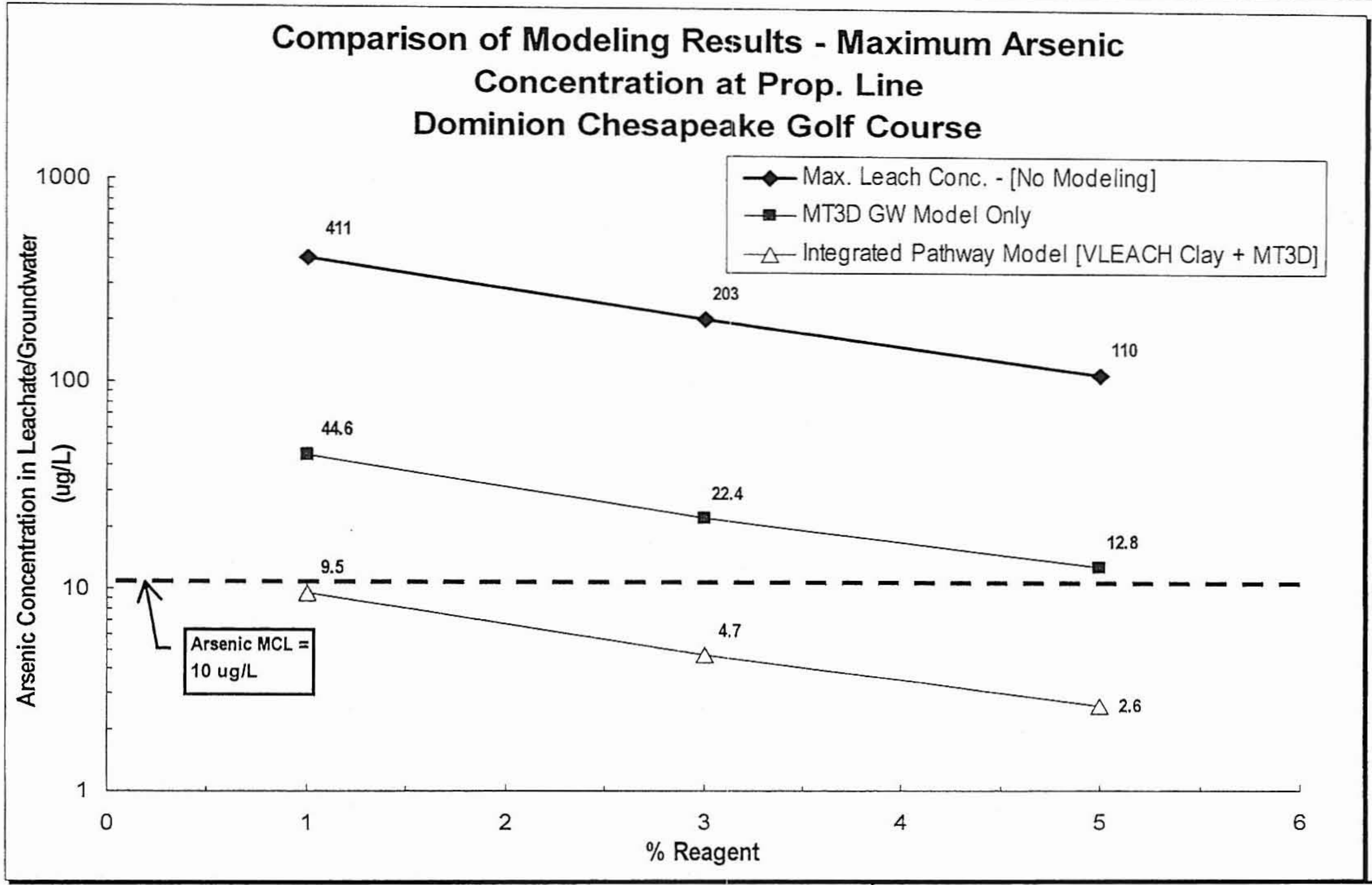

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Figure 3-23

URS

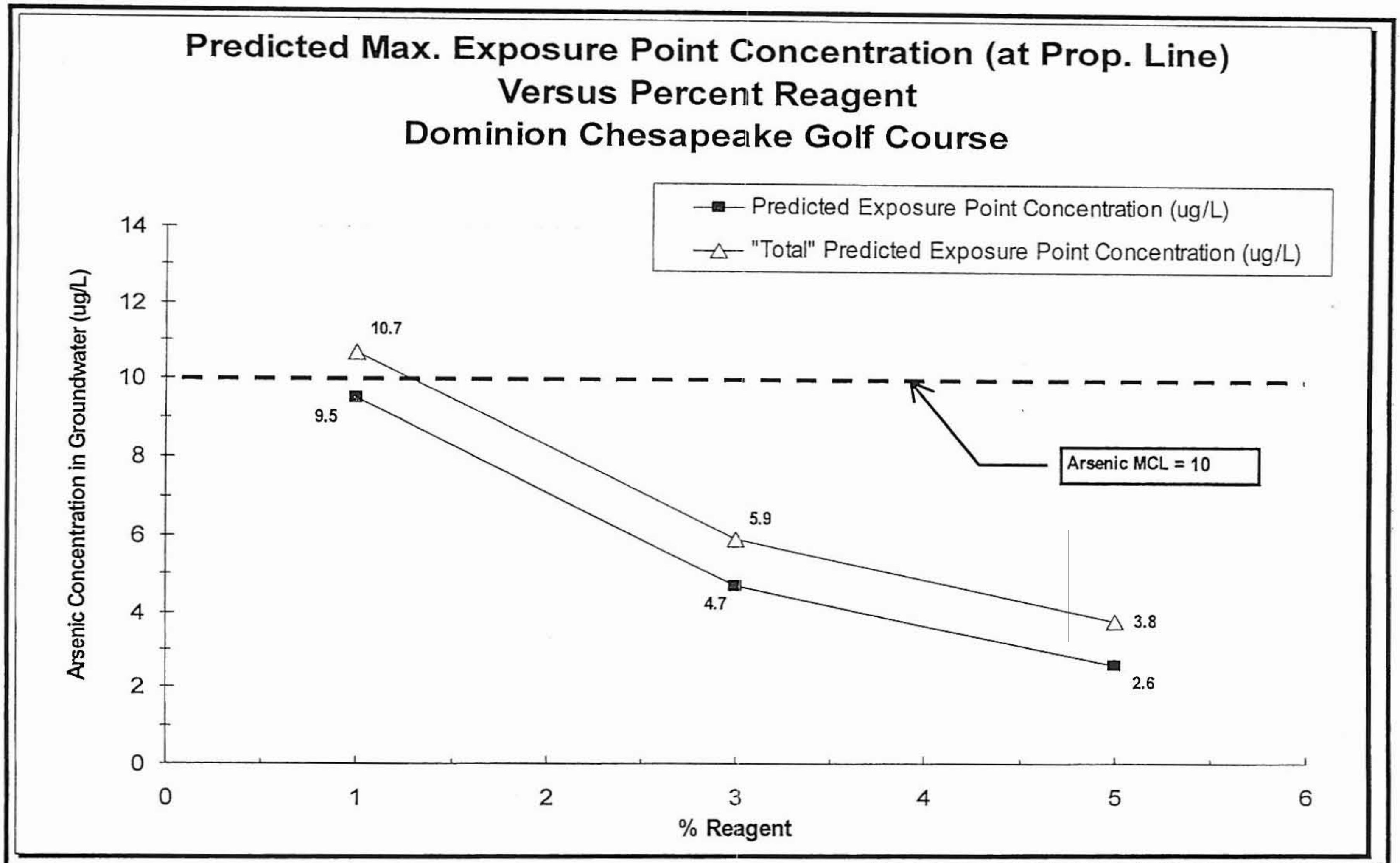

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Figure 4-1

URS

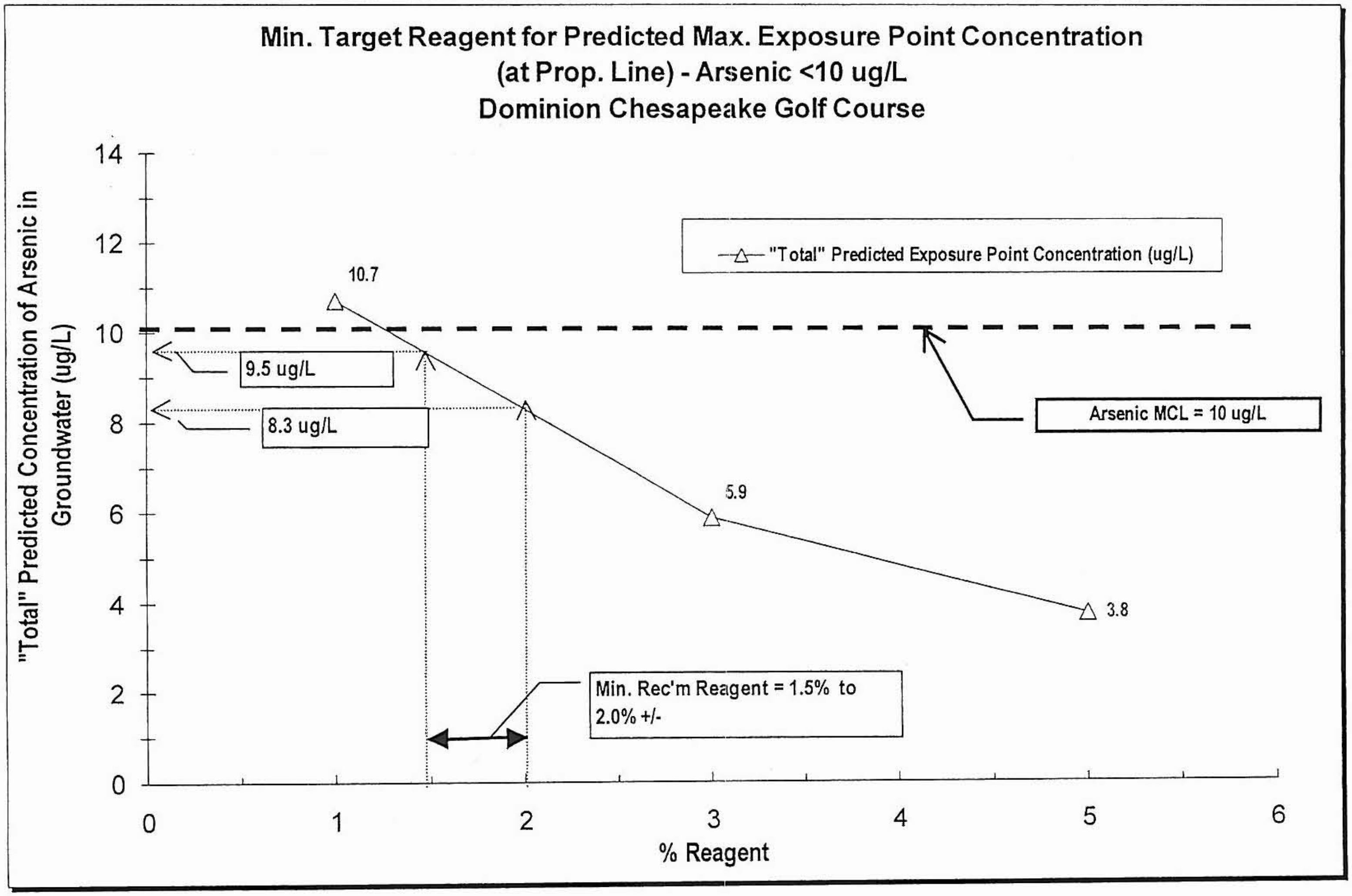

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Figure 5-1

URS