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GROUNDWATER REPORT VOLUME 2 OF 13 DECEMBER 1990

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REPORT

GROUNDWATER REPORT

FEED MATERIALS PRODUCTION CENTER
FERNALD, OHIO

REMEDIAL INVESTIGATION AND FEASIBILITY STUDY

VOLUME 2 OF 13

December 1990

U.S. DEPARTMENT OF ENERGY
OAK RIDGE OPERATIONS OFFICE

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PUBLIC ENVIRONMENTAL INFORMATION CENTER
10845 Hamilton Cleves Highway
Harrison, Ohio 45030
(513) 738-0164

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15.0 GROUNDWATER QUALITY

This section addresses the groundwater quality of the glacial overburden perched zones and the Great Miami Aquifer. The objective of this section is to determine which, if any, chemical constituents detected in groundwater can be related to FMPC activities. This is accomplished primarily by the statistical analysis of FMPC groundwater chemistry data.

The section begins with a review of the hydrogeological features of the FMPC area that are critical to the evaluation of groundwater quality. Background water quality is established and the methodology for the analysis of groundwater quality data is developed. Finally, a detailed discussion of the results of the groundwater quality analysis is presented.

Groundwater quality data collected during RI/FS sampling and discussed in this section are presented in Appendix E. These data were collected during RI/FS sampling from the period March 1988 through August 1989. Also included in Appendix E are groundwater quality data collected from June 1989 through January 1990 as part of the ongoing RCRA compliance program.

Due to the low percentage of detected organic compounds, statistical analysis of organic-compound data was not performed. To supplement RI/FS sampling results for organic compounds, the results of RCRA sampling for organic compounds are also presented. The RCRA data were collected from August 1985 through December 1987 and from June 1989 through January 1990. Also, to supplement limited RI/FS and RCRA data for aluminum, vanadium and zinc, groundwater concentrations of these metals are compared with background data from the Air, Soil, Water, and Health Risk Assessment Report in the vicinity of the FMPC (IT 1986).

15.1 HYDROGEOLOGIC FRAMEWORK FOR GROUNDWATER QUALITY ANALYSIS

The analysis of groundwater quality must begin by understanding the hydrogeology under the FMPC and vicinity. Groundwater flow directions must be known in order to examine the relationship between upgradient water quality, contaminant sources, and downgradient water quality. Regional hydrogeology is discussed in detail in Section 12.0. What follows in this section is a discussion of hydrogeologic features that are critical to the analysis of groundwater quality.

15.1.1 Glacial Overburden

As discussed in Section 12.2, perched groundwater zones are isolated and not interconnected in the vicinity of the FMPC reservation. Thus, by the nature of the perched zones in the glacial overburden, groundwater defined as upgradient under the strictest interpretation may not exist to

establish background water quality. Nevertheless, wells that are considered to be representative of typical water quality for perched groundwater have been selected as background wells. These are addressed in Section 15.2.1.1.

15.1.2 Great Miami Aquifer

Three flow systems of the Great Miami Aquifer converge in the vicinity of the FMPC reservation. As shown in Figures 15-1 and 15-2, groundwater in the Dry Fork Section of the New Haven Trough generally flows from west to east. Groundwater in the Shandon Tributary to the New Haven Trough generally flows to the southeast, and groundwater in the Ross Section of the New Haven Trough generally flows to the southwest. Figures 15-1 and 15-2 show that a flow divide is located in the southern portion of the FMPC and separates Dry Fork Section groundwater from Shandon Tributary groundwater. The location of the divide fluctuates, depending on flow conditions; therefore, mixing occurs along the divide.

Groundwater from the Ross Section does not enter the FMPC. A flow divide separating the Ross Section groundwater from Shandon Tributary groundwater is located east of the FMPC, as shown in Figures 15-1 and 15-2. This divide is influenced by pumping of the collector wells located within and near the "big bend" in the Great Miami River.

The two groundwater flow divides that separate these three flow systems were delineated using the FMPC RI/FS groundwater flow model and the particle tracking program, STLINE. A sensitivity analysis using a range of groundwater flow conditions showed that the location of the flow divides could fluctuate for average precipitation conditions up to 750 feet on either side of the divides shown in Figures 15-1 and 15-2.

Due to the hydrogeological configuration in the vicinity of the FMPC, wells are grouped prior to the analysis of water quality based on where they are located with respect to the groundwater flow divides shown in Figures 15-1 and 15-2.

15.2 BACKGROUND CONDITIONS

This section describes background groundwater quality and the rationale for grouping background wells based on groundwater flow.

15.2.1 Background Well Selection

Background wells were selected to establish upgradient groundwater quality that is not influenced by operations at the FMPC. Wells designated as background were sampled from the second quarter

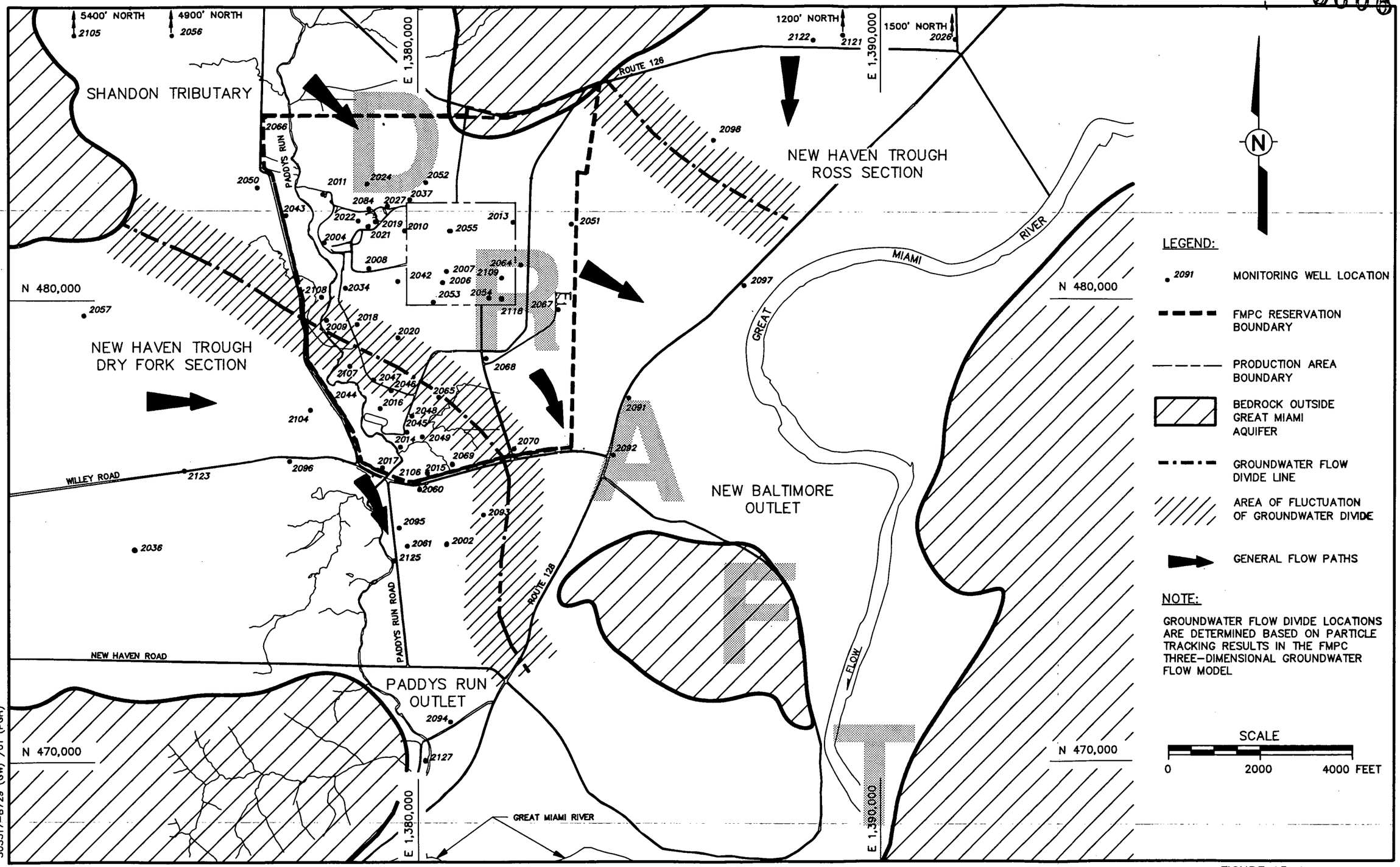
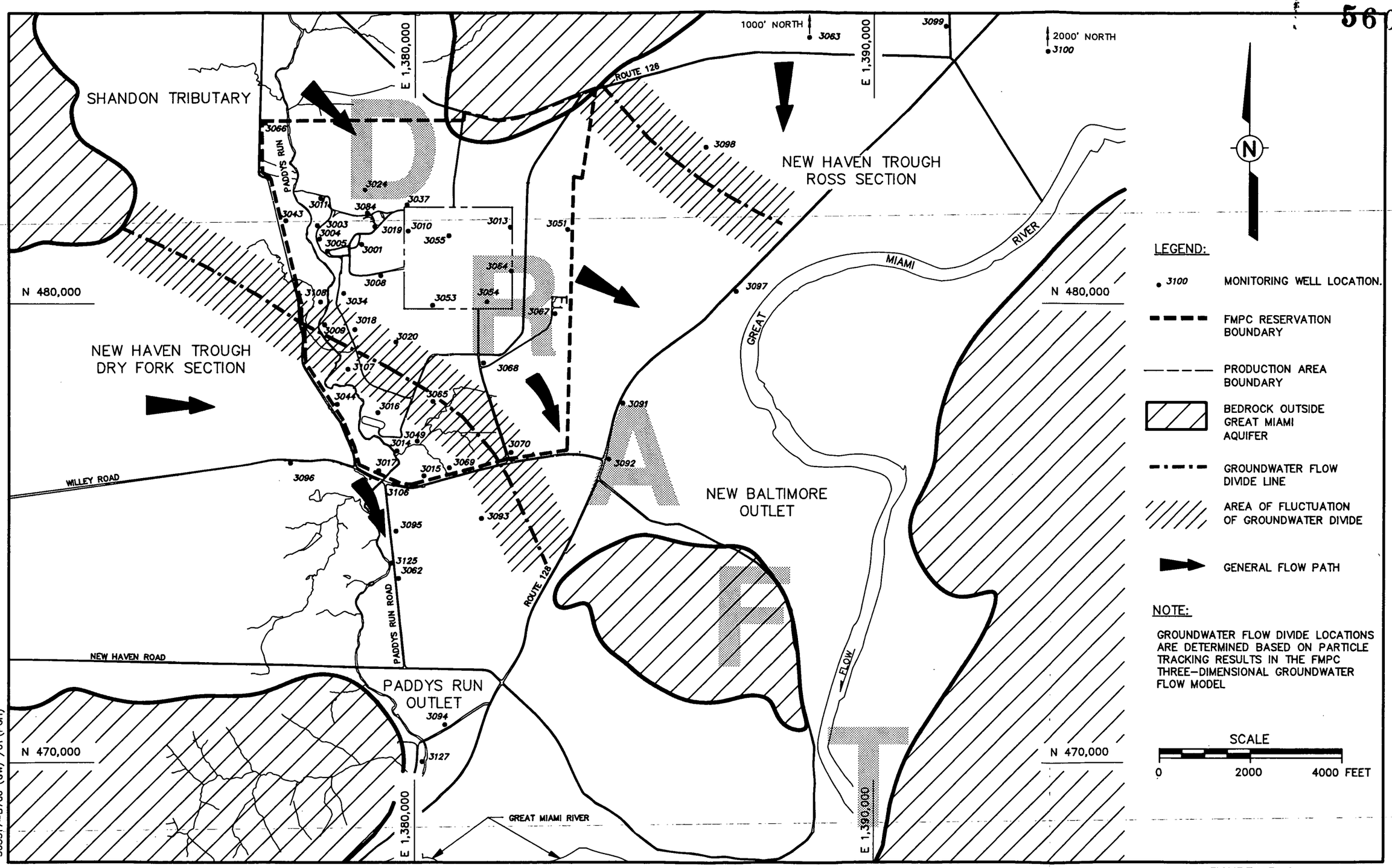


FIGURE 15-1
GROUNDWATER FLOW DIVIDES
2000 SERIES WELLS

303317-B729 (GW) /01,100% BLUE
303317-B729 (GW) /01 (PGH)



LEGEND:

- 3100 — MONITORING WELL LOCATION.
- FMPC RESERVATION BOUNDARY
- PRODUCTION AREA BOUNDARY
- BEDROCK OUTSIDE GREAT MIAMI AQUIFER
- GROUNDWATER FLOW DIVIDE LINE
- AREA OF FLUCTUATION OF GROUNDWATER DIVIDE
- GENERAL FLOW PATH

NOTE:

GROUNDWATER FLOW DIVIDE LOCATIONS ARE DETERMINED BASED ON PARTICLE TRACKING RESULTS IN THE FMPC THREE-DIMENSIONAL GROUNDWATER FLOW MODEL

SCALE

0 2000 4000 FEET

FIGURE 15-2
GROUNDWATER FLOW DIVIDES
3000 SERIES WELLS

303317-B706 (GW) /01 100% BLUE
303317-B706 (GW) /01 (PGH)

1988 through the first quarter 1989 as part of the RI and some were sampled from June 1989 through January 1990 as part of the ongoing RCRA compliance program. Thus, the average values reported in this section are based on at least four quarterly samples from each well.

15.2.1.1 Glacial Overburden

Wells 1024, 1059, 1060, and 1065 are designated as background wells for glacial overburden groundwater because they are located in areas that are not likely to have been impacted by FMPC activities (refer to Figure 6-2 for locations). Due to the isolated nature of perched groundwater zones in the glacial overburden (Section 12.2), these wells are not truly upgradient wells. However, it is assumed that the groundwater chemistry data from these wells is representative of typical groundwater quality in glacial overburden not affected by FMPC releases.

15.2.1.2 Great Miami Aquifer

As discussed in Section 15.2.2, the chemistry of groundwater from each flow system in the Great Miami Aquifer is distinctly different from that of the other flow systems. Because of the heterogeneity of upgradient groundwater chemistry, it is not possible to derive a single representative background water chemistry for the Great Miami Aquifer groundwater. Thus, the background wells are divided into Ross, Dry Fork, and Shandon background groups before the analysis of groundwater quality proceeds.

Wells 2026, 2121, 2122, 3063, 3099, and 3100 are designated as background wells for groundwater from the Ross Section of the Great Miami Aquifer. These wells, which are referred to as Ross background wells, are located upgradient of the FMPC and are shown in Figures 15-1 and 15-2. All of these wells are private wells and may be used for drinking water supply.

Wells 2036, 2057, and 2123 are designated as background wells for groundwater from the Dry Fork Section of the Great Miami Aquifer. These wells, which are referred to as Dry Fork background wells, are located upgradient of the FMPC and are shown in Figure 15-1. These wells are also private wells, and probably are used for drinking water supply. Note that Wells 2096, 3096, 4096, and 2104 also appear to be upgradient of the FMPC. Well 4096 is located adjacent to Wells 2096 and 3096. However, as discussed in Section 12.5.2, a seasonal groundwater mound develops beneath Paddys Run during periods of high precipitation. While the mound persists, groundwater may flow toward the west, and potentially carry contaminated water toward Wells 2096, 3096, 4096, and 2104. The mounding is a transient effect; but, because the magnitude of the impact is not fully understood, these wells are not designated as background.

Wells 2050, 2056, 2066, and 2105 are designated as background wells for groundwater from the Shandon Tributary of the Great Miami Aquifer. These wells, which are referred to as Shandon background wells, are located upgradient of the FMPC and are shown in Figure 15-1. Well 2066 is a monitoring well installed during the RI/FS program. The other wells are private wells, and all but 2050 may be used for drinking water supply. Well 2050 is a water supply well for a light industry. Well 3066 is not included in the Shandon background well group. The groundwater chemistry of samples from this well is anomalous compared with that of samples from other Shandon background wells. A portion of the groundwater sampled at Well 3066 may flow through bedrock and acquire the anomalous chemical characteristics. Alternatively, the chemistry may be influenced by the greater percentage of clay minerals in the Shandon Tributary, relative to the rest of the aquifer.

15.2.2 Background Groundwater Quality

Groundwater from glacial overburden and the Great Miami Aquifer is classified as fresh water since total dissolved solids (TDS) generally range from 500 to 700 milligrams per liter (mg/L). In general, calcium is the dominant cationic constituent, and bicarbonate is the dominant anionic constituent. Generally, background groundwater is saturated with respect to calcium carbonate.

15.2.2.1 Glacial Overburden

Table 15-1 lists radionuclide, metal, and general chemical concentrations observed in groundwater samples from glacial overburden background wells. The concentration range of most constituents is greater compared with groundwater from the Great Miami Aquifer. This is most likely due to the diverse mineralogy of glacial overburden materials and the interreaction between these materials and groundwater. Calcium, followed by magnesium, sodium, potassium, and iron, are the dominant metals found in glacial overburden groundwater. Trace concentrations of several other metals are also observed. Bicarbonate, followed by sulfate and chloride, are dominant anionic constituents. Nitrate and ammonia concentrations are low in glacial overburden groundwater. The pH of glacial overburden groundwater is neutral; the average pH is 7.2. The average TDS concentration is 665 mg/L.

Some radionuclides are shown to be naturally present at detectable, but low activity concentrations. In particular, radium, thorium, and uranium were observed at detectable levels.

15.2.2.2 Great Miami Aquifer

Ross and Dry Fork Section groundwater chemistries are similar; however, Shandon Tributary groundwater chemistry is quite different from that of the other parts of the aquifer. The

TABLE 15-1
GROUNDWATER QUALITY, GLACIAL OVERBURDEN BACKGROUND WELLS

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Parameter	GLACIAL OVERBURDEN BACKGROUND Average[a]	Standard Deviation[a]	Range[a]
Radionuclides (pCi/L)			
Cs-137	20 U	-	20 U - 20 U
Np-237	1 U	-	1 U - 1 U
Pu-238	1 U	-	1 U - 1 U
Pu-239/240	1 U	-	1 U - 1 U
Ra-226	1 U	-	1 U - 1 U
Ra-228	3.2	0.7	3 U - 5.2
Ru-106	150 U	-	150 U - 150 U
Sr-90	5 U	-	5 U - 5 U
Tc-99	30 U	-	30 U - 30 U
Th-228	1.1	0.2	1 U - 1.6
Th-230	1.1	0.3	1 U - 2
Th-232	1 U	-	1 U - 1 U
Th-Total[b]	13 U	-	13 U - 13 U
U-234	1.1	0.3	1 U - 1.9
U-235/236	1 U	-	1 U - 1 U
U-238	1.0	0.1	1 U - 1.5
U-Total[b]	1.6	1.2	1 U - 5.3
Metals (mg/L)			
Al	0.12	0.00	0.1206 - 0.123
As	0.002	0.00	0.002 U - 0.003
Ba	0.069	0.022	0.05 U - 0.112
Cd	0.003	0.002	0.002 U - 0.007
Ca	96.38	17.41	74.4 - 130
Cr	0.023	0.005	0.02 U - 0.0345
Cu	0.016	0.010	0.01 U - 0.044
Fe	0.689	1.345	0.005 - 4.9
Pb	0.003	0.001	0.002 U - 0.006
Mg	34.28	9.39	20.4 - 47.8
Mn	0.044	0.038	0.003 - 0.1
Hg[b]	0.22	0.06	0.2 U - 0.4
Mo	0.02 U	-	0.02 U - 0.02 U
Ni	0.02	0.00	0.02 U - 0.026
K	8.90	9.07	0.891 - 31.5
Se	0.002 U	-	0.002 U - 0.002 U

See footnotes at end of table.

TABLE 15-1
(continued)

GLACIAL OVERBURDEN BACKGROUND					
Parameter	Average[a]	Standard Deviation[a]	Range[a]		
Metals (Continued)					
Ag	0.015	0.013	0.01 U	-	0.052
Na	21.33	16.80	5.71	-	56.3
V	0.02	0.00	0.018	-	0.0195
Zn	0.04	0.01	0.0317	-	0.042
General Chemistry (mg/L)					
Ammonia as N	0.18	0.17	0.1 U	-	0.58
Chloride	12.19	12.20	1.4	-	35.7
Conductance[c]	640.3	157.6	470	-	975
Fluoride	0.62	0.34	0.22	-	1.3
Nitrate as N	0.15	0.07	0.1 U	-	0.3
Total phosphorus	0.07	0.04	0.05 U	-	0.18
Sulfate	71.15	49.52	3.24	-	175
Total organic halide	0.05	0.01	0.05 U	-	0.067
Alkalinity as HCO3	416.3	49.7	334	-	501.1
pH	7.2	-	7	-	7.4
Total dissolved solids[d]	663.40	76.35	441.35	-	986.89

[a]Outliers and inconsistent MDLs are excluded. Nondetects are set equal to the MDLs and included in the calculation of averages and standard deviations (Section 15.3.2.2).

U = below the MDL shown.

[b]In ug/L.

[c]In umhos/cm.

[d]TDS is calculated based on following assumptions:

The mass contributed by radionuclides is negligible.
Phosphorus is present as phosphate.

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background groundwater chemistry of each section of the Great Miami Aquifer is discussed separately below.

Ross Section

Table 15-2 lists radionuclide, metal, and general chemical concentrations observed in groundwater samples from Ross background wells. Since both 2000 and 3000 Series wells are present in the Ross background area, Ross background water quality data are separated in Table 15-2 into Ross background from 2000 Series wells and Ross background from 2000 and 3000 Series wells. Calcium, followed by magnesium, sodium, and potassium, are the dominant metals found in Ross groundwater. Trace concentrations of many other metals were also detected. Bicarbonate, followed by sulfate, chloride, and nitrate, are the dominant anionic constituents. The pH of Ross groundwater is neutral; the average pH is 7.1. The average TDS concentration is 544 mg/L.

Low activity concentrations of radium-226 (Ra-226), radium-228 (Ra-228), thorium-228 (Th-230), and thorium-230 were detected in Ross background groundwater. Low concentrations of total uranium were also detected. These radionuclides are naturally occurring, but their presence may also be due to contaminant sources upgradient of the Ross area. For unknown reasons, the average arsenic concentration, 0.106 mg/L, calculated for Ross groundwater is high relative to the average arsenic concentrations of other background groundwater. Background concentrations for aluminum, vanadium, and zinc for the Ross section groundwater could not be established because of the absence of data.

Dry Fork Section

Table 15-3 lists the average and range of radionuclide, metal, and general chemical concentrations observed in groundwater samples from Dry Fork background wells. Calcium, followed by magnesium, sodium, and potassium, are the dominant metals found in Dry Fork groundwater. Trace concentrations of several other metals were also observed. Bicarbonate, followed by sulfate, chloride, and nitrate, are dominant anionic constituents. The pH of Dry Fork groundwater is neutral; the average pH is 7.2. Dry Fork background groundwater generally contains less dissolved solids than other background groundwaters; the average TDS concentration is 500 mg/L.

Other than a one-time, low-level detection of Th-230 at Well 2123, no radionuclides were detected in groundwater samples from Dry Fork background wells. Data from the IT (1986) report were used to establish background concentrations of aluminum, vanadium, and zinc in Dry Fork groundwater.

TABLE 15-2

GROUNDWATER QUALITY, GREAT MIAMI AQUIFER ROSS BACKGROUND WELLS

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GREAT MIAMI AQUIFER ROSS BACKGROUND (2000 AND 3000 SERIES WELLS)

Parameter	Average[a]	Standard Deviation[a]	Range[a]
Radionuclides (pCi/L)			
Cs-137	20 U	-	20 U - 20 U
Np-237	1 U	-	1 U - 1 U
Pu-238	1 U	-	1 U - 1 U
Pu-239/240	1 U	-	1 U - 1 U
Ra-226	1.0	0.0	1 U - 1.1
Ra-228	3.1	0.4	3 U - 4.5
Ru-106	150 U	-	150 U - 150 U
Sr-90	5 U	-	5 U - 5 U
Tc-99	30 U	-	30 U - 30 U
Th-228	1.0	0.0	1 U - 1.2
Th-230	1.0	0.2	1 U - 1.7
Th-232	1 U	-	1 U - 1 U
Th-Total[b]	13 U	-	13 U - 13 U
U-234	1 U	-	1 U - 1 U
U-235/236	1 U	-	1 U - 1 U
U-238	1 U	-	1 U - 1 U
U-Total[b]	1.0	0.2	1 U - 2.0
Metals (mg/L)			
Al[c]	--	--	-- --
As	0.106	0.189	0.002 U - 0.55
Ba	0.051	0.010	0.035 - 0.073
Cd	0.005	0.001	0.005 U - 0.01
Ca	92.48	8.51	70.2 - 110
Cr	0.021	0.003	0.02 U - 0.03
Cu	0.047	0.052	0.01 U - 0.176
Fe	0.028	0.039	0.005 U - 0.145
Pb	0.002	0.001	0.002 U - 0.004
Mg	26.00	5.43	17 - 33
Mn	0.004	0.004	0.001 U - 0.016
Hg[b]	0.21	0.04	0.2 U - 0.4

See footnotes at end of table.

TABLE 15-2
(continued)

GREAT MIAMI AQUIFER ROSS BACKGROUND
(2000 AND 3000 SERIES WELLS)

Parameter	Average[a]	Standard Deviation[a]	Range[a]		
Metals (Continued)					
Mo	0.02	0.02	0.02 U	-	0.092
Ni	0.02	0.00	0.02 U	-	0.024
K	2.62	0.03	2.07	-	3.28
Se	0.002 U	-	0.002 U	-	0.002 U
Ag	0.011	0.005	0.01 U	-	0.033
Na	15.25	3.96	9.41	-	26.7
V[c]	--	--	--		--
Zn[c]	--	--	--		--
General Chemistry (mg/L)					
Ammonia as N	0.11	0.03	0.1 U	-	0.2
Chloride	29.14	12.20	7	-	49.5
Conductance[d]	585.8	109.1	400	-	800
Fluoride	0.25	0.06	0.18	-	0.365
Nitrate as N	4.11	3.78	0.28	-	12.4
Total Phosphorus	0.05	0.00	0.05 U	-	0.065
Sulfate	47.05	18.15	3.08	-	72
Total Organic Halides	0.05 U	-	0.05 U	-	0.05 U
Alkalinity as HCO3	312.6	34.3	236.8	-	363.55
pH	7.1	-	6.64	-	7.7
Total Dissolved Solids[e]	544.25	45.30	347.44	-	714.95

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See footnotes at end of table.

TABLE 15-2
(continued)

GREAT MIAMI AQUIFER ROSS BACKGROUND
(2000 SERIES WELLS)

Parameter	Average[a]	Standard Deviation[a]	Range[a]
Radionuclides (pCi/L)			
Cs-137	20 U	-	20 U - 20 U
Np-237	1 U	-	1 U - 1 U
Pu-238	1 U	-	1 U - 1 U
Pu-239/240	1 U	-	1 U - 1 U
Ra-226	1.0	0.0	1 U - 1.1
Ra-228	3.1	0.4	3 U - 4.5
Ru-106	150 U	-	150 U - 150 U
Sr-90	5 U	-	5 U - 5 U
Tc-99	30 U	-	30 U - 30 U
Th-228	1 U	-	1 U - 1 U
Th-230	1.1	0.2	1 U - 1.7
Th-232	1 U	-	1 U - 1 U
Th-Total[b]	13 U	-	13 U - 13 U
U-234	1 U	-	1 U - 1 U
U-235/236	1 U	-	1 U - 1 U
U-238	1 U	-	1 U - 1 U
U-Total[b]	1.1	0.3	1 U - 2.0
Metals (mg/L)			
Al[c]	–	–	–
As	0.143	0.232	0.002 U - 0.55
Ba	0.047	0.012	0.035 - 0.073
Cd	0.005 U	-	0.005 U - 0.005 U
Ca	87.04	7.88	70.2 - 95.2
Cr	0.02	-	0.02 U - 0.02
Cu	0.050	0.063	0.01 U - 0.176
Fe	0.011	0.015	0.005 U - 0.05
Pb	0.002 U	-	0.002 U - 0.002 U
Mg	22.95	5.92	17 - 33
Mn	0.002	0.001	0.001 U - 0.003
Hg[b]	0.2 U	-	0.2 U - 0.2 U
Mo	0.02	0.01	0.02 U - 0.04
Ni	0.02 U	-	0.02 U - 0.02 U
K	2.67	0.32	2.07 - 3.01
Se	0.002 U	-	0.002 U - 0.002 U

See footnotes at end of table.

TABLE 15-2
(continued)

GREAT MIAMI AQUIFER ROSS BACKGROUND
(2000 SERIES WELLS)

Parameter	Average[a]	Standard Deviation[a]	Range[a]		
Metals (Continued)					
Ag	0.012	0.007	0.01 U	-	0.033
Na	16.20	5.79	9.41	-	26.9
V	--	--	--	-	--
Zn	--	--	--	-	--
General Chemistry (mg/L)					
Ammonia as N	0.11	0.03	0.1 U	-	0.2
Chloride	24.06	13.02	7	-	45.15
Conductance[d]	564.17	116.95	400	-	800
Fluoride	0.26	0.06	0.2	-	0.35
Nitrate as N	4.35	3.46	0.4	-	8.4
Total Phosphorus	0.05	0.00	0.05 U	-	0.055
Sulfate	37.36	17.13	3.08	-	58
Total Organic Halide	0.05 U	-	0.05 U	-	0.05 U
Alkalinity as HCO3	294.7	37.2	236.8	-	343.2
pH	7.1	-	7	-	7.7
Total Dissolved Solids[e]	505.18	47.03	348.00	-	643.45

[a]Outliers and inconsistent MDLs are excluded. Nondetects are set equal to the MDLs and included in the calculation of averages and standard deviations (Section 15.3.2.2).

U = below the MDL shown.

[b]In ug/L.

[c]Ross background wells have not been analyzed for aluminum, vanadium, and zinc.

[d]In umhos/cm.

[e]TDS is calculated based on following assumptions:

The mass contributed by radionuclides is negligible.
Phosphorus is present as phosphate.

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TABLE 15-3

GROUNDWATER QUALITY, GREAT MIAMI AQUIFER, DRY FORK BACKGROUND WELLS

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Parameter	GREAT MIAMI AQUIFER DRY FORK BACKGROUND				
	Average[a]	Standard Deviation[a]	Range[a]		
Radionuclides (pCi/L)					
Cs-137	20 U	-	20 U	-	20 U
Np-237	1 U	-	1 U	-	1 U
Pu-238	1 U	-	1 U	-	1 U
Pu-239/240	1 U	-	1 U	-	1 U
Ra-226	1 U	-	1 U	-	1 U
Ra-228	3 U	-	3 U	-	3 U
Ru-106	159 U	-	159 U	-	159 U
Sr-90	5 U	-	5 U	-	5 U
Tc-99	30 U	-	30 U	-	30 U
Th-228	1 U	-	1 U	-	1 U
Th-230	1.1	0.3	1 U	-	2.1
Th-232	1 U	-	1 U	-	1 U
Th-Total[b]	13 U	-	13 U	-	13 U
U-234	1 U	-	1 U	-	1 U
U-235/236	1 U	-	1 U	-	1 U
U-238	1 U	-	1 U	-	1 U
U-Total[b]	1 U	-	1 U	-	1 U
Metals (mg/L)					
Al	0.1 U	-	0.1 U	-	0.1 U
As	0.002	0.001	0.002 U	-	0.004
Ba	0.038	0.004	0.034	-	0.045
Cd	0.005 U	-	0.005 U	-	0.005 U
Ca	89.10	5.43	81.7	-	99.4
Cr	0.021	0.003	0.02 U	-	0.03
Cu	0.016	0.008	0.01 U	-	0.033
Fe	0.015	0.009	0.005 U	-	0.03
Pb	0.002	0.001	0.002 U	-	0.004
Mg	23.20	1.71	20.7	-	26.2
Mn	0.201	0.237	0.001 U	-	0.48
Hg[b]	0.2 U	-	0.2 U	-	0.2 U
Mo	0.02 U	-	0.02 U	-	0.02 U
Ni	0.02 U	-	0.02 U	-	0.02 U
K	1.28	0.24	1 U	-	1.7

See footnotes at end of table.

TABLE 15-3
(continued)

GREAT MIAMI AQUIFER DRY FORK BACKGROUND

Parameter	Average[a]	Standard Deviation[a]	Range[a]		
Metals (Continued)					
Se	0.002 U	-	0.002 U	-	0.002 U
Ag	0.012	0.008	0.01 U	-	0.034
Na	3.23	0.59	1.96	-	3.99
V	0.01 U	-	0.01 U	-	0.01 U
Zn	0.17	-	0.17	-	0.17
General Chemistry (mg/L)					
Ammonia as N	0.11	0.03	0.1 U	-	0.2
Chloride	9.20	4.09	3	-	13
Conductance[c]	535.0	108.2	370	-	750
Fluoride	0.25	0.08	0.1	-	0.38
Nitrate as N	4.64	4.87	0.1 U	-	11.4
Total Phosphorus	0.08	0.05	0.05 U	-	0.195
Sulfate	29.59	6.80	21	-	39.9
Total Organic Halide	0.05 U	-	0.05 U	-	0.05 U
Alkalinity	322.1	24.0	285.2	-	344.3
pH	7.2	-	7.0	-	7.6
Total Dissolved Solids[d]	499.56	33.71	415.85	-	581.23

[a]Outliers and inconsistent MDLs are excluded. Nondetects are set equal to the MDLs and included in the calculation of averages and standard deviations (Section 15.3.2.2).

U = below the MDL shown

[b]In ug/L.

[c]In umhos/cm.

[d]TDS is calculated based on following assumptions:

The mass contributed by radionuclides is negligible.
Phosphorus is present as phosphate.

While Dry Fork background chemistry is very similar to Ross background chemistry, concentrations of sodium, chloride, sulfate, and TDS appear to be slightly higher in Ross groundwater. This may be due to greater urbanization of the Ross Section of the aquifer relative to the Dry Fork Section.

Shandon Tributary

Table 15-4 lists the average and range of radionuclide, metal, and general chemistry concentrations observed in groundwater samples from Shandon background wells. Calcium is the dominant metal in Shandon groundwater, followed by sodium, magnesium, iron, and potassium. Trace concentrations of other metals were also observed. Bicarbonate, followed by chloride, ammonia, and sulfate, are dominant anionic constituents. The pH of Shandon groundwater is neutral; the average pH is 6.9. The average TDS concentration is 598 mg/L.

Ra-226 and total uranium were the only radionuclides detected in groundwater samples from Shandon background wells. The detections were low and inconsistent. Detections of Ra-226 were 1.8 and 2 pCi/L from Wells 2050 and 2056, respectively, during second quarter 1988 sampling. Detections of total uranium occurred in Wells 2056 and 2066 with concentrations of 1.85 and 3 ug/L, respectively. Data from the IT (1986) report were used to establish background concentrations of aluminum, and zinc in Shandon groundwater.

Calcium and bicarbonate are the dominant dissolved constituents of all background groundwaters. The second most abundant ionic constituents in the Shandon groundwater are sodium and chloride; for Ross and Dry Fork groundwater, they are magnesium and sulfate. The barium concentration of Shandon groundwater is approximately an order of magnitude greater than that of other background groundwaters. It is believed that low sulfate concentrations most likely prevent the precipitation of barite, $BaSO_4$, from controlling barium concentrations in Shandon groundwater.

Differences in the geology and the chemical redox state of groundwater in the Shandon Tributary, compared with other areas of the aquifer, most likely contribute to the observed groundwater chemistry differences. The Shandon Tributary portion of the aquifer is mantled by thick deposits of undissected glacial overburden (Section 11.5); therefore, surface water infiltration may be slow. Shandon Tributary groundwater may have lower dissolved oxygen levels; i.e., is chemically reduced, relative to groundwater from other areas. Nitrogen is present in Shandon Tributary groundwater predominantly in the reduced form as ammonia (NH_3), whereas nitrogen is present in an oxidized form as nitrate (NO_3) in groundwater from other areas of the aquifer (Tables 15-2 and 15-3). Reduced iron (Fe^{+2}) concentrations also appear to be higher in Shandon groundwater relative to other background groundwaters.

TABLE 15-4

GROUNDWATER QUALITY, GREAT MIAMI AQUIFER SHANDON BACKGROUND WELLS

DRAFT

Parameter	GREAT MIAMI AQUIFER SHANDON BACKGROUND				
	Average[a]	Standard Deviation[a]	Range[a]		
Radionuclides (pCi/L)					
Cs-137	20 U	-	20 U	-	20 U
Np-237	1 U	-	1 U	-	1 U
Pu-238	1 U	-	1 U	-	1 U
Pu-239/2401	1 U	-	1 U	-	1 U
Ra-226	1.1	0.3	1 U	-	2
Ra-228	3 U	-	3 U	-	3 U
Ru-106	150 U	-	150 U	-	150 U
Sr-90	5 U	-	5 U	-	5 U
Tc-99	30 U	-	30 U	-	30 U
Th-228	1.1	0.2	1 U	-	1.99
Th-230	1 U	-	1 U	-	1 U
Th-232	1 U	-	1 U	-	1 U
Th-Total[b]	13 U	-	13 U	-	13 U
U-234	1 U	-	1 U	-	1 U
U-235/236	1 U	-	1 U	-	1 U
U-238	1 U	-	1 U	-	1 U
U-Total[b]	1.2 U	0.5	1 U	-	3
Metals (mg/L)					
Al	0.11	0.02	0.1 U	-	0.143
As	0.021	0.012	0.004	-	0.042
Ba	0.638	0.104	0.48	-	0.8
Cd	0.005	0.000	0.005 U	-	0.006
Ca	74.55	13.69	30.4	-	89
Cr	0.021	0.000	0.02 U	-	0.03
Cu	0.01 U	-	0.01 U	-	0.01 U
Fe	2.272	0.664	1.6	-	3.55
Pb	0.001	0.005	0.002 U	-	0.015
Mg	26.06	3.54	15.8	-	31.3
Mn	0.047	0.035	0.014	-	0.113
Hg[b]	0.26	0.20	0.2 U	-	1
Mo	0.02	0.00	0.02 U	-	0.03
Ni	0.02 U	-	0.02 U	-	0.02 U

See footnotes at end of table.

TABLE 15-4
(continued)

GREAT MIAMI AQUIFER SHANDON BACKGROUND

Parameter	Average[a]	Standard Deviation[a]	Range[a]
Metals (Continued)			
K	1.49	0.28	1.13 - 2
Se	0.002 U	-	0.002 U - 0.002 U
Ag	0.010	0.001	0.01 U - 0.014
Na	41.09	6.83	28.1 - 48
V	0.014	0.004	0.01 U - 0.0167
Zn	0.03	0.02	0.01 - 0.052
General Chemistry (mg/L)			
Ammonia as N	3.15	1.14	0.3 - 4.22
Chloride	61.01	11.80	39.5 - 78.9
Conductance[c]	624.2	86.11	450 - 850
Fluoride	0.85	0.22	0.4 - 1.25
Nitrate as N	0.1	-	0.1 U - 0.1
Total Phosphorus	0.10	0.03	0.05 U - 0.18
Sulfate	3.66	3.25	2 U - 13.4
Total Organic Halide	0.05 U	-	0.05 U - 0.05 U
Alkalinity	387.0	13.8	369.2 - 416.6
pH	6.9	-	6 - 7.6
Total Dissolved Solids[d]	603.79	24.28	489.87 - 691.77

[a]Outliers and inconsistent MDLs are excluded. Nondetects are set equal to the MDLs and included in the calculation of averages and standard deviations (Section 15.3.2.2).

U = below the MDL shown.

[b]In ug/L.

[c]In umhos/cm.

[d]TDS is calculated based on following assumptions:

The mass contributed by radionuclides is negligible.
Phosphorus is present as phosphate.

15.3 GROUNDWATER QUALITY DATA ANALYSIS - METHODOLOGY

The methodology used to evaluate groundwater quality is presented in this section. The statistical methods employed to analyze groundwater quality data are also presented. The rationale for grouping downgradient wells prior to statistical analysis is addressed.

15.3.1 Downgradient Well Groups

Because of the hydrogeologic configuration of the Great Miami Aquifer and groundwater chemistry heterogeneities, downgradient 2000 and 3000 Series wells are placed into groups for the purpose of analyzing downgradient groundwater quality. Each downgradient well group generally receives groundwater from only one Great Miami Aquifer flow system. Throughout this section, wells referred to as downgradient are those that are located hydrologically downgradient of potential FMPC contaminant sources.

15.3.1.1 1000 Series Wells

The 1000 Series wells are not divided into separate downgradient groups. The groundwater chemistry of all 1000 Series wells which may be potentially impacted by the FMPC is compared to composite background water chemistry developed from several wells as described in Section 15.2.1.1.

15.3.1.2 2000 and 3000 Series Wells

As discussed in Section 15.1.2, the upper Great Miami Aquifer in the vicinity of the FMPC receives groundwater from three distinct flow systems. Each downgradient 2000 Series well was placed into one of the three groups--Dry Fork, Shandon, and Ross--based on the well location relative to the location of the groundwater flow divides. Downgradient 3000 Series wells were also placed into the three groups. However, wells located in the area of fluctuation between the Dry Fork and Shandon sections may receive groundwater from either section, depending on flow conditions.

15.3.1.3 4000 Series Wells

Groundwater in the lower part of the Great Miami Aquifer flows from west to east in the vicinity of the FMPC. Thus, the lower part of the aquifer receives groundwater from only the Dry Fork Section, and the grouping of downgradient 4000 Series wells prior to statistical analysis was not necessary.

15.3.2 Statistical Methods

The statistical methods used in the analysis of FMPC groundwater quality data are from the guidance document "Statistical Analysis of Ground-Water Monitoring Data at RCRA Facilities," EPA 530-SW-89-026 (U.S. EPA 1989a). Upper tolerance limits are generated using background water quality data; downgradient groundwater quality data are then compared to these tolerance limits. A concentration is considered statistically elevated relative to background when it exceeds the upper tolerance limit.

15.3.2.1 Treatment of Data for Statistical Analysis

Prior to the statistical analysis of groundwater quality, the data set was inspected for outliers and a consistent detection limit was assigned to each chemical constituent. Also, a test of normality was applied to the background water quality data.

Detection Limits

During RI/FS and RCRA sampling, minimum detection limits (MDLs) for a given constituent have varied from one sampling round to the next and have even varied within a single sampling round. Because data reported as nondetect (ND) are included in the statistical analysis, each constituent must have only one MDL. Thus, for each constituent, one MDL was selected in such a manner as to keep modifications of the original groundwater quality data set to a minimum, and the data were adjusted using the following procedure. Each reported ND that was greater than the selected MDL was omitted from the statistical analysis because of the uncertainty about the actual concentration of the ND. The true value may be higher than the selected MDL. However, any ND that was less than the selected MDL was given a value equal to the selected MDL and still denoted as ND. Any detections that were lower than the selected MDL were also set equal to the selected MDL and denoted as ND. To maintain consistency, both background and downgradient data were adjusted using this procedure.

The statistical procedures used in this analysis are dependent on the magnitude of the results more so than on whether or not the result is a detected concentration. As such, the modifications made to the original data set were generally conservative and did not decrease average concentrations. The MDLs used in this analysis are listed in Table M-1 of Appendix M.

Outliers

An analytical result may be an outlier if, for a given well and a given constituent, the analytical result detected during one sampling round is very different compared with the analytical results observed during other sampling rounds. A suspect result is labeled as an outlier only if it can be

proven statistically to be an outlier. A student's t-test described by Skoog (1985) was used to determine whether an observation was an outlier. For a given well and constituent, a 95 percent confidence interval was generated using all data except the result in question. The 95 percent confidence interval is defined as:

$$(\overline{X} - \frac{tS}{\sqrt{n}}, \overline{X} + \frac{tS}{\sqrt{n}})$$

where $\overline{X}$ is the sample mean, n is the number of samples excluding the suspect outlier, t is a published factor based on 95 percent confidence interval and n-1 degrees of freedom, and S is the sample standard deviation. The values of $\overline{X}$ and S are calculated as:

$$\overline{X} = (\sum X)/n \qquad S = \sqrt{\frac{\sum (X^2) - (\sum X)^2/n}{(n-1)}}$$

If the suspect outlier fell outside the interval, it was labeled as an outlier and not used in calculating an upper tolerance limit or an average concentration at the given well.

When a constituent was detected only once during three or more sampling events, the student's t-test could not be applied. In this case, the detection was considered to be an outlier if it was ten times greater than the MDL of the analysis.

Test of Normality

A condition that must be met prior to the construction of tolerance intervals is that background water quality data are normally distributed. The coefficient of variation test was applied to background water quality data as a check on the hypothesis that the data are normally distributed. The coefficient of variation, CV, is calculated as:

$$CV = S / \overline{X}$$

where S and $\overline{X}$ are as defined previously. If the CV of a constituent was greater than one, it indicated that the data are not normally distributed. For background data, the CV exceeded one only for those constituents that are highly dependent on the redox condition of the groundwater; e.g., nitrogen species and manganese. Tolerance limits were constructed for these constituents, and it is noted in the results when the CV exceeded one.

15.3.2.2 Tolerance Limits

A tolerance interval based on the normal distribution was established for each constituent of concern from background water quality data. A tolerance coefficient of 95 percent was used to construct tolerance intervals with 95 percent coverage. This means that 95 percent of the background observations would be within the tolerance interval at a 95 percent confidence level if the observations would be randomly taken. A minimum number of three observations was required for the construction of a tolerance interval, and a sample size of eight or more was considered large enough to establish an adequate tolerance interval (U.S. EPA 1989a).

Since contamination in groundwater is generally indicated by elevated concentrations, upper tolerance limits, which are one-sided tolerance intervals, were generated from the background water quality data. To construct the upper tolerance limit (UTL) of a constituent, the mean, $\overline{X}$, and the sample standard deviation, S, were calculated from the background groundwater analytical results. The one-sided UTL was constructed as

$$UTL = \overline{X} + (K * S)$$

where K is the one-sided normal tolerance factor (U.S. EPA 1989a).

If all observations of a constituent from a background well group were at the MDL, S could not be calculated and the UTL for that constituent could not be determined. In this case, the UTL was set equal to the MDL, and any detection of the constituent in a downgradient groundwater sample that was greater than the MDL was considered to be statistically elevated relative to background.

A two-sided tolerance limit was established for pH based on a 95 percent confidence interval, with a 2.5 percent error level on each side. A t-distribution was used to construct the tolerance interval as:

$$\left(\overline{X} - \frac{tS}{\sqrt{n}}, \overline{X} + \frac{tS}{\sqrt{n}}\right)$$

where $\overline{X}$ and S are as previously defined, n is the number of sample observations of background data, and t is a factor based on the 95 percent confidence interval with n-1 degrees of freedom (U.S. EPA 1989a).

As discussed in Section 15.2.2.2, background water quality data from IT (1986) were used to supplement RI/FS and 1989 RCRA background data for aluminum, vanadium, and zinc. Most

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groundwater samples were not analyzed for these metals; therefore, inclusion of these data is necessary to generate UTLs for the metals. Some of the sampled wells for the Interim Risk Assessment were also sampled during the RI/FS and are shown in Figures 15-1 and 15-2. Wells sampled during the Interim Risk Assessment that are upgradient of the FMPC and can be considered background for the Dry Fork background area are Wells State-16 and 12-3, which are equivalent to RI/FS Wells 2057 and 2123, respectively. Wells State-8 and State-10, which are equivalent to RI/FS Wells 2056 and 2105, respectively, and Well H-113 were considered for the Shandon background area. Well H-113 is located near Well 2050.

Upper tolerance limits are generated for aluminum, vanadium, and zinc only for the Shandon downgradient well group. Upper tolerance limits for these metals could not be determined for the 1000 Series wells or the Dry Fork downgradient well group due to limited background data. No data are available to define concentrations of aluminum, vanadium, or zinc for the Ross downgradient well group (Section 15.2.2.2).

Groundwater quality data from each downgradient well were checked for outliers, as described in Section 15.3.2.1. For each chemical constituent, an average concentration was calculated in each downgradient well after the removal of outliers. As discussed above, NDs were set equal to the MDL and included in the average. The average concentration was then compared to the established UTL to determine whether the average concentration exceeded the UTL. If the average concentration from a given well exceeded the UTL, this was taken as statistical evidence of contamination by a particular constituent in groundwater at the given well. Concentrations of the constituent observed at the given well were, therefore, termed statistically elevated with respect to background concentrations.

15.3.3 Factors Influencing Groundwater Quality

When comparing downgradient to background groundwater quality using the statistical methods just described, it is being assumed that statistically elevated concentrations are related to contamination from the FMPC and associated waste storage units. However, several factors independent of the FMPC may also influence groundwater quality. These factors are considered during the analysis of FMPC groundwater quality data so that conclusions reached do not overstate the impact that the FMPC has had on groundwater quality. The non-FMPC factors that can influence water quality are discussed below.

Almost all background wells are private homeowner or industrial wells. Materials used in the construction of these wells may not be consistent with the materials used to construct RI/FS

000025

monitoring wells. Due to access restrictions, systems used to pump groundwater from the private wells are not consistent with those used to sample RI/FS monitoring wells. Also, some of the background well systems include pressure tanks. Groundwater pumped into a tank prior to sampling could be partially oxidized, affecting the concentration of redox sensitive parameters such as iron and manganese.

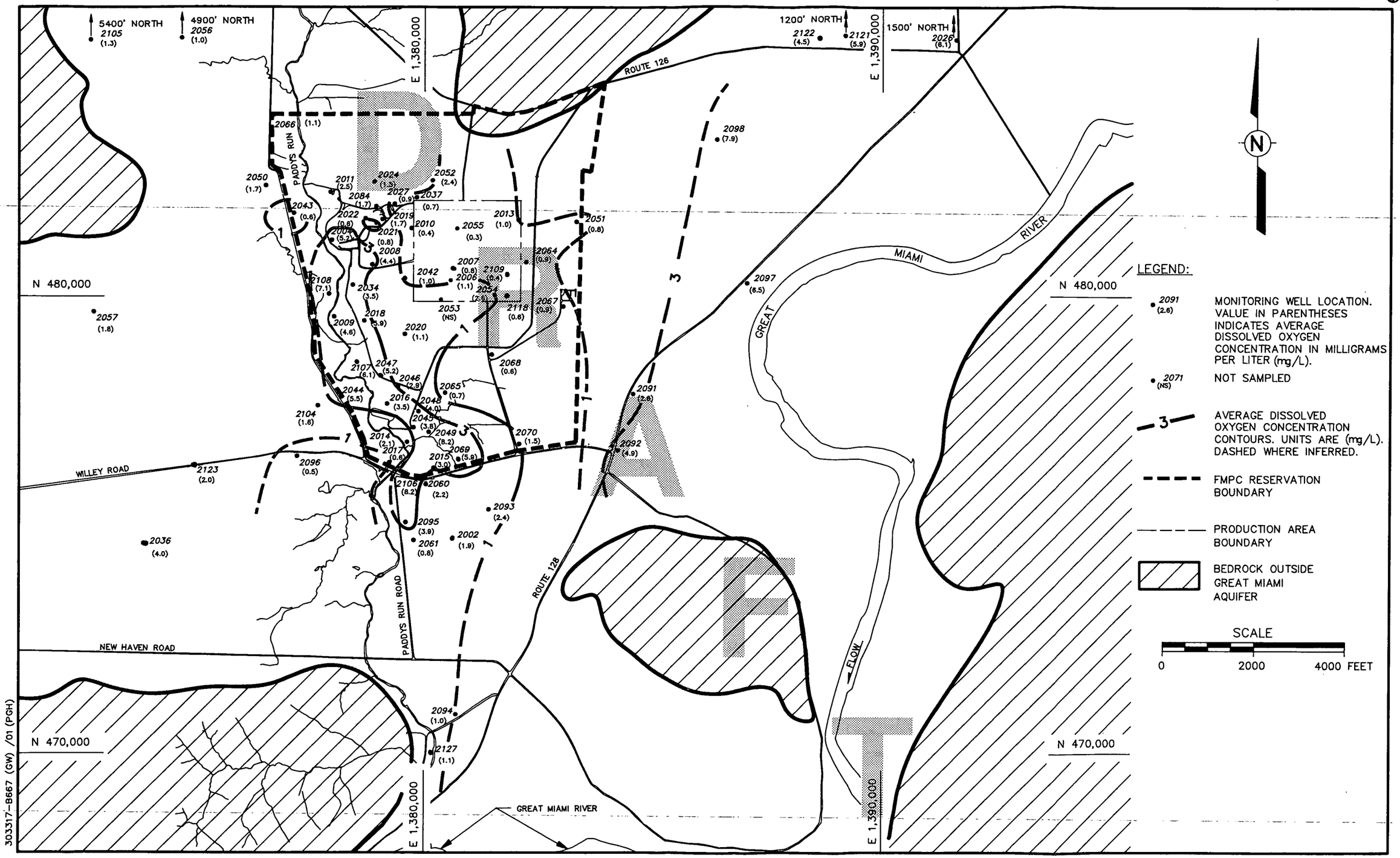
As discussed in Section 12.5.2, surface water from Paddys Run recharges the Great Miami Aquifer. From a location near the metal oxide silo to the southern FMPC boundary, Paddys Run has eroded through the glacial overburden and flows directly above the Great Miami Aquifer. The water table in the aquifer is below the level of the stream bed. These conditions are conducive to the infiltration of oxygenated stream water during periods when Paddys Run is flowing.

Figures 15-3 and 15-4 show the average dissolved oxygen concentration of groundwater measured during RI/FS and 1989 RCRA sampling of 2000 and 3000 Series wells. Groundwater in an area of the Great Miami Aquifer beneath and east of Paddys Run contains dissolved oxygen concentrations exceeding 1.0 mg/L. This dissolved oxygen distribution, as well as hydrogeologic data presented in Section 12.5.2, strongly indicate that stream water is infiltrating into the aquifer. As groundwater from the Shandon and Dry Fork Section flow into the relatively oxidized region of the aquifer, the groundwater chemistry of redox sensitive parameters will change. Also, mixing of recharge water from Paddys Run with groundwater will result in changes in groundwater chemistry downgradient from the recharge zone.

Certain constituents, such as ammonia, nitrate, phosphorus, and sulfate, are common contaminants in rural areas due to agricultural activities and septic systems. It is possible that sources of some of these constituents are located between the background and downgradient wells. Statistically elevated concentrations of these constituents observed at some downgradient wells may actually be due to sources that are upgradient from the FMPC.

Several industrial facilities are located south of the FMPC, along Paddys Run Road. The Paddys Run Road site RI/FS is currently under way to investigate whether the industrial facilities are contributing contamination to the Great Miami River. Contamination would be superimposed on the area under investigation by the FMPC RI/FS. Thus, statistically elevated concentrations of certain constituents may be related to the industrial facilities, rather than the FMPC.

303317-B667 (GW) /01 100% GRN.



303317-B667 (GW) /01 (PGH)

FIGURE 15-3
AVERAGE DISSOLVED OXYGEN IN GROUNDWATER
2000 SERIES WELLS
SECOND QUARTER 1988 THROUGH
FOURTH QUARTER 1989

303317-B675 (GW) /01,100% GRN

303317-B675 (GW) /01 (PGH)

FIGURE 15-4
AVERAGE DISSOLVED OXYGEN CONCENTRATION
IN GROUNDWATER, 3000 SERIES WELLS
SECOND QUARTER 1988 THROUGH
FOURTH QUARTER 1989

15.4 GROUNDWATER QUALITY DATA ANALYSIS - RESULTS

The remainder of this chapter presents the results of the FMPC groundwater quality data analysis. The groundwater quality of perched zones in the glacial overburden is addressed first, followed by a discussion of the groundwater quality of the Great Miami Aquifer. Summary sections are provided.

Appendix M contains background and average groundwater concentration tables for all radionuclides, metals, and general chemistry parameters. Tables M-2 through M-5 list the averages and UTLs generated for perched groundwater and groundwater from the various background well groups of the Great Miami Aquifer. Also presented in these tables are the average concentrations of radionuclides, metals, and general chemical parameters in groundwater from downgradient wells. Appendix M contains background and average groundwater concentration tables for all radionuclides, metals, and general chemistry parameters. The table format presents the well number in the left column and the data for each parameter in successive columns to the right. Each parameter column is headed by the parameter symbol followed by the concentration units, the background average concentration, and the upper tolerance limit for the parameter. Listed adjacent to each well is the average concentration for that parameter for all samples from that well.

Tables M-6 through M-9 in Appendix M list outliers that were excluded from the statistical analysis.

15.4.1 Glacial Overburden

As discussed in Section 6.1.2.1, 1000 Series wells are screened in the glacial overburden which overlies the Great Miami Aquifer. Since the glacial overburden is the uppermost geologic deposit over most of the FMPC facility, the Production Area and Waste Storage areas are in direct contact with this unit. Radiological or chemical spills or leaks can potentially infiltrate to the perched groundwater zone through the glacial overburden.

15.4.1.1 Radionuclides

Uranium

Uranium is, by far, the most widespread contaminant attributable to the FMPC in glacial overburden groundwater. Total uranium data are used to evaluate the spatial and temporal distribution of uranium in glacial overburden groundwater. Extremely variable, but elevated, total uranium concentrations are observed in perched groundwater beneath the Production and Waste Storage

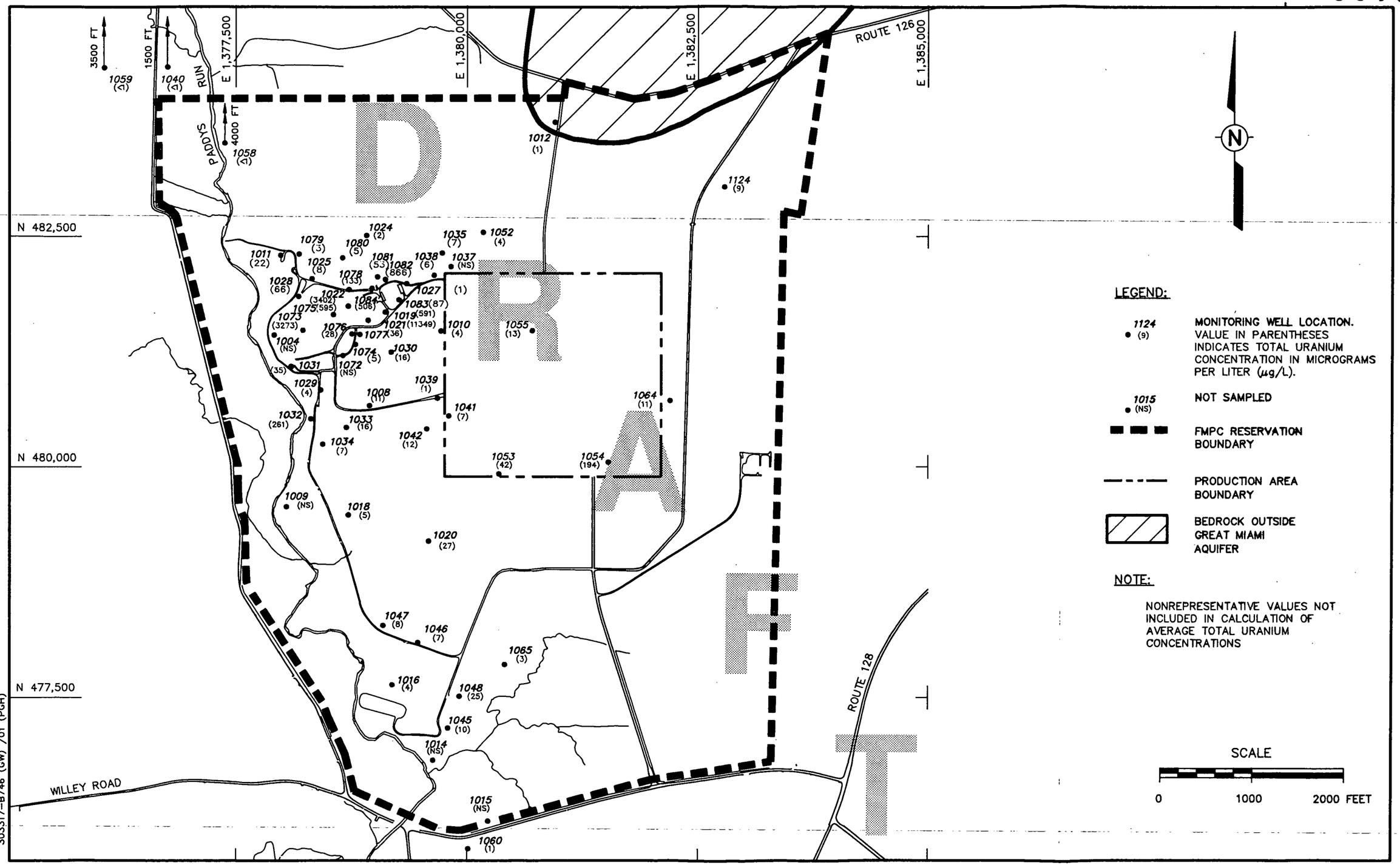
areas. The UTL for total uranium in perched groundwater is 4.7 ug/L. The uranium concentration of perched groundwater samples from most 1000 Series wells exceeded this tolerance limit. Figure 15-5 shows the average total uranium concentration of groundwater in 1000 Series wells calculated from RI/FS and RCRA sampling data collected from the second quarter 1988 to the fourth quarter 1989. Figures 15-6 and 15-7 show the total uranium concentration of perched groundwater samples collected from 1000 Series piezometers located in the Production Area and sampled as part of the RI/FS Production and Additional Suspect Areas Program during the period of April 1989 to February 1990. Figure 15-8 presents the average total uranium concentration of perched groundwater samples collected from 1000 Series piezometers located in the vicinity of Plant 2/3. In Appendix L, Figures L-1 through L-7 show the total uranium concentration data of groundwater samples collected from 1000 Series wells during individual sampling rounds.

When appropriate in the following sections, isotopic uranium data are used to examine uranium distribution patterns in groundwater. In particular, the U-238/U-234 activity ratios of groundwater samples and waste material are used to determine if specific waste storage units can be identified as sources of specific areas of observed groundwater uranium contamination. If it is assumed that the U-238/U-234 activity ratio observed in groundwater is the same as that of the source material, then the source of uranium in groundwater may be identified for those cases with exceptional activity ratios.

The isotopic activity of U-238 and its daughter product U-234 should be in secular equilibrium since the half-life of U-238 is much longer than the half-life of U-234. Secular equilibrium implies that the activity ratio U-238/U-234 is one. However, due to preferential leaching at radiation-damaged sites of minerals, the observed U-238/U-234 activity ratio is commonly less than one in groundwater that contains uranium from natural sources. When a U-238 atom present in a mineral decays to U-234, radiation from the decay damages the mineral structure at the site where the uranium atom is present. This uranium atom, which is now a U-234 atom, is more easily leached from the mineral relative to U-238 atoms that are present at undamaged sites. The result is that there would be an overabundance of U-234 atoms in solution.

The sources of elevated uranium found in groundwater at the FMPC are not natural, and the preferential leaching mechanism observed in natural materials would most likely not apply to the FMPC uranium sources. Possible FMPC source materials include liquids resulting from the dissolution of feed materials and leachable solid wastes resulting from FMPC operations. Locally unique U-238/U-234 activity ratios are therefore possible, particularly for those release points involving depleted uranium.

303317-B746 (GW) /01 100% ORG.



303317-B746 (GW) /01 (PGH)

FIGURE 15-5
AVERAGE URANIUM CONCENTRATION
IN GROUNDWATER, 1000 SERIES WELLS
SECOND QUARTER 1988 THROUGH
FOURTH QUARTER 1989

303317-B725 (GW) /01,100% ORG

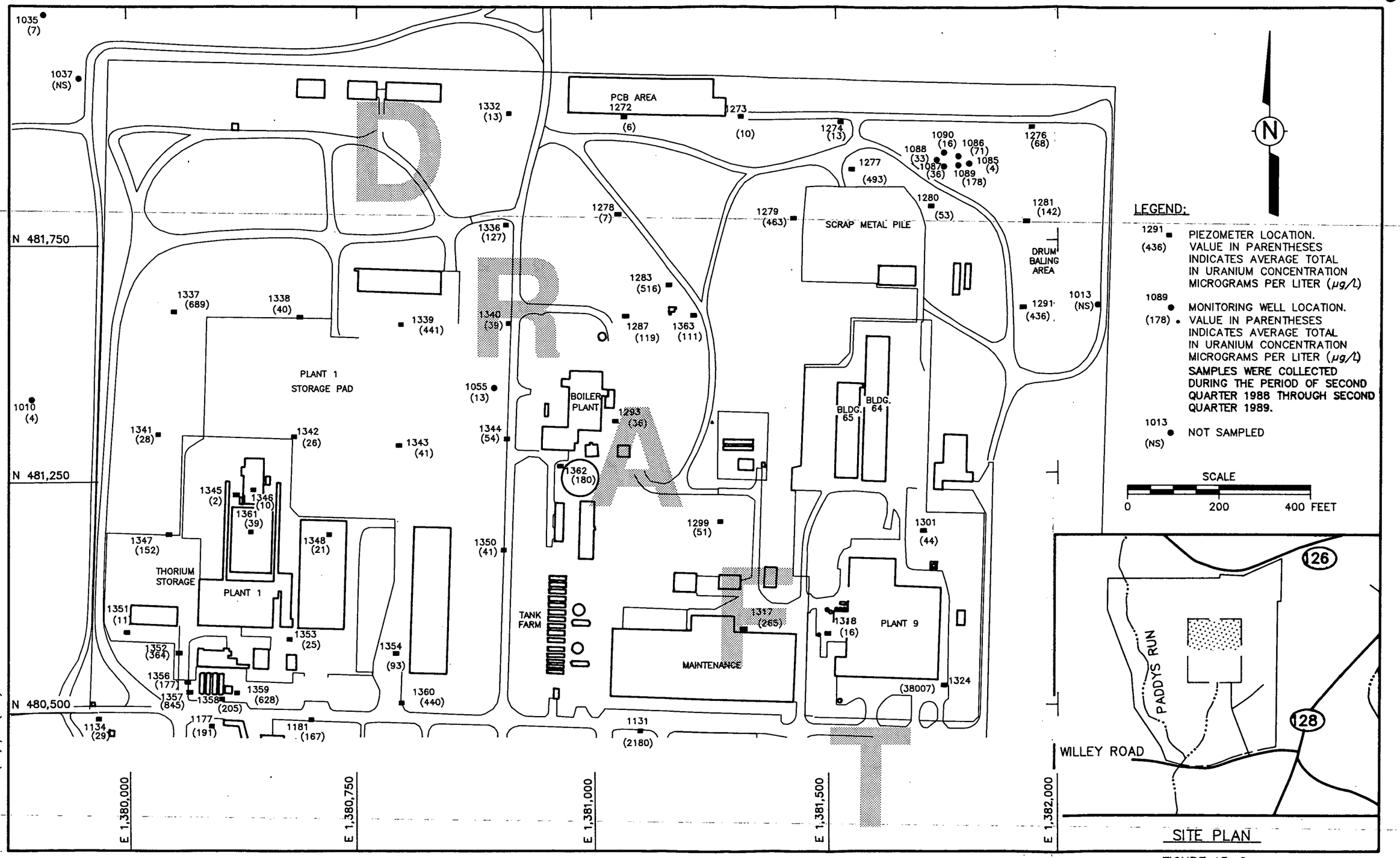


FIGURE 15-6
AVERAGE URANIUM CONCENTRATION IN GROUNDWATER
1000 SERIES WELLS, FMPC PRODUCTION AREA,
NORTHERN HALF, APRIIL 1989 – JANUARY 1990

303317-B725 (GW) /01 (PGH)

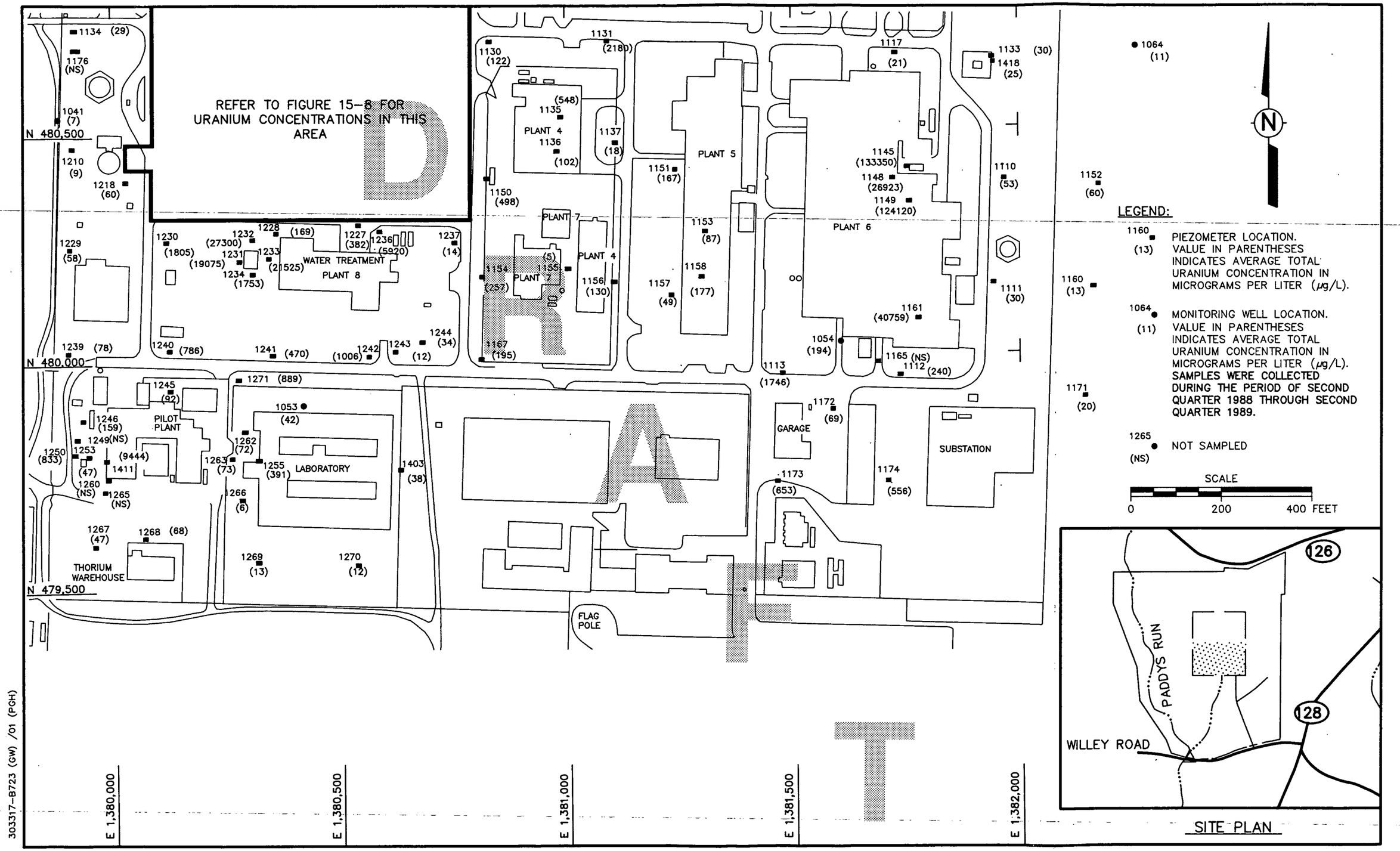
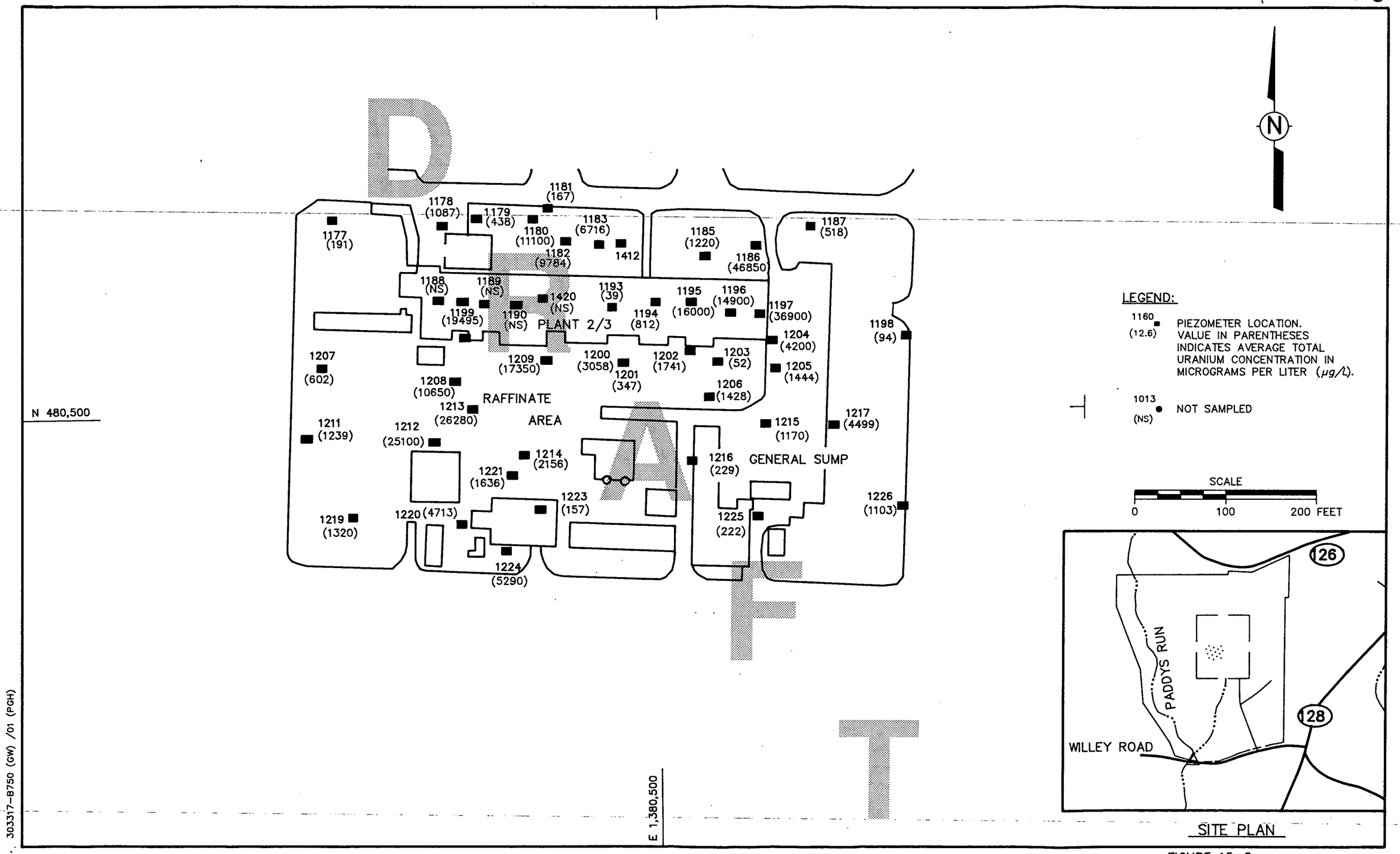


FIGURE 15-7
AVERAGE URANIUM CONCENTRATION
IN GROUNDWATER, 1000 SERIES WELLS
FMPC PRODUCTION AREA, SOUTHERN HALF
APRIL 1989 – JANUARY 1990

303317-B723 (GW) /01,100% ORG

303317-B723 (GW) /01 (PGH)

303317-B750 (GW) /01,100% ORG



303317-B750 (GW) /01 (PGH)

FIGURE 15-8
AVERAGE URANIUM CONCENTRATION IN GROUNDWATER
1000 SERIES WELLS, FMPC PRODUCTION AREA,
PLANT 2/3 AND VICINITY
APRIL 1989 – JANUARY 1990

Production Area

Statistically elevated total uranium concentrations are observed in perched groundwater samples from almost every 1000 Series piezometer located in the Production Area. The total uranium concentration in perched groundwater samples from the Production Area ranges from 2 to 696,000 ug/L. A large area where total uranium concentrations are greater than 100 ug/L occurs in the perched groundwater beneath the southern half of the Production Area (Figures 15-7 and 15-8). The facilities that converted feed materials to metallic uranium products are situated in the southern half of the Production Area, and spills and leaks of uranium-bearing materials probably occurred in this area. Total uranium concentrations greater than 100 ug/L are also observed in groundwater samples from piezometers located near the scrap metal pile and the graphite furnace/oil burner area (Figure 15-6).

Total uranium concentrations greater than 1000 ug/L occur in perched groundwater beneath Plant 6, a large area beneath Plant 2/3, the Raffinate Area, and the General Sump (Figures 15-7 and 15-8). Soluble uranium forms such as uranyl nitrate and uranium-organic extraction liquids were processed in these areas. Spills or leaks of the soluble uranium forms from process facilities/equipment are the likely source of the observed uranium concentrations.

Total uranium concentrations greater than 10,000 ug/L occur in perched groundwater samples from piezometers located near the pickling area on the east side of Plant 6, Plant 2/3, and Raffinate areas, and the west side of Plant 8 (Figures 15-7 and 15-8). Soluble forms of uranium were handled in these areas. A total uranium concentration of 696,000 ug/L was detected in groundwater from Piezometer 1324, which is located near the southeastern corner of Plant 9. A subsequent sample contained 431,000 ug/L total uranium. The source of this uranium is considered to be a nearby sump that overflowed.

Waste Storage Area

The Waste Storage Area is the restricted area that contains the waste pits, Clearwell, K-65 and metal oxide silos, the sanitary landfill, and Lime Sludge ponds (Figure 15-9). Total uranium concentrations observed in perched groundwater samples from 1000 Series wells located in the Waste Storage Area vary over several orders of magnitude (Figure 15-5). Concentrations ranged from undetected, at an MDL of 1 ug/L, to 15,330 ug/L. Total uranium concentrations were statistically elevated in all 1000 Series wells located in the Waste Storage Area, with the exception of Wells 1010, 1027, 1029, 1039, and 1079 (Table M-2).

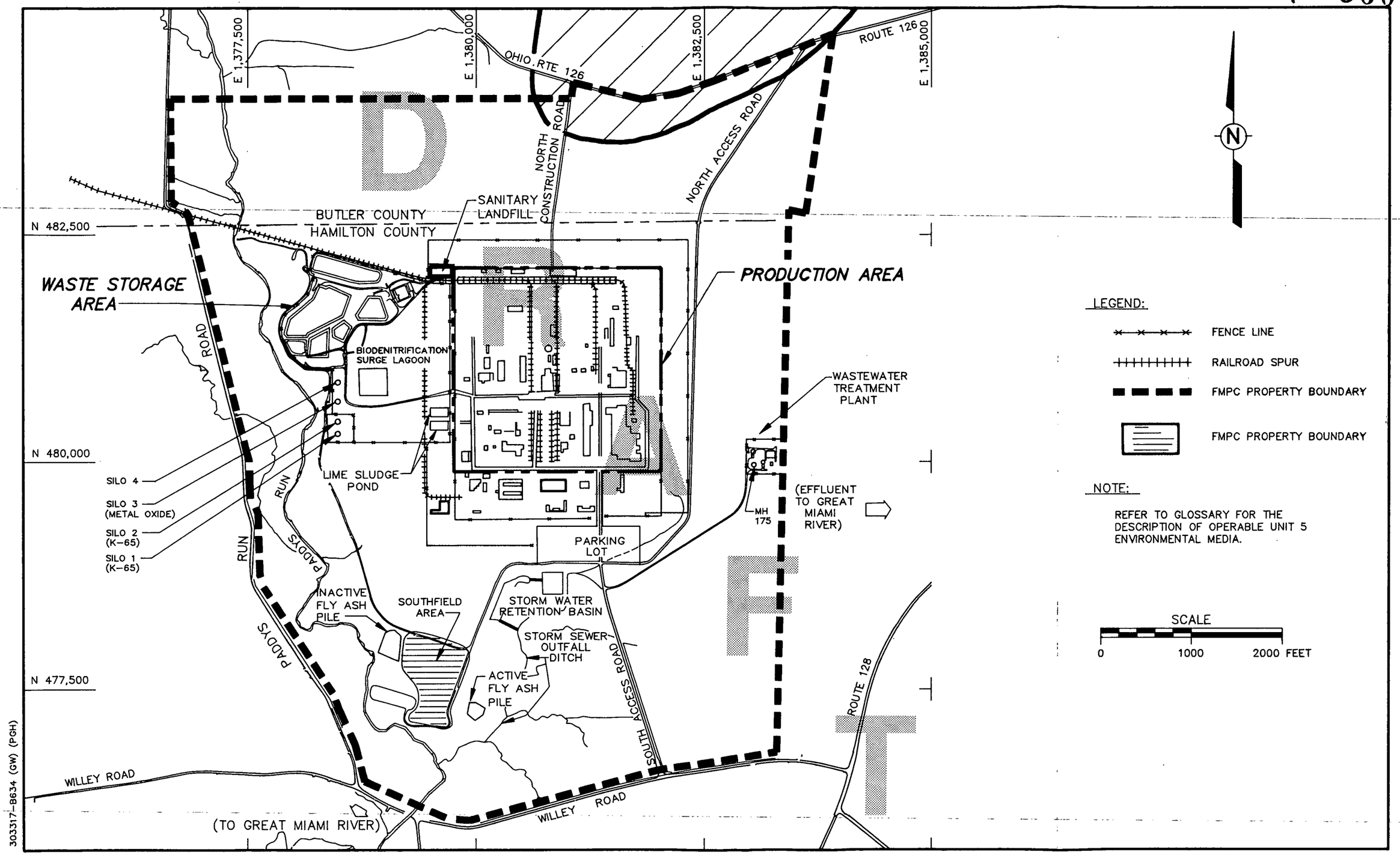
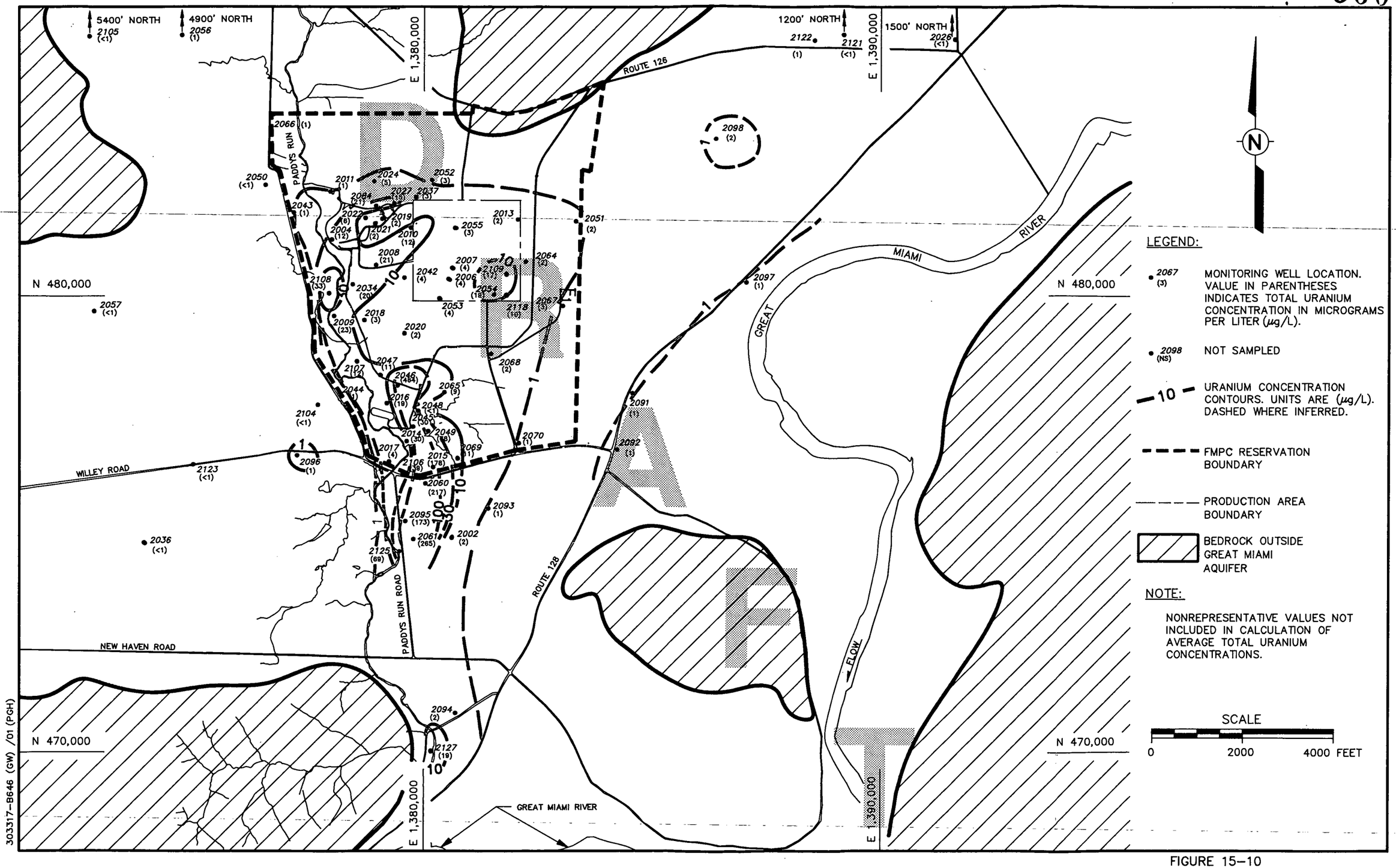


FIGURE 15-9
POTENTIAL SOURCE AREAS OF THE FEED MATERIALS PRODUCTION CENTER

303317-B646 (GW) /01,100% ORG.



303317-B646 (GW) /01 (PGH)

FIGURE 15-10
AVERAGE URANIUM CONCENTRATION
IN GROUNDWATER, 2000 SERIES WELLS
SECOND QUARTER 1988 THROUGH
FOURTH QUARTER 1989

Total uranium concentrations of groundwater samples from Wells 1019, 1021, 1022, 1073, 1075, 1082, and 1084 were repeatedly greater than 500 ug/L (Figure 15-5). These wells are all located near waste pits. Well 1073 is located between Pits 1 and 3, and Well 1075 is located between Pits 2, 3, and the burn pit. It is likely that the uranium observed in groundwater from Wells 1073 and 1075 has a source in these pits.

Leakage from Pits 4 and 6 may be the source of the elevated uranium concentrations which occur in Wells 1019, 1021, 1022, and 1084. The average total uranium concentration of surface water samples from these pits is approximately 34,000 and 1,000 ug/L, respectively (Weston 1987). Uranium isotopic data further support this statement, since the uranium isotopic ratio (U-238/U-234) of groundwater samples from these wells matches the ratio observed in Pits 4 and 6. The groundwater U-238/U-234 activity ratio ranges from 1.96 to 5.51, indicative of a depleted uranium source. The U-238/U-234 activity ratio calculated for waste samples from Pits 4 and 6 ranges from 1.7 to 10.3, and the average U-238/U-234 activity ratios calculated for surface water samples from Pits 4 and 6 are 3.4 and 5.7, respectively (Weston 1987).

Enough total uranium concentration data have been collected from Wells 1019, 1021, and 1022 to permit the analysis of the data as a function of time. Plots of total uranium concentration versus time for these wells during the period from early 1985 through 1989 are presented in Appendix O. Sources of the data are the Task C Report (Dames and Moore 1985), RCRA Groundwater Monitoring Reports Rounds 1-5 (Dames and Moore 1986a, 1986b, 1987a, 1987b, 1987c), and RI/FS and 1989 RCRA sampling. The total uranium concentration of groundwater varied greatly in each of the wells over the period of observation (Figures O-1, O-3, and O-4). The total uranium concentration of perched groundwater in Well 1019 has fluctuated, but no concentration trend is evident for the observation period. Groundwater total uranium concentrations in Well 1021 have generally increased during the observation period. The total uranium concentration of groundwater in Well 1022 increased from February 1985 to May 1987; however, it has decreased since May 1987. Because Wells 1019, 1021, and 1022 are located around the perimeter of Waste Pit 4, the concentrations observed in groundwater at these wells may decrease in the future as a result of the installation of a clay cap on this pit during the spring of 1989.

The U-238/U234 activity ratio calculated for groundwater samples from Wells 1025 and 1080 located near Pit 5 ranges from 0.48 to 1.18. The average activity ratio of Pit 5 waste samples is calculated as 1.1, and the average U-238/U234 activity ratio of surface water samples from Pit 5 has been calculated as 0.93 (Weston 1987). Thus, Pit 5 may be the source of the low levels of uranium observed in Wells 1025 and 1080. Though the uranium concentration of surface water in

Pit 5 is approximately 1,000 ug/L, concentrations in groundwater samples from Wells 1025 and 1080 ranged from only 3 to 11 ug/L. This may indicate that the leakage rate is slow due to the impermeable liner in Pit 5.

The U-238/U-234 activity ratio of groundwater from Well 1081, located between Pits 5 and 6, was approximately 1.0 during the first two RI/FS sampling rounds, and then increased to approximately 2.0 during later sampling rounds. This could indicate that both Pits 5 and 6 are sources of the uranium found in Well 1081 groundwater at an average concentration of 53 ug/L. The temporal change in the activity ratio may be because the relative contribution from each pit is a function of groundwater flow conditions.

The U-238/U-234 activity ratio of fourth quarter 1988 and first quarter 1989 groundwater samples from Well 1031 are 3.12 and 2.87, respectively. The average uranium concentration is 35 ug/L. Well 1031 is located near Waste Pit 1 and the Clearwell (Figures 15-5 and 15-9). Waste material from both Pit 1 and the Clearwell have U-238/U-234 activity ratios significantly greater than 1.0 (Weston 1987). Also, the average U-238/U-234 activity ratio calculated for surface water samples taken from the Clearwell was 3.1 (Weston 1987), signifying the presence of depleted uranium. While not conclusive, these findings combined with pH data (Section 15.4.1.4) indicate that the Clearwell is a likely source of the observed uranium contamination found in Well 1031.

Relatively low total uranium concentrations occur in groundwater samples from 1000 Series wells located near the sanitary landfill, Lime Sludge ponds, and K-65 and metal oxide silos (Figures 15-5 and 15-9). These and 15-9) waste storage units may not be a current source of uranium to the glacial overburden. Total uranium concentrations ranging from 196 to 337 ug/L are found in groundwater samples from Well 1032, however, located west of the silos. As discussed in the Production and Additional Suspect Areas Work Plan (ASI/IT 1989), Well 1032 is screened in an area of fill that is located between Paddys Run and the K-65 silos. The source of the elevated uranium in Well 1032 may be the fill material rather than the silos.

The U-238/U-234 activity ratios of waste material in the K-65 silos and the metal oxide silo are 0.9 and 1.0, respectively, based on average values of U-238 and U-234 obtained during 1989 sampling by WMCO. The U-238/U-234 activity ratio of groundwater samples from 1000 Series wells located in the vicinity of the silos is approximately 1.0. However, a definitive relationship cannot be shown for the elevated uranium concentrations, since the observed U-238/U-234 activity ratio of almost all perched groundwater samples outside the Waste Pit Area is approximately 1.0.

Figures presented in Appendix N show the total uranium concentration in groundwater plotted versus time and groundwater elevation for 1000 Series wells located in the Waste Storage Area. The total uranium data are from RI/FS and 1989 RCRA sampling. Based on the Appendix N figures, it appears that the uranium concentrations observed in 1000 Series wells are not directly related to groundwater elevation. These figures indicate that the total uranium concentration of groundwater in Wells 1042 and 1073 has increased during the period of observation, and the total uranium concentration of groundwater in Well 1082 has decreased. These trends are not conclusive, however.

Other Areas

Although still statistically elevated, the total uranium concentration of perched groundwater samples from wells located outside the Production and Waste Storage areas are generally much lower than the concentration observed in samples from wells located inside these areas. Total uranium data from 1000 Series wells located in the southern half of the FMPC reservation, and Wells 1011, 1012, 1064, and 1124 located to the north and east of the Production Area, are discussed in this section. Total uranium concentrations observed in perched groundwater samples collected from these wells were all statistically elevated relative to background, with the exception of the concentrations occurring at Well 1012. Total uranium concentrations ranged from undetected, at an MDL of 1 ug/L, to 203 ug/L (Figure 15-5). However, the concentration of 203 ug/L observed during second quarter 1988 sampling of Well 1046 and the concentration of 198 ug/L observed during first quarter 1989 sampling of Well 1020 are outliers. Excluding the outliers, the total uranium concentration range observed in groundwater samples from all 1000 Series wells that are located outside the Production and Waste Storage areas is less than 1 to 80 ug/L.

The U-238/U-234 activity ratio calculated for the fly ash pile and Southfield wastes ranged from 1.0 to 2.8. The U-238/U-234 activity ratio of groundwater in 1000 Series wells in the vicinity of these waste storage units was approximately 1.0. The activity ratio observed in groundwater samples from Well 1047 was consistently less than one. A relationship between elevated uranium concentrations in perched groundwater and the fly ash and Southfield wastes cannot, therefore, be shown based on the uranium isotopic data available.

Uranium versus time plots for Wells 1020, 1060, and 1124 are presented in Figures O-2, O-5, and O-6. The observation period for Wells 1020 and 1060 is February 1985 to February 1989, and for Well 1124 it is January 1982 to January 1989. Sources of the data are the Task C Report (Dames and Moore 1985), RCRA Groundwater Monitoring Reports Rounds 1-5 (Dames and Moore 1986a, 1986b, 1987a, 1987b, 1987c), and RI/FS and 1989 RCRA sampling. NLO environmental

monitoring data (1983 to 1985) provided additional data for Well 1124. Figure O-2 shows that, except for the outlier of 198 ug/L discussed above, the total uranium concentration of perched groundwater at Well 1020, located north of Southfield, has remained relatively constant during the period of observation. The concentration of uranium in perched groundwater at Well 1060, located south of the FMPC, has generally decreased during the observation period (Figure O-5). Total uranium concentrations observed in groundwater samples from Well 1124 have fluctuated during the observation period (Figure O-6), with concentrations recorded during the most recent sampling generally lower than those occurring in earlier sampling. Well 1124 is located to the northeast of the Production Area.

Thorium

Thorium-bearing wastes are present in the Waste Storage Area, and materials containing thorium are stored in the Production Area. Relatively high thorium concentrations were observed in waste solids from Pits 1, 2, 3, and 5, the Clearwell, and Silos 1 and 2 (Weston 1987; Dettorre et al. 1981).

The UTL computed for total thorium in perched groundwater is 13 ug/L. Total thorium detections in perched groundwater samples ranged from 14 to 27.5 ug/L. The maximum total thorium concentration observed in perched groundwater samples was from the third quarter 1988 sampling of Well 1021. Statistically elevated total thorium concentrations were observed in perched groundwater samples from Wells 1010, 1021, 1022, 1073, and 1084 (Table M-2). However, only at Wells 1021 and 1022 was total thorium detected during more than one sampling round.

Statistically elevated Th-228, Th-230, and Th-232 activity concentrations were observed in perched groundwater samples from several 1000 Series wells. As expected, most isotopic thorium detections were also one-time, low-level occurrences; detections are not repeated in subsequent sampling rounds. As will be indicated below, the presence and level of individual thorium isotopes at specific well locations was on occasion inconsistent with the pattern of detection of total thorium. This is due to the variance in the analytical results combined with the fact that most readings were close to either the laboratory detection limit or the UTL. In other words, analytical uncertainty of many sample results spanned the UTL criterion; thus, the results would not be considered statistically elevated.

The UTL for Th-228 in perched groundwater is 1.5 pCi/L. Th-228 activity concentrations in perched groundwater samples ranged from less than the MDL of 1 pCi/L to 4.4 pCi/L, with the

maximum occurring in a sample collected from Well 1038 during fourth quarter 1989 RCRA sampling. Among the wells with Th-228 activity concentration greater than the UTL, only in Wells 1022, 1035, and 1055 was Th-228 detected during more than one sampling round (Appendix E).

The UTL for Th-230 in perched groundwater is 1.7 pCi/L. Th-230 activity concentrations in perched groundwater samples ranged from less than the MDL of 1 pCi/L to 7.76 pCi/L, with the maximum occurring in a sample collected from Well 1046 during fourth quarter 1989 RCRA sampling. Outliers of 23.3 and 10.3 pCi/L were observed in the first quarter 1989 sampling of Well 1073 and the third quarter 1988 sampling of Well 1077, respectively. Among the wells with statistically elevated Th-230 activity concentrations, only at Wells 1021, 1022, 1035, 1046, 1074, and 1076 was Th-230 detected during more than one sampling round (Appendix E).

The UTL for Th-232 in perched groundwater is 1 pCi/L. Th-232 activity concentrations in perched groundwater samples ranged from less than the MDL of 1 pCi/L to 9.8 pCi/L, with the maximum occurring in a sample collected from Well 1021 during second quarter 1988 sampling. Outliers of 23.1 and 9.9 pCi/L were observed in second quarter 1988 samples from Wells 1022 and 1073, respectively. Only at Wells 1021, 1022, 1073, and 1084 was Th-232 detected during more than one sampling round (Appendix E).

The Waste Storage Area appears to be a source of thorium for perched groundwater based on the location of statistically elevated total and isotopic thorium detections. Due to the insolubility of thorium and the low concentrations detected, it is unlikely that detectable levels of thorium would migrate to groundwater in the Great Miami Aquifer below the Waste Storage Area.

Radium-226 and Radium-228

Radium-bearing waste material is stored in the waste pits and Clearwell (Weston 1987) and in the silos (Dettore et al. 1981). Ra-226 was a significant component of the FMPC feed material that could not be completely removed during the formation of yellowcake ore concentrates (ASI/IT 1987).

Although statistically elevated Ra-226 activity concentrations were detected in perched groundwater from several 1000 Series wells, most Ra-226 detections were one-time, low-level occurrences; detections were not repeated in subsequent sampling rounds. The UTL for Ra-226 in perched groundwater is 1 pCi/L. Radium-226 activity concentrations in perched groundwater samples ranged from less than the MDL of 1 pCi/L to 5.1 pCi/L, with the maximum activity concentration

occurring in a sample collected from Well 1064 during second quarter 1989 sampling. Only at Wells 1020, 1031, 1053, 1073, and 1075 was Ra-226 detected during more than one sampling round (Appendix E). Based on the locations of 1000 Series Wells with multiple detections of Ra-226, the Waste Storage Area and possibly the Production Area are sources of radium for perched groundwater in the glacial overburden.

No statistically elevated Ra-228 activity concentrations are observed in perched groundwater samples taken from 1000 Series wells. The UTL for Ra-228 in perched groundwater is 5.1 pCi/L. All Ra-228 detections were one-time occurrences and ranged from 3.1 to 5 pCi/L, with the maximum occurring in a sample collected from Well 1035 during second quarter 1988 sampling.

Strontium-90

Strontium-90 (Sr-90) is a product of nuclear fission that was contained in reprocessed nuclear fuel material received by the FMPC (ASI/IT 1987). Based on radiochemical analysis of waste solids, Sr-90 is present in the Waste Storage Area, particularly in Pits 3, 5, and the Clearwell (Weston 1987).

Sr-90 is not detected in any of the background water samples from the perched zone (Table 15-1). Therefore, the UTL for Sr-90 in perched groundwater is set at the MDL of 5 pCi/L. The only occurrences of Sr-90 in perched groundwater are one-time detections of 6.0 and 5.3 pCi/L from third quarter 1988 sampling of Well 1032 and first quarter 1989 sampling of Well 1076, respectively. Both of these detections are considered statistically elevated, even though close to the MDL.

The isolated, nonrecurrent nature of Sr-90 detections suggests that the FMPC and associated waste storage units are not contributing significant amounts of the radionuclide to the glacial overburden groundwater.

Technetium-99

Technetium-99 (Tc-99) is a fission product detected in solids stored in the FMPC waste pits (Weston 1987). The UTL for Tc-99 in perched groundwater is 30 pCi/L. The range of Tc-99 detected is 30.3 to 2,805 pCi/L, with the maximum occurring in a sample collected from Well 1073 near Pits 1 and 3 during third quarter 1989 sampling. Statistically elevated Tc-99 activity concentrations are observed in perched groundwater samples from Wells 1025, 1054, 1073, 1075, 1079, 1082, and 1083. Repeated detections were observed in samples from Wells 1025, 1073, 1075, and 1083 located in the Waste Storage Area (Appendix E).

Based on the repeated detections of Tc-99 in groundwater samples and the magnitude of the levels found in some wells, the Waste Storage Area is considered to be a source of Tc-99 to groundwater in the glacial overburden groundwater.

Other Radionuclides

No detections of cesium-137 (Cs-137), neptunium-237 (Np-237), plutonium-238 (Pu-238), plutonium-239/240 (Pu-239/240), or ruthenium-106 (Ru-106) occurred in RI/FS or 1989 RCRA perched groundwater samples.

15.4.1.2 Metals

Aluminum

Aluminum concentration data were collected during the second quarter 1988 RI/FS sampling and second and third quarter 1989 RCRA sampling. Because of the limited data, a statistical analysis of aluminum concentration data was not performed.

Levels of aluminum in perched groundwater range from 0.0144 to 2.16 mg/L, with the maximum occurring in a third quarter 1989 sample from Well 1081 north of Pit 6 near the Sanitary Landfill. Levels greater than 0.25 mg/L, which is twice the average aluminum concentration in background wells (Table 15-1), were observed in perched groundwater samples from Wells 1025, 1028, 1052, 1073, 1080, 1081, 1082, and 1083. Excluding Well 1052, these wells are all located in the vicinity of the Waste Storage Area. Well 1052 is located northeast of the Waste Storage Area.

Based on the limited data available, it cannot be determined whether the FMPC and associated waste storage units are a significant source of aluminum to groundwater in the glacial overburden.

Arsenic

The UTL for arsenic in perched groundwater is 0.004 mg/L. Excluding outliers, detections of arsenic ranged from 0.002 to 0.122 mg/L. The maximum arsenic concentration observed is from a third quarter sample from Well 1040, which is located off-FMPC property to the northwest. Concentrations of arsenic which are statistically elevated relative to background occurred at Wells 1010, 1011, 1078, and 1080 located within the FMPC reservation, and Wells 1040 and 1058, located northwest of the reservation in the Shandon Tributary (Figure M-2). The locations of these wells are shown in Figure 15-5.

Because detections of arsenic were isolated, nonrecurrent, and of low concentration, it appears that the FMPC and associated waste storage units are not contributing arsenic to groundwater in the

glacial overburden. The highest levels of arsenic were detected in an off-FMPC well not hydraulically connected to the Waste Storage or Production areas.

Barium

Barium chloride, BaCl, is a documented waste component of Waste Pit 4 (ASI/IT 1987). The UTL for barium in perched groundwater is 0.125 mg/L. Detected barium concentrations ranged from 0.025 to 0.459 mg/L, with the maximum level occurring in a second quarter 1988 sample from Well 1039 located in the southwest corner of the Waste Storage Area near the Lime Sludge ponds. Statistically elevated barium concentrations occurred in perched groundwater samples from many 1000 Series wells located in and around the Waste Storage Area (Table M-2). Statistically elevated barium concentrations were also observed in Production Area Well 1053 and in Shandon Tributary Wells 1040 and 1058, which are located off-FMPC property.

Elevated barium concentrations in perched groundwater beneath the Waste Storage Area may be related to the waste material in this area. Elevated barium concentrations occurring in Wells 1039 and 1040 may be related to the material stored in the unlined Lime Sludge ponds. However, because two off-FMPC 1000 Series wells also showed statistically elevated barium concentrations, the natural variability of barium in perched groundwater may be greater than predicted by background data and there may be no relationship between the FMPC waste material and the occurrence of barium in 1000 Series wells.

Cadmium

The UTL for cadmium in perched groundwater is 0.011 mg/L. Excluding outliers, the range of detected cadmium concentrations was 0.002 to 0.031 mg/L. The maximum cadmium concentration occurred in perched groundwater in the first quarter 1989 sampling of Well 1025 located north of Pit 5 and the Waste Storage Area. Statistically elevated concentrations of cadmium did not occur in perched groundwater samples taken from 1000 Series wells during RI/FS sampling or third quarter 1989 RCRA sampling.

Based on the paucity of cadmium detections and the low concentrations when detected, the FMPC and associated waste storage units do not appear to contribute significant amounts of cadmium to the groundwater in the glacial overburden.

Calcium

During FMPC operations, lime (calcium oxide) was utilized to neutralize process waste solids (Battelle 1981). Fly ash, which contains calcium, is also a documented material stored in the

Waste Storage Area, and spent lime from FMPC Water Treatment Plant operations is stored in the Lime Sludge ponds (ASI/IT 1987).

The UTL for calcium in perched groundwater is 141 mg/L. Excluding outliers, the range of calcium concentrations detected is 12.9 to 1800 mg/L, with the maximum concentration occurring in first quarter 1989 sampling of Well 1075 located south of Pit 5. Statistically elevated calcium concentrations occur in groundwater samples from several 1000 Series wells located in the Waste Storage Area and from Well 1016, which is located immediately south of the Inactive Fly Ash area.

It appears that the calcium-bearing wastes stored in the Waste Storage Area are contributing calcium to the glacial overburden groundwater. The elevated calcium concentration observed at Well 1016 may be due to contributions from the fly ash in the Inactive Fly Ash area.

Chromium

Groundwater samples collected during the RI/FS and 1989 RCRA programs were analyzed for total chromium. The UTL for total chromium in perched groundwater is 0.035 ug/L. Excluding outliers, the range of detected total chromium is 0.0027 to 0.066 mg/L. The maximum total chromium concentration found in perched groundwater samples is from first quarter 1989 sampling of Well 1025. Only three 1000 Series wells, Wells 1025, 1028, and 1030, showed statistically elevated chromium concentrations relative to background (Table M-2).

Most chromium detections were isolated and of low concentration; therefore, the FMPC and associated waste storage units do not appear to be significant sources of chromium to groundwater in the glacial overburden.

Copper

The UTL for copper in perched groundwater is 0.043 mg/L. Excluding outliers, the range of detected copper concentrations is 0.01 to 0.06 mg/L; the maximum concentration occurred in a first quarter 1989 sample from Well 1075. Statistically elevated concentrations of copper were not present in perched groundwater samples taken from 1000 Series wells during the RI/FS or 1989 RCRA sampling.

Based on the isolated, nonrecurrent, and low-level nature of copper detections in perched groundwater, the FMPC and associated waste storage units are not significant sources of copper to the glacial overburden.

Iron

The UTL for iron in perched groundwater is 4.2 mg/L. However, the coefficient of variation for iron exceeds 1.0 in groundwater samples from the background wells. This implies that the background data are not normally distributed. Dissolved iron concentrations are often bimodally distributed, since iron solubility is highly sensitive to the level of dissolved oxygen in groundwater. In oxygenated water, dissolved iron will probably not be detected; however, in anaerobic waters containing no dissolved oxygen, dissolved iron will be present in detectable concentrations.

Detected iron concentrations ranged from 0.005 to 27 mg/L. The maximum iron concentration was observed in the first quarter 1989 sampling of Well 1011 located northwest of the Waste Storage Area. High iron concentrations, such as detected in Well 1011, indicate that iron attributed to the dissolved form may actually be present in groundwater in colloidal form. Colloids in groundwater can enhance the mobility of other metals and radionuclides.

Since almost all iron detections in perched groundwater samples were below the UTL, it appears that the FMPC and associated waste storage units are not significant sources of iron to groundwater in the glacial overburden.

Lead

The UTL for lead in perched groundwater is 0.007 mg/L. Detected lead concentrations ranged from 0.002 to 0.118 mg/L, and the maximum concentration occurred in a second quarter 1989 sample from Well 1064, located near the eastern boundary of the Production Area. Statistically elevated lead concentrations in perched groundwater were observed in Wells 1010, 1029, 1048, 1064, 1079, and 1080 (Table M-2). However, only at Wells 1010, 1048, 1079, and 1080 was lead detected during more than one round of sampling (Appendix E).

Based on the isolated, nonrecurrent, and low-level nature of lead detections in most perched groundwater samples, it appears that the FMPC and associated waste storage units have not contributed significant levels of lead to the perched zones in the glacial overburden.

Magnesium

Magnesium fluoride (MgF_2) is among the waste materials disposed in the Waste Storage Area (Battelle 1981 and ASI/IT 1987). The UTL for magnesium in perched groundwater is 57.98 mg/L. The range of detected magnesium concentrations is 5.68 to 698 mg/L. The maximum concentration was observed in a third quarter 1988 sample from Well 1075, located south of Pit 5. Magnesium concentrations in perched groundwater were statistically elevated relative to background at several

1000 Series well locations in and around the Waste Storage Area (Table M-2). Magnesium concentrations in samples from these 1000 Series wells range from 70 to 698 mg/L.

Based on the distribution of elevated magnesium concentrations, the Waste Storage Area is a likely source of magnesium to perched groundwater in the underlying glacial overburden.

Manganese

The UTL for manganese in perched groundwater is 0.150 mg/L. Manganese, like iron, is redox sensitive and may be bimodally distributed. Detected manganese concentrations ranged from 0.003 to 38 mg/L in perched groundwater. The maximum manganese concentration occurred in a third quarter 1988 sample from Well 1075, and was much higher than other manganese concentrations observed during RI/FS and 1989 RCRA sampling. Groundwater samples from several 1000 Series wells located in and around the Waste Storage Area contained statistically elevated concentrations of manganese (Table M-2).

Waste material may be the source of the statistically elevated manganese concentrations found in groundwater samples collected in and around the Waste Storage Area. For Well 1075, this almost certainly is the case. However, other statistically elevated manganese concentrations may be due to dissolution of manganese-bearing minerals naturally present in the glacial overburden. RI/FS analyses of overburden solids indicate that greater than 3.0 mg/L of manganese can be leached from the solids using an acidic leaching procedure. Rainfall infiltration through waste material may produce localized areas of chemically reduced and possibly acidic groundwater in the perched water-bearing zone. Naturally occurring manganese-bearing solids could become unstable in these areas, dissolve, and contribute manganese to the perched groundwater.

Mercury

The UTL for mercury in perched groundwater is 0.00037 mg/L. Excluding outliers, detections of mercury in 1000 Series wells ranged from 0.0002 to 0.0012 mg/L. The maximum mercury concentration in perched groundwater samples was from first quarter 1989 sampling of Wells 1038 and 1082.

Statistically elevated mercury concentrations occurred in perched groundwater from several 1000 Series wells which are located on the FMPC reservation. However, among these wells, mercury was detected in more than one sampling round only at Wells 1020, 1064, and 1073. These wells are widely spaced throughout the FMPC and cannot be related to a single source. Based on the distribution of wells in which statistically elevated concentrations of mercury were observed, it

cannot be determined whether the FMPC and associated waste storage units are a source of mercury to groundwater in the glacial overburden.

Molybdenum

The UTL computed for molybdenum in perched groundwater is 0.02 mg/L. Detections of molybdenum ranged from 0.008 to 0.678 mg/L, with the maximum concentration occurring in a fourth quarter 1988 sample from Well 1031. Statistically elevated molybdenum concentrations occurred in perched groundwater samples from 1000 Series wells located in the Waste Storage Area, Production Area, and off-FMPC to the northwest. Repeated detections of statistically elevated molybdenum occurred in wells located in the Waste Storage Area and Production Area. The highest statistically elevated molybdenum concentrations, greater than 0.030 mg/L, were observed in Wells 1031, 1038, 1073, 1075, 1082, and 1084 (Table M-2), which are all located in the Waste Storage Area.

Based on the distribution of 1000 Series wells showing statistically elevated concentrations of molybdenum, it appears that the Waste Storage Area and possibly the Production Area are sources of the low levels of molybdenum observed in perched groundwater. However, the detection of statistically elevated molybdenum in wells located northwest of the FMPC reservation suggests that the natural variation of molybdenum in perched groundwater is greater than that accounted for in the tolerance limit calculation for background conditions.

Nickel

The UTL for nickel in perched groundwater is 0.026 mg/L. Detected nickel concentrations ranged from 0.0138 to 0.981 mg/L, with the maximum concentration observed in a second quarter 1989 sample from Well 1028 located in the northwest corner of the Waste Storage Area. Statistically elevated concentrations of nickel were observed in perched groundwater samples from several 1000 Series wells located in and around the Waste Storage Area and from Well 1064 along the eastern boundary of the Production Area. The highest statistically elevated nickel concentrations in perched groundwater, greater than 0.1 mg/L, were observed in Waste Storage Area Wells 1028, 1030, 1073, 1074, and 1075. The locations of these wells are shown in Figure 15-5.

The waste material in the Waste Storage Area appears to be the source of the statistically elevated nickel concentrations observed in perched groundwater in the underlying glacial overburden.

Potassium

The UTL for potassium in perched groundwater is 33.1 mg/L. Potassium concentrations (excluding outliers) ranged from 0.34 to 455 mg/L in perched groundwater. The maximum potassium concentration occurred in a third quarter 1989 sample from Well 1031. Statistically elevated concentrations of potassium were observed only in perched groundwater samples from Wells 1031 and 1075 (Table M-2).

The waste material in the Waste Storage Area appears to be the source of the elevated potassium concentrations observed at Wells 1031 and 1075. However, elevated potassium concentrations are not widespread in groundwater in the glacial overburden beneath the Waste Storage Area.

Selenium

The UTL for selenium in perched groundwater is 0.002 mg/L. Detected selenium concentrations ranged from 0.002 to 0.019 mg/L, with the maximum concentration occurring in a second quarter 1988 sample from Well 1055 in the Production Area. Statistically elevated selenium concentrations were observed in perched groundwater samples from several 1000 Series wells which are located within the FMPC reservation. However, only at Wells 1016 and 1046 was selenium detected more than once. Figure 15-5 shows that these wells are located in the Inactive Fly Ash Storage Area.

Based on the isolated, nonrecurrent, and low-level nature of selenium detections, the FMPC and associated waste storage areas are probably not contributing significant levels of selenium to the groundwater in the glacial overburden. However, the repeated low-level selenium detections observed at Wells 1016 and 1046 may be due to the proximity of these wells to the Inactive Fly Ash Storage Area. Selenium contamination of groundwater by fly ash has been documented by Spencer and Drake (1987).

Silver

The UTL for silver in perched groundwater is 0.050 mg/L. Excluding outliers, detected silver concentrations ranged from 0.01 to 0.174 mg/L. The maximum concentration was from second quarter 1988 sampling of Well 1022. The silver concentration of perched groundwater samples was statistically elevated only at Well 1022.

Based on the lack of statistically elevated silver concentrations and the isolated, nonrecurrent, and low-level nature of the detections, the FMPC and associated waste storage units are probably not contributing significant amounts of silver to perched groundwater in the glacial overburden.

DRAFT

Sodium

Sodium carbonate, Na_2CO_3, was used at the FMPC to treat the organic solvent utilized in the uranium extraction process (Harrington and Ruehle 1959). Sodium hydroxide, NaOH, was used to adjust the pH of aqueous waste streams resulting in waste sludges (Battelle 1981). Sodium-bearing residuals from these processes are present in the Waste Storage Area.

The UTL for sodium in perched groundwater is 66.2 mg/L. Detected sodium concentrations ranged from 1.6 to 1300 mg/L in perched groundwater, with the maximum concentration occurring in a first quarter 1989 sample from Well 1075. Statistically elevated sodium concentrations occurred in perched groundwater samples from eight 1000 Series wells in the Waste Storage Area (Table M-2). Statistically elevated concentrations also occurred in Well 1012, which is located on the hillside to the north of the Production Area. Sodium concentrations were repeatedly greater than 100 mg/L in samples from the eight Waste Storage Area wells and Well 1012.

The Waste Storage Area appears to be the source of the elevated sodium concentrations observed in perched groundwater in the underlying glacial overburden. Well 1012 is located near a paved road and may be impacted by runoff containing dissolved salts from road deicing. Alternatively, groundwater sampled at Well 1012 may be partially derived from bedrock. As the bedrock is predominantly shale (Section 11.1), groundwater in contact with the bedrock may contain relatively high sodium concentrations.

Vanadium

Vanadium concentration data were collected only during the second quarter 1988 RI/FS, second quarter 1989 RCRA, and third quarter 1989 RCRA sampling rounds. Because of the limited data, a statistical analysis of vanadium concentration data was not performed.

The MDL for vanadium in perched groundwater samples is 0.01 mg/L. Detected vanadium concentrations ranged from 0.0043 to 0.51 mg/L, with the maximum concentration occurring in a third quarter 1989 RCRA sample from Well 1080.

Based on the limited data, it is difficult to determine whether the FMPC and the associated facilities are contributing vanadium to perched groundwater in the glacial overburden.

Zinc

Zinc concentration data were also collected during the second quarter 1988 RI/FS, second quarter 1989 RCRA, and third quarter 1989 RCRA sampling rounds. Because of the limited data, a statistical analysis of zinc concentration data was not performed.

The MDL for zinc in perched groundwater samples is 0.01 mg/L. Detected zinc concentrations ranged from 0.0048 to 0.501 mg/L, with the maximum concentration observed in a third quarter 1989 RCRA sample from Well 1081. Levels of zinc greater than 0.04 mg/L occurred in perched groundwater samples from Wells 1029, 1052, 1048, 1075, 1076, 1081, and 1082 (Appendix E).

Based on the limited data, it appears that the waste storage units are contributing zinc to perched groundwater in the glacial overburden.

15.4.1.3 Organics

Total Organic Halides

Total organic halides (TOX) is a parameter that indicates the presence of halogenated hydrocarbons in groundwater. Chlorinated solvents were used in degreasing operations at the FMPC, and the FMPC waste stream included dry cleaning sludges containing tetrachloroethene (ASI/IT 1987).

The UTL for TOX in perched groundwater is 0.069 mg/L. Detected TOX concentrations ranged from 0.0039 to 0.896 mg/L, with the maximum TOX concentration occurring in the first quarter 1989 sample from Well 1031 located south of the Clearwell. Statistically elevated concentrations of TOX occurred in perched groundwater samples from Wells 1011, 1016, 1019, 1031, 1073, and 1075 (Table M-2). TOX was repeatedly detected only at Wells 1019, 1031, 1073, and 1075 (Appendix E). Concentrations detected in these four wells ranged from 0.09 to 0.896 mg/L.

The distribution and occurrence of TOX in perched groundwater samples indicate that the Waste Storage Area is a source of halogenated hydrocarbons.

Volatile and Semivolatile Organics, Pesticides, and PCBs

Table 15-5 presents the concentrations of organic compounds detected during RI/FS and 1989 RCRA sampling of groundwater from 1000 Series wells. Sampling was conducted during the second quarter 1988 through third quarter 1989. Compounds associated with method blank contamination are excluded in Table 15-5.

TABLE 15-5

DETECTED ORGANIC COMPOUNDS (ug/L) IN GROUNDWATER FROM 1000 SERIES WELLS[a] SECOND QUARTER 1988 TO THIRD QUARTER 1989

Well	1010	1016	1019	1025		1031	1052
Sampling Quarter	2nd/88	2nd/88	2nd/88	2nd/88	3rd/89	3rd/89	3rd/89
Volatile Organic Compounds							
Bromochloromethane	49.7						
1,1,1-Trichloroethane		2 J		2 J			
1,1-Dichloroethane			5			30	
Total 1,2-Dichloroethene			25				
Acetone[b]			7 J			15	2 J
Methylene chloride			18				
Vinyl chloride			2 J			16	
Diethyl phthalate					3 J		
1,1-Dichloroethene						5	
1,2-Dichloroethane						2 J	
2-Butanone						3 J	
4-Methyl-2-pentanone						6 J	
Benzene						3 J	
Chloroform						2 J	
Tetrachloroethene						300 E	
Toluene						2 J	
Total xylenes						2 J	

See footnotes at end of table.

TABLE 15-5
(continued)

Well	1010	1016	1019	1025		1031	1052
Sampling Quarter	2nd/88	2nd/88	2nd/88	2nd/88	3rd/89	3rd/89	3rd/89
Volatile Organic Compounds							
Trichloroethene						530 E	
trans-1,2-Dichloroethene						18	
Carbon disulfide							4 J
Semivolatile Organic Compounds							
Butyl benzyl phthalate				1 J			
2,3,4,6-Tetrachlorophenol						4 J	
2-Nitrophenol						12	
4-Nitrophenol						11 J	
N-Nitrosodiethylamine						4 J	
N-Nitrosodiephenylamine						3 J	2 J
Naphthalene						2 J	
Pentachlorophenol						3. J	
2-Methyl-4,6-dinitrophenol						6 J	
Phenol							2 J
Diethyl phthalate							
bis(2-Ethylhexyl)phthalate							

See footnotes at end of table.

TABLE 15-5
(continued)

Well	1073	1074	1078	1079		1080	1083
Sampling Quarter	2nd/88	2nd/88	2nd/88	2nd/88	3rd/89	3rd/89	3rd/89
Volatile Organic Compounds							
1,1-Dichloroethane	76					2 J	
Acetone[b]					3 J		
Methylene chloride							12
1,1,1-Trichloroethane	5 J						
1,1-Dichloroethene	2 J						
Trichloroethene						29	
Carbon disulfide		2 J		130			
Semivolatile Organic Compounds							
Butyl benzyl phthalate	3 J		3 J				
N-Nitrosodiphenylamine			2 J				
Diethyl phthalate				4 J			
bis(2-Ethylhexyl)phthalate				4 J	3 J		6 J

[a]This table does not include compounds found in method blanks. Second and third quarter 1989 results are from RCRA sampling program. Second quarter 1988 results are from RI/FS sampling.

[b]Also known as 2-Propanone.

Qualifier:

J - Compound detected below the contract required quantitation limit. The value presented is an estimate.

E - Concentration exceeds calibration range of GC/MS instrument.

Prior to fourth quarter 1987, RCRA groundwater samples were collected using methods and QA/QC procedures inconsistent with the subsequently approved RI/FS QAPP. In addition, the analytical work done on these samples, including fourth quarter 1984 samples, were performed by a laboratory not participating in the U.S. EPA Contract Laboratory Program (CLP). CLP methods,including QA/QC protocols, were not followed. Table 15-6 lists organic compounds detected during the early RCRA sampling; i.e., third quarter 1985 through fourth quarter 1987. The following discussion of detected organic compounds is based only on data collected during RI/FS and 1989 RCRA sampling, which were generated using procedures outlined in the QAPP (Table 15-5).

Volatile organic compounds (VOCs) and semivolatile organic compounds (semi-VOCs) were detected in several 1000 Series wells. No pesticides or polychlorinated biphenyls (PCBs) were detected in perched groundwater samples.

Several volatile compounds were detected during a second quarter 1988 sampling of Well 1019. Concentrations ranged from 2 to 25 ug/L. The detected compounds are 1,1-dichloroethane, total 1,2-dichloroethene, acetone, methylene chloride, and vinyl chloride (Table 15-5).

Various volatile and semivolatile compounds were detected in Well 1031 from third quarter 1989 sampling (Table 15-5). Concentrations of these organic compounds ranged from 2 to 530 ug/L. This well is located in the vicinity of the Clearwell and may intercept leakage from this source.

Detections of volatile compounds 1,1-dichloroethane, 1,1,1-trichloroethane, 1,1-dichloroethene, and semivolatile butyl benzyl phthalate occurred at Well 1073 during the second quarter 1988 RI/FS sampling (Table 15-5). Concentrations ranged from 2 to 76 ug/L.

Given the detections of various volatile and semivolatile compounds at Wells 1019, 1031, and 1073, which are located in the Waste Storage Area, this area may be contributing trace amounts of organic compounds to the underlying perched groundwater.

15.4.1.4 General Chemistry

Alkalinity

The UTL for alkalinity, reported as bicarbonate, is 549 mg/L. Alkalinity measurements in perched groundwater ranged from 196 to 1161 mg/L, with the maximum measurement observed in a second quarter 1988 sample from Well 1075. Statistically elevated alkalinity measurements occurred in perched groundwater samples from several 1000 Series wells which are located on the FMPC

TABLE 15-6

DETECTED ORGANIC COMPOUNDS (ug/L) IN GROUNDWATER FROM 1000 SERIES WELLS[a] THIRD QUARTER 1985 TO FOURTH QUARTER 1987

Well	1012					1019			1020	1022		1060	
Sampling Quarter	3rd/85	2nd/87	2nd/87[b]	3rd/85	3rd/86	4th/86	2nd/87	4th/87	3rd/86	3rd/86	4th/86	3rd/86	4th/86
Volatile Organic Compounds													
Benzene	14.8												
Xylene	11.8												
Acetone[c]		84.4	694							73	4770	88	4480
Cyclohexane		15	10.2										
1,1-Dichloroethane				3.4	17.1	1	15.5	15					
Methylene chloride				(trace)									
cis-1,2-Dichloroethene					5.1		25.8	29.5					
Trichloroethene						0.5							
Tetrachloroethene						8.2							
Bromoform						7		0.91					
Chloroethene							0.8	0.87					
1,1,1-Trichloroethane									16.1				
2-Propanol										21.5	8500		1420
1,1,2,2-Tetrachloroethane								1.54					
1,2-Dichlorobenzene								0.83					

[a]Third quarter 1985 through fourth quarter 1987 results from RCRA sampling program. Pre-Quality Assurance Project Plan analytical results; qualifiers are not included.

[b]Duplicate analysis.

[c]Also known as 2-propanone.

000037

reservation. Alkalinity measurements exceeding 600 mg/L were observed at Wells 1011, 1016, 1048, 1075, 1076, 1078, and 1081 (Appendix E). See Figure 15-5 for the locations of these wells. The Waste Storage Area appears to be the source of the elevated alkalinity in groundwater in the glacial overburden. As discussed in Section 15.4.1.2, sodium carbonate was used at the FMPC to treat the organic solvent extract, and residue from this process may be present in the waste storage pits. Elevated alkalinity levels observed at Wells 1016 and 1048 may be related to fly ash wastes which are stored in the vicinity of these wells. As alkalinities measured during the RI/FS program were total alkalinities, dissociated organic acids may be contributing to elevated alkalinity values.

Ammonia

Nitric acid was used at the FMPC to digest uranium-bearing feed material, and ammonia was used in the uranium reduction process (Battelle 1981).

The UTL for ammonia in perched groundwater is 0.614 mg/L (reported as nitrogen). Detected ammonia concentrations ranged from 0.1 to 253 mg/L, with the maximum concentration occurring in a fourth quarter 1988 sample from Well 1075. Statistically elevated ammonia concentrations occurred in Wells 1012, 1031, 1053, 1073, and 1075, which are located on the FMPC reservation and in Wells 1040 and 1058 which are located northwest of the FMPC in the Shandon Tributary. While the ammonia concentration of perched groundwater samples from Wells 1012, 1040, 1053, and 1058 ranged from 0.4 to 4.5 mg/L, the concentration at Wells 1031, 1073, and 1075 ranged from 36.6 to 253 mg/L. The latter group of wells are located in the Waste Storage Area (Figure 15-9).

Based on the high ammonia concentrations observed in wells located in the Waste Storage Area, it appears that waste material is contributing nitrogen, in the form of ammonia, to groundwater in the underlying glacial overburden.

Chloride

The UTL for chloride in perched groundwater is 45.57 mg/L. Detected concentrations of chloride ranged from 0.5 to 6300 mg/L, with the maximum value occurring in a first quarter 1989 sample from Well 1075. Statistically elevated chloride concentrations were observed in perched groundwater samples from several 1000 Series wells located in and adjacent to the Waste Storage Area. Chloride concentrations exceeding 1000 mg/L were observed in groundwater samples from Wells 1019, 1025, 1031, 1039, 1073, and 1075 (Appendix E). Statistically elevated chloride concentrations were also observed at Well 1012, located along the northern property boundary of the FMPC.

Waste material in the Waste Storage Area is probably the source of the elevated chloride concentrations observed in perched groundwater samples from the underlying glacial overburden. As previously discussed, Well 1012 may be impacted by dissolved road salt from an adjacent paved road, or groundwater sampled at this well may receive chloride as it flows through shale bedrock.

Cyanide

Cyanide concentration data were collected only during the second quarter 1988 RI/FS and second quarter 1989 RCRA sampling rounds. Because of the limited data, a statistical analysis of cyanide concentration data was not performed.

Cyanide was detected at a concentration of 0.0794 mg/L from second quarter 1989 RCRA sampling at Well 1031. Cyanide was not detected in any other perched groundwater sample collected from 1000 Series wells.

Fluoride

Fluoride was used in two steps of the uranium reduction process (Battelle 1981), and both magnesium fluoride (MgF_2), and green salt (UF_4) are waste constituents of the waste pits.

The UTL for fluoride in perched groundwater is 1.49 mg/L. Detected fluoride concentrations ranged from 0.1 to 7.25 mg/L, with the maximum value occurring in a first quarter 1989 sample from Well 1073 in the middle of the Waste Storage Area. Statistically elevated fluoride concentrations were observed in perched groundwater samples from Wells 1021, 1031, 1073, and 1075 (Table M-2). As shown in Figure 15-5, all of these wells are located in the Waste Storage Area.

Although concentrations are low, it appears that the waste pits may be the source of elevated fluoride concentrations in perched groundwater samples.

Nitrate

Nitric acid was used at the FMPC in the feed material digestion process. The Characterization Investigation Study (Weston 1987) reports that nitrate concentrations greater than 1000 mg/L were detected in surface water samples collected from Pits 5 and 6.

The UTL for nitrate in perched groundwater is 0.33 mg/L (reported as nitrogen). Detected concentrations of nitrate range from 0.012 to 197 mg/L, with the maximum concentration occurring in a first quarter 1989 sample from Well 1073. Statistically elevated nitrate concentrations were

observed in perched groundwater samples from several 1000 Series wells located in the Waste Storage Area and Wells 1016, 1054, and 1064 (Table M-2). Nitrate concentrations were repeatedly elevated only in wells surrounding the waste pits and Well 1054 adjacent to Plant 6. Nitric acid was a major component of Plant 6 operations.

The Waste Storage Area and possibly the Production Area appear to be sources of the elevated nitrate concentrations observed in perched groundwater in the glacial overburden.

pH

The tolerance interval for the pH of perched groundwater is 7.13 to 7.27. The pH of groundwater samples from many 1000 Series wells were statistically different from background. However, the two-sided pH tolerance interval may be too narrow and not representative of the actual range of pH in perched groundwater not affected by FMPC operations. The pH data from 1000 Series wells did not exhibit any spatial trends; i.e., no high or low pH plumes were observed. Only at Well 1031, where groundwater pH was consistently greater than 9, is the pH considered to be significantly different than background.

The pH of surface waters collected from Waste Pits 5 and 6 and the Clearwell ranged from 9.0 to 11.7, based on data gathered during the Characterization Investigation Study (Weston 1987). However, other than at Well 1031, abnormally high pH values were not observed in groundwater samples from wells located in proximity to the pits. This may indicate that significant buffering or dilution of waste pit leachate occurs as the leachate infiltrates into the glacial overburden. This also further indicates that Well 1031 is receiving leakage directly from the adjacent Clearwell.

Phosphorus

The UTL for total phosphorus in perched groundwater is 0.16 mg/L. Excluding outliers, detections of total phosphorus ranged from 0.02 to 7.4 mg/L, with the maximum concentration occurring in a second quarter 1988 sample from Well 1055, located in the Production Area. Statistically elevated total phosphorus concentrations occurred in groundwater samples from the majority of the 1000 Series wells which are located on the FMPC reservation. High total phosphorus concentrations of greater than 0.5 mg/L are repeatedly observed in perched groundwater samples from Wells 1025, 1035, 1038, 1064, and 1081 (Appendix E). Total phosphorus concentrations fluctuated greatly from one sampling round to the next.

The FMPC and associated waste storage units may be a source of phosphorus for perched groundwater in the underlying glacial overburden. Tributyl phosphate was used at the FMPC in the

uranium extraction process (ASI/IT 1987). However, though statistically elevated, the widespread low-level total phosphorus concentrations may also be due to natural variation.

Sulfate

The UTL computed for sulfate in perched groundwater is 198.2 mg/L. Detections of sulfate ranged from 2 to 1180 mg/L, with the maximum concentration occurring in a second quarter 1988 sample from Well 1022. Statistically elevated sulfate concentrations occurred in perched groundwater samples from several wells in the Waste Storage Area and Wells 1055 and 1064. Average sulfate concentrations of greater than 300 mg/L were observed in samples from Wells 1022, 1028, 1032, 1073, 1075, 1081, 1084, and 1089 (Table M-2). With the exception of Well 1089, all of these wells are located in the vicinity of the waste pits (Figure 15-9).

Although sulfate is not listed in reports or literature which describe the FMPC waste streams, it appears that the Waste Storage Area and possibly the Production Area are sources of sulfate for perched groundwater in the glacial overburden.

15.4.1.5 Summary

The chemistry of the perched groundwater in the glacial overburden is impacted by FMPC activities in the vicinity of the Waste Storage Area, the Production Area, and possibly the fly ash area/Southfield. Uranium concentrations exceeding background were observed in perched groundwater in the Waste Storage and Production areas. The concentration of some inorganic constituents and a small number of volatile organic compounds were also elevated in perched groundwater in the vicinity of the waste pits.

The total uranium concentration of perched groundwater samples from most 1000 Series wells was statistically higher than the UTL. Total uranium concentrations greater than 100 ug/L occurred in perched groundwater below much of the southern half of the Production Area (Figures 15-7 and 15-8), beneath isolated locations in the northern half of the Production Area (Figure 15-6), and beneath parts of the Waste Storage Area (Figure 15-5). Total uranium concentrations exceeding 1000 ug/L were observed in perched groundwater beneath parts of the southern half of the Production Area (Figures 15-7 and 15-8), and in several samples collected from wells located in the vicinity of the waste pits (Figure 15-5). Total uranium concentrations greater than 10,000 ug/L were observed in perched groundwater below the Production Area in the vicinity of Plants 2/3, 6, 8, and the Raffinate Area (Figures 15-7 and 15-8).

Total uranium concentrations occurring in 1000 Series wells located outside of the Production and Waste Storage areas were slightly elevated relative to background. However, these concentrations were lower than those in wells located in the Production and Waste Storage areas.

Based on the location of uranium-contaminated groundwater and a review of U-238/U-234 activity ratios, the FMPC and the associated waste storage units are contributing uranium to the glacial overburden. Possible sources include spills and leaks of uranium-bearing materials in the Production Area, and infiltration of leachate containing uranium from the waste pits and Clearwell. The leakage of contaminated materials from the fly ash area and Southfield is suspected, but not proven, given the available data. Leakage from the silos is not substantiated by the groundwater data.

Repeated detections of thorium and radium concentrations occurred in perched groundwater in the vicinity of the Waste Storage Area. Repeated detections of Th-228 and Ra-226 were also observed in a few 1000 Series wells located in the Production Area. This indicates that the FMPC Waste Storage and Production areas are probably contributing various radionuclides other than uranium to perched groundwater in the glacial overburden. However, concentrations of radionuclides other than uranium are relatively low, and detections were isolated in occurrence.

Statistically elevated concentrations of various chemical constituents were observed in groundwater samples from 1000 Series wells located in the vicinity of the Waste Storage Area and some 1000 Series wells located in the Production Area. These chemical constituents include alkalinity, ammonia, barium, calcium, chloride, fluoride, magnesium, manganese, molybdenum, nickel, nitrate, phosphorus, potassium, sodium, and sulfate. Elevated levels of major dissolved constituents were confirmed by high conductance measurements in perched groundwater samples from wells located in the Waste Storage Area and Wells 1010 and 1041 located along the western edge of the Production Area (Table M-2). Infiltration of soluble waste constituents and possibly leaks or spills of chemicals used in FMPC operations are considered to be the sources of the statistically elevated concentrations of inorganic constituents observed in glacial overburden groundwater in these areas.

Statistically elevated concentrations of selenium that occur in groundwater at Wells 1016 and 1046 may be related to the fly ash stored in the vicinity of these wells. Also, the statistically elevated alkalinity levels observed at Wells 1016 and 1048 may be due to the proximity of fly ash and waste materials stored in Southfield.

Elevated pH values and the presence of detectable levels of many chemicals in Well 1031 provide evidence of direct leakage from the Clearwell. However, the near-neutral pH measured in groundwater from the majority of 1000 Series wells appears to indicate that significant buffering or dilution of waste leachate occurs in the glacial overburden. Elevated total dissolved solids found in perched groundwater samples could be the result of buffering reactions between leachate and glacial overburden solids.

Elevated concentrations of sodium and chloride at Well 1012 and elevated ammonia concentrations at Wells 1012, 1040, and 1058 are likely not related to FMPC activities.

Repeated detections of TOX in perched groundwater occurred at Wells 1019, 1031, 1073, and 1075, which are all located in the vicinity of the waste pits. Various volatile and semivolatile compounds were detected or estimated in single samples from Wells 1010, 1016, 1019, 1025, 1031, 1052, 1073, 1074, 1078, 1079, 1080, 1083, and 1088 (Table 15-5). Based on the limited detections of TOX, volatiles, and semivolatile compounds, the Waste Storage Area is probably the source of localized, low-level releases of organic compounds to perched groundwater in the underlying glacial overburden.

15.4.2 Great Miami Aquifer

The 2000 Series wells are screened at the water table of the Great Miami Aquifer (Section 6.1.2.1) and are used to sample groundwater from the uppermost part of the aquifer. Contaminants can potentially infiltrate from perched groundwater in the glacial overburden to the Great Miami Aquifer. As discussed in Section 12.5, contaminant-bearing surface water can also infiltrate directly to the Great Miami Aquifer in areas where Paddys Run and tributaries, particularly the Storm Sewer Outfall Ditch, have eroded through the glacial overburden. Glacial overburden is also not present east of the FMPC in the Great Miami River flood plain, where only a layer of flood plain sediments overlies the Great Miami Aquifer. Contaminants present in surface water may also reach the Great Miami Aquifer in this area.

As discussed in Section 6.1.2.1, 3000 Series wells are screened in the Great Miami Aquifer, immediately above a semiconfining clay interbed. In those areas where the interbed was not present, 3000 Series wells were screened at the elevation where the interbed would have been expected. It was thought that the major clay interbed, if present, would hydrogeologically and geochemically retard the downward migration of any contamination. Thus, 3000 Series wells were installed to evaluate whether contamination had reached and concentrated within the portion of the Great Miami Aquifer that is immediately above the clay interbed.

The 4000 Series wells are screened immediately above the contact between bedrock and the outwash sands and gravel of the Great Miami Aquifer (Section 6.1.2.1). Where these wells are screened below the clay interbed, it is assumed there is a poor hydraulic communication with potential contaminant sources.

Data from Wells 2026, 2121, and 2122 were used to generate the UTLs for statistical comparisons of groundwater for the 2000 Series wells associated with the Ross Section of the Great Miami Aquifer. Data from Wells 3063, 3099, and 3100, in addition to the three 2000 Series wells, were used to calculate the background UTLs for statistical comparisons of groundwater from the 3000 Series wells within the same area. Data from Wells 2050, 2056, 2066, and 2105 were used to compute the UTLs for statistical comparisons of groundwater from 2000 and 3000 Series wells associated with the Shandon Tributary. Groundwater data from Wells 2036, 2057, and 2123 were used to construct the background UTLs for statistical comparisons of 2000 and 3000 Series Wells in the Dry Fork Section of the downgradient wells, and all 4000 Series downgradient wells.

15.4.2.1 Radionuclides

Uranium

Two well-defined uranium plumes are present in the Great Miami Aquifer. One plume appears to originate under the Waste Storage Area and is moving to the east. The other one, the South Plume, appears to have originated along Paddys Run and the Storm Sewer Outfall Ditch, and is extending to the south of the FMPC reservation. The South Plume appears to be predominantly due to historical releases of uranium, while contamination under the Waste Storage Area appears to be due to continuing releases.

The UTLs for total uranium are as follows:

- Ross 2000 Series 1.5 ug/L
- Ross 3000 Series 1.5 ug/L
- Shandon 2000 and 3000 Series 2.3 ug/L
- Dry Fork 2000 and 3000 Series 1.0 ug/L
- 4000 Series 1.0 ug/L

2000 Series Wells

Statistically elevated total uranium concentrations were observed in groundwater samples from many 2000 Series wells. Total uranium concentrations in groundwater samples from downgradient 2000 Series wells ranged from undetected, at an MDL of 1.0 ug/L, to 851 ug/L.

Figures L-8 through L-14 show the total uranium concentration of groundwater samples collected from 2000 Series wells during second quarter 1988 to fourth quarter 1989 RI/FS and RCRA sampling. Complete contours are not constructed because of insufficient data in Figures L-12 through L-14, which show uranium concentrations in 2000 Series wells from second to fourth quarter 1989. Figure 15-10 shows the average total uranium concentration of groundwater in 2000 Series wells calculated from the RI/FS and RCRA sampling data. These figures show that groundwater with total uranium concentrations exceeding 1 ug/L is present at the water table of the Great Miami Aquifer over a large area. This area underlies the Waste Storage and Production areas, much of the southern half of the FMPC reservation, and an area to the south of the reservation boundary.

Total uranium concentrations greater than 10 ug/L are present in groundwater at the water table in a large part of the aquifer on the west side of the reservation, beneath and immediately east of Paddys Run. As illustrated in Figure 15-9, the area exceeding 10 ug/L begins beneath the Waste Storage Area and extends southward, paralleling Paddys Run. An area of total uranium concentrations in groundwater greater than 10 ug/L is also present at the water table of the Great Miami Aquifer beneath the southeastern corner of the Production Area. A narrow area of the Great Miami Aquifer with groundwater uranium concentrations exceeding 100 ug/L at the water table extends southward from Well 2046. This area widens near the southern facility boundary and extends off the FMPC property.

Total uranium concentrations observed in 2000 Series wells located in the Waste Storage Area ranged from undetected, at an MDL of 1 ug/L, to 78 ug/L. Total uranium concentrations observed in 2000 Series wells located in the Production Area ranged from below the MDL to 36 ug/L. While most wells located in these areas showed statistically elevated total uranium concentrations, they were orders of magnitude lower than concentrations observed in perched groundwater in the Waste Storage and Production areas. This implies that during transport to the Great Miami Aquifer, the infiltrating water and associated uranium are either physically or chemically attenuated. Physical attenuation may be due to low hydraulic conductivity in the glacial overburden, resulting in very slow infiltration to the regional water table and subsequent dilution. Chemical attenuation may be due to precipitation reactions or adsorption of uranium onto overburden solids.

If dilution is the dominant reason for the decrease in uranium concentrations, then the decrease in the concentration of other constituents should be proportional to the decrease in uranium concentrations. However, this is not the case. Concentrations of other constituents in groundwater from 2000 Series wells located in the Waste Storage and Production areas were generally no more

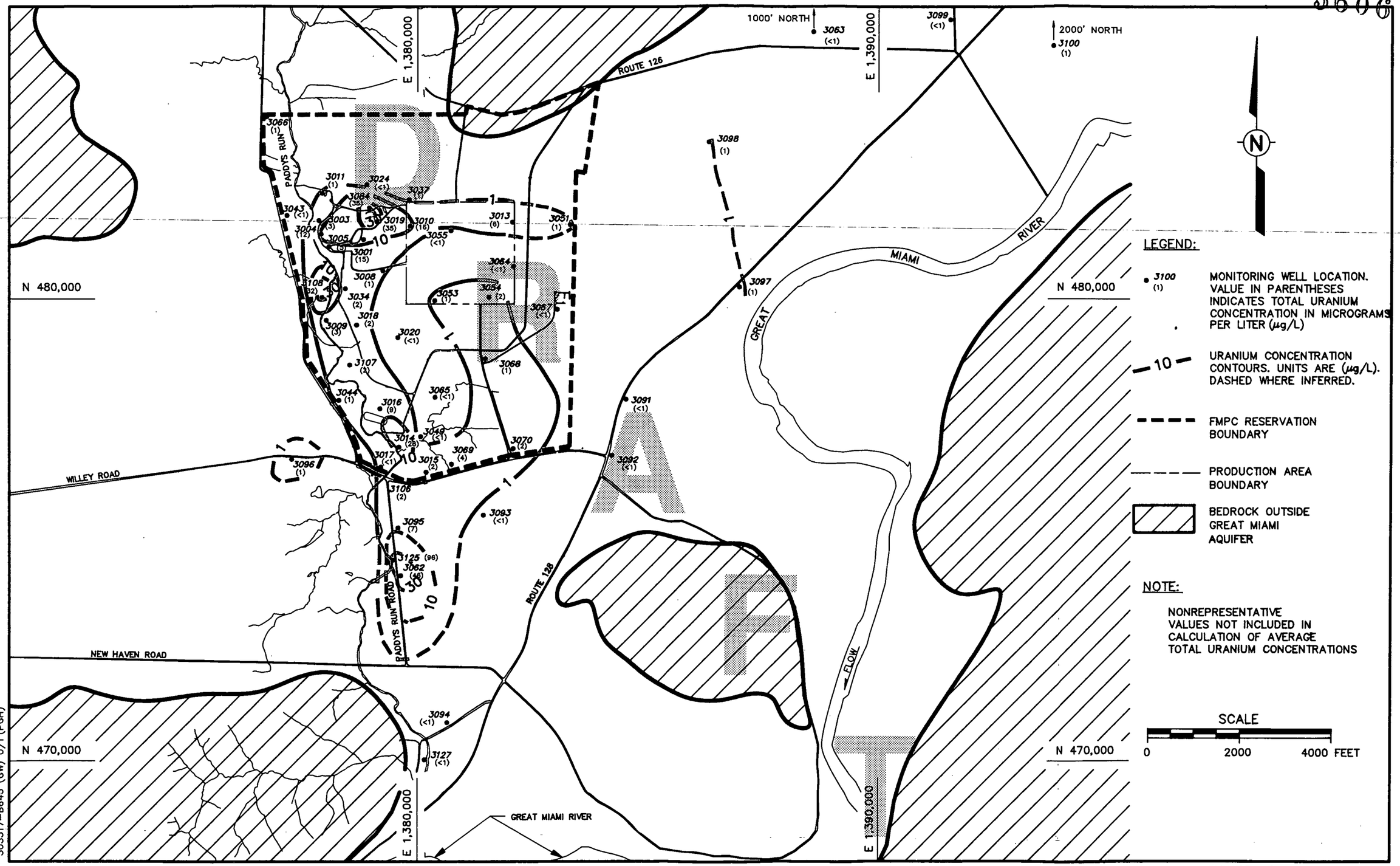
than one order of magnitude less than concentrations observed in perched groundwater. This implies that chemical attenuation of uranium is occurring in the glacial overburden or unsaturated zone of the outwash sands and gravels.

As shown in Figure 15-10, elevated total uranium concentrations were observed in Wells 2009, 2016, and 2034 located immediately east of Paddys Run. This provides confirmatory evidence that direct infiltration along the stream is a pathway for contamination reaching the Great Miami Aquifer. As discussed above, uranium concentrations may be attenuated during infiltration through the glacial overburden. However, because Paddys Run flows directly on the Great Miami Aquifer along the southern two-thirds of the facility (Section 12.5.2), uranium may directly enter the aquifer without the benefit of chemical attenuation by the glacial overburden. The U-238/U-234 activity ratio of groundwater in Wells 2009 and 2014 is similar to the ratio calculated for Paddys Run surface water. This, coupled with the high total dissolved oxygen concentrations in these wells (Figure 15-3), indicates that surface water is directly entering the Great Miami Aquifer through the Paddys Run channel. Historic sources of uranium in Paddys Run water were most likely uncontrolled runoff from the Waste Storage Area and possibly part of the Production Area, and the past discharge of groundwater pumped from test wells located in the Waste Storage Area (NLO 1972 ACCR). In addition, in November 1976, a few thousand gallons of liquid reportedly overflowed and leaked from the K-65 waste slurry trench and ran off into Paddys Run (NLO 1977 ACCR). Uncontrolled runoff from the FMPC may be an ongoing source of uranium in Paddys Run water (Section 14.1.2).

The U-238/U-234 activity ratio of groundwater samples from Wells 2009 and 2014 ranged from 1.23 to 1.59 and was consistently greater than one. Both of these wells are located immediately east of Paddys Run, in an area of the regional aquifer which receives infiltration from the stream. The average U-238/U-234 activity ratios of RI/FS surface water samples from Paddys Run Locations W-7, W-10, and W-11 (Figure 14-1), were 1.82, 2.14, and 1.40, respectively (ASI/IT 1990). The source of at least some of the uranium observed in groundwater at Wells 2009 and 2014 is therefore, likely to be Paddys Run stream water.

The highest concentrations of uranium found in the Great Miami Aquifer were observed in an elongated plume which begins near Well 2046 north of Southfield and stretches southward beyond the reservation boundary to at least Well 2061 (Figure 15-10). The elevated uranium concentrations in the South Plume are probably due to the historic discharge of uranium-bearing water into Paddys Run and the Storm Sewer Outfall Ditch, and subsequent infiltration along the Paddys Run and Storm Sewer Outfall Ditch beds. Peak discharges of contaminated water to the Storm Sewer

303317-B645 (GW) /01,100% ORG



303317-B645 (GW) 0/1 (PGH)

FIGURE 15-11
AVERAGE URANIUM CONCENTRATION
IN GROUNDWATER, 3000 SERIES WELLS
SECOND QUARTER 1988 THROUGH
FOURTH QUARTER 1989

Outfall Ditch occurred from 1958 to 1969 (NLO 1962; NLO 1965-1970). The elevated total uranium concentrations observed in groundwater from Wells 2045 and 2046 are probably not due to the discharge of uranium-bearing water from the Storm Sewer Outfall Ditch. These wells are upgradient of the area of the aquifer that would receive recharge from the Storm Sewer Outfall Ditch. Elevated uranium concentrations in groundwater from these wells may be due to infiltration from Southfield, leakage from abandoned farm drain tiles (NLO 1972 ACCR), or the past infiltration and eastward movement of uranium-bearing surface water from Paddys Run at locations west of Well 2046. The validity of the last scenario is supported by the travel time of groundwater from Paddys Run to Well 1046 in relation to the earlier periods of maximum uranium release.

Figure 15-10 shows that statistically elevated uranium concentrations occurred during RI/FS sampling of groundwater from Well 2127.

Total uranium concentrations observed in this well ranged from 6 to 37 ug/L (Appendix E). Well 2127 is located south of New Haven Road, immediately west of Paddys Run. The uranium observed in this well is most likely due to the localized infiltration of uranium-bearing surface water from Paddys Run.

Plots of groundwater total uranium concentration versus time for the period 1985 to mid-1989 are presented in Figures O-7 through O-16, Appendix O, for those 2000 Series wells with a sufficiently long record of total uranium data.. Sources of the data are the Task C Report (Dames and Moore 1985), RCRA Groundwater Monitoring Reports Rounds 1-5 (Dames and Moore 1986a, 1986b, 1987a, 1987b, 1987c), and RI/FS sampling. The longer periods of observation for Wells 2004, 2060, and 2061, from early 1982 to mid-1989 (Figures O-7, O-15, and O-16), are due to the availability of additional data (NLO 1983-1986; WMCO 1987-1989). Interim Air, Soil, Water, and Health Risk Assessment Report data (IT 1986) were also used in the concentration versus time plots for Wells 2013, 2060, and 2061 (Figures O-8, O-15, and O-16).

Uranium concentrations in groundwater have been relatively steady with time in most of the 2000 Series wells. The total uranium concentration of groundwater at Well 2015 has increased from 132 to 186 ug/L during the observation period (Figure O-10), which could reflect the transient movement of north-south trending plume that originated in Paddys Run in the 1950s and 1960s. The uranium concentration of groundwater at Well 2060, has generally ranged from 200 to 300 ug/L. However, from April to October 1987, concentrations measured at this well were greater than 300 ug/L (Figure O-15). The reason for this apparently temporary increase in uranium concentrations is not known. The total uranium concentration of groundwater at Well 2061 was

greater than 400 ug/L from early 1982 to mid-1983 (Figure O-16). Since mid-1983, the uranium concentration of groundwater at Well 2061 has been less than 400 ug/L, and generally less than 350 ug/L. Total uranium concentrations in this well have generally decreased over the period of observation, which could reflect the passage of the maximum concentration plume.
Figures N-33 to N-64 (Appendix N) again show the groundwater total uranium concentration plotted versus time for selected 2000 Series wells. Also plotted on these figures is groundwater elevation versus time to evaluate and correlation between uranium level and groundwater elevation. The total uranium data plotted are from RI/FS and 1989 RCRA sampling. The uranium concentration in groundwater at most 2000 Series wells does not appear to be related to groundwater elevation. However, the total uranium concentration at Wells 2010 and 2049 appears to be bimodally distributed (Figures N-36 and N-50). This implies that short-term changes in groundwater elevation, flow direction, or infiltration rates may greatly affect the uranium concentration of groundwater in these wells. Such a relationship would be true if the plume had a high concentration gradient in the horizontal and/or vertical directions. Well 2049 is located on the northern edge of the South Plume, and Well 2010 is located in the Waste Storage Area (Figure 15-10).

The U-238/U-234 activity ratio of groundwater samples from Well 2004 along the western edge of the Waste Storage Area ranged from 1.49 to 2.87. As discussed previously, the U-238/U-234 activity ratio of waste material in Pit 1 and the Clearwell was observed to be significantly greater than one. Possibly one or both of these waste storage units are contributing to the uranium observed in groundwater at Well 2004.

A U-238/U-234 activity ratio of 3.22 was obtained from second quarter 1988 groundwater sampling of Well 2021, which is located south of Well 1084 in the vicinity of Pits 4 and 6. This ratio is similar to that observed in samples from Well 1084. In subsequent sampling of Well 2021, uranium concentrations were much lower, and the U-238/U-234 activity ratio could not be calculated. Thus, even though evidence exists that uranium in perched groundwater in the vicinity of Pits 4 and 6 is reaching the regional aquifer, this cannot be substantiated.

3000 Series Wells

Statistically elevated total uranium concentrations were observed in groundwater samples from less than half of the 3000 Series wells. All, except three, of these wells are located inside the FMPC property boundary. The three off-FMPC wells are located in the South Plume area. Total uranium concentrations in groundwater samples from downgradient 3000 Series wells ranged from undetected, at a MDL of 0.1 ug/L, to 490 ug/L. The maximum concentration, from first quarter 1989 sampling of Well 3013, was determined to be an outlier. Excluding this outlier, the total

uranium concentration at Well 3013 from second quarter 1988 to fourth quarter 1989 ranged from 4 to 12 ug/L. Excluding outliers, the maximum total uranium concentration observed in groundwater samples from the 3000 Series wells was 218 ug/L from second quarter 1988 sampling of Well 3084 located in the Waste Storage Area near Pits 4 and 6.

Figures L-15 through L-21 show the total uranium concentration of groundwater samples collected from 3000 Series wells during second quarter 1988 to fourth quarter RI/FS and RCRA sampling. Contours are not fully developed for Figures L-19 through L-21, which show uranium concentrations from 3000 Series wells from second to fourth quarter 1989, because of insufficient data.

Figure 15-11 shows the average total uranium concentration of groundwater in 3000 Series wells calculated from the RI/FS and RCRA sampling data. A large area with total uranium concentrations greater than 1 ug/L is present in the portion of the Great Miami Aquifer sampled by 3000 Series wells. This area underlies much of the FMPC reservation and extends off-FMPC to the south. However, this area is not as extensive as the area defined by 2000 Series wells.

Several areas of the aquifer contain groundwater total uranium concentrations in excess of 10 ug/L at the horizon sampled by 3000 Series wells. Moving from north to south in Figure 15-11, these areas are located in the Waste Storage Area, around Well 3108, near the confluence of the Storm Sewer Outfall Ditch with Paddys Run, and off-FMPC in the South Plume. These areas are much smaller and more discontinuous than the areas with uranium concentrations greater than 10 ug/L that are delineated by 2000 Series wells.

Average total uranium concentrations greater than 30 ug/L in 3000 Series wells occurred only in Wells 3019 and 3084, located near the waste pits; Well 3108, located on the west side of Paddys Run; and Well 3062, located off-FMPC in the South Plume (Figure 15-11).

Total uranium concentrations in 3000 Series wells located in the Waste Storage Area were generally higher than those in 2000 Series wells. The average total uranium concentration of groundwater from both Wells 3019 and 3084 was 35 ug/L. The magnitude of the vertical hydraulic gradient that would be necessary to transport uranium from the pits downward to the point of sampling in the middle of the Great Miami Aquifer is much higher than would naturally occur. However, at least one test well located near the waste pits was pumped from 1965 to 1974, or longer, in an attempt to prevent contamination of the Great Miami Aquifer (NLO 1965-1974). Induced downward transport due to this pumping of the test wells could be responsible for the observed

uranium levels in Wells 3019 and 3084. As illustrated by Figures L-15 through L-21, groundwater containing elevated uranium concentrations may be moving slowly eastward from beneath the Waste Storage and Production areas in the Great Miami Aquifer at the 3000 Series level. However, statistically elevated concentrations are not observed at Well 3051, indicating that uranium contamination has not yet reached the eastern FMPC boundary.

Groundwater samples from Wells 3019 and 3084 near Pits 4 and 6 showed similar patterns of U-238/U-234 activity ratios. The activity ratio observed in groundwater from both wells was significantly higher than 1.0 in samples from the second quarter 1988 and first quarter 1989 sampling rounds. However, groundwater from third and fourth quarter 1988 sampling of these wells had a U-238/U-234 activity ratio approximately equal to 1.0. This similar behavior was observed in adjacent 2000 Series Wells. Based on these observations, a possible source of elevated uranium found in groundwater at Wells 3019 and 3084 is the waste in Pits 4 and 6.

The U-238/U-234 activity ratio of several 3000 Series wells located in the Waste Pit Area is slightly less than 1.0. The activity ratio of Wells 3001, 3003, 3004, and 3010, located in the western and northeastern portions of the Waste Storage Area, ranged from 0.66 to 0.98. This indicates that the predominant sources of uranium found in these wells are other than Pits 4 and 6, since, as discussed in Section 15.4.1.1, the uranium activity of waste material from Pits 4 and 6 was much higher than one.

Dissolved oxygen concentrations measured in 3000 Series wells (Figure 15-4), show that infiltrating surface water is reaching the middle of the Great Miami Aquifer in some areas near Paddys Run. Statistically elevated total uranium concentrations in groundwater samples from the same 3000 Series wells further indicate surface water infiltrates to the middle of the aquifer along parts of Paddys Run. Except at Wells 3108 and 3014, uranium concentrations in 3000 Series wells located along Paddys Run were lower than those observed in 2000 Series wells.

The U-238/U-234 activity ratio of groundwater samples from Well 3014 ranged from 1.23 to 1.37. As discussed previously, the activity ratio of Paddys Run stream water is significantly greater than 1.0. The elevated U-238/U-234 activity ratio observed at Well 3014 is indicative, therefore, of a source due to the infiltration of stream water into the regional aquifer.

Statistically elevated total uranium concentrations were observed off-FMPC property in South Plume Wells 3062, 3095, and 3125. As discussed above, most uranium in the South Plume is present near the water table of the Great Miami Aquifer. Well 3062 is a production well for Albright &

Wilson Co. Groundwater containing elevated uranium concentrations may flow downward into the middle of the aquifer due to pumping of this well.

The U-238/U-234 activity ratio for samples from many 3000 Series wells could not be calculated, as isotopic uranium activity concentrations were below the MDL in these wells. The U-238/U-234 activity ratio of groundwater from most 3000 Series wells was approximately equal to one. The U-238/U-234 activity ratio of groundwater from 3000 Series wells located in the South Plume was also approximately one. The activity ratio of groundwater samples from Wells 3062, 3095, and 3125 ranged from 0.81 to 1.47.

Plots of total uranium concentration in groundwater versus time from early 1981 to 1989 for the 3000 Series wells with sufficient data are presented in Figures O-17 to O-26. Sources of the data are the FMPC environmental monitoring program (NLO 1983-1986; WMCO 1987-1989), the Task C Report (Dames and Moore 1985), RCRA Groundwater Monitoring Reports Rounds 1-5 (Dames and Moore 1986a, 1986b, 1987a, 1987b, 1987c), and RI/FS sampling. The period of observation for Wells 3013, 3014, and 3019 is shorter, from early 1985 to 1989 (Figures O-23, O-24, and O-25), due to the inavailability of FMPC environmental monitoring data during the earlier years. Also, data from the Interim Air, Soil, Water, and Health Risk Assessment Report (IT 1986) were also used in the concentration versus time plots for Wells 3013 and 3062 (Figures O-23 and O-26). Uranium concentrations in groundwater are relatively steady in most of the 3000 Series wells presented in Appendix O. The total uranium concentration of groundwater at Well 3001 has increased slightly during the observation period (Figure O-17), while the uranium concentration at Well 3010 has decreased during the same observation period (Figure O-22).

Figures N-32 to N-64 show total uranium concentration and groundwater elevation versus time for selected 3000 Series wells. The total uranium data plotted are from RI/FS and 1989 RCRA sampling. The uranium concentration of groundwater at most 3000 Series wells does not appear to be related to groundwater elevation as would be expected due to the location of the sampling points in the middle of the aquifer. Relative to the uranium concentration of the companion 2000 Series well in each well cluster, the uranium concentration observed in the 3000 Series well is lower and less variable. However, the total uranium concentration groundwater at Wells 3013, 3019, and 3084 are generally greater than the concentration observed in the companion 2000 Series wells (Figures N-37, N-41, and N-56).

4000 Series Wells

Statistically elevated total uranium concentrations were observed in groundwater samples from approximately half of the 4000 Series wells. Total uranium concentrations in groundwater samples from these wells ranged from undetected, at an MDL of 0.1 ug/L, to 86 ug/L. The maximum concentration is an outlier, detected during second quarter 1989 sampling of Well 4013. Excluding this outlier, the maximum uranium concentration observed in groundwater samples from 4000 Series wells was 5 ug/L in Well 4013. Only at Wells 4008, 4013, and 4097 was uranium detected repeatedly (Appendix E). With reference to Figure 11-3, the clay interbed thickness at these wells was approximately 10, 0, and 10 feet, respectively. Areas west and upgradient from Well 4097 have no clay interbed present.

Figures L-22 through L-28 show the total uranium concentration of groundwater samples collected from 4000 Series wells during second quarter 1988 to fourth quarter 1989 RI/FS and RCRA sampling. Figure 15-12 shows the average total uranium concentration of groundwater in 4000 Series wells calculated from the RI/FS and RCRA sampling data.

Appendix O, Figures O-27 to O-31, are plots of total uranium concentration versus time for groundwater samples from Wells 4001, 4008, 4101, 4102, and 4103 for the period early 1982 through late 1989. Concentrations vary somewhat; however, the uranium concentration of these wells has not noticeably increased or decreased during the period of sampling.

Uranium isotopes were detected in only a few groundwater samples from 4000 Series wells. Due to the paucity of isotopic uranium data, uranium activity ratios in groundwater from 4000 Series wells could not be calculated.

Thorium

The UTLs for total and isotopic thorium are as follows:

	Total Thorium ug/L	Th-228 pCi/L	Th-230 pCi/L	Th-232 pCi/L
Ross 2000 Series	13	1.0	1.6	1.0
Ross 3000 Series	13	1.1	1.4	1.0
Shandon 2000 and 3000 Series	13	1.6	1.0	1.0
Dry Fork 2000 and 3000 Series	13	1.0	2.0	1.0
4000 Series	13	1.0	2.0	1.0

303317-B741 (GW) /01.100% ORG

303317-B741 (GW) /01 (PGH)

FIGURE 15-12
AVERAGE URANIUM CONCENTRATION
IN GROUNDWATER, 4000 SERIES WELLS
SECOND QUARTER 1988 THROUGH
FOURTH QUARTER 1989

2000 Series Wells

No statistically elevated total thorium concentrations were observed in Great Miami Aquifer groundwater sampled by 2000 Series wells. Only five nonrecurrent total thorium detections, ranging from 3 to 13 ug/L, occurred during RI/FS sampling. The detections were not spatially related. Statistically elevated Th-228, Th-230, and Th-232 activity concentrations were observed in groundwater samples from several 2000 Series wells (Table M-3). Th-228 detections ranged from 1.0 to 2.9 pCi/L in groundwater samples from 2000 Series wells. Repeated detection of Th-228 was observed only at Well 2049.

Th-230 detections ranged from 1.0 to 2.5 pCi/L in groundwater samples from 2000 Series wells. The only repeated detections of Th-230 were observed at Wells 2004, 2042, and 2065.

Th-232 detections ranged from 1.1 to 1.6 pCi/L in groundwater samples from 2000 Series wells. No detections of Th-232 were repeated in previous or subsequent sampling rounds.

Given the paucity of total and isotopic thorium detections, and given the low-level, nonrecurrent nature of the few detections observed, the FMPC and associated waste storage units are not contributing significant amounts of thorium to the Great Miami Aquifer at the depths and locations sampled by the 2000 Series wells.

3000 Series Wells

No statistically elevated total thorium concentrations were observed in regional aquifer groundwater sampled by 3000 Series wells. Total thorium levels of 2 to 7 ug/L were detected in five wells. The detections were not spatially related. Only at Well 3069 was total thorium detected more than once.

Statistically elevated Th-228, Th-230, and Th-232 activity concentrations were observed in groundwater samples from several 3000 Series wells (Table M-4). Except for the repeated occurrence of Th-230 in Well 3044, all isotopic thorium detections were nonrecurrent.

Given the low-level, nonrecurrent nature of the few detections of thorium isotopes, the FMPC and associated waste storage units are not contributing significant amounts of thorium to the Great Miami Aquifer at an elevation slightly above the clay interbed.

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4000 Series Wells

No total thorium is detected at or above the MDL of 13 ug/L in samples from any 4000 Series well. The only statistically elevated Th-228 concentration from 4000 Series wells was observed at Well 4091, where no clay interbed is present. This nonrecurrent detection occurred during first quarter 1989 RI/FS sampling.

Based on the low-level, nonrecurrent, thorium detections from 4000 Series wells, the FMPC is not contributing a significant amount of thorium to the bottom of the Great Miami aquifer.

Radium-226 and Radium-228

The UTLs of radium isotopes for downgradient 2000, 3000, and 4000 Series wells are as follows:

	Ra-226 pCi/L	Ra-228 pCi/L
Ross 2000 Series	1.1	4.3
Ross 3000 Series	1.1	4.1
Shandon 2000 and 3000 Series	1.9	3.0
Dry Fork 2000 and 3000 Series	1.0	3.0
4000 Series	1.0	3.0

2000 Series Wells

Statistically elevated Ra-226 activity concentrations were found in groundwater samples from only six 2000 Series wells. The six wells, 2014, 2015, 2044, 2060, 2094, and 2095, are all in the Dry Fork downgradient well group near and beyond the southern boundary of the FMPC (Figure 6-3). Repeated Ra-226 detections were observed only at Wells 2014 and 2094. The maximum Ra-226 activity concentration detected in groundwater sampled by 2000 Series wells was 6.3 pCi/L from third quarter 1988 sampling of Well 2094, which is located south of New Haven Road (Figure 6-3).

Statistically elevated Ra-228 activity concentrations were observed in groundwater from several 2000 Series wells. However, for the majority of these wells, the statistically elevated status is assigned because of a low-level, nonrecurrent Ra-228 detection. Only at Wells 2044 and 2094 was Ra-228 detected during more than one sampling round. The maximum Ra-228 activity concentration observed in groundwater sampled by 2000 Series wells was 5.9 pCi/L from third quarter 1988 sampling of Well 2052, located north of the Production Area.

The locations of 2000 Series wells in which radium isotopes were detected indicates that the observed low-level activity of radium may be related to the infiltration of Paddys Run stream water.

The presence of radium in Well 2094 would be difficult to justify as a result of the historically-formed South Plume, because radium would not be expected to migrate more quickly than uranium (see Chapter 16.0). It should be noted that Well 2094 is also downgradient of several industrial operations along Paddys Run Road. In general, based on the isolated, nonrecurrent, and low-level nature of detections, it appears that the FMPC and associated waste storage units are not contributing significant quantities of radium to the portion of the Great Miami Aquifer sampled by 2000 Series wells.

3000 Series Wells

Statistically elevated Ra-226 activity concentrations were observed only at Wells 3043 and 3094 (Table M-4). The only repeated Ra-226 detection occurred in Well 3055. Statistically elevated Ra-228 activity concentrations were observed in Wells 3005, 3018, 3043, and 3094. Repeated Ra-228 detections only occurred in Well 3094.

The lack of recurrent Ra-226 and Ra-228 detections shows that the FMPC and associated waste storage units are not contributing significant levels of radium to the Great Miami Aquifer at the depths and locations sampled by 3000 Series wells.

4000 Series Wells

Statistically elevated Ra-226 activity concentrations were observed in groundwater samples from Wells 4015 and 4101, but the low-level detections are nonrecurrent. No clay interbed is present at Well 4015, but a 16-foot layer of clay was found at Well 4101.

Statistically elevated Ra-228 activity concentrations were observed in groundwater samples from Wells 4091, 4101, and 4103. The detections of Ra-228 in these wells were low-level and nonrecurrent. The Ra-228 detections ranged from 3.0 to 4.3 pCi/L.

Based on the isolated, nonrecurrent, and low-level detections of Ra-226 and Ra-228, it appears that the FMPC facilities are not contributing a significant amount of radium to the bottom portion of the Great Miami Aquifer.

Strontium-90

The UTL of Sr-90 is 5.0 pCi/L for the downgradient wells from all background areas.

2000 Series Wells

Statistically elevated Sr-90 activity concentrations were detected in groundwater from 2000 Series Wells 2004, 2021, and 2109 (Table M-3). Only at Well 2021 was Sr-90 detected during more than one sampling round. Sr-90 detections in groundwater sampled by 2000 Series wells ranged from 7.9 to 38.5 pCi/L, with the maximum observed in fourth quarter 1988 sample from Well 2021 located in the middle of the Waste Storage Area.

As discussed for 1000 Series wells, Sr-90 was reported in samples of waste material stored in the waste pits, and this is probably the source of the Sr-90 repeatedly detected at Well 2021.

3000 Series Wells

Sr-90 was not detected in any of the 3000 Series wells. This indicates that the FMPC and the associated waste storage units are not contributing Sr-90 to those levels of the Great Miami Aquifer sampled by 3000 Series wells.

4000 Series Wells

Sr-90 was not detected in any 4000 Series wells. Thus, the FMPC and the associated facilities are not contributing Sr-90 to the lowest depths of the Great Miami Aquifer where groundwater is sampled by 4000 Series wells.

Technetium-99

The UTL of Tc-99 for all downgradient wells is 30 pCi/L.

2000 Series Wells

Statistically elevated Tc-99 activity concentrations were detected in groundwater samples from Wells 2019, 2021, 2022, 2043, and 2109 (Table M-3). Except for Wells 2043 and 2109, detections were recurrent and ranged from 166 to 5510 pCi/L. The maximum detection occurred in third quarter 1988 RI/FS sampling of Well 2021.

The source of the repeated Tc-99 detections in 2000 Series wells is most likely the waste pits. Tc-99 is a documented constituent of FMPC waste solids. It should be noted that Tc-99 is a relatively mobile radionuclide and detectable quantities of Tc-99 were also observed in the glacial overburden beneath the Waste Storage Area.

3000 Series Wells

Statistically elevated, but nonrecurrent Tc-99 activity concentrations were detected in Wells 3009 and 3034. The lack of detection shows that the FMPC and the associated facilities are not contributing Tc-99 to the Great Miami Aquifer at the depths where 3000 Series wells are screened.

4000 Series Wells

No statistically elevated Tc-99 activity concentration was detected in groundwater from any 4000 Series wells. Thus, there is no indication that the FMPC and the associated waste storage units are contributing Tc-99 to the bottom of the Great Miami Aquifer.

Other Radionuclides

No detections of Cs-137, Np-237, Pu-238, Pu-239/240, or Ru-106 were observed in groundwater samples collected from wells screened in the Great Miami Aquifer during RI/FS and 1989 RCRA sampling.

15.4.2.2 Metals

Aluminum

As discussed in Section 15.3.2.2, aluminum data from the Interim Air, Soil, Water, and Health Risk Assessment Report (IT 1986) were used to generate UTLs for various downgradient well groups.

No aluminum UTL was developed for the Ross Section well group, since no groundwater samples from wells located in the Ross background area were analyzed for aluminum. The UTL of aluminum is 0.19 mg/L for downgradient 2000 and 3000 Series wells in the Shandon Tributary. Insufficient aluminum concentration data were available from the Dry Fork Section of the New Haven Trough to create UTLs for Dry Fork downgradient 2000 and 3000 Series wells, and 4000 Series wells.

2000 Series Wells

Limited data were available for aluminum in groundwater from 2000 Series wells. Samples analyzed for aluminum were collected during RI/FS and second to third quarter 1989 RCRA sampling rounds.

Statistically elevated aluminum concentrations were observed in groundwater collected from several 2000 Series wells. Among these wells, aluminum concentrations greater than 0.20 mg/L were observed in groundwater samples from Wells 2008, 2014, 2016, 2027, 2045, 2084, and 2109 (Appendix E). The aluminum concentration of groundwater from 2000 Series wells ranges from

less than the MDL of 0.1 mg/L to 0.755 mg/L, with the maximum concentration from the second quarter 1988 sampling of Well 2014, located near the confluence of Paddys Run and the Storm Sewer Outfall Ditch.

Statistically elevated aluminum concentrations in groundwater from 2000 Series wells may be related to the FMPC and associated waste storage units. However, due to the lack of data, the dispersed location of wells with statistically elevated concentrations, and the low concentrations, correlation to the FMPC is uncertain.

3000 Series Wells

Aluminum in 3000 Series wells was analyzed only in some samples taken during second quarter 1988 RI/FS, second to third quarter 1989 RI/FS, and second to third quarters 1989 RCRA sampling rounds.

Statistically elevated aluminum concentrations were observed in Wells 3019, 3037, 3055, and 3084 (Table M-4). Aluminum concentration in 3000 Series wells ranged from 0.0199 to 0.224 mg/L, which is only slightly above the UTL. The maximum concentration occurred in a sample from Well 3037 in the third quarter of 1989.

Given the limited data and the low-level detection in 3000 Series wells, there is no clear evidence that the FMPC is contributing to aluminum concentrations in Great Miami Aquifer groundwater at the elevation where 3000 Series wells are screened.

4000 Series Wells

Groundwater samples from seven 4000 Series wells were analyzed for aluminum. Average aluminum concentrations of greater than 0.10 mg/L were observed in Wells 4001, 4008, 4013, 4064, and 4101 (Appendix E). Aluminum concentrations in 4000 Series wells ranged from 0.0144 to 0.238 mg/L, with the maximum concentration observed during third quarter 1989 RCRA sampling of Well 4013.

Given the limited data for aluminum concentration from 4000 Series wells, no correlation to the FMPC and associated facilities is possible.

Arsenic

The UTLs for arsenic are as follows:

Ross 2000 Series	1.004 mg/L
Ross 3000 Series	0.639 mg/L
Shandon 2000 and 3000 Series	0.060 mg/L
Dry Fork 2000 and 3000 Series	0.006 mg/L
4000 Series	0.006 mg/L

The background data used to construct the UTLs for the Ross 2000 and 3000 Series well groups have a coefficient of variation greater than one.

2000 Series Wells

Statistically elevated arsenic concentrations were observed in groundwater samples from Wells 2094 and 2104. With the exception of the repeated detection of arsenic at Well 2094, detections in the portion of the regional aquifer sampled by 2000 Series wells were isolated, nonrecurrent, and of low concentration. Excluding outliers, arsenic detection in 2000 Series wells ranged from 0.002 to 0.275 mg/L. The maximum arsenic concentration was from fourth quarter 1988 sampling of Well 2094.

As discussed earlier, Well 2094 is downgradient of several industrial sites located along Paddys Run Road, which may be the source of the elevated arsenic found in groundwater from the well.

3000 Series Wells

Statistically elevated arsenic concentrations were not observed in any 3000 Series downgradient well. Arsenic detection in 3000 Series wells ranged from 0.002 to 0.31 mg/L. The maximum concentration is an outlier and was excluded from the data analysis. Excluding outliers, the maximum arsenic concentration occurring in 3000 Series downgradient groundwater was 0.02 mg/L from Wells 3001, 3054, and 3097. Recurrent detections of arsenic in 3000 Series wells occurred only at Well 3043 and were below the UTL for the associated Shandon Tributary.

Given the isolated and low-level detections of arsenic in groundwater sampled by 3000 Series wells, the FMPC and associated waste storage units are not contributing significant amounts of arsenic to the aquifer at the depth where water is sampled by 3000 Series wells.

4000 Series Wells

Statistically elevated arsenic concentrations were observed in two 4000 Series wells, Wells 4067 and 4091. Arsenic detections from 4000 Series wells ranged from 0.002 to 0.015 mg/L. Groundwater from Well 4067 was tested only once for arsenic during the RI/FS and RCRA sampling program; thus, the concentration reported in Well 4067 may not be representative. The source of the

consistently elevated arsenic concentrations observed in Well 4091 is not known; however, the well is in an area that has been used for agriculture and is adjacent to State Highway 128. Therefore, the presence of arsenic in this well may not be related to FMPC operations.

Barium

The UTLs for barium are as follows:

Ross 2000 Series	0.079 mg/L
Ross 3000 Series	0.077 mg/L
Shandon 2000 and 3000 Series	0.900 mg/L
Dry Fork 2000 and 3000 Series	0.048 mg/L
4000 Series	0.048 mg/L

2000 Series Wells

Statistically elevated barium concentrations were observed in groundwater samples from several 2000 Series wells which are located in the southern portion of the reservation and off-FMPC to the southwest and south. Among these, Wells 2046, 2065, 2070, and 2093 lie within an area of the aquifer which may receive water from both the Shandon Tributary and Dry Fork Section. Barium concentrations from these wells were considered statistically elevated when compared to the Dry Fork Section background data with a much lower UTL. Barium detection in 2000 Series wells ranged from 0.014 to 1.25 mg/L, with the maximum observed during first quarter 1989 sampling of Well 2094.

The FMPC and associated waste storage areas do not appear to contribute significant barium to the portion of the aquifer sampled by 2000 Series wells. Elevated barium concentrations in groundwater samples from 2000 Series wells, except for Well 2094, were low enough to be due to natural variation. Barium concentrations at Well 2094 were consistently greater than 1.0 mg/L; however, these concentrations appear to be due to contamination from a source other than the FMPC. Well 2094 is downgradient of industrial sites located along Paddys Run Road. The elevated barium concentrations at the wells located in the fluctuation area may be due to mixing of groundwater from the Shandon Tributary and Dry Fork Section and not due to FMPC-related contamination. Elevated barium concentrations observed in 1000 Series wells located in the Waste Storage Area were not found in samples from the corresponding clustered 2000 Series wells.

3000 Series Wells

Statistically elevated barium concentrations were observed in groundwater samples from several 3000 Series wells. Wells 3018, 3020, 3065, 3070, and 3093 lie within an area of the aquifer which may receive water from both the Shandon Tributary and the Dry Fork Section. Barium

concentrations in these wells were considered statistically elevated when compared to the lower-UTL Dry Fork background data; however, concentrations are not considered elevated when compared to the Shandon Tributary UTL. Barium detection in 3000 Series wells ranged from 0.006 to 0.393 mg/L, with the maximum observed during the second quarter 1989 sampling of Well 3094.

Statistically elevated barium concentrations at wells located in the fluctuation area may be due to the mixing of groundwaters from the Shandon Tributary and Dry Fork Section and not due to FMPC-related contamination. Other statistically elevated barium concentrations in groundwater samples from 3000 Series wells, except for Well 3094, were low enough that they may be due to natural variation. The barium concentration at Well 3094 is almost an order of magnitude higher than that from the background area. The elevated barium concentration in Well 3094 may be due to operations at the industrial facilities that are located along Paddys Run Road south of the FMPC.

4000 Series Wells

Statistically elevated barium concentrations were observed in almost all 4000 Series wells. However, the natural variation of barium is considered greater than the UTL. Barium detection in 4000 Series wells ranged from 0.028 to 0.45 mg/L. Four wells located directly downgradient and to the east of the waste pits, Wells 4010, 4013, 4064, and 4101, indicate barium concentrations that are higher than twice that of the UTL.

The lower barium concentrations observed in 3000 Series wells located in the vicinity of the waste pits suggest that groundwater chemistry near the bottom of the aquifer may be influenced by bedrock more so that surface releases of waste constituents. Thus, the elevated barium concentrations may be due to natural variation and not the operation of the FMPC.

Cadmium

The UTLs for cadmium are:

Ross 2000 Series	0.005 mg/L
Ross 3000 Series	0.008 mg/L
Shandon 2000 and 3000 Series	0.006 mg/L
Dry Fork 2000 and 3000 Series	0.005 mg/L
4000 Series	0.005 mg/L

2000 Series Wells

Statistically elevated cadmium concentrations were observed in groundwater samples from 2000 Series wells that are located in the southern half of the Production Area, near the fly ash piles and Southfield, and Well 2094 (Table M-3). Among these wells, repeated detections occurred only in Wells 2046, 2048, 2053, 2094, and 2109. Cadmium concentrations in groundwater from 2000 Series wells ranged from 0.002 to 0.045 mg/L. The maximum reported concentration was from the second quarter 1989 sampling of Well 2055 in the Production Area, and was nonrecurrent. Excluding this observation, the maximum cadmium concentration in the 2000 Series wells was 0.0118 mg/L from the second quarter 1989 sampling of Well 2094.

Given the low cadmium concentrations from most 2000 Series wells, it is reasonable to conclude that the FMPC and associated waste storage areas are not significant sources of cadmium to the portion of the Great Miami Aquifer sampled by 2000 Series wells. The occurrence of elevated cadmium in Well 2094 may be related to the industries operating along Paddys Run Road rather than the FMPC.

3000 Series Wells

Statistically elevated cadmium concentrations occurred in groundwater samples from several 3000 Series wells (Table M-4). Cadmium detection in 3000 Series wells ranged from 0.002 to 0.032 mg/L, with the maximum concentration from the third quarter 1988 sampling of Well 3034. Except for Wells 3015, 3084, and 3094, statistically elevated cadmium concentrations represented nonrecurrent detections in each well.

The low concentrations and nonrepeated detections of cadmium from 3000 Series wells provide no clear evidence that the FMPC and associated facilities are contributing significant amounts of cadmium to the portion of the Great Miami Aquifer sampled by 3000 Series wells.

4000 Series Wells

Statistically elevated cadmium concentrations were observed in several 4000 Series wells (Table M-5). No recurrent elevated cadmium concentration was observed in these wells. Cadmium detection in groundwater from 4000 Series wells ranged from 0.0021 to 0.012 mg/L, with the maximum from second quarter 1989 sampling of Well 4013 and fourth quarter 1988 sampling of Well 4015.

Based on the nonrecurrent, low-level cadmium concentrations in 4000 Series wells, the FMPC and associated facilities do not appear to contribute significant amounts of cadmium to the bottom part of the Great Miami Aquifer.

Calcium

The UTLs for calcium are:

Ross 2000 Series	109.23 mg/L
Ross 3000 Series	112.30 mg/L
Shandon 2000 and 3000 Series	108.58 mg/L
Dry Fork 2000 and 3000 Series	104.38 mg/L
4000 Series	104.38 mg/L

2000 Series Wells

Statistically elevated calcium concentrations were detected in groundwater samples from many 2000 Series wells located both within and off-FMPC property (Table M-3). The calcium concentrations of approximately half of these wells are close enough to the UTLs that the concentrations can be ascribed to natural variation. However, groundwater samples containing substantially elevated calcium concentrations were also found in several 2000 Series wells located in the Waste Storage Area and southwestern portion of the Production Area. Calcium concentrations in 2000 Series wells ranged from 10.1 mg/L to 449 mg/L, with the minimum an outlier and the maximum from third quarter 1989 sampling of Well 2027 located in the northeast corner of the Waste Storage Area.

Elevated calcium concentrations observed in 2000 Series wells located in the vicinity of the waste pits are probably due to the downward movement of perched groundwater containing elevated calcium levels or direct leakage from the waste pits. Elevated calcium concentrations observed in 2000 Series wells located in the southeast portion of the Production Area may be due to the dissolution of calcium-bearing minerals, chiefly calcite and dolomite, contained in the glacial overburden. Spills of acidic solutions, such as those used in Plant 6 pickling operations, would be neutralized by reaction with these minerals, increasing dissolved calcium concentrations. Elevated calcium concentrations could also be due to the leaching of concrete within the Production Area.

3000 Series Wells

Statistically elevated calcium concentrations were observed in groundwater sampled from several 3000 Series wells located within the FMPC reservation (Table M-4). Groundwater samples containing elevated calcium concentrations were found in 3000 Series wells located immediately east of the Waste Storage Area and extending into the northern portion of the Production Area.

Wells 3017 and 3070, which are located at the southern boundary of the reservation, were also found to contain moderately elevated calcium concentrations. Calcium concentrations in groundwater samples from 3000 Series wells ranged from 7.9 to 337 mg/L, with the maximum concentration from fourth quarter 1988 sampling of Well 3010 located along the eastern boundary of the Waste Storage Area.

Elevated calcium concentrations in 3000 Series wells located east of the waste pits are probably due to the eastward and downward movement of groundwater which carries calcium-bearing solutions from the waste pits. Elevated calcium concentrations observed in Well 3017 may be due to the infiltration of surface water from the fly ash area/Southfield. The elevated calcium concentration observed in Well 3070 may be caused by groundwater flow from Southfield or the Active Fly Ash pile. Large quantities of concrete were disposed in Southfield.

4000 Series Wells

Statistically elevated calcium concentrations were observed in five 4000 Series wells, 4013, 4015, 4016, 4064, and 4101. Calcium concentrations in 4000 Series wells ranged from 68.5 to 270 mg/L, with the maximum concentration from third quarter 1989 RCRA sampling of Well 4013 located at the eastern edge of the Production Area where the clay interbed is absent.

The elevated calcium concentrations observed at Wells 4015 and 4016 are very close to the UTL and may be due to natural variation. The elevated calcium concentrations observed at Wells 4013, 4064 and 4101, which are located in the Production Area, are correlated with the locations of elevated calcium concentrations observed in 2000 and 3000 Series wells, and may be related to the operation of the Waste Storage Area and/or the Production Area.

Chromium

All chromium concentrations given in the following discussion are total chromium concentrations.

The UTLs for total chromium are:

Ross 2000 Series	0.020 mg/L
Ross 3000 Series	0.027 mg/L
Shandon 2000 and 3000 Series	0.028 mg/L
Dry Fork 2000 and 3000 Series	0.029 mg/L
4000 Series	0.029 mg/L

2000 Series Wells

Statistically elevated total chromium concentrations were observed in groundwater samples from several 2000 Series wells (Table M-3). Elevated, repeated detections of total chromium were observed in Wells 2017, 2048, and 2065 in the southern portion of the reservation; Wells 2027 and 2084 in the Waste Storage Area; Wells 2006, 2007, 2054, 2053, 2055, and 2109 in the Production Area; and off-FMPC Wells 2098 and 2127. In these wells, total chromium detections ranged from 0.0023 to 0.0918 mg/L, with the maximum detection from second quarter 1989 sampling of Well 2109 located near Plant 6.

The Waste Storage and Production areas may be the source of the elevated chromium concentrations in 2000 Series wells in these areas. Based on the concentrations of chromium relative to the tolerance limits and on the widespread distribution of wells exhibiting elevated concentrations; however, it is uncertain whether the FMPC has contributed chromium to the Great Miami Aquifer.

3000 Series Wells

Statistically elevated chromium concentrations were observed in groundwater samples from Wells 3019, 3024, 3037, 3055, 3084, 3065, and 3108 located within the FMPC reservation (Table M-4). Among these wells, repeated detections of chromium only occurred in Wells 3019, 3024, 3037, 3055, 3065, and 3084. However, except for Well 3065, where only two samples were analyzed for chromium, the detections in these wells were not consistent. Excluding outliers, chromium detection in 3000 Series wells ranged from 0.0071 to 0.0774 mg/L, with the maximum detection from second quarter 1989 RCRA sampling of Well 3084.

The Waste Storage Area and Production Area may be the sources of the elevated chromium concentrations observed from 3000 Series wells. However, again, based on the inconsistent and low-level chromium detections, the correlation is uncertain.

4000 Series Wells

Statistically elevated chromium concentrations were observed in two 4000 Series wells, Wells 4013 and 4064 (Table M-5). Chromium detections observed in groundwater samples from 4000 Series wells ranged from 0.0072 to 0.062 mg/L, with the maximum detection from second quarter 1989 sampling of Well 4013. The average chromium concentrations calculated for Wells 4013 and 4064 are less than twice the UTL.

As discussed previously, groundwater chemistry at the base of the aquifer may be different from the upper portion of the aquifer due to the influence of the bedrock. The slightly elevated chromium concentrations observed in Wells 4013 and 4064 do not geographically correlate with chromium concentrations observed in 2000 and 3000 Series wells. Thus, the source of chromium observed in Wells 4013 and 4064 may not be related to the FMPC facilities.

Copper

The UTLs for copper are:

Ross 2000 Series	0.227 mg/L
Ross 3000 Series	0.168 mg/L
Shandon 2000 and 3000 Series	0.010 mg/L
Dry Fork 2000 and 3000 Series	0.039 mg/L
4000 Series	0.039 mg/L

The UTL of copper for Ross 3000 Series wells was calculated based on a coefficient of variation greater than 1.0.

2000 Series Wells

Statistically elevated copper concentrations in groundwater were observed in 2000 Series wells located in the Shandon Tributary well group (Table M-3). Excluding outliers, copper detected in 2000 Series wells ranged from 0.0046 to 0.0697 mg/L, with the maximum detection in the second quarter 1988 sample from Well 2021.

Most 2000 Series wells in the Shandon Tributary group exhibited statistically elevated copper concentrations. However, only Wells 2054, 2084, 2097, and 2109 showed repeated detections (Appendix E). The average copper concentrations calculated for groundwater from 2000 Series downgradient wells are all below 0.02 mg/L, and are therefore less than twice the UTL for the Shandon Tributary. Thus, the total variations of copper concentrations in groundwater may be greater than accounted for in the Shandon Tributary UTL, and the FMPC and associated waste units are probably not a source of copper for the portion of the Great Miami Aquifer sampled by 2000 Series wells.

3000 Series Wells

Statistically elevated copper concentrations were observed in many 3000 Series wells within the reservation and several 3000 Series wells off-FMPC (Table M-4). Except for Well 3106, none of these wells showed consistently repeated and significant copper concentrations. The two samples

taken from Well 3106 showed results of 0.015 and 0.100 mg/L. Wells 3065, 3070, and 3108, which are located in the area that may receive groundwater from the Shandon Tributary and/or the Dry Fork Section of the New Haven Trough, show elevated copper concentrations only when compared to the Shandon background copper concentration. Copper detections in groundwater from 3000 Series wells ranged from 0.0057 to 0.1 mg/L, with the maximum concentration from second quarter 1989 RI/FS sampling of Well 3106.

The elevated copper concentrations from the wells located in the fluctuation areas may be due to groundwater flows from the Dry Fork Section, which has a higher background copper concentration than groundwater from the Shandon Tributary. Also, as discussed above, the natural variability of copper concentrations in Great Miami Aquifer groundwater may be greater than the variability observed in background data. Given the nonrecurrent and low concentrations of copper, the FMPC and associated waste storage units are probably not contributing significant amounts of copper to the Great Miami Aquifer at depths sampled by 3000 Series wells.

4000 Series Wells

Statistically elevated copper concentrations were not detected in any 4000 Series wells. Copper concentrations in 4000 Series wells ranged from 0.0051 to 0.104 mg/L, with the maximum concentration from fourth quarter 1988 sampling of Well 4023 which is located in Ross and represents background conditions.

The FMPC and the associated waste storage units are not contributing copper to the lower portion of the Great Miami Aquifer where groundwater is sampled by 4000 Series wells.

Iron

The UTLs for iron are:

Ross 2000 Series	0.057 mg/L
Ross 3000 Series	0.127 mg/L
Shandon 2000 and 3000 Series	3.948 mg/L
Dry Fork 2000 and 3000 Series	0.043 mg/L
4000 Series	0.043 mg/L

It is noted that the UTL for the 3000 Series Ross Section wells was calculated based on a coefficient of variation greater than one.

Iron concentrations are sensitive to redox conditions. As discussed previously, groundwater in the Shandon Tributary is most likely chemically reduced, which allows higher iron concentrations and

therefore a higher UTL. Given the behavior of iron in the aquifer and the complexity of surface water and groundwater interactions in the study area, the statistical analysis for this parameter is of questionable validity.

2000 Series Wells

Statistically elevated iron concentrations were observed in groundwater samples from approximately half of the 2000 Series wells (Table M-3). Iron levels in all but two of these wells were statistically elevated only when compared to the UTL calculated from the Dry Fork Section background data. Iron in 2000 Series wells ranged from below an MDL of 0.005 mg/L to 21.2 mg/L, with the maximum concentration from first quarter 1989 sampling of Well 2094 located far to the south of the FMPC.

Iron concentrations greater than 1.0 mg/L were commonly observed in 2000 Series wells which contained less than 1.0 mg/L of dissolved oxygen, while iron concentrations much less than 1.0 mg/L were commonly observed in 2000 Series which contained greater than 1.0 mg/L of dissolved oxygen. Thus, in areas of the aquifer that receive recharge from surface water, such as the area immediately downgradient from Paddys Run, iron concentrations generally are low. This is due to the oxidation of ferrous iron (Fe^{+2}) to ferric iron (Fe^{+3}) and subsequent precipitation within the matrix of the aquifer. Over short time frames of a few years, this reaction is reversible. Therefore, iron may be fixed in the aquifer during parts of a year, or for several years, and then remobilized.

Because of the behavior of iron in the aquifer, the contribution, if any, of the metal from the FMPC and associated waste storage units cannot be assessed. The iron concentrations of 15.8 to 21.2 mg/L from Well 2094 should be considered anomalous and may be due to other industrial operations in the area.

3000 Series Wells

Statistically elevated iron concentrations were observed in groundwater samples from many 3000 Series wells (Table M-4). Iron levels in all but three of these wells were statistically elevated only when compared to the UTL calculated from the Dry Fork Section background data. Excluding outliers, iron concentrations in 3000 Series wells ranged from 0.02 to 16.9 mg/L, with the maximum concentration from fourth quarter 1988 sampling of Well 3037.

Given the behavior of iron in nature, the elevated iron concentrations observed in groundwater samples from most western downgradient wells can be attributed to natural variation. However,

slightly elevated iron concentrations in Wells 3017, 3062, 3095, and 3106 may be related to the leaching of waste materials in Southfield into the aquifer through surface water infiltration. The elevated iron concentration observed at Well 3094 is probably related to the operation of the industrial facilities along Paddys Run Road. Significantly elevated iron concentrations observed at Wells 3013, 3037, and 3055, which are located downgradient from the Waste Pit Area, may be due to eastward and downward groundwater flow from the waste pits.

4000 Series Wells

Statistically elevated iron concentrations were observed in all 4000 Series wells (Table 5-M). Iron concentrations in 4000 Series wells ranged from 0.03 to 9.64 mg/L, with the maximum concentration from third quarter 1989 sampling of Well 4013.

Groundwater at the bottom of the regional aquifer receives the least amount of direct stream or surface water infiltration, and thus is chemically reduced relative to groundwater in the upper portion of the aquifer. The elevated iron concentration observed in the majority of 4000 Series wells is most likely due to the chemically reduced state of groundwater in the deeper portion of the aquifer. However, significantly elevated iron concentrations observed in Wells 4001, 4010, 4013, 4064, 4067, and 4101, which are located downgradient from the waste pits, may be partially related to releases from the waste storage units. This is supported by the elevated iron concentrations from 2000 and 3000 Series wells that are located in the vicinity of the waste pits.

Lead

The UTLs for lead are:

Ross 2000 Series	0.002 mg/L
Ross 3000 Series	0.004 mg/L
Shandon 2000 and 3000 Series	0.019 mg/L
Dry Fork 2000 and 3000 Series	0.004 mg/L
4000 Series	0.004 mg/L

2000 Series Wells

Statistically elevated lead concentrations were observed in groundwater samples from Wells 2017, 2018, 2044, 2045, and 2046 within the reservation and Wells 2002 and 2093 off the FMPC. Among these wells, repeated detection of lead occurred in groundwater samples from only Wells 2044, 2045, 2046, and 2093. The lead concentrations were statistically elevated by comparison with Dry Fork background well data (Table M-3). Excluding outliers, lead detected in 2000 Series wells ranged from 0.002 to 0.026. The maximum concentration was observed during fourth quarter 1988 sampling of Well 2017.

Based on the nonrecurrent and low-level nature of lead detections, it appears that the FMPC and associated waste storage units are not contributing significant quantities of lead to the portion of the regional aquifer sampled by 2000 Series wells. Repeated detections of lead in samples from Wells 2045 and 2046 may be related to historic infiltration of contaminated stream water through the Paddys Run bed or infiltration through Southfield or the fly ash area.

3000 Series Wells

Statistically elevated lead concentrations were observed in groundwater samples from several 3000 Series wells (Table M-4). Among these wells, repeated detections of lead occurred only in Wells 3016, 3062, 3069, and 3094. Excluding outliers, lead detected in 3000 Series wells ranged from 0.002 to 0.078 mg/L, with the maximum concentration from fourth quarter 1988 sampling of Well 3017.

Based on the isolated, nonrecurrent, and low-level lead detections, the FMPC and associated waste storage units are not contributing significant amounts of lead to the Great Miami Aquifer at depths where groundwater is sampled by 3000 Series wells.

4000 Series Wells

Statistically elevated lead concentrations were observed in Wells 4014, 4096, 4102, and 4103 (Table M-5). However, no recurrent elevated lead concentrations were observed in these wells. Excluding outliers, lead concentrations detected in 4000 Series wells ranged from 0.002 to 0.05 mg/L, with the maximum concentration from second quarter 1988 sampling of Well 4103.

Based on the low-level, nonrecurrent lead detections in 4000 Series wells, the FMPC and the associated facilities are probably not contributing significant amounts of lead to the bottom of the Great Miami Aquifer.

Magnesium

The UTLs for magnesium are:

Ross 2000 Series	40.90 mg/L
Ross 3000 Series	38.87 mg/L
Shandon 2000 and 3000 Series	34.86 mg/L
Dry Fork 2000 and 3000 Series	27.88 mg/L
4000 Series	27.88 mg/L

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2000 Series Wells

Statistically elevated magnesium concentrations were observed in groundwater samples from several 2000 Series wells (Table M-3). Magnesium concentrations in approximately half of these wells were close to the UTL and may be due to natural variation. However, magnesium concentrations in groundwater samples from Wells 2065 and 2094; Waste Storage Area Wells 2022, 2027, and 2084; and Production Area Wells 2054, 2109, and 2118 are higher than can be ascribed to natural variation. Excluding outliers, magnesium concentrations in groundwater from 2000 Series wells ranged from 18 to 100 mg/L, with the maximum concentration from third quarter 1989 RCRA sampling of Well 2084 located east of Pit 5.

As discussed earlier, elevated magnesium concentrations were found in perched groundwater beneath the Waste Storage Area. Downward movement of groundwater from the perched aquifer or direct leakage from the waste pits may contribute to the statistically elevated magnesium concentrations in Wells 2022, 2027, and 2084.

Elevated magnesium concentrations in the regional aquifer beneath the southeastern corner of the Production Area may be due to the Plant 6 pickling operations. Neutralization of an acidic plume could result in the dissolution of native magnesium-bearing minerals or concrete, or the desorption or exchange of magnesium absorbed on other minerals. The presence of magnesium under Plant 6, therefore, is an indicator that the effects of spillage in the plant area have impacted the aquifer.

Elevated magnesium concentrations observed in Well 2094 may be related to the industrial facilities along Paddys Run Road.

3000 Series Wells

Statistically elevated magnesium concentrations were found in several 3000 Series wells (Table M-4). The magnesium concentration in groundwater moving from 3000 Series wells ranged from 0.5 to 232 mg/L. The minimum observed concentration is an outlier. The maximum concentration, from Well 3108, appears to be a data transcription error that could not be substantiated. Excluding this value, the maximum magnesium concentration was 85.8 mg/L from second quarter 1989 RCRA sampling of Well 3084, which is located east of Pit 5.

The correlation between the locations of 2000 and 3000 Series wells which exhibited elevated magnesium concentrations indicates that groundwater moving downward and eastward from the glacial overburden and the aquifer beneath the waste pits is contributing magnesium to Wells 3010, 3013, 3037, 3051, and 3084. Elevated magnesium concentrations observed at Wells 3018 and 3095

may be ascribed to natural variation, and the anomalous magnesium concentrations observed at Well 3094 may be due to the operation of the industrial facilities along Paddys Run Road.

4000 Series Wells

Statistically elevated magnesium concentrations were found in Wells 4010, 4013, 4064, and 4101, which are located in the Production Area (Table M-5). Magnesium concentrations observed in groundwater samples from these wells ranged from 28.5 to 91.7 mg/L. Magnesium concentrations in samples from all 4000 Series wells ranged from 18 to 91.7 mg/L, with the maximum concentration from the third quarter 1989 sampling of Well 4013.

The correlation between the locations of 4000 Series wells, and 2000 and 3000 Series wells that contain elevated groundwater magnesium concentrations, suggests that the anomalous magnesium concentrations observed in Wells 4010, 4013, 4064, and 4101 may be derived from the Waste Storage and/or Production areas.

Manganese

The UTLs for manganese are:

Ross 2000 Series	0.004 mg/L
Ross 3000 Series	0.015 mg/L
Shandon 2000 and 3000 Series	0.134 mg/L
Dry Fork 2000 and 3000 Series	0.919 mg/L
4000 Series	0.919 mg/L

It should be noted that the UTL of 0.919 mg/L, calculated for the Dry Fork Section, was based on a coefficient of variation greater than one. The variation of manganese concentrations in Dry Fork background groundwater is very high because of high concentrations observed at Well 2057. This leads to a high UTL for the Dry Fork well group.

Similar to iron, the manganese concentration in the aquifer is sensitive to groundwater redox conditions. Thus, the application of statistical analysis to manganese data is not as straightforward as for other parameters.

2000 Series Wells

Statistically elevated manganese concentrations were detected in the majority of 2000 Series wells in the Shandon Tributary group (Table M-3). Excluding outliers, manganese concentrations in 2000 Series wells ranged from 0.001 to 1.86 mg/L, with the maximum concentration being from the second quarter 1989 sampling of Well 2109.

As with iron, interpretation of the dissolved manganese distribution is complicated by the presence of relatively reduced and oxidized areas within the Great Miami Aquifer. Despite the added complexity of redox sensitive behavior, it appears that the FMPC and associated waste storage units are not a significant source of manganese for the portion of the aquifer sampled by 2000 Series wells. Only at Wells 2054, 2109, and 2118, located in the vicinity of Plant 6, were observed manganese concentrations high enough to suggest contributions from the FMPC.

3000 Series Wells

Statistically elevated manganese concentrations were found in groundwater from a majority of 3000 Series wells in the Shandon downgradient well group (Table M-4). Manganese concentrations in 3000 Series wells ranged from 0.006 to 4.32 mg/L, with the maximum from fourth quarter 1988 RI/FS sampling of Well 3010.

Given the behavior of manganese in the aquifer, the majority of the elevated manganese concentrations may be due to natural variation. However, the significantly elevated manganese concentration of groundwater in Wells 3010, 3019, and 3084, which are located in the vicinity of the waste pits, may indicate a contribution from the pits.

4000 Series Wells

A statistically elevated concentration of manganese was found only in groundwater from Well 4016 (Table M-5). Excluding outliers, manganese concentrations in 4000 Series wells ranged from 0.009 to 1.02 mg/L, with the maximum manganese concentration from first quarter 1989 sampling of Well 4016.

The average manganese concentration of 0.939 mg/L in Well 4016 is only slightly greater than the UTL. Because of the relatively reduced conditions present in the deeper portion of the aquifer, this value is not considered an indication of FMPC-related contamination.

Based on the manganese concentrations observed in 4000 Series wells, the FMPC and the associated facilities are not contributing manganese to the area of the Great Miami Aquifer sampled by 4000 Series wells.

Mercury

The UTLs for mercury are:

Ross 2000 Series	0.20 ug/l
Ross 3000 Series	0.32 ug/l

Shandon 2000 and 3000 Series	0.76 ug/l
Dry Fork 2000 and 3000 Series	0.20 ug/l
4000 Series	0.20 ug/l

2000 Series Wells

Statistically elevated mercury concentrations were found in several 2000 Series wells in the Dry Fork downgradient group and at Well 2098 (Table M-3). Among these wells, repeated detections of mercury occurred only at Wells 2014, 2016, 2018, and 2044 within the reservation boundary, and Wells 2094, 2095, and 2096 off the FMPC. Excluding outliers, mercury detected in 2000 Series wells ranged from 0.2 to 2.6 ug/L, with the maximum mercury concentration from first quarter 1989 sampling of Well 2044.

Recurring statistically elevated mercury concentrations were observed in seven of the 2000 Series wells. These may be related to the infiltration of surface water from Paddys Run, since five of the seven wells are located within a portion of the aquifer where surface water infiltration occurs. Elevated mercury concentration was also detected in Well 2094, located downgradient of the Paddys Run Road industrial sites. Because most mercury detections are nonrecurrent and low-level, the FMPC and associated waste storage areas are probably not a significant source of mercury for the portion of the Great Miami Aquifer sampled by 2000 Series wells.

3000 Series Wells

Statistically elevated mercury concentrations were found in a majority of the Dry Fork 3000 Series wells (Table M-4) and four wells located in the fluctuation area (Figure 15-2), Wells 3018, 3020, 3093, and 3108. Among these wells, mercury was recurrently detected in Wells 3014, 3017, 3018, 3044, 3095, and 3108. Excluding outliers, mercury detected in 3000 Series wells ranged from 0.2 to 7.5 ug/L, with the maximum concentration from second quarter 1989 sampling of Well 3108.

Because most mercury detections were low-level and did not consistently recur, the FMPC and associated waste storage facilities are probably not contributing a significant amount of mercury to the Great Miami Aquifer at depths where groundwater is sampled by 3000 Series wells.

4000 Series Wells

Statistically elevated mercury concentrations were observed in several 4000 Series wells (Table M-5). Recurrent mercury detections occurred only at Well 4014. Mercury detection in groundwater from 4000 Series wells ranged from 0.2 to 6.4 ug/L, with the maximum concentration from the first quarter 1989 sampling of Well 4014.

Elevated mercury concentrations were also observed in Wells 2014 and 3014, located near the confluence of the Storm Sewer Outfall Ditch and Paddys Run. Thus, the elevated mercury concentrations in Well 4014 may be related to the vertical migration of contaminated surface water.

Molybdenum

The UTLs for molybdenum are:

Ross 2000 Series	0.04 mg/L
Ross 3000 Series	0.06 mg/L
Shandon 2000 and 3000 Series	0.03 mg/L
Dry Fork 2000 and 3000 Series	0.02 mg/L
4000 Series	0.02 mg/L

2000 Series Wells

Statistically elevated molybdenum concentrations were observed in groundwater samples from Wells 2021, 2064, 2069, and 2070 within the reservation and Wells 2061, 2097, and 2104 off the FMPC (Table M-3). Molybdenum detections were not recurrent in any 2000 Series well. Molybdenum detected in the 2000 Series wells ranged from 0.003 to 0.057 mg/L, with the maximum concentration from fourth quarter 1988 sampling of Well 2098 which is receiving water from the Ross area.

Based on the paucity of molybdenum detections and the isolated, nonrecurrent, and low-level nature of the detections, the FMPC and associated waste storage units are not a source of significant molybdenum to the upper portion of the Great Miami Aquifer that is sampled by 2000 Series wells.

3000 Series Wells

Statistically elevated molybdenum concentrations were observed at Wells 3013, 3044, 3049, 3064, 3094, and 3106 (Table M-4), but molybdenum detections were not recurrent in any of these wells. Molybdenum detected in 3000 Series wells ranged from 0.007 to 0.2 mg/L, with the maximum concentration from the third quarter 1988 sampling of Well 3064.

Elevated molybdenum concentrations in Wells 3013 and 3064, which are located around the eastern boundary of the Production Area, are possibly due to the location of these wells downgradient from the Waste Storage and Production areas. The elevated molybdenum concentrations observed in Wells 3049 and 3106 may be related to groundwater flow and stream infiltration from the fly ash piles and Southfield. However, given the low-level and nonrecurrent detections of molybdenum, any relationship between the operation of the FMPC facilities and the observed elevated molybdenum concentrations is uncertain.

4000 Series Wells

Statistically elevated molybdenum concentrations were observed in Wells 4001, 4008, 4013, 4015, 4097, 4101, and 4103 (Table M-5). Among these wells, repeatedly elevated molybdenum concentrations occurred only in groundwater samples from Wells 4001, 4008, 4013, and 4101. Molybdenum detections in 4000 Series wells ranged from 0.004 to 0.06 mg/L, with the maximum concentration from the first quarter 1989 sampling of Well 4013.

Elevated molybdenum concentrations observed in Wells 4001, 4008, 4013, and 4101, which are located directly downgradient from the Waste Storage Area, may be related to the eastward and downward groundwater movement of molybdenum-bearing groundwater from this area. However, given the low-level and isolated detections of molybdenum, the FMPC and associated facilities may not be contributing a significant quantity of molybdenum to the portion of the Great Miami Aquifer sampled by 4000 Series wells.

Nickel

The UTLs for nickel are:

Ross 2000 Series	0.020 mg/L
Ross 3000 Series	0.024 mg/L
Shandon 2000 and 3000 Series	0.020 mg/L
Dry Fork 2000 and 3000 Series	0.020 mg/L
4000 Series	0.020 mg/L

2000 Series Wells

Statistically elevated nickel concentrations were detected in groundwater samples from several 2000 Series wells (Table M-3). However, only at Wells 2027, 2043, 2051, 2054, 2084, 2094, and 2109 was nickel detected in more than one sampling round. Excluding outliers, nickel detected in 2000 Series wells ranged from 0.0105 to 0.084 mg/L. The maximum concentration occurred during third quarter 1989 sampling of Well 2014, which is the only time nickel was detected in that well.

Repeated nickel detections in Wells 2054 and 2109 may be due to contributions from the Production Area. The repeated detection of nickel at Well 2094 may be related to location of the well, downgradient from industrial facilities situated along Paddys Run Road. It appears that the FMPC and associated facilities are not significant sources of nickel for the portion of the Great Miami Aquifer sampled by 2000 Series wells.

3000 Series Wells

Statistically elevated concentrations of nickel were observed in several 3000 Series wells (Table M-4), but recurrent nickel detections only occurred in Wells 3084 and 3094 (Appendix E). Excluding outliers, nickel detected in 3000 Series wells ranged from 0.0105 to 0.066 mg/L, with the maximum nickel concentration from first quarter 1989 sampling of Well 3094.

Given the nonrecurrent and low-level detections of nickel in 3000 Series wells, other than Well 3094, the FMPC is most likely not contributing a significant amount of nickel to the Great Miami Aquifer at depths where groundwater is sampled by 3000 Series wells. The nickel detected in Well 3094 may be due to the operation of the industrial facilities located along Paddys Run Road.

4000 Series Wells

Statistically elevated nickel concentrations were observed in groundwater samples from Wells 4013, 4015, and 4064 (Table M-5). No recurrent detection occurred in any of these wells. Excluding outliers, nickel detected in 4000 Series wells ranged from 0.014 to 0.0365 mg/L, with the maximum concentration from second quarter 1989 RI/FS sampling of Well 4013.

Because of the nonrecurrent, low-level detections of nickel from 4000 Series wells, the FMPC and the associated facilities are most likely not contributing significant amounts of nickel to the lower portion of aquifer that is sampled by the 4000 Series wells.

Potassium

The UTLs for potassium are:

Ross 2000 Series	3.55 mg/L
Ross 3000 Series	3.29 mg/L
Shandon 2000 and 3000 Series	2.24 mg/L
Dry Fork 2000 and 3000 Series	1.96 mg/L
4000 Series	1.96 mg/L

2000 Series Wells

Potassium concentrations in groundwater samples from most 2000 Series wells were statistically elevated relative to background (Table M-3). Only at Wells 2021, 2022, 2084, and 2094 was the average potassium concentration greater than 5.00 mg/L. Except for these four wells, the elevated potassium concentrations observed in groundwater samples from 2000 Series wells may be due to natural variation. Potassium concentrations in 2000 Series wells ranged from 0.169 to 214 mg/L. The minimum concentration is an outlier. The maximum potassium concentration was from the fourth quarter 1988 sampling of Well 2094.

Statistically elevated potassium concentrations observed at Wells 2021, 2022, and 2084 are most likely due to the downward movement of potassium-bearing groundwater from the glacial overburden or direct leakage from the waste pits. Samples of surface water from Waste Pits 5 and 6 and the Clearwell contain potassium concentrations ranging from 26.0 to 230 mg/L (Weston 1987).

The extremely elevated potassium concentration at Well 2094, which averaged 199.8 mg/L, does not occur in samples from any other 2000 Series well. The source of this potassium is located downgradient of the FMPC.

3000 Series Wells

Statistically elevated potassium concentrations were found in most 3000 Series wells (Table M-4). Excluding outliers, potassium concentrations in 3000 Series wells ranged from 0.479 to 230 mg/L, with the maximum concentration from third quarter 1988 sampling of Well 3094. Except for Wells 3010, 3013, 3019, 3037, 3053, 3054, 3055, 3084, and 3094, where potassium concentrations were consistently greater than 3.0 mg/L, the elevated potassium concentrations were most likely due to natural variation.

Elevated potassium concentrations in 3000 Series wells located downgradient of the Waste Storage and Production areas are most likely due to sources in these areas. The elevated potassium concentration observed in Well 3094 appears to be related to the industrial facilities along Paddys Run Road, which are downgradient of the FMPC.

4000 Series Wells

Statistically elevated potassium concentrations were detected in several 4000 Series wells (Table M-5). Potassium concentrations in 4000 Series wells ranged from 0.85 to 8.75 mg/L, with the maximum concentration from fourth quarter 1988 sampling of Well 4001.

Except for Wells 4001, 4013, and 4101, the slightly elevated potassium concentrations in the 4000 Series wells can be attributed to natural variation. The elevated potassium concentrations in Wells 4001, 4013, and 4101 appear to be related to sources in the Waste Storage Area. This is supported by the presence of elevated potassium concentrations in the adjacent 2000 and 3000 Series Wells.

Selenium

With the exception of an outlier, selenium was not detected in groundwater samples from any background well. Consequently, the UTL for selenium is 0.002 mg/L for all 2000, 3000, and 4000 Series downgradient wells.

2000 Series Wells

Statistically elevated selenium concentrations were detected in groundwater samples from several 2000 Series wells (Table M-3). However, selenium was not detected during more than one sampling round at any 2000 Series wells except for Wells 2091 and 2098, where average selenium concentrations were 0.003 and 0.004 mg/L, respectively. Selenium detected in groundwater samples from 2000 Series wells ranged from 0.0018 to 0.022 mg/L, with the maximum concentration from fourth quarter 1988 sampling of Well 2010.

Based on the low-level, nonrecurrent nature of selenium detections, the FMPC and associated facilities are not sources of significant amounts of selenium to the portion of the Great Miami Aquifer sampled by 2000 Series wells.

3000 Series Wells

Statistically elevated selenium concentrations were found in groundwater samples from several 3000 Series wells (Table M-4). However, detections of selenium in these wells were not recurrent except for Well 3014, where the average selenium concentration was 0.003 mg/L. Excluding outliers, selenium detected in groundwater samples from 3000 Series wells ranged from 0.002 to 0.017 mg/L, with the maximum concentration from fourth quarter 1988 sampling of Well 3054.

Given the low-level, nonrecurrent detections of selenium in 3000 Series wells, the FMPC and associated facilities are not contributing significant levels of selenium to the Great Miami Aquifer at depths where groundwater is sampled by 3000 Series wells.

4000 Series Wells

Statistically elevated selenium concentrations were found only in Wells 4014 and 4097 (Table M-5). However, selenium detection was not recurrent in these two wells. Selenium detected in 4000 Series wells ranged from 0.002 to 0.003 mg/L, with the maximum concentration from first quarter 1989 sampling of Wells 4014 and 4097.

Given the low-level, nonrecurrent detections of selenium in 4000 Series wells, the FMPC and associated waste storage units are not contributing significant amounts of selenium to the lower portion of the aquifer that is monitored by 4000 Series wells.

Silver

The UTLs for silver are:

Ross 2000 Series	0.032 mg/L
Ross 3000 Series	0.023 mg/L
Shandon 2000 and 3000 Series	0.013 mg/L
Dry Fork 2000 and 3000 Series	0.034 mg/L
4000 Series	0.034 mg/L

2000 Series Wells

Statistically elevated silver concentrations were detected in groundwater samples from eleven 2000 Series wells (Table M-3). Production Area Wells 2007, 2053, and 2055 are the only wells in which silver was detected during more than one sampling round. Excluding outliers, silver detected in 2000 Series wells ranged from 0.01 to 0.034 mg/L, with the maximum silver concentration from second quarter 1988 sampling of Well 2057. Well 2057 is a background well in the Dry Fork section of the New Haven Trough (Figure 6-3).

Based on the locations of the three wells with repeated, statistically elevated silver detections, the Production Area may be a limited source of silver for the upper portion of the regional aquifer sampled by the 2000 Series wells. However, other areas of the FMPC facility, including the Waste Storage Area, are probably not contributing significant amounts of silver to the upper portion of the Great Miami Aquifer.

3000 Series Wells

A statistically elevated average silver concentration was observed in groundwater only at Well 3034 (Table M-4). However, silver detection was not recurrent in this well. Silver detection in 3000 Series Wells ranged from 0.01 to 0.024 mg/L, with the maximum concentration from the second quarter 1988 sampling of Well 3034.

Based on the low-level and nonrecurrent detections of silver in 3000 Series wells, the FMPC and associated facilities are not contributing significant amounts of silver to the Great Miami Aquifer at depths where groundwater is sampled by 3000 Series wells.

4000 Series Wells

Statistically elevated silver concentrations were not found in any 4000 Series well (Table M-5). Silver detections in 4000 Series wells ranged from 0.0103 to 0.09 mg/L, with the maximum concentration from the second quarter 1988 RI/FS sampling of Well 4103.

From the analytical results of groundwater samples from 4000 Series wells, there is no evidence that the FMPC and associated waste storage units are contributing silver to the bottom of the Great Miami Aquifer.

Sodium

The UTLs for sodium concentration are:

Ross 2000 Series	34.66 mg/L
Ross 3000 Series	24.55 mg/L
Shandon 2000 and 3000 Series	59.33 mg/L
Dry Fork 2000 and 3000 Series	4.85 mg/L
4000 Series	4.85 mg/L

The UTL of sodium calculated from the Shandon Tributary background well data is more than one order of magnitude greater than that from the Dry Fork Section background well data.

2000 Series Wells

If compared with the sodium UTL of 4.9 mg/L from the Dry Fork Section, every well in the downgradient group would be statistically elevated (Table M-3). However, no wells in the Shandon Tributary group exceeded the UTL of 59.3 mg/L.

Sodium concentrations in groundwater samples from 2000 Series wells were typically less than 20 mg/L. Sodium concentrations greater than 20 mg/L were found in groundwater at Wells 2019, 2021, 2022, 2027, and 2084 located in the Waste Storage Area. Sodium concentrations of greater than 20 mg/L were also found in Wells 2043, 2068, 2092, 2093, and 2094. Excluding outliers, sodium concentrations from 2000 Series wells ranged from 4.28 to 82.7 mg/L, with the maximum concentration from the fourth quarter 1988 sampling of Well 2092.

The waste pits may be the source of the elevated sodium concentrations observed in groundwater at the regional aquifer water table in the Waste Storage Area. Rock salt used to deice roads during winter months may be a source of the elevated sodium concentrations observed at Wells 2043, 2068, and 2092. Wells 2093 and 2094 are located downgradient from both major roadways and commercial/industrial facilities.

3000 Series Wells

Statistically elevated sodium concentrations were found in every 3000 Series well in the Dry Fork group (Table M-4). Sodium concentrations of greater than 40 mg/L were detected in Wells 3010, 3013, 3019, 3037, 3055, and 3084 located downgradient from the Waste Storage and/or Production areas. In addition, sodium concentrations of greater than 40 mg/L also occurred in Wells 3108 and 3094. Excluding outliers, sodium concentrations in 3000 Series wells ranged from 2.9 to 110 mg/L, with the maximum concentration from the third quarter 1988 sampling of Well 3010.

The waste pits may be the source of significant amounts of sodium in groundwater from 3000 Series wells located immediately downgradient of the area. The high sodium concentration in Well 3108 was a one-time detection of 90.6 mg/L, during the first quarter sampling of 1989; the elevated result was not confirmed by other samplings. The elevated sodium concentration found in Well 3094 is a further indication that the industrial facilities along Paddys Run have an impact on the Great Miami Aquifer monitored by Well 3094.

4000 Series Wells

Statistically elevated sodium concentrations were detected in almost all 4000 Series downgradient wells (Table M-5). Average sodium concentrations of greater than 20.0 mg/L were found in Wells 4010, 4013, 4064, and 4101, which are located downgradient from the Waste Storage Area. Sodium concentration in 4000 Series wells ranged from 3.93 to 70.7 mg/L, with the maximum from the second quarter 1989 sampling of Well 4013.

Elevated sodium concentrations in a majority of the 4000 Series wells may be due to natural variation. However, the elevated sodium concentrations observed in Wells 4010, 4013, 4064, and 4101 are probably related to the waste pits. This conclusion is supported by the elevated sodium concentrations observed in the associated 2000 and 3000 Series wells.

Vanadium

Groundwater data from the Interim Air, Soil, Water, and Health Risk Assessment Report (IT 1987) were used to supplement background data for vanadium. Groundwater samples from Ross Section background wells were not analyzed for vanadium; therefore, no UTL was developed for Ross Section wells. The UTL for vanadium is 0.042 mg/L for 2000 and 3000 Series Shandon Tributary wells. Vanadium concentration data from the Dry Fork Section were insufficient to compute a UTL for 2000 and 3000 Series Dry Fork wells, and 4000 Series well; however, by incorporating vanadium data from wells located near the divide with the Shandon Tributary background, a UTL of 0.032 mg/L was determined.

2000 Series Wells

Limited data are available for vanadium in groundwater from 2000 Series wells. Samples analyzed for vanadium were collected during second quarter 1988 to third quarter 1989 RI/FS sampling and second to third quarter 1989 RCRA sampling rounds. Statistical analysis of vanadium concentration data from 2000 Series Ross Section wells was not performed due to the absence of background data.

Vanadium detections in 2000 Series wells ranged from 0.0034 to 0.0548 mg/L, with the maximum from the second quarter 1989 sampling of Well 2084. A statistically elevated vanadium concentration occurred only in Well 2084 (Table M-3).

Based on the limited data, it cannot be concluded that the observed vanadium concentrations are due to contributions from the FMPC.

3000 Series Wells

Groundwater samples from 3000 Series wells were analyzed for vanadium only during three RI/FS and two RCRA sampling rounds. Groundwater samples from 3000 Series Ross Section wells were not analyzed for vanadium. Statistically elevated vanadium concentration did not occur in any 3000 Series well (Table M-4). Vanadium detected in 3000 Series wells ranged from 0.01 to 0.049 mg/L, with the maximum concentration from the second quarter 1989 RCRA sampling of Well 3084.

4000 Series Wells

Groundwater samples from only eight 4000 Series wells were analyzed for vanadium. A statistically elevated vanadium concentration occurred only at Well 4013 (Table M-5). Vanadium detection in 4000 Series wells ranged from 0.0136 to 0.038 mg/L, with the maximum concentration from second quarter 1989 RCRA sampling of Well 4013.

Based on the limited data, it appears that operations of the FMPC are not significantly impacting vanadium concentrations in the aquifer.

Zinc

Groundwater data from the Interim Air, Soil, Water, and Health Risk Assessment Report (IT 1987) were used to supplement background data for zinc. Groundwater samples from Ross Section background wells and downgradient wells were not analyzed for zinc. The UTL of zinc concentration for 2000 and 3000 Series Shandon wells is 0.12 mg/L. Zinc data from Dry Fork

Section background wells were insufficient for UTL calculation. The only zinc data available from the Dry Fork Section background is a reading of 0.17 mg/L from the Interim Risk Assessment Well State-16, which is equivalent to RI/FS Well 2057. This value is used for the statistical comparison of downgradient 2000 and 3000 Series Dry Fork Section wells, and 4000 Series wells.

2000 Series Wells

Zinc detection in groundwater samples collected by 2000 Series wells ranged from 0.0119 to 0.268 mg/L. The maximum zinc concentration was observed during first quarter 1989 RI/FS sampling of Well 2109. A statistically elevated zinc concentration was detected only at Well 2109, but this is based on only one sampling result.

Based on the limited zinc data, it is difficult to conclude whether the FMPC and associated facilities have impacted zinc concentrations in the upper Great Miami Aquifer sampled by 2000 Series wells.

3000 Series Wells

Zinc was analyzed in groundwater samples collected from eleven 3000 Series downgradient wells. A statistically elevated zinc concentration occurred only at Well 3037 (Table M-4). Zinc detected in the 3000 Series wells ranged from 0.017 to 0.267 mg/L, with the maximum detection from the second quarter 1989 RCRA sampling of Well 3037. The other analytical result from Well 3037 exhibited a zinc concentration of only 0.039 mg/L during the third quarter 1989 RCRA sampling.

Given the limited zinc data from 3000 Series wells, it is difficult to determine whether the FMPC and associated waste storage facilities have influenced zinc concentrations in the portion of the Great Miami Aquifer sampled by 3000 Series wells.

4000 Series Wells

Groundwater samples from five 4000 Series wells were analyzed for zinc. Detections of zinc in these wells ranged from 0.0156 to 0.108 mg/L, which are less than the only analytical result from a background well, 0.17 mg/L. Consequently, the FMPC facilities do not appear to have any impact on zinc concentrations in the lower portion of the Great Miami Aquifer that is monitored by 4000 Series wells.

15.4.2.3 Organics

Total Organic Halides

Since TOX was not detected above the MDL of 0.05 mg/L in any background well, the UTL of TOX is set at 0.05 mg/L for the 2000, 3000, and 4000 Series downgradient wells.

2000 Series Wells

Statistically elevated TOX concentrations were detected in groundwater from Wells 2014, 2043, and 2094 (Table M-3). Total organic halide concentrations in these wells ranged from below the MDL of 0.05 mg/L to 17.7 mg/L. Detection of TOX from all 2000 Series wells ranged from 0.0015 to 45.1 mg/L. The maximum TOX concentration was from second quarter 1989 sampling of Well 2095 and is an outlier. A TOX concentration of 2.9 mg/L was measured in a duplicate sample from Well 2095. The reason for the wide range of TOX concentration at a single well is not known. Excluding outliers, the maximum TOX concentration in 2000 Series downgradient wells was 17.7 mg/L.

The sporadic and nonrecurrent detection of TOX in groundwater indicates that halogenated hydrocarbon compounds are probably not present in groundwater at the Great Miami Aquifer water table, except in a few isolated areas. The source of the repeated TOX detections observed in Well 2014 is not known, but could be related to the fly ash piles, Southfield, or infiltration from Paddys Run. The chemistry of groundwater from Well 2094 is anomalous; the source of the TOX concentrations in this well may be the industries downgradient from the FMPC.

3000 Series Wells

Statistically elevated TOX concentrations were found in groundwater from Wells 3013, 3014, 3017, 3019, and 3037 (Table M-4). TOX concentrations in these wells ranged from below the MDL of 0.05 mg/L, to 1.31 mg/L, with the maximum concentration from the first quarter 1989 RI/FS sampling of Well 3014. Except for Well 3014, detections of TOX in 3000 Series wells were not recurrent.

Based on the isolated and nonrecurrent TOX detections, the FMPC facilities are probably not contributing halogenated hydrocarbon compounds to the Great Miami Aquifer to the depths where groundwater is sampled by 3000 Series wells. One exception is Well 3014. The source of the slightly elevated TOX concentration in Well 3014 may be the fly ash or Southfield waste materials, or infiltration of surface water from Paddys Run.

4000 Series Wells

Statistically elevated TOX concentrations were detected in groundwater from Wells 4013 and 4067 (Table M-5). However, the detections in these two wells did not recur. The TOX detection in groundwater from 4000 Series wells ranged from 0.0135 to 0.07 mg/L, with the maximum concentration from the second quarter 1989 RI/FS sampling of Well 4013, located in the Northeast corner of the Production Area.

Based on the low-level, isolated, and nonrecurrent detections of TOX in groundwater from 4000 Series wells, the FMPC and the associated facilities are apparently not contributing a significant amount of halogenated hydrocarbon compounds to the lower Great Miami Aquifer.

Volatile and Semivolatile Organics, Pesticides, and PCBs

2000 Series Wells

Table 15-7 presents organic compounds detected during RI/FS and 1989 RCRA sampling of groundwater from 2000 Series wells. Note that detections associated with method blank contamination have been excluded from Table 15-7.

Table 15-8 lists organic compounds detected in groundwater samples from 2000 Series wells collected during third quarter 1985 through fourth quarter 1987 RCRA monitoring. CLP and QAPP procedures were not followed during the sampling and/or analysis and qualifiers were not included with the results (Section 15.4.1.3). The following discussion of detected organic compounds is based only on data collected during RI/FS and 1989 RCRA sampling (Table 15-7).

Volatile and semivolatile organic compounds were detected in groundwater samples from several 2000 Series wells, and pesticides were detected in a sample from one 2000 Series well. PCBs were not detected in groundwater samples collected from 2000 Series wells. Various volatiles and semivolatiles were detected at Well 2011 during second quarter 1988 RI/FS sampling (Table 15-7). This well is located northwest of the Waste Storage Area. Volatiles toluene, carbon disulfide, and methylene chloride, and semivolatiles 3-nitroaniline, isophorone, and N-nitrosodiphenylamine were detected in Well 2011 at concentrations near or below the contract required quantitation limit (CRQL). Only acetone was detected at a concentration greater than twice the CRQL, and acetone is a common laboratory contaminant.

TABLE 15-7

DETECTED ORGANIC COMPOUNDS (ug/L) IN GROUNDWATER FROM 2000 SERIES WELLS[a] SECOND QUARTER 1988 TO THIRD QUARTER 1989

Well	2006	2007	2010	2011	2014	2015	2016
Sampling Quarter	2nd/89	2nd/89	2nd/88	2nd/88	2nd/88	2nd/88	2nd/88
Volatile Organic Compounds							
2-Butanone	4 J	2 J					
Toluene	3 J			5			
Acetone[c]			8 J	37	2 J		
Benzene			1 J				
Carbon disulfide				8			
Methylene chloride				1 J			
1,1,1-Trichloroethane						6	
Semivolatile Organic Compounds							
Phenol		14					
Diethyl phthalate			7 J				
3-Nitroaniline				50			
Isophorone				2 J			
N-nitrosodiphenylamine				10			
bis(2-Ethylhexyl)phthalate					2 J	1 J	4 J

See footnotes at end of table.

TABLE 15-7
(continued)

Well	2019		2021	2022	2027	2034	2037		2042
Sampling Quarter	2nd/88	3rd/89	2nd/88	2nd/88	3rd/89	2nd/88	2nd/88	3rd/89	2nd/88
Volatile Organic Compounds									
Acetone[c]	13	2 J							7 J
Benzene							1 J		
Carbon disulfide					10			11	
Methylene chloride									9
Ethylbenzene						2 J			
Total xylenes						3 J			
Semivolatile Organic Compounds									
Diethyl phthalate			1 J						
N-nitrosodiphenylamine					3 J				
bis(2-Ethylhexyl)phthalate		4 J			3 J		7 J	4 J	
4-Nitrophenol						1 J			
Butyl benzyl phthalate									
Pesticides									
Aldrin				0.11					
Heptachlor				0.07					
alpha-BHC				0.1					

See footnotes at end of table.

TABLE 15-7
(continued)

Well	2051	2052	2053	2054	2065	2066	2094		2109
Sampling Quarter	3rd/89	3rd/88	2nd/89	1st/89	2nd/88	3rd/89	3rd/88	4th/88	1st/89
Volatile Organic Compounds									
2-Butanone									
Toluene			2 J						
Acetone[c]	3 J	4 J				6 J	4 J		
Benzene								160	
Ethylbenzene								1 J	
Total xylenes							7	189	
Semivolatile Organic Compounds									
Diethyl phthalate					20				
Di-n-butyl phthalate				4 J					10

[a]This table does not include compounds found in method blanks. Second and third quarter 1989 results are from RCRA sampling program. Second quarter 1988 and first quarter 1989 results are from RI/FS sampling.

[b]Also known as 2-Propanone

Qualifier:

J - Compound detected; below the contract required quantitation limit. The value presented is an estimate.

TABLE 15-8

DETECTED ORGANIC COMPOUNDS (ug/L) IN GROUNDWATER FROM 2000 SERIES WELLS[a] THIRD QUARTER 1985 TO FOURTH QUARTER 1987

Well	2004		2011	2013				2015		
Sampling Quarter	2nd/86	2nd/87	3rd/86	3rd/86	2nd/87	2nd/86	3rd/86	4th/86	2nd/87	4th/87
Volatile Organic Compounds										
Trichloroethene	4.4						3.7			
Acetone[b]		580		735	1,800			120	104	
Cyclohexane		600			300				20	
Carbon disulfide			6.3							
Butanol			195							
2-Propanol				29,600						
1,1,1-Trichloroethane						5.1	2.9	7	18.9	5.13
1,1-Dichloroethane								1.1	2.3	

Well	2016	2017	2018	2021			2060			2061	
Sampling Quarter	2nd/87	4th/86	2nd/87	3rd/86	4th/86	3rd/86	4th/86	2nd/87	2nd/87	2nd/87[b]	
Volatile Organic Compounds											
Acetone[b]	930	17,770	24,500					34.2		40.5	
Cyclohexane			900						42	12.5	
2-Propanol		45,000									
1,1,1-Trichloroethane				10.5		2	8.7		87		
Tetrachloroethene					11.9						

[a]Third quarter 1985 through fourth quarter 1987 results from RCRA sampling program. Pre-Quality Assurance Project Plan analytical results; qualifiers are not included.

[b]Duplicate analysis.

[c]Also known as 2-Propanone.

5606
000112

A low-level detection of 1,1,1-trichloroethane occurred in a groundwater sample from second quarter 1988 sampling of Well 2015 (Table 15-7), which is located near the southern boundary of the FMPC reservation. This well has a water supply submersible pump installed in it.

Aldrin, heptachlor, and alpha-BHC were detected in a sample from Well 2022 at low concentrations of 0.11, 0.07, and 0.1 ug/L, respectively. The source of these contaminants, which are common in pesticides and herbicides is unknown. The adjacent burn pit is a possible source. Bis(2-ethylhexyl)phthalate was detected at Well 2037 near the Sanitary Landfill during second quarter 1988 and again during third quarter 1989 sampling. In both cases, the compound was detected atestimated concentrations below the CRQL.

Total xylene was repeatedly detected at Well 2094 during third and fourth quarters 1988 (Table 15-7). Toluene was detected only during third quarter 1988 sampling and benzene and ethylbenzene were detected only during fourth quarter 1988 sampling of Well 2094. Well 2094 is located downgradient of several industrial facilities along Paddys Run Road. The detection of VOCs at Well 2094 may be attributable to these sources.

Based on the limited data available, with the exception of Well 2094, the FMPC and associated facilities are the likely sources of the very low concentrations of organic contaminants observed in groundwater samples from 2000 Series wells. However, these detections were spatially and temporally dispersed, and no single event or facility can be targeted as a continuing source of organic compound release.

3000 Series Wells

Table 15-9 presents the organic compounds detected during 1988 RI/FS and 1989 RCRA sampling of groundwater from 3000 Series wells. Note that detections associated with method blank contamination were excluded from Table 15-9.

Table 15-10 lists organic compounds detected during third quarter 1985 through fourth quarter 1987 RCRA samplings. As addressed previously, CLP and the QAPP were not followed during the sampling and/or sampling analysis conducted during early RCRA monitoring. The following discussion of detected organic compounds is based only on data collected during RI/FS and 1989 RCRA sampling (Table 15-9).

Excluding possible false detections due to laboratory contamination, detections of organic compounds occurred only in groundwater samples from Wells 3001, 3010, 3019, 3037, and 3043.

TABLE 15-9

DETECTED ORGANIC COMPOUNDS (ug/L) IN GROUNDWATER FROM 3000 SERIES WELLS[a] SECOND QUARTER 1988 TO THIRD QUARTER 1989

Well	3001	3010	3019	3037		3043		3051		3066	3084
Sampling Quarter	2nd/88	2nd/89	3rd/89	2nd/89	3rd/89	2nd/89	3rd/89	2nd/89	3rd/89	3rd/89	3rd/89
Volatile Organic Compounds											
4-Methyl-2-pentanone	3 J										
Benzene	4 J										
Toluene	10	2 J		3 J		3 J		8			
Trichloroethene		2 J		3 J		2 J		9			
Carbon disulfide							3 J				
Acetone[b]									8 J	4 J	6 J
Semivolatile Organic Compounds											
bis(2-ethylhexyl)phthalate			2J		5 J						
Butyl benzyl phthalate					3 J						
Diethyl phthalate					2 J						
Phenol					17						
Di-n-butyl phthalate											2 J

[a]This table does not include compounds found in method blanks. Second and third quarter 1989 results are from RCRA sampling program. Second quarter 1988 results are from RI/FS sampling.

[b]Also known as 2-Propanone.

Qualifier:

J - Compound detected; below the contract required quantitation limit. The value is an estimate.

DETECTED ORGANIC COMPOUNDS (ug/L) IN GROUNDWATER FROM 3000 SERIES WELLS[a] THIRD QUARTER 1985 TO FOURTH QUARTER 1987

Well	3008			3009			3010		3013	
Sampling Quarter	2nd/86	3rd/86	2nd/87	3rd/86	4th/86	2nd/87	3rd/86	2nd/87	4th/86	2nd/87
Pesticides										
2,4-D	4.06									
2,4,5-TP, Silvex	0.694									
Volatile Organic Compounds										
1,1,1-Trichloroethane		6								
Acetone[b]			40	8,670	12	29.8		10,000	1,030	34.2
Cyclohexane			22			10.2		5,000		
2-Propanol				26,400	4,000					
Trichloroethene							1.7			

Well	3014		3016	3017			3019	3020
Sampling Quarter	3rd/86	4th/86	2nd/84	3rd/86	4th/86	2nd/87	3rd/86	3rd/86
Volatile Organic Compounds								
Acetone[b]			40.8	58.5		1,040	490	3,600
Cyclohexane			60			5,000		
2-Propanol				130	578		830	8,390
Trichloroethene		1						
Tetrachloroethene	1.1							
Bromoform		2.4						

[a]Third quarter 1985 through fourth quarter 1987 results from RCRA sampling program. Pre-Quality Assurance Project Plan analytical results; qualifiers are not included.

[b]Also known as 2-Propanone.

As shown in Table 15-9, most organic compounds were detected at estimated concentrations below the CRQL. The two exceptions were toluene at 10 ug/L in Well 3001, and phenol at 17 ug/L in third quarter 1989 sampling of Well 3037. Well 3037 was also sampled during second quarter 1989, and phenol was not detected.

Detections of organic compounds in groundwater samples from 3000 Series wells were isolated and nonrecurrent. Based on the limited data available, it appears that the FMPC and associated waste storage units are not contributing volatile, semivolatile compounds, pesticides, or PBCs to the portion of the regional aquifer sampled by the 3000 Series wells.

4000 Series Wells

Table 15-11 presents organic compounds detected prior to 1988. The CLP and QAPP procedures were not followed during this early RCRA sampling. Since the beginning of RI/FS sampling, only toluene has been detected at 2 ug/L in Well 4001 during the second quarter 1989 sampling. This is an estimated concentration and is below the CRQL.

Based on the limited data available, it appears that no significant levels of organics are present in the portion of the regional aquifer sampled by 4000 Series wells. The FMPC and associated waste storage units are apparently not contributing detectable quantities of organic compounds to the bottom of the Great Miami Aquifer.

15.4.2.4 General Chemistry

Alkalinity

All alkalinity measurements are reported as bicarbonate. The UTLs for alkalinity are:

Ross 2000 Series	399.5 mg/L
Ross 3000 Series	393.1 mg/L
Shandon 2000 and 3000 Series	421.8 mg/L
Dry Fork 2000 and 3000 Series	387.9 mg/L
4000 Series	387.9 mg/L

2000 Series Wells

Statistically elevated levels of alkalinity were measured in groundwater samples from Wells 2017, 2043, 2046, 2065, 2084, and 2118 within the FMPC property, and Wells 2092 and 2094 off the reservation (Table M-3). Excluding outliers, alkalinity measurements at these wells ranged from 227.5 to 808.4 mg/L, with the maximum measurement from fourth quarter 1988 sampling of Well 2094.

TABLE 15-11

DETECTED ORGANIC COMPOUNDS (ug/L) IN GROUNDWATER FROM 4000 SERIES WELLS[a]

Well	4001		4008		4428
Sampling Quarter	3rd/86	2nd/87	3rd/86	2nd/87	2nd/87
Volatile Organic Compounds					
1,1,1-Trichloroethane	2				
Acetone[b]		15.6		21.3	30.8
Cyclohexane		20		12	
Carbon disulfide			32.3		
Butanol			180		

[a]Third quarter 1986, second quarter 1987, and second quarter 1989 results from RCRA sampling program.

[b]Also known as 2-Propanone.

DRAFT

The elevated alkalinity at Well 2084 may be due to the downward movement of high alkalinity groundwater from the perched zones beneath the Waste Storage Area. The elevated alkalinity at Well 2118 may be due to the dissolution of the carbonate minerals, calcite and dolomite, by acid solutions from Plant 6. Groundwater chemistry of Well 2094 is probably controlled more by the location of this well, downgradient from several industrial facilities, than by FMPC releases. Sources of elevated alkalinity in Wells 2017, 2043, 2046, 2065, and 2092 are not readily identifiable. As alkalinities measured during the RI/FS program were total alkalinities, dissociated organic acids may be contributing to elevated alkalinity values.

3000 Series Wells

Statistically elevated alkalinity values were measured in groundwater samples from Wells 3013 and 3037 within the FMPC property, and Well 3094 off the reservation (Table M-4). Alkalinity in 3000 Series wells ranged from 175.3 to 784.9 mg/L. The minimum is from third quarter 1988 sampling of Well 3095 and is an outlier. Excluding outliers, the minimum alkalinity is 210.2 mg/L. The maximum is from second quarter 1989 sampling of Well 3094.

The elevated alkalinity measured at Wells 3037 and 3013 may be caused by the eastward and downward movement of high alkalinity groundwater from the glacial overburden beneath the Waste Storage Area. The highly elevated alkalinity in Well 3094 is probably related to the industrial facilities upgradient from the well.

4000 Series Wells

Statistically elevated alkalinity values were measured in Wells 4001, 4010, 4013, 4101, and 4102, which are located downgradient from the Waste Storage Area (Figure 6-5). The alkalinity from 4000 Series wells ranged from 260.3 to 581.2 mg/L, with the minimum and maximum from third quarter 1988 sampling of Well 4023 and first quarter 1989 sampling of Well 4013, respectively.

Elevated alkalinity values in the vicinity of the waste pits and Production Area are probably related to the waste materials stored in the pits. Elevated alkalinity values in the 2000 and 3000 Series wells located in the same vicinity provide evidence of these FMPC-related sources.

Ammonia

Ammonia concentrations are reported as nitrogen. The UTLs for ammonia are:

Ross 2000 Series	0.19 mg/L
Ross 3000 Series	0.18 mg/L
Shandon 2000 and 3000 Series	6.19 mg/L

Dry Fork 2000 and 3000 Series	0.20 mg/L
4000 Series	0.20 mg/L

The high ammonia concentrations in the Shandon Tributary background wells may be due to the presence of specialized bacteria that convert nitrate to ammonia in low oxygen environments. Concentrations of nitrate in these wells were very low when compared to other background areas. In addition, localized contamination by septic systems and/or agricultural practices may contribute to the relatively high ammonia concentrations observed in these wells.

2000 Series Wells

Ammonia concentrations observed in groundwater samples from Wells 2022, 2094, 2107, and 2108 are statistically elevated relative to background (Table M-3). In addition, the average ammonia concentrations at Wells 2019, 2021, 2043, and 2084 exceeded 1.0 mg/L and are also considered elevated. Ammonia concentrations observed in any single well were highly variable. Ammonia concentrations in 2000 Series wells ranged from 0.029 to 11.7 mg/L, with the maximum concentration from fourth quarter 1988 sampling of Well 2022, located in the middle of the Waste Storage Area.

Downward movement of groundwater containing high ammonia concentrations or direct leakage from the waste pits is probably the source of the elevated ammonia concentrations in the 2000 Series wells in the Waste Storage Area. Elevated ammonia concentrations present in groundwater near the reservation boundaries or beneath areas used for grazing dairy cattle may be related to agricultural activities. Wells 2107 and 2108 are in areas where surface water runoff from grazing areas can be introduced to the aquifer as infiltration from Paddys Run and its tributaries. The groundwater chemistry at Well 2094 is most likely influenced by sources located downgradient from the FMPC.

3000 Series Wells

Statistically elevated ammonia concentrations were observed in Wells 3037, 3094, and 3107 (Table M-4). In addition, Wells 3043 and 3055 showed average ammonia concentrations of 3.59 and 5.09 mg/L, respectively, and are considered elevated. Ammonia detections in 3000 Series wells ranged from 0.04 to 18.8 mg/L, with the maximum concentration from fourth quarter 1988 sampling of Well 3037.

Well 3107 is located in an area that may receive groundwater from Shandon Tributary and/or Dry Fork Section of the New Haven Trough. The ammonia concentration observed in Well 3107 is

elevated only when compared to Dry Fork background data. Thus, the slightly higher ammonia concentration in Well 3107 may be due to the mixing of groundwater from the two background areas. Elevated ammonia concentrations in Wells 3037 and 3055 may be related to the vertical migration of contaminants released from the waste pits. The elevated ammonia concentration in Well 3043 may be related to the agricultural activities along the reservation boundary. The ammonia concentration in Well 3094 is likely related to other industrial facilities or the application of fertilizers on the agricultural field adjacent to the well.

4000 Series Wells

Statistically elevated ammonia concentrations were found in Wells 4001, 4008, 4010, 4013, 4064, 4067, 4101, and 4102 (Table M-5). Among these wells, ammonia concentrations of greater than 1.0 mg/L were found only in Wells 4010, 4013, and 4101, which are located along the west and east boundaries of the Production Area. Ammonia detection in 4000 Series wells ranged from 0.1 to 6.63 mg/L, with the maximum concentration from the fourth quarter 1988 sampling of Well 4101.

Nitrogen, in the oxidized form of nitrate, can be reduced to ammonia at the bottom of the aquifer. This is shown by the lower nitrate concentrations observed in groundwater samples from 4000 Series wells. Given this condition and the distribution of the wells that contain elevated ammonia concentrations, the Waste Storage and Production areas are most likely the source of elevated ammonia concentrations observed in 4000 Series wells. Nitric acid and ammonia are both reagents used in FMPC operations. The historic agricultural use of the FMPC property could also represent a source of nitrogen components to the aquifer.

Chloride

The UTLs for chloride are:

Ross 2000 Series	63.52 mg/L
Ross 3000 Series	60.17 mg/L
Shandon 2000 and 3000 Series	91.85 mg/L
Dry Fork 2000 and 3000 Series	21.60 mg/L
4000 Series	21.60 mg/L

Similar to sodium, the UTL calculated from the Shandon Tributary background well data is much greater than that from the Dry Fork Section.

2000 Series Wells

Statistically elevated chloride concentrations were detected in groundwater from Well 2084 and most of the 2000 Series wells located in the Dry Fork downgradient group (Table M-3). Average chloride concentrations of greater than 60 mg/L were found in Wells 2022, 2027, 2084, 2092, 2093, and 2094. Chloride concentrations at these wells ranged from 45.99 to 250 mg/L, with the maximum concentration from third quarter 1989 RCRA sampling of Well 2084.

Elevated chloride concentrations in the 2000 Series wells located in the Waste Storage Area are probably due to the downward movement of chloride-contaminated groundwater from the glacial overburden or direct leakage from the waste pits. The elevated chloride detected in Well 2094 is another indication of the anomalous groundwater chemistry observed in that well. The sources of the elevated chloride concentrations at Wells 2092 and 2093 are not known; however, the effects of deicing road salt are suspected for Well 2092, and could influence the chloride concentrations in Wells 2093 and 2094.

3000 Series Wells

Statistically elevated chloride concentrations were detected in groundwater samples from two wells located in the vicinity of the waste pits and most of the 3000 Series wells in the Dry Fork downgradient group (Table M-4). The majority of the 3000 Series downgradient wells have average chloride concentrations less than 50 mg/L. However, average chloride concentrations of greater than 50 mg/L were detected in Wells 3010, 3013, 3019, 3037, 3051, 3055, and 3084, which are located immediately downgradient from the Waste Storage Area, and Well 3094. Excluding outliers, chloride concentrations in the 3000 Series wells ranged from 0.75 to 250 mg/L, with the maximum chloride concentration from the third quarter 1989 sampling of Well 3037.

Elevated chloride concentrations downgradient from the waste pits may be due to groundwater movement from the perched zone or direct leakage from the waste pits. The ultimate source of elevated chloride concentrations in Well 3094 is likely either the upgradient industries or the application of road salt to New Haven Road.

4000 Series Wells

Statistically elevated chloride concentrations were found in Wells 4001, 4010, 4013, 4064, 4101, and 4102, which are located downgradient from the Waste Storage and/or the Production areas, and Well 4015, which is located within the South Plume area. Average chloride concentrations of greater than 35 mg/L are observed in groundwater from Wells 4010, 4013, 4064, and 4101.

Chloride concentrations in 4000 Series wells ranged from 3 to 472 mg/L, with the maximum concentration from second quarter 1989 sampling of Well 4013.

Elevated chloride concentrations in the vicinity of the Waste Storage and Production areas may be due to groundwater movement from the waste storage units and/or the Production Area.

Cyanide

Cyanide was not measured in samples from any background well. Thus, a statistical analysis was not performed for cyanide concentration in the Great Miami Aquifer.

2000 Series Wells

A limited number of groundwater samples from 2000 Series wells were collected and analyzed for cyanide. Well 2054 contained a cyanide concentration of 0.0013 mg/L from first quarter 1989 RI/FS sampling, and is the only cyanide detection in 2000 Series wells.

Based on the paucity of cyanide detections in 2000 Series wells, the FMPC and associated waste storage units are probably not contributing cyanide to the portions of the regional aquifer sampled by the 2000 Series wells.

3000 Series Wells

The only sample analyzed for cyanide was from Well 3001 during the second quarter 1988 program. No cyanide was detected.

4000 Series Wells

A sample from Well 4101 was the only sample analyzed for cyanide. No cyanide was detected.

Fluoride

The UTLs for fluoride are:

Ross 2000 Series	0.44 mg/L
Ross 3000 Series	0.38 mg/L
Shandon 2000 and 3000 Series	1.42 mg/L
Dry Fork 2000 and 3000 Series	0.48 mg/L
4000 Series	0.48 mg/L

2000 Series Wells

Statistically elevated fluoride concentrations were detected in groundwater from Wells 2017 and 2060 (Table M-3). In addition to these wells, average groundwater fluoride concentrations of

greater than 0.50 mg/L were found in Wells 2022 and 2043. Excluding outliers, fluoride detections in 2000 Series wells ranged from 0.1 to 1.25 mg/L, with the maximum concentration from third quarter 1988 sampling of background Well 2056.

Most fluoride concentrations were at or near background levels in groundwater samples from 2000 Series downgradient wells. The source of slightly elevated fluoride concentrations at Well 2022 may be the waste pits. However, in general, the FMPC and associated waste storage units are probably not contributing significant quantities of fluoride to the Great Miami Aquifer.

3000 Series Wells

Statistically elevated fluoride concentrations were detected in groundwater from Well 3017. Fluoride detections in 3000 Series wells ranged from 0.06 to 1.9 mg/L, with the maximum concentration from the third quarter 1988 RI/FS sampling of Well 3024.

Most fluoride concentrations from 3000 Series wells were at or near background levels. Based on the statistical comparisons, the FMPC and associated waste storage facilities are most likely not contributing significant amounts of fluoride to the Great Miami Aquifer.

4000 Series Wells

Statistically elevated fluoride concentrations were found in Wells 4010 and 4013 (Table M-5). Fluoride detections in 4000 Series wells ranged from 0.62 to 0.74 mg/L, with the maximum concentration from third quarter 1988 RI/FS sampling of Well 4091.

The slightly elevated fluoride concentrations in Wells 4010 and 4013, which are located immediately downgradient from the waste pits, may reflect natural variations given the UTL of 1.42 mg/L in 2000 and 3000 Series wells in the Shandon Tributary that feeds the Production Area.

Nitrate

All nitrate concentrations are reported as nitrogen. The UTLs for nitrate are:

Ross 2000 Series	15.40 mg/L
Ross 3000 Series	13.27 mg/L
Shandon 2000 and 3000 Series	0.10 mg/L
Dry Fork 2000 and 3000 Series	19.42 mg/L
4000 Series	19.42 mg/L

The UTL calculated from the Dry Fork Section background data is based on a coefficient of variation greater than 1.0.

Nitrate concentrations correlate positively with dissolved oxygen content. Groundwater in the Shandon Tributary background area is lower in dissolved oxygen relative to the Dry Fork Section area. This indicates that the more reduced forms of nitrogen will dominate; thus, the background concentrations and UTL for nitrate in the Shandon Tributary area are exceptionally low. It should be noted that the ammonia UTL calculated for the Shandon background wells was high compared to other background areas. The application of statistical analysis to nitrate concentrations may not be technically valid if the dissolved oxygen level and, therefore, the redox potential, changes between upgradient and downgradient wells.

2000 Series Wells

Because the UTL calculated for the 2000 Series wells in the Shandon background area is so low, most downgradient wells exhibited statistically elevated groundwater nitrate concentrations (Table M-3). Nitrate was detected at greater than 1.0 mg/L in upper portions of the aquifer throughout the vicinity of Paddys Run, where direct recharge of oxygenated water occurs. Elevated nitrate was also observed in the uppermost regional aquifer where the glacial overburden has been removed by the Great Miami River near the eastern site boundary. In areas where nitrate was not detected or was detected in low concentrations, dissolved oxygen was absent or existed in low enough concentrations to allow nitrate to be converted to ammonia by bacteria that live in anaerobic environments. Excluding outliers, nitrate detected in the 2000 Series wells ranged from 0.014 to 38.6 mg/L, with the maximum concentration from the second quarter 1989 sampling of Well 2021. Most nitrate concentrations in groundwater at 2000 Series wells were lower than those present in Paddys Run stream water, indicating that infiltration of surface water could be the source of the elevated nitrate concentrations. However, nitrate concentrations at Wells 2019 and 2084 ranged from 2.8 to 24.6 mg/L, which exceeded nitrate concentrations found in surface water. These levels may be due to contamination from the waste pits.

3000 Series Wells

Statistically elevated nitrate concentrations were observed in groundwater from most 3000 Series downgradient wells, including four wells located in the area that may receive water from either the Shandon Tributary and/or the Dry Fork Section (Table M-4). This exceedance by a large number of wells is due to the use of the Shandon Tributary UTL for the statistical comparisons. However, nitrate concentrations in Wells 3010, 3019, and 3084 significantly exceeded the UTL values. Excluding outliers, nitrate detections in 3000 Series wells ranged from 0.012 to 22.8 mg/L, with the maximum nitrate concentration from second quarter 1988 sampling of Well 3010.

The Waste Storage Area may be contributing nitrate to the Great Miami Aquifer, as illustrated by the elevated nitrate concentrations observed in Wells 3010, 3019, and 3084.

4000 Series Wells

No statistically elevated nitrate concentration was observed in any 4000 Series well (Table M-5). Excluding outliers, nitrate detections in 4000 Series wells ranged from 0.1 to 7.2 mg/L, with the maximum concentration from first quarter 1989 sampling of Well 4023.

Given the low concentrations of nitrate in the 4000 Series wells, the FMPC is not contributing significant amounts of nitrate to the bottom portion of the regional aquifer. However, chemically reducing conditions at the bottom portion of the aquifer may allow the transformation of nitrate influxed from the upper portion of the aquifer to reduced nitrogen forms, such as ammonia.

pH

The computed tolerance intervals for pH are:

Ross 2000 Series	7.02 - 7.24
Ross 3000 Series	7.00 - 7.23
Shandon 2000 and 3000 Series	6.66 - 7.63
Dry Fork 2000 and 3000 Series	7.06 - 7.33
4000 Series	7.06 - 7.33

It is expected that the natural variation of pH in the aquifer may actually be greater than the calculated tolerance intervals.

2000 Series Wells

The pH of groundwater samples from several 2000 Series Wells was statistically different than background (Table M-3). The pH of groundwater from 2000 Series wells ranged from 6 to 8.1, and the average pH calculated for these wells ranged from 6.25 to 7.68. No spatial trend, such as a plume, was evident from the pH data for the 2000 Series wells.

The mineralogy of the glacial overburden and the proglacial sands and gravels most likely buffers the pH of the groundwater. Evidence of this buffering capacity may be found in the regional aquifer below the southeast corner of the Production Area, where elevated concentrations of calcium, magnesium, and bicarbonate may have resulted from the neutralization of an acidic release.

3000 Series Wells

The pH of groundwater samples from several 3000 Series wells was statistically different than background (Table M-4). The pH value measured in groundwater from 3000 Series downgradient wells ranged from 4.50 to 8.82. The lowest pH was measured during second quarter 1989 in Well 3065. This appeared to be an erroneous reading since there was no reason for low pH groundwater to occur south of the stormwater retention basin. The other measurement in Well 3065 during first quarter 1989 was 7.4.

Based on the pH measurements from 3000 Series wells, the pH of the aquifer at the locations sampled by these wells is not significantly affected by the FMPC facilities. This is most likely attributable to the buffering capacity provided by the glacial overburden and the aquifer sand and gravel sediments.

4000 Series Wells

Statistically different pH values were only measured in Wells 4001, 4102, and 4103 (Table M-5). However, the tolerance interval of pH used for 4000 Series wells may be too narrow. Average pH values in Wells 4001, 4102, and 4103 ranged from 7.39 to 7.55. The pH measured from all 4000 Series wells ranged from 6.75 to 7.92, with the most acidic measurement from second quarter 1989 sampling of Well 4097 and the most basic measurement taken from the third quarter 1988 sample of Well 4096.

Given the pH measurements from 4000 Series wells, the pH at the bottom portion of the aquifer is not affected by the operation of the FMPC and the associated facilities.

Phosphorus, Total

The UTLs for total phosphorus are:

Ross 2000 Series	0.05 mg/l
Ross 3000 Series	0.06 mg/l
Shandon 2000 and 3000 Series	0.20 mg/l
Dry Fork 2000 and 3000 Series	0.22 mg/l
4000 Series	0.22 mg/l

The natural variation of total phosphorus concentration in the aquifer may actually exceed the calculated UTLs. In addition, sources other than the FMPC, such as agricultural activities, may be influencing the distribution of phosphorus in the regional aquifer.

2000 Series Wells

Statistically elevated total phosphorus concentrations were found in groundwater samples from several 2000 Series wells (Table M-3). As they did in 1000 Series wells, total phosphorus concentrations in samples from 2000 Series wells varied greatly from one sampling round to the next. Excluding outliers, detections of total phosphorus in 2000 Series wells ranged from 0.01 to 18.9 mg/L, with the maximum concentration from the second quarter 1989 sampling of Well 2055.

Total phosphorus concentrations were repeatedly greater than 0.5 mg/L in Wells 2006, 2007, and 2053, which are in the Production Area. These elevated concentrations may be related to spills or leaks of tributyl phosphate in the Plant 2/3 area. Total phosphorus concentrations repeatedly in excess of 0.5 mg/L were also detected in several wells located in the vicinity of Paddys Run; i.e., Wells 2095, 2107, and 2108. Because the total phosphorus concentration in surface water samples from Paddys Run was well below 0.1 mg/L (ASI/IT 1990), the source of the elevated phosphorus in these wells may be earlier releases that are no longer detectable in Paddys Run surface water.

Total phosphorus concentrations exceeding 1.0 mg/L were repeatedly observed in groundwater samples from Wells 2094 and 2127. These wells are downgradient from several industrial sites south of the FMPC.

3000 Series Wells

Statistically elevated total phosphorus concentrations were detected in several 3000 Series wells (Table M-4). Excluding outliers, total phosphorus detected in 3000 Series wells ranged from 0.01 to 1.5 mg/L, with the maximum concentration from the third quarter 1988 sampling of Well 3094.

Statistically elevated total phosphorus concentrations in Wells 3004, 3011, and 3043 are probably due to the agricultural activities located upgradient from the FMPC. Elevated phosphorus concentrations in downgradient 3000 Series wells, except for Well 3094, can be ascribed as natural variation. Total phosphorus in Well 3094 ranged from 0.88 mg/L to 1.50 mg/L during the third quarter 1988 to the second quarter 1989 sampling programs. The phosphorus in Well 3094 is most likely due to the industrial sites upgradient of the well and/or agricultural activities in the area.

4000 Series Wells

Statistically elevated total phosphorus concentrations were found in Wells 4008, 4010, 4064, 4067, and 4103. Excluding outliers, total phosphorus concentrations in 4000 Series wells ranged from 0.02 to 0.54 mg/L, with the maximum concentration from the third quarter 1988 sampling of Well 4008.

The elevated phosphorus concentrations observed in 4000 Series wells were very close to the UTL, and may be due to natural variation. Because elevated phosphorus concentrations in 4000 Series wells are very close to the UTL, the FMPC and the associated facilities may not be contributing significant amounts of phosphorus to the lower Great Miami Aquifer.

Sulfate

The UTLs for sulfate are:

Ross 2000 Series	87.22 mg/l
Ross 3000 Series	91.03 mg/l
Shandon 2000 and 3000 Series	12.15 mg/l
Dry Fork 2000 and 3000 Series	49.39 mg/l
4000 Series	49.39 mg/l

The natural variation of sulfate concentration may exceed the calculated UTLs, particularly in wells downgradient from the Shandon Tributary wells.

2000 Series Wells

Statistically elevated sulfate concentrations were detected in almost every 2000 Series well, indicating that the computed UTLs are probably too low (Table M-3).

Average sulfate concentrations in groundwater sampled by 2000 Series wells were generally higher in wells located north of and in the Production Area, compared to wells located in the South Plume area and southern and western portions of the FMPC reservation.

Average sulfate concentrations of approximately 50 mg/L, which are similar to those in Paddys Run surface water, were observed in groundwater from the 2000 Series wells located in the relatively oxidized groundwater zone beneath and east of Paddys Run (Section 15.3.3). Average sulfate concentrations of greater than 100 mg/L were found in wells located in the Production Area, wells located immediately northeast of the waste pits, and Wells 2017, 2051, 2065, 2068, 2096, and 2097. Sulfate detected in 2000 Series wells ranged from 1 to 837 mg/L. The maximum was from second quarter 1989 sampling of Well 2109 near Plant 6.

Elevated sulfate concentrations in perched groundwater beneath the Waste Storage Area or direct leakage from the waste pits represent the probable source of elevated sulfate concentrations at Wells 2027 and 2084. The Production Area is also probably a source of elevated sulfate concentrations in the uppermost regional aquifer. Elevated sulfate concentrations at Well 2065 may

be due to infiltration of surface water from the Storm Sewer Outfall Ditch or the storm water retention basin.

3000 Series Wells

Statistically elevated sulfate concentrations were observed in all 3000 Series wells, except for Wells 3043 and 3098 (Table M-4). Average sulfate concentrations greater than 200 mg/L were observed in groundwater samples from Wells 3010, 3013, 3019, 3037, 3055, and 3084. These wells are located beneath or east of the Waste Storage Area. Wells 3053, 3054, 3064, 3067, 3068, and 3091, located in the southern portion of the Production Area and to the southeast of the Production Area, exhibited average sulfate concentrations ranging from 73.7 to 112.5 mg/L. Wells located in the Dry Fork Section, Wells 3003, 3004, 3011, 3018, 3106, and 3095, had average sulfate concentrations ranging from 71.8 to 95.7 mg/L. Sulfate detected in all 3000 Series wells ranged from 4.75 to 800 mg/L, with the maximum concentration from the second quarter 1988 sampling of Well 3013.

The elevated sulfate concentrations observed in 3000 Series wells located downgradient of the Waste Storage Area is probably due to the migration of contaminated groundwater from the perched zone beneath the waste storage units. The Production Area may also be contributing to the elevated sulfate concentrations observed in groundwater from 3000 Series wells. Elevated sulfate concentrations from the wells located east of Paddys Run and south of the storm water retention ponds may be related to stream and surface water infiltration.

4000 Series Wells

Statistically elevated sulfate concentrations were observed in approximately half of the 4000 Series wells (Table M-5). Average sulfate concentrations greater than 100 mg/L were found in Wells 4013, 4016, 4064, and 4101. Sulfate detected in 4000 Series wells ranged from 5 to 355 mg/L, with the maximum concentration from the second quarter 1989 sampling of Well 4013. This well also exhibited the highest concentration of chloride. Located in the northeast corner of the Production Area, elevated anionic constituents indicate that the well is directly downgradient from a facility using or storing inorganic salts, such as the Waste Storage Area.

Elevated sulfate concentrations at Wells 4013, 4064, and 4101, which are located in or near the Production Area, are probably due to the downward and downgradient movement of groundwater from the Waste Storage and/or the Production areas.

15.4.2.5 Summary

2000 Series Wells

As shown in Figure 15-10, total uranium concentrations greater than 100 ug/L were detected in Wells 2015, 2045, 2046, 2060, 2061, and 2095, which are located in an area extending from Southfield to a location off the southern boundary of the reservation. This area of elevated uranium concentrations is called the South Plume. Total uranium concentrations of greater than 10 ug/L were found in the 2000 Series wells located in the Waste Storage Area, along and immediately east of Paddys Run, and the southeastern quarter of the Production Area. The elevated total uranium concentrations in the South Plume are probably related to the discharge of uranium-bearing water from Paddys Run and the Storm Sewer Outfall Ditch and also possibly leakage from the fly ash piles/Southfield and old farm drain tiles.

The sources of elevated uranium concentrations in the Waste Storage Area are most likely the waste pits and possibly the Clearwell. The elevated uranium concentrations which occur in 2000 Series wells along Paddys Run are due to stream water infiltration; Paddys Run receives uncontrolled runoff from the Waste Storage Area and possibly part of the Production Area. Elevated total uranium concentrations in the 2000 Series wells along Paddys Run indicate that stream infiltration is a pathway for contaminants. Generally, in the vicinity of the Waste Storage and Production areas, total uranium concentrations observed at the water table of the Great Miami Aquifer are significantly lower than those observed in the glacial overburden. The glacial overburden has most likely contributed to the chemical attenuation of uranium.

In addition to uranium, statistically elevated concentrations of thorium isotopes, radium isotopes, Sr-90, and Tc-99 were found in several 2000 Series wells. However, recurrent detections of these radionuclides were only observed at a few wells, as listed below:

- Th-230 — Well 2065
- Ra-226 — Wells 2014 and 2094
- Ra-228 — Wells 2044 and 2094
- Sr-90 — Well 2021
- Tc-99 — Wells 2019, 2021, and 2022

Based on the location of these wells, the repeated detections of Th-230 from Well 2065 may be due to historic releases of stormwater prior to the construction of the retention basins. The recurrent detections of Ra-226 and Ra-228 in Wells 2014 and 2044, respectively, may be associated with stream infiltration from Paddys Run. The repeated detections of Ra-226 and Ra-228 in Well 2094 may be related to the industrial facilities south of the FMPC. The repeated Sr-90 and Tc-99 detections in the Waste Storage Area are associated with leaching of the waste materials

stored in the pits. Thorium and radium isotopes are naturally occurring, and most detections, even repeated detections, are slightly above the minimum detection level. This implies that most detections of these isotopes are not related to the FMPC and associated facilities.

Statistically elevated concentrations of various chemical constituents were detected in various 2000 Series wells located in the southeast portion of the Production Area, including calcium, chromium, magnesium, manganese, nickel, alkalinity, and sulfate. The anomalous chemistry in this area is also reflected by higher conductance measurements from the uppermost portion of the Great Miami Aquifer. The source of these chemical constituents is most likely the Production Area, mainly the pickling operation in Plant 6 and the dissolution of geologic materials in the glacial overburden due to the release of acidic solutions.

Elevated concentrations of calcium, magnesium, potassium, sodium, alkalinity, ammonia, chloride, nitrate, and sulfate were found in various 2000 Series wells located directly beneath and east of the Waste Storage Area. These findings are consistent with the observed chemistry of perched groundwater beneath the waste pits, indicating that the FMPC waste pits are contributing various chemical constituents to the uppermost portion of the Great Miami Aquifer.

Elevated concentrations of lead in Wells 2045 and 2046 may be associated with surface water infiltration from Paddys Run or wastes present in Southfield and the fly ash area. Repeated detections of vanadium from Wells 2047 and 2048 may be related to the waste materials in Southfield and the fly ash area.

The elevated concentrations of calcium, magnesium, alkalinity, and sulfate, repeated detections of cadmium and chromium, and the corresponding high conductance from Well 2065 are probably related either to leakage of the storm water retention basin or historic releases of contaminated runoff prior to the construction of the retention basin.

Statistically elevated sodium concentrations from Wells 2043, 2092, 2093, and 2094, and chloride concentrations from Wells 2092 and 2093 may be related to contamination from the rock salt used to deice roads in the vicinity of the FMPC. Well 2043 is located adjacent to Paddys Run Road, Well 2092 is situated along Route 128, and Wells 2093 and 2094 are downgradient from Willey Road and New Haven Road, respectively.

The source of consistently elevated concentrations of arsenic, barium, cadmium, iron, magnesium, mercury, nickel, potassium, sodium, alkalinity, ammonia, chloride, and phosphorus, and the

corresponding high measurements of conductance in Well 2094 is probably the industrial facilities situated along Paddys Run Road and New Haven Road. Agricultural land use between the FMPC and Well 2094 may also contribute some of these chemicals to the regional aquifer.

The pH of the uppermost portion of the Great Miami Aquifer is not affected by the operation of the FMPC facilities. The pH buffering capacity of the glacial overburden and the sand and gravels would tend to offset the effects of any acidic discharges.

Statistically elevated TOX concentrations were observed in groundwater from Wells 2014 and 2094. In addition, various organic compounds were periodically detected in various 2000 Series wells. Most detections were nonrecurrent and only a few detections were above the CRQL. The pesticides aldrin, heptachlor, and alpha-BHC were detected in Well 2022 during the second quarter of 1988. No PCBs were detected in any 2000 Series well. The source for the detections of volatile compounds, semivolatile compounds, and pesticides in the vicinity of the Waste Storage Area cannot be conclusively determined. The statistically elevated TOX concentrations and the detections of the VOCs in Well 2094 likely have the same source or sources as the abnormally high inorganic chemical concentrations observed in this well.

3000 Series Wells

As shown in Figure 15-11, statistically elevated total uranium concentrations were detected in less than half of the 3000 Series wells. Total uranium concentrations greater than 10 ug/L in 3000 Series wells produced a more discontinuous pattern than in the 2000 Series wells. Readings greater than 10 ug/L occurred at the Waste Storage Area, Well 3108, near the confluence of the Storm Sewer Outfall Ditch and Paddys Run, and also in the South Plume area. Among these wells, total uranium concentrations greater than 30 ug/L occurred only in Wells 3019, 3062, 3084, 3108, and 3125. The unexpectedly high uranium concentrations in Wells 3019 and 3084, which were higher than those from Wells 2019 and 2084, can possibly be attributed to the vertical hydraulic gradient induced by the pumping of test extraction wells near the waste pits from 1965 to 1974. The sources of elevated total uranium concentrations observed in the 3000 Series wells are analogous to those contributing uranium to the portion of the Great Miami Aquifer sampled by the 2000 Series wells. However, the middle portion of the aquifer monitored by 3000 Series wells is less extensively contaminated than the uppermost portion of the aquifer. This may be due to chemical attenuation of uranium in the aquifer, or may indicate that the plume has not yet fully migrated vertically.

Except for uranium isotopes, most of the statistically elevated concentrations of other radionuclides in 3000 Series wells were low-level, isolated, and nonrecurrent. Thus, the FMPC facilities are not contributing significant amounts of radionuclides, except for uranium, to the middle and lower portions of the Great Miami Aquifer.

The groundwater chemistry observed in Wells 3010, 3013, 3019, 3037, 3055, and 3084 is characterized by elevated concentrations of calcium, magnesium, potassium, sodium, chloride, and sulfate. Elevated concentrations of iron, manganese, alkalinity, ammonia, and nitrate are also observed in some of these wells. These wells are located directly downgradient from the Waste Storage Area and within or near the Production Area. Based on the magnitude and distribution of various constituents in these wells, the waste pits and the Production Area appear to be contributing a variety of inorganic chemicals to the portion of the Great Miami Aquifer sampled by the 3000 Series wells.

Elevated concentrations of total phosphorus in Wells 3004, 3011, and 3043, which are located along Paddys Run, may be related to agricultural activities to the north of the site and not the operation of the FMPC facilities. Elevated sulfate concentrations observed in Wells 3003, 3004, 3011, 3018, 3106, and 3095 east of Paddys Run are probably due to stream water infiltration from Paddys Run.

Elevated concentrations of various chemical constituents, including barium, iron, magnesium, nickel, potassium, sodium, alkalinity, ammonia, chloride, total phosphorus, and sulfate are observed in groundwater at Well 3094. The abnormal chemistry observed at this well is apparently related to the industrial facilities situated south of the FMPC. The anomalous chemistry of groundwater from Well 3094 is similar to that observed in Well 2094.

The pH values measured in groundwater immediately above the clay interbed indicate that the FMPC facilities are not influencing the pH of the Great Miami Aquifer at this level.

Repeated detections of TOX are observed in Well 3014, which is located south of the fly ash piles and Southfield. The source of the TOX is probably the waste materials disposed in these areas or surface water infiltration. Other than at this location, volatile and semivolatile organic compounds detected in 3000 Series wells were isolated and nonrecurrent. No pesticides or PCBs were detected in any 3000 Series wells. Thus, the FMPC and the associated waste storage units are not contributing volatile organics, semivolatile compounds, pesticides, or PCBs to the deeper portions of the aquifer.

4000 Series Wells

Radionuclide detections in groundwater samples from 4000 Series wells were all low-level and nonrecurrent. Thus, it appears that the FMPC and the associated waste storage units are not contributing any radionuclides, including uranium, to the bottom portion of the Great Miami Aquifer.

Elevated concentrations of various chemical constituents were detected in Wells 4010, 4013, 4064, and 4101 located immediately east of the waste pits. These chemical constituents included barium, calcium, iron, magnesium, potassium, sodium, alkalinity, ammonia, chloride, fluoride, and sulfate. Except for barium, most of the elevated concentrations of chemical constituents can be related to FMPC waste materials and the observations in the 2000 and 3000 Series wells. Elevated barium concentrations observed in 4000 Series wells may be due to barium dissolution from bedrock, since no correlative barium concentrations existed in the corresponding 2000 and 3000 Series wells. Barium would likely have been detected in the upper wells if Pit 4 was the source.

The pH at the bottom of the Great Miami Aquifer shows no significant difference from background; thus, the pH at the base of the aquifer is probably not affected by the FMPC and associated facilities.

No significant levels of any organic compounds were observed in groundwater from any 4000 Series well sampled during RI/FS and 1987 RCRA monitoring. It appears that the FMPC and the associated facilities are not contributing significant quantities of volatiles, semivolatiles, PCBs, or pesticides to the Great Miami Aquifer.

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16.0 GEOCHEMICAL PARAMETER ESTIMATION

16.1 INTRODUCTION

Field, laboratory, literature and modeling studies were conducted to obtain estimates of the geochemical parameter values required for the FMPC solute transport modeling, geochemical, and risk assessment programs. Because of the dominance of uranium as a contaminant of concern at the FMPC, the groundwater and related modeling studies were initially focused on the nature, extent, and potential migration of uranium plumes. A geochemical program was developed in a separate 1988 work plan to address distinct issues of importance to the understanding of uranium behavior within the FMPC environment. These issues were intended to address:

- Uranium concentrations in glacial overburden and Great Miami Aquifer solids
- The leachable fraction of uranium present in these solids
- The distribution of uranium valence in groundwater
- The uranium distribution coefficient applicable to the site-specific characteristics of the Great Miami Aquifer

Appendix P contains a more complete description of these issues within the context of the geochemical program. Results obtained to date are also documented in detail in Appendix P.

The subsequent issuance of new RI/FS and risk assessment guidance by the EPA redefined the concept of contaminants of concern and required that numerous radionuclides and hazardous chemicals be more fully analyzed in the FMPC RI/FS. As a consequence, the fate and transport modeling associated with the risk assessment program required literature searches to obtain information on the retardation of additional contaminants by subsurface soils in the glacial overburden. Geochemical modeling also had to be extended to estimate leachate concentrations introduced into the regional aquifer from the waste units comprising Operable Units 1 through 4. This chapter will focus on the geochemical studies used to support the groundwater modeling and risk assessment programs.

16.2 FIELD ANALYTICAL PARAMETERS

Observed total uranium and U^{+4} concentrations were used to evaluate uranium reduction-oxidation (redox) reactions in groundwater. A redox reaction can be described as a gain (reduction) and loss (oxidation) of electrons between constituents in groundwater. A complete redox reaction must involve both reduction and oxidation half-reactions, as the total number of electrons in the

groundwater must remain constant to maintain electrical neutrality. Other general water quality parameters such as pH and dissolved oxygen are discussed in Chapter 15.0. Analytical results for total uranium and U^{+4}, and a more detailed discussion of these results, are provided in Appendix P.

The analytical methods for obtaining the fraction of U^{+4} and total uranium from groundwater samples has been discussed in Section 6.3.7. Table 16-1 presents the results of several methods used to determine the Eh of groundwater. Based on these results, the Eh range determined by measurement with a platinum electrode and the Eh range calculated using the NH_4^+/NO_3^- ratio are in agreement. However, the Eh range calculated from the U^{+4}/U^{+6} ratio appears to be too low. The low Eh values calculated by the U^{+4}/U^{+6} ratio are attributed to the adsorption of U^{+6} species, like $UO_2(CO_3)_2^{-2}$, by iron-bearing colloidal particles. Colloidal particles are widely defined as particles with diameters between 1 and 0.001 micron (um). Consequently, a fraction of the iron-bearing colloidal particles that adsorbed U^{+6} species would not have passed through the 0.1 um filter used in the sampling procedure and would have become a portion of the filtered residue analyzed for U^{+4} (Appendix P). This led to unrepresentatively high concentrations of U^{+4} being measured in groundwater and, ultimately, unrepresentatively low Eh values based on the U^{+4}/U^{+6} ratio.

16.3 LABORATORY STUDIES

Thirty-one subsurface solid samples obtained from drilling cores and cuttings during the installation of monitoring wells underwent testing to characterize physical and chemical properties of the glacial overburden and aquifer sands and gravels. The solids were characterized by measuring the following parameters: total organic carbon (TOC); total cation exchange capacity (CEC); grain-size variation; leachable fraction of uranium, iron and manganese; total uranium (only 13 of 31 samples were tested); and differential leaching of uranium (only 7 of 31 samples were tested). These analytical parameters were measured to evaluate uranium adsorption on subsurface solids and to support the estimation of a uranium adsorption coefficient for the Great Miami Aquifer. The analytical approach and methods have been presented in Section 6.3.7.

The principal conclusion reached by these studies was that less than 3 ug/kg (ppb) of uranium was adsorbed onto the subsurface solids tested. Analytical results for these parameters, and a detailed account of the results, can be found in Appendix P.

TABLE 16-1
COMPARISON OF METHODS USED TO OBTAIN GROUNDWATER Eh VALUES

Method	Eh (millivolts)
Platinum electrode	+75 to +454
NH_4^+/NO_3^- ratio	+324 to +350
U^{+4}/U^{+6} ratio	-120 to -160

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Based on a sieve analysis of the subsurface solids, clay comprised greater than 50 weight percent of the glacial overburden and major clay interbed samples, and silt plus clay comprised less than 25 weight percent of the regional aquifer samples. The leachable fraction of iron, manganese and uranium in the majority of subsurface-solid samples, and the amount of TOC, were not a function of the particle size. The range of values for these parameters was similar for glacial overburden clays and silts and outwash sands and gravels. This observation suggests that the majority of iron and manganese solids contacted by the groundwater are present as oxyhydroxide coatings along fractures in the fluid-flow path, rather than as detrital grains or surface coatings on the individual particles (see Appendix P for details).

Oxyhydroxide coatings of iron and manganese are of interest because their surfaces will generally have a net positive charge below a pH of 8, and anionic forms of uranium (e.g., $UO_2(CO_3)_2^{-2}$) can adsorb to these positive surfaces. However, a lack of correlation between the sum of leachable iron and manganese versus total uranium indicates that adsorption of uranium on iron- and manganese-oxyhydroxide coatings may not be occurring in the aquifer. Another key observation of these studies was the correlation of high TOC values to high leachable iron values, which implies that iron is associated with or adsorbed on organic materials in the aquifer solids.

The CEC of a soil or sediment is dependent on the mineralogy and size of the particles, and varies from less than 0.01 meq/g for quartz and feldspar to greater than 1 meq/g for vermiculite, zeolites and organic matter (Carroll 1960, 1970). The analytical results on the CEC of the aquifer solids indicated that CEC is a direct function of the silt and clay content of the aquifer. Samples containing less than 25 weight percent of silt and clay had a total CEC of less than 0.04 milliequivalents of charge per gram of material (meq/g), whereas those samples with greater than 50 weight percent silt and clay had CEC values of up to 0.28 meq/g.

16.4 LITERATURE STUDIES

A literature search was conducted to obtain estimates of adsorption distribution coefficients (Table 16-2), which are used to quantitatively describe for contaminant retardation along the flow path through the glacial overburden. Note that the adsorption distribution coefficient for uranium is based on site-specific data, and details of the development of this value are presented in Issue 5 of Appendix P. The adsorption distribution coefficient is commonly referred to as K_d, and has the units of liter per kilogram (L/kg). It is defined as the concentration of an element adsorbed on the solid (mg/kg) divided by the concentration of the element in solution (mg/L).

TABLE 16-2

ADSORPTION COEFFICIENTS K_d FOR CERTAIN INORGANIC ELEMENTS

Element	K_d (L/kg)	Reference
Co	56	Hawkins (1964)
Cd	500	Gerrotse et al. (1982)
Cs	1800	Gillham et al. (1981b)
Pb	3000	Gerritse et al. (1982)
Pu	1700	Glover et al. (1976)
Ra	696	Gillham et al. (1981a)
Sr	10	Gillham et al. (1981b)
Tc	0.118	Balogh and Grigal (1980)
U	1.8	Appendix P

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The literature search was focused on adsorption studies conducted with soils that had physical and chemical properties similar to those of the glacial overburden, as measured by the fraction of sand, silt and clay, pH, CEC, and TOC. When a range of K_d values was reported for a given soil, the lowest, therefore most conservative, value was chosen. The lowest K_d is considered most conservative because, when input into a transport model, it will result in the relatively highest peak concentration of a contaminant, and the peak will occur after the relatively shortest time interval, once the contaminant is released from the waste source. Additionally, the literature search attempted to focus on studies conducted with solutions that were similar in composition to groundwater in the glacial overburden. Reasonable matches between documented studies and conditions at the FMPC were found for the soil properties, but not for the solutions. Most of the documented studies used dilute solutions that were made by adding a small volume of acid solution spiked with the contaminant of interest, such as uranium in nitric acid, to distilled water. These "simple" laboratory solutions do not simulate the groundwater composition at the FMPC accurately. Nevertheless, K_d values obtained from these experiments were useful as initial estimates of contaminant retardation in the glacial overburden. Excluding uranium, the K_d values represent literature values derived from batch sorption tests with solids that have physical and chemical properties similar to solids in the glacial overburden (e.g., weight fraction of clay, cation exchange capacity, etc.). However, it was not possible to match the chemical composition of laboratory solutions to groundwater in the glacial overburden. In general, the laboratory solutions were found to overlap the pH range reported for groundwater in the glacial overburden, but the laboratory solutions did not contain all the ligands (e.g., SO_4^{-2}, PO_4^{-3}, etc.) and metals (e.g., NA^+, K^+, Mg^{+2}, etc.) found in the groundwater. Therefore, the K_d values in Table P-1 (except uranium) do not reflect competition among ions for available sorption sites, and Table 1 values may overestimate the K_d for the given contaminants. At this time, the site-specific sorption tests are being planned to remove the uncertainty associated with literature values; but, until these tests are completed, Table P-1 contains the best available data.

Uranium K_d values were reported for the glacial overburden and regional aquifer in Appendix P. The literature value was discarded in favor of Appendix P because the latter value is derived from site-specific data on leachable uranium and uranium speciation in groundwater. Therefore, the K_d value of 1.8 was selected for uranium and was used in the modeling. Further refinement of the K_d values was achieved through model calibration and additional studies are planned. These studies will include laboratory measurements of K_d using soils and groundwaters from the FMPC.

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The K_d values for organic chemical constituents were calculated by using the following empirical formula (Mills et al. 1985):

$$K_d = K_{ow}\ (0.63)\ [0.2\ (1\text{-}f)\ (X_{soc}) + (f)\ (X_{foc})]$$

where:

K_{ow} = octanol-water partition coefficient
f = mass fraction of silt or clay
X_{soc} = organic carbon content in sand fraction
X_{foc} = organic carbon content in silt-clay

For the organic constituents, the K_{ow} was taken from the U.S. EPA Treatability Database and Volumes I and III of the Handbook of Environmental Fate Exposure Data for Organic Chemicals (Howard 1990). The other parameters were estimated based on characteristics of geologic units. The calculated K_d values are shown in Chapter 22.0.

16.5 GEOCHEMICAL MODELING

Geochemical modeling was conducted to investigate uranium speciation such as the distribution of UO_2^{+2}, UO_2F^+, $UO_2(CO_3)_2^{-2}$, and $UO_2(CO_3)_2^{-2}$ in groundwater, and to estimate leachate concentrations that could develop at the base of each waste unit and the base of the glacial overburden. The results obtained from modeling the uranium speciation in groundwater were used to estimate the retention of uranium by adsorption onto subsurface solids. Leachate concentrations derived from the modeling are being used as input data to fate and transport modeling to estimate the contaminant dose received at the 100-year (off-FMPC) and 500-year (on-FMPC) human receptor sites.

16.5.1 Geochemical Computer Codes

Geochemical modeling was conducted with the EQ3/6 geochemical code (Wolery 1983; 1984) and was restricted to inorganic systems, because thermodynamic data on organic constituents of interest are not available. The discussion which follows is focused on the derivation of leachate concentrations. The speciation study is addressed and documented in Appendix P.

EQ3/6, an industry-standard geochemical code, performs solubility, speciation, and reaction-path calculations. Solubility and speciation calculations reveal the maximum concentration a contaminant can have in solution and the aqueous form(s) of that contaminant for a specific solid/liquid/gas system. Reaction-path calculations enable a solution to migrate through, and equilibrate with,

different solids. This can be used to simulate groundwater that moves through compositionally distinct stratigraphic horizons.

The EQ3/6 package was developed at Lawrence Livermore National Laboratory for predicting the behavior of metals, radionuclides, and other contaminants in the natural environment. The code accesses a data base containing the thermodynamic properties of 47 elements, 686 aqueous species, 713 minerals, and 11 gases. This data base includes 49 uranium-bearing aqueous species and 53 uranium-bearing minerals, constituting the most complete data base available for modeling the behavior of uranium in natural waters. EQ3/6 has been validated using standard geochemistry problems, such as the speciation of sea water (Nordstrom 1979), basalt/sea water interactions (Bowers et al. 1985), and numerous comparisons with experimentally determined mineral solubilities (Jackson 1988). Benchmark comparisons with the results of similar codes (e.g., PHREEQE) have been performed by INTERA (1983), Nordstrom (1979), Kincaid and Morey (1984), and Kerrisk (1981).

The PHREEQE (Parkhurst et al. 1980) geochemical code was used to carry out solubility and speciation calculations for waste constituents not contained in the EQ3/6 thermodynamic data base, such as cadmium and molybdenum. PHREEQE is also an industry-standard program which has been validated against standard geochemistry problems.

16.5.2 Limitations and Assumptions

The geochemical codes used to develop the estimates of leachate concentrations were based on the following limitations and assumptions:

- Only inorganic systems can be modeled, and this can lead to low estimates of leachate concentrations for some constituents if organic complexation is significant
- Mineral phases reacted with the leaching fluid in the model are assumed to represent the actual solid phases in the waste material and glacial overburden
- Dissolution and precipitation kinetics are assumed to be instantaneous, which can lead to estimates of concentrations that are too high or too low
- Adsorption (includes ion exchange) processes are not considered
- Modeled concentrations are site-specific solubility limits, and in most cases these concentrations are the highest concentrations which can exist in solution
- Rain water, or groundwater, is assumed to enter the waste units to act as the leaching fluid

The first four limitations and assumptions introduce the greatest uncertainty in the calculated leachate concentrations and merit further discussion. With respect to the first assumption, for example, a solution containing trace amounts of tributyl phosphate will contain a greater concentration of uranium relative to a solution with no tributyl phosphate. If a contaminant is present in a metastable, amorphous form rather than a crystalline mineral, using a mineral phase for the solid will usually lead to low estimates of the contaminant concentration. This occurs because crystalline phases generally contain shorter and stronger element bonds relative to amorphous solids, and this leads to a lower solubility for the crystalline phase. Dissolution of crystalline solids is rarely instantaneous or complete in the natural environment, except for some highly soluble salts like sodium chloride, and this can lead to overestimates of contaminant concentrations. On the other hand, the assumption of instantaneous precipitation of mineral phases can lead to low estimates of element concentrations. This is particularly true if the mineral is difficult to nucleate and crystallize in the natural environment. Finally, the calculated concentrations of soluble species may be too high because adsorption reactions are not considered. Adsorption processes could lower some contaminant concentrations substantially below the calculated solubility limit.

16.5.3 Geochemical Modeling of Leachate Concentrations

Geochemical modeling was conducted with the EQ3/6 geochemical code to estimate leachate concentrations at the base of each defined waste unit within an operable unit and at the base of the glacial overburden. Leachate values at the base of the glacial overburden were used as input to the long-term fate and transport modeling (Chapter 22.0) to estimate the contaminant dose received at the 100-year (off-FMPC) and 500-year (on-FMPC) human receptor sites.

16.5.3.1 Development of the Conceptual Model

The conceptual model, as illustrated in Figure 16-1 accounts for infiltration of rain water or groundwater into the waste storage unit, its reaction with inorganic waste solids present in the unit, and subsequent reaction of the leachate derived from the waste unit with minerals in the underlying glacial overburden. For some waste units, solution analyses were available to describe the actual leachate derived from the waste unit, and only the reaction of the leachate with the glacial overburden mineralogy was modeled.

The path used by the rain water or groundwater to enter the waste unit is not an issue for the geochemical modeling and is not addressed here. Once the rain water or groundwater has reached

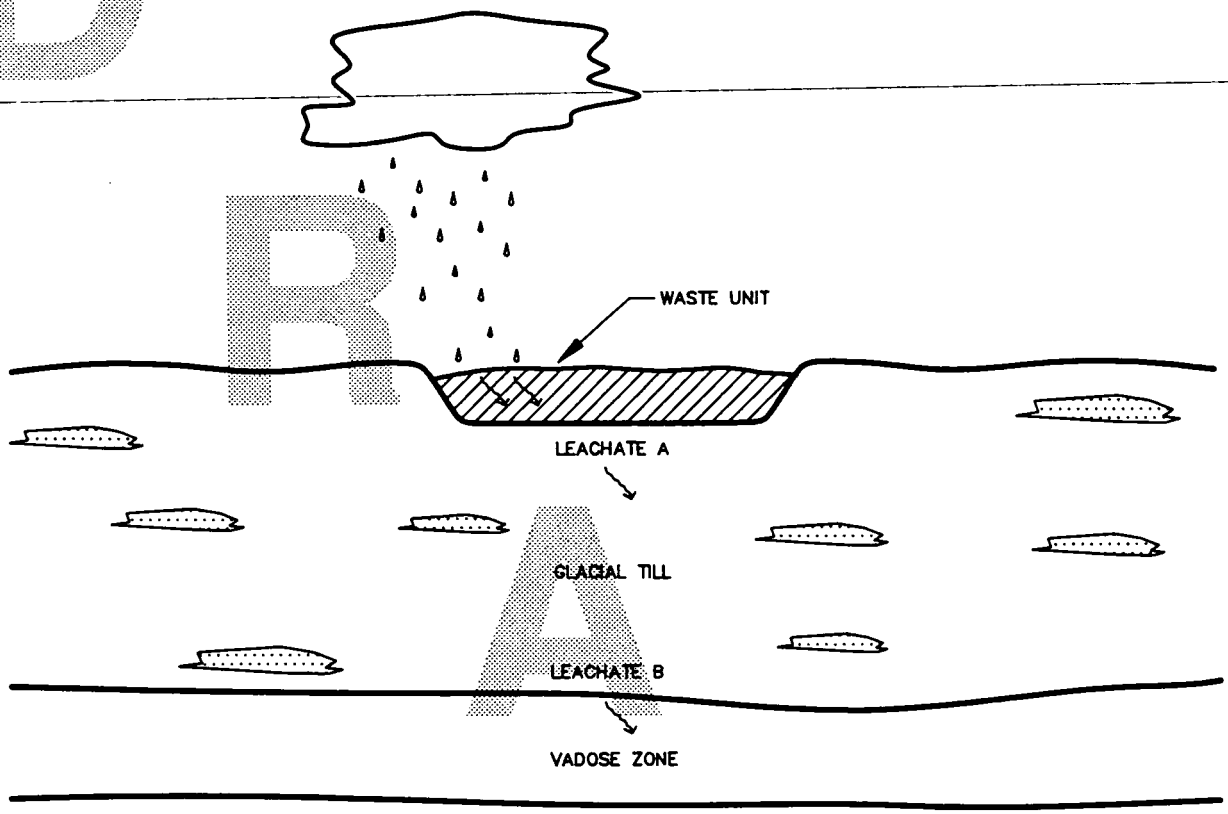

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FIGURE 16-1
CONCEPTUAL MODEL FOR GEOCHEMICAL MODELING

the waste, it is assumed that the water instantaneously equilibrates with the waste solids. This assumption is a requirement of the mathematical model because kinetic data on the waste solids are not available to assess the time-dependent equilibration. As the initial solution equilibrates with the waste solids, it acquires the characteristics of a leachate. This leachate exits at the base of the waste unit, where it is referred to as Leachate A (Figure 16-1). Leachate A continues to migrate downward through the glacial overburden underlying the waste unit, and is assumed to instantaneously equilibrate with the minerals in the glacial overburden. Again, this assumption is a requirement of the mathematical model because of the lack of kinetic data on minerals in the glacial overburden. Along this portion of the flow path, Leachate A is free to react with minerals in the glacial overburden, possibly reducing the concentration of contaminants by precipitation of solids. At the base of the glacial overburden, the modified leachate is referred to as Leachate B (Figure 16-1). The concentrations of contaminants in Leachate B are then used as input to the fate and transport modeling in the regional aquifer.

16.5.3.2 Development of the Leachate Composition at the Base of the Waste Unit

Two different scenarios were used to estimate the composition of Leachate A. The first scenario involved the simulated titration of mineral phases present in the waste unit into modified rain water or groundwater to develop Leachate A. Titration is the process of adding mineral constituents to a solution in their molar proportions. For instance, if one mole of dolomite, $CaMg(CO_3)_2$, is titrated into solution, the mass of calcium and magnesium in the solution increases by one mole and carbonate by two moles. A second scenario used the chemical analysis of water obtained from within the waste unit to represent Leachate A. Because the second scenario required no modeling to develop Leachate A, only the first scenario is discussed below.

For the simulated titration, waste characterization data were reviewed to estimate the mineralogy within each waste unit. The data consisted of descriptions of the processes which generated the waste, general descriptions of the waste forms, elemental analyses of the waste, and, infrequently, actual data on the mineral phases within the waste. All types of data were not available for each waste unit and, in general, the data were not adequate to conclusively determine the mineralogy of the waste. Therefore, to account for waste solids represented by the elemental analysis of the waste, many mineral phases had to be assumed to be present. The assumed mineral phases were chosen based on the elemental concentrations reported for the waste, and were formed by combining metals and radionuclides with reported ligands such as bicarbonate, sulfate, and phosphate. For example, barium was combined with sulfate to form the mineral barite ($BaSO_4$).

After selecting the waste-bearing mineral phases, the minerals were numerically titrated into rain water or groundwater at rates proportional to their molar abundance in the waste. The molar abundance of the mineral in the waste was determined directly from analytical data by converting the mass concentrations to molar concentrations. The titration step was used to increase the concentrations of dissolved constituents in solution. As solution concentrations increased, solubility limits were reached and solid phases were precipitated from the solution. When the system reached equilibrium, the modeling was stopped. The solution composition at the termination of modeling was assumed to represent the leachate composition at the base of the waste unit, except for the metals Ag, As, Ba, Cr, Hg, Pb and Se, which were estimated using the maximum value obtained from EP toxicity or TCLP tests, when available.

16.5.3.3 Development of the Leachate Composition at the Base of the Glacial Overburden

The leachate composition at the base of the waste unit, Leachate A, was modified by simulating its reaction with the minerals assumed to be present in the glacial overburden. Minerals in the glacial overburden underlying the waste units were assumed to be similar to analyses of glacial overburden material recovered from a terminal lobe of similar age in South Dakota (Barari et al. 1987). This assumption was required because mineralogic data for the glacial overburden at the FMPC site was not available. It should be noted that any errors introduced by this assumption are not significant relative to the conservation in the results introduced by the assumptions and limitations outlined in Section 16.5.2.

The minerals in the glacial overburden were numerically titrated into Leachate A at rates proportional to their molar abundance. As the titration proceeded, the composition of Leachate A was modified by the dissolution of minerals in the glacial overburden and precipitation of both glacial overburden and secondary mineral phases. The secondary mineral phases represent solids that are stable in the presence of leachate and glacial overburden, but which are not present in the glacial overburden prior to the introduction of leachate. When the system reached equilibrium, the modeling was stopped. The modified leachate composition at the termination of modeling was assumed to represent the leachate composition at the base of the glacial overburden, and is referred to as Leachate B (Figure 16-1).

16.5.3.4 Modeling Results

Leachate concentrations derived from the geochemical modeling are given in Section 22.1.3. It is reemphasized that these concentrations were derived with conservative assumptions, and they represent estimates of the maximum concentrations expected in leachates present at the base of the waste unit and at the base of the glacial overburden. Actual values would likely be lower than

these model results; however, it is extremely unlikely that the actual values will be higher than the model predictions.

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17.0 OPERABLE UNIT AREA HYDROGEOLOGY

Operable Units 1 through 4 are potential sources of contamination to the glacial overburden and the Great Miami Aquifer. Chapter 17.0 presents the features of these operable units, including a detailed description of the glacial overburden and the perched water system underlying each operable unit.

17.1 OPERABLE UNIT 1

17.1.1 Operable Unit Features

Operable Unit 1 (OU1) includes many of the waste disposal units which lie to the west-northwest of the Production Area. Within OU1 are the six waste pits, the burn pit, and the Clearwell (Figure 17-1). These units were constructed over a 27-year period, from 1952 to 1979. A brief description and history of each follows:

- Waste Pit 1, constructed in 1952, was excavated into an existing clay lens and was lined with clay excavated from the burn pit (H&R 1986). The bottom of Pit 1 is estimated to be at an elevation of 558 feet. Expansion of the pit in 1957 provided a total capacity of 40,000 cy. The waste material that was disposed in the waste pit consisted primarily of neutralized waste filter cakes, production plant sump cakes, depleted slag, scrap graphite, contaminated brick, and sump liquor. The quantity of uranium disposed in the pit is estimated to be 52,000 kg (114,400 lbs). Waste Pit 1 was closed in 1959, backfilled, and covered with clean fill dirt (NLO 1985c). Surface water runoff is diverted to the Clearwell prior to its discharge to the Great Miami River.

- Waste Pit 2 was constructed in 1957 and was operated from 1957 to 1964 (H&R 1986). This pit was constructed in a small pond east of Waste Pit 1 and was lined with a compacted clay layer (Dames and Moore, n.d.). The bottom of the pit is estimated to be at an elevation of 562 feet. Waste Pit 2 received primarily dry, low-level radioactive wastes consisting of neutralized waste filter cakes, sump cakes from production plants, depleted slag, scrap graphite, contaminated brick, sump liquor, and concentrated raffinate residues. The pit holds approximately 13,000 cy of wastes that contain about 1,206,000 kg (2,653,200 lbs) of uranium and approximately 400 kg (880 lbs) of thorium. The waste pit has been covered with clean fill and graded to provide surface drainage to the Clearwell for subsequent discharge to the Great Miami River.

- Waste Pit 3 was constructed in 1959 by excavating through the glacial overburden and placing a layer of clay along the pit walls (Dames and Moore, n.d.). The estimated pit bottom elevation is at an elevation of 548 feet. Waste Pit 3 was operated as a settling basin, from 1959 to 1968, receiving wet waste streams consisting of lime-neutralized waste solution from the FMPC production plants (H&R 1986; NLO 1985c).

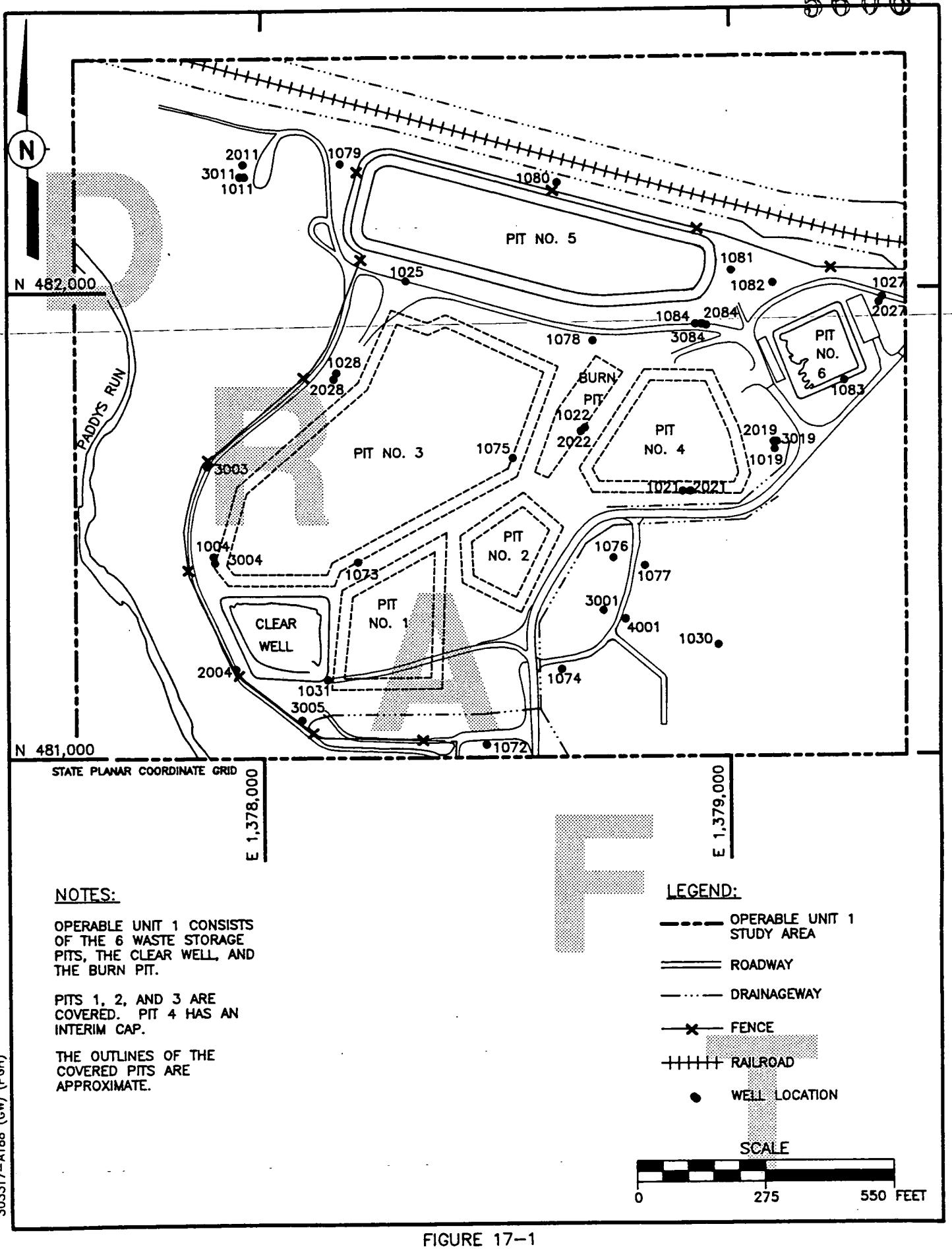

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FIGURE 17-1
LOCATION MAP OF OPERABLE UNIT 1
WASTE DISPOSAL PITS AND
MONITORING WELLS

From 1975 to 1977, the waste pit was used to dispose slag leach residue, filter cakes, fly ash, and lime sludges. The pit contains an estimated 227,000 cy of wastes, including 129,000 kg (285,600 lbs) of uranium and 400 kg (880 lbs) of thorium. The pit was retired in 1977 and clean fill was placed over the area. Surface water runoff is diverted to the Clearwell prior to discharge to the Great Miami River.

- Waste Pit 4 was constructed in 1960 and was used until May 1986. This pit was constructed similar to Pit 3, but with a bottom elevation of about 560 feet. Waste Pit 4 received process residues, filter cakes, slurries, raffinates, graphite, barium chloride waste sludges (received from RMI from 1981 - 1983), noncombustible trash, and asbestos (H&R 1986; NLO 1985c). The pit contains an estimated 563,000 cy, including more than 3 million kg (6.6 million lbs) of uranium and 61,800 kg (135,960 lbs) of thorium. Primarily due to the presence of barium, an interim RCRA cap was placed over Pit 4.

- Waste Pit 5 was constructed in 1968 and was operated from 1968 to 1983 (H&R 1986; NLO 1985c). The bottom elevation is 558.5 to 560 feet. The pit is lined with a 60-mil-thick Royal-Seal EPDM Elastomeric Membrane. This liner has had occasional joint failures and tears (NLO 1985c) which have been repaired. Like Waste Pit 3, this waste pit received liquid waste slurries from the Refinery and the Recovery Plant, including neutralized raffinate solids, slag leach slurry, sump slurries, and lime sludges. The waste volume consists of approximately 102,500 cy, containing 50,309 kg (110,680 lbs) of uranium and 17,000 kg (37,400 lbs) of thorium (H&R 1986). Between 1983 and 1987, Pit 5 received only clear decant from the General Sump, filtrate from the Recovery Plant, or nonradioactive slurries, such as blowdown from the Boiler Plant and Water Treatment Plant. Only rainfall has entered Pit 5 since it became inactive in 1989.

- Waste Pit 6 was constructed in 1979 and operated until 1985 (H&R, 1986). It was constructed in the same manner as Waste Pit 5 and lined with a similar synthetic liner, although no joint failures or tears have been observed (NLO 1985c). The bottom of Pit 6 is at an elevation of 560 feet. Solid wastes, including green salt, filter cakes, and process residues, containing elevated levels of uranium, have been disposed in this pit. Rainfall that is collected in the pit is pumped to Waste Pit 5 for settling and discharge to the Clearwell. The current waste volume is approximately 9,000 cy, which consists of 843,142 kg (1,854,972 lbs) uranium (H&R 1986). No materials have been added to the pit since April 1985.

- The burn pit was first used as a clay borrow area to line Waste Pits 1 and 2. The resulting excavation was subsequently used to dispose of laboratory chemicals and to burn combustible materials, including pyrophoric and reactive chemicals, oils, and other low-level mixed combustible materials (H&R 1986). The actual inventory of materials or chemicals that was disposed in the burn pit is unknown. The boundaries and bottom elevation of the burn pit are also unknown.

- The Clearwell was first used in 1954. It continues in use as a temporary holding facility for storm water runoff. It has a bottom elevation of

548 feet. The Clearwell served as the second solid settling tank in the FMPC liquid waste treatment system and also receives runoff from the general site area. Effluent from the Clearwell is discharged to the Great Miami River through the main effluent line. The Clearwell was periodically dredged to remove accumulated solids. The actual amount or inventory of materials or chemicals that remain in the Clearwell is unknown.

Topographic relief of the OU1 area is moderate, with variations of about 10 feet across the Waste Storage Area. The original topography has been highly modified by pit construction and capping, including filling and rerouting of a tributary to Paddys Run which once flowed in the vicinity of Waste Pits 3 and 5.

17.1.2 Glacial Overburden

Depositional characteristics of the glacial overburden specific to OU1 are shown on the Operable Unit 1 Glacial Overburden Fence Diagram (Plate Q-1, Appendix Q). The Geologic Cross Section Location Map (Figure 17-2) shows that Cross Section M-M' (Figure 17-3) and Cross Section N-N' (Figure 17-4) pass orthogonally through OU1. The stratigraphic descriptions given on the cross sections are based on lithology and coloration as seen in borehole samples. The units generally correlate with depositional units described by other researchers (Leggett 1976; Cravens 1987; Barari 1988; Brockman 1988; Hendry 1988).

The glacial overburden unconformably overlies the sands and gravels of the Great Miami Aquifer and ranges from 29 to 43 feet thick in OU1. The gray and brown clays and silts represent the Wisconsin glacial till. The variations in color indicated in Plate Q-1 are due to differential weathering of the unit rather than depositional differences. Based on the descriptions found in the U.S. Army Corps of Engineers' boring program conducted during plant construction, it appears that near-surface brown silts represent either lacustrine deposits associated with a glacial lake, or postglacial retreat loess deposits. Sand and gravel stringers and beds found within the glacial overburden are undifferentiated glaciofluvial outwash deposits.

The depth of weathering in the glacial overburden varies across the Waste Storage Area from west to east. In the western portion of the operable unit, the weathered zone extends approximately 40 feet deep, nearly to the top of the regional aquifer, as seen in Wells 3004, 1031, 1073 and 1079. This is because Paddys Run has eroded a channel almost to the base of the glacial overburden and drains almost the entire thickness of the glacial overburden. One the eastern side of the operable unit, typical weathering depths are 7 to 15 feet and the gray unweathered clay is relatively thick.

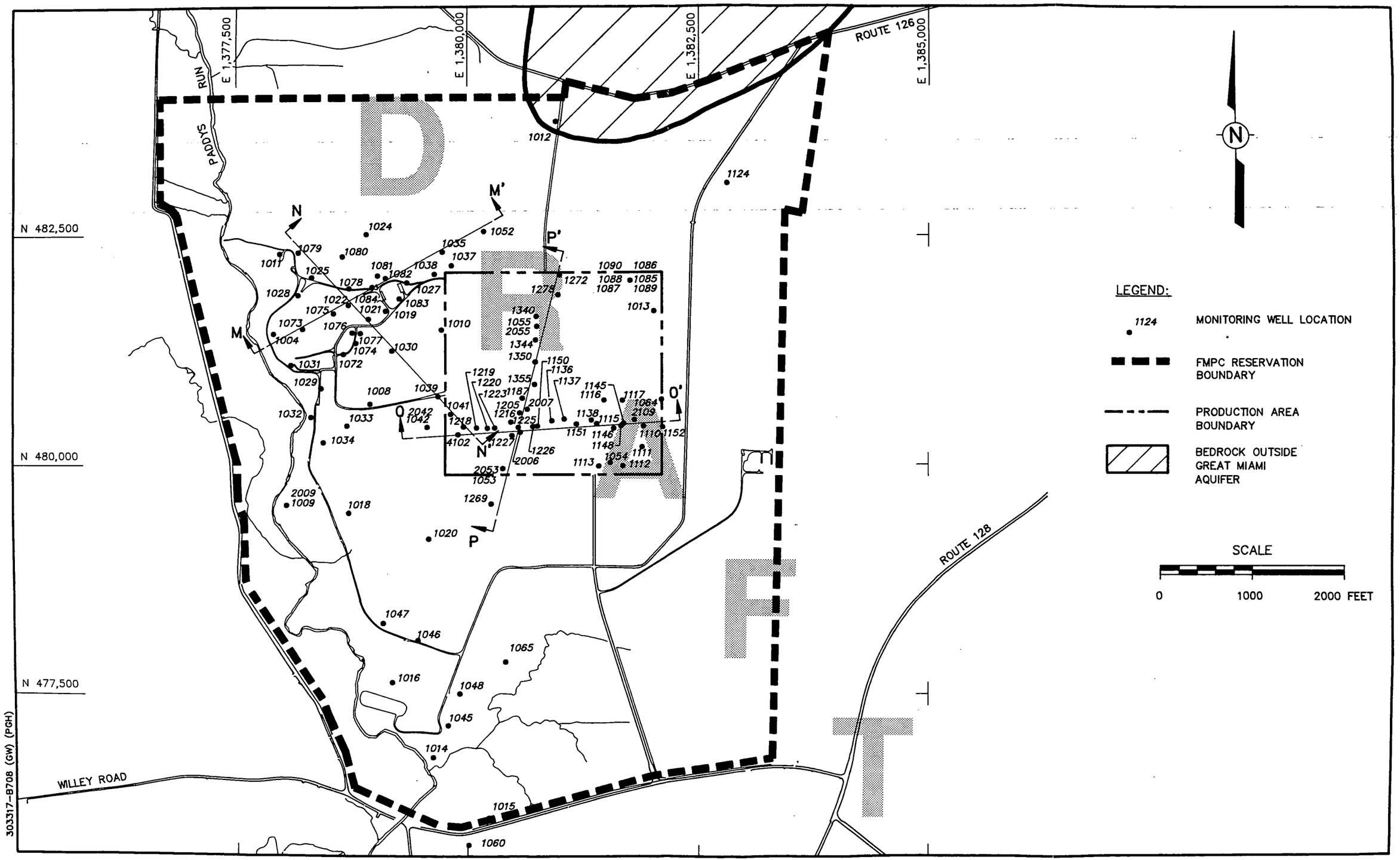
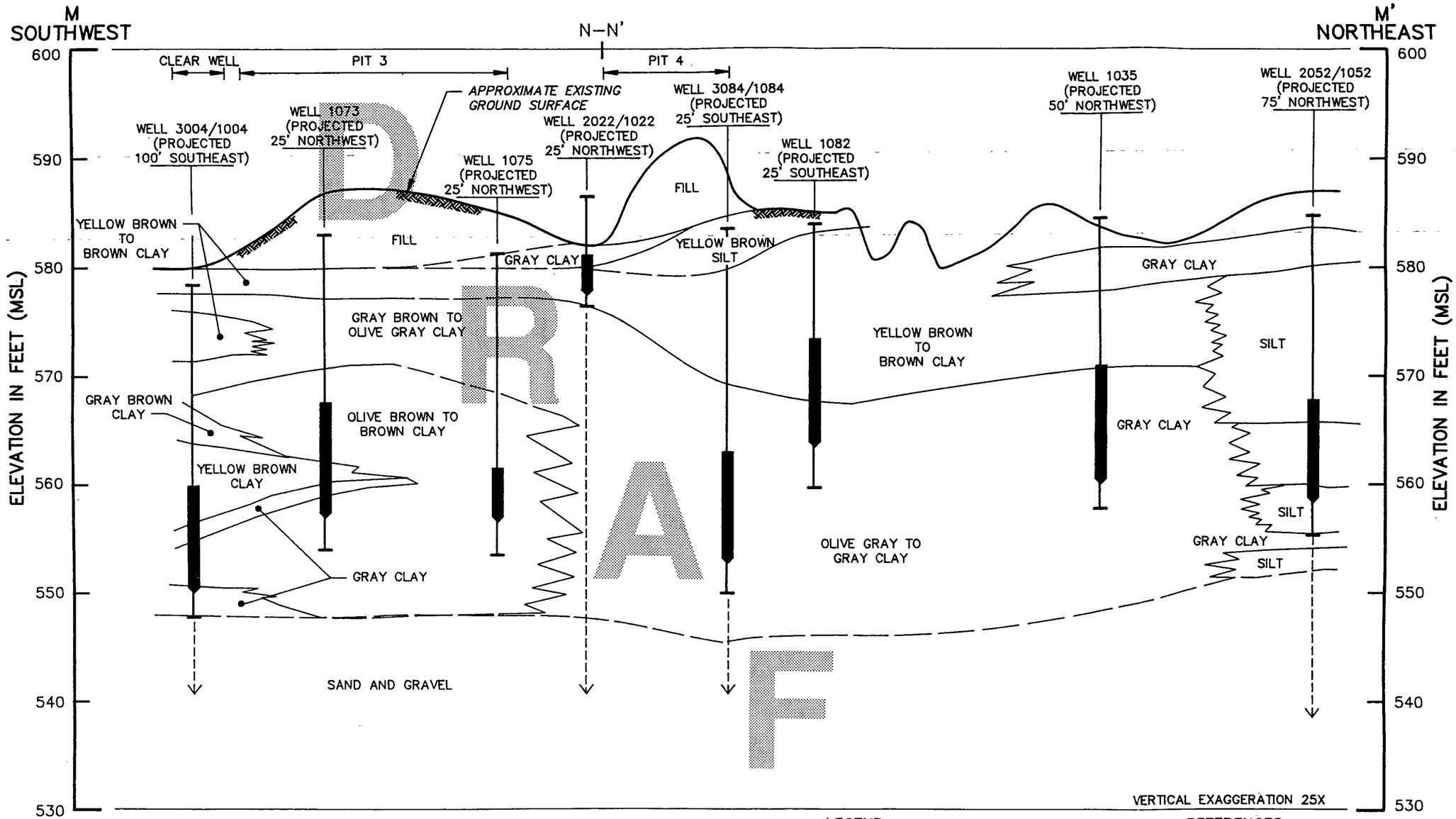


FIGURE 17-2
GLACIAL OVERBURDEN
GEOLOGIC CROSS SECTION LOCATIONS



THE BORING LOGS AND RELATED INFORMATION DEPICT SUBSURFACE CONDITIONS ONLY AT THE SPECIFIC LOCATIONS AND DATES INDICATED. SOIL CONDITIONS AND WATER LEVELS AT OTHER LOCATIONS MAY DIFFER FROM CONDITIONS OCCURRING AT THESE BORING LOCATIONS. ALSO THE PASSAGE OF TIME MAY RESULT IN A CHANGE IN THE CONDITIONS AT THESE BORING LOCATIONS.

THE DEPTH AND THICKNESS OF THE SUBSURFACE STRATA INDICATED ON THE SECTIONS WERE GENERALIZED FROM AND INTERPOLATED BETWEEN THE TEST BORINGS. INFORMATION ON ACTUAL SUBSURFACE CONDITIONS EXISTS ONLY AT THE LOCATION OF THE TEST BORINGS AND IT IS POSSIBLE THAT SUBSURFACE CONDITIONS BETWEEN THE TEST BORINGS MAY VARY FROM THOSE INDICATED.

303317-B531 (GW) (PGH)

CROSS SECTION M–M'

(LOOKING NORTHWEST)

VERTICAL SCALE

0 10 20 FEET

HORIZONTAL SCALE

0 250 500 FEET

NOTE:

FOR LOCATION OF CROSS SECTION M–M', SEE FIGURE 17–2

LEGEND:

INFERRED CONTACT

INDICATES THAT BORING EXTENDS BEYOND THE VERTICAL BOUNDS OF THE CROSS SECTION

SCREENED INTERVAL FOR 1000 SERIES WELL WHERE A DEEPER WELL HAS BEEN USED TO DETERMINE LITHOLOGY, BOTH WELL NUMBERS ARE LISTED.

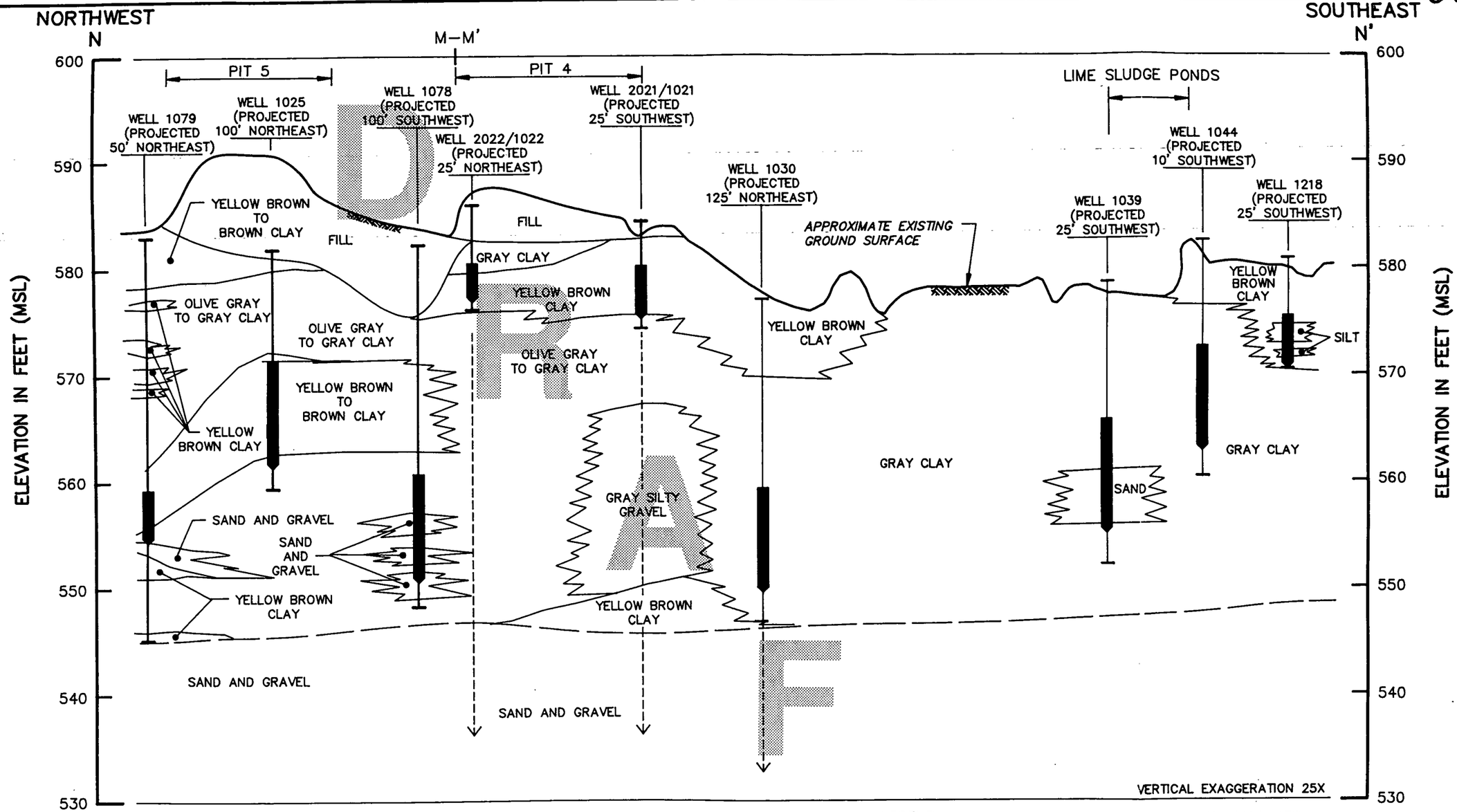
REFERENCES:

MODIFIED FROM U.S.G.S. PROFESSIONAL PAPER NO. 605–A USING AVAILABLE OHIO STATE WATER WELL RECORDS, SOWC WATER WELL DATA, DAMES & MOORE'S "FEED MATERIALS PRODUCTION CENTER GROUNDWATER STUDY, TASK C REPORT" FOR THE DOE (1985), AND IT (1986) RI/FS BORING LOGS.

DRAFT

FIGURE 17–3

WASTE PIT/SANITARY LANDFILL
GLACIAL OVERBURDEN
CROSS SECTION M–M'



THE BORING LOGS AND RELATED INFORMATION DEPICT SUBSURFACE CONDITIONS ONLY AT THE SPECIFIC LOCATIONS AND DATES INDICATED. SOIL CONDITIONS AND WATER LEVELS AT OTHER LOCATIONS MAY DIFFER FROM CONDITIONS OCCURRING AT THESE BORING LOCATIONS. ALSO THE PASSAGE OF TIME MAY RESULT IN A CHANGE IN THE CONDITIONS AT THESE BORING LOCATIONS.

THE DEPTH AND THICKNESS OF THE SUBSURFACE STRATA INDICATED ON THE SECTIONS WERE GENERALIZED FROM AND INTERPOLATED BETWEEN THE TEST BORINGS. INFORMATION ON ACTUAL SUBSURFACE CONDITIONS EXISTS ONLY AT THE LOCATION OF THE TEST BORINGS AND IT IS POSSIBLE THAT SUBSURFACE CONDITIONS BETWEEN THE TEST BORINGS MAY VARY FROM THOSE INDICATED.

CROSS SECTION N–N'

(LOOKING NORTHEAST)

VERTICAL SCALE

0 10 20 FEET

HORIZONTAL SCALE

0 250 500 FEET

NOTE:

FOR LOCATION OF CROSS SECTION N–N', SEE FIGURE 17–2

LEGEND:

INFERRED CONTACT

INDICATES THAT BORING EXTENDS BEYOND THE VERTICAL BOUNDS OF THE CROSS SECTION

SCREENED INTERVAL FOR 1000 SERIES WELL. WHERE A DEEPER WELL HAS BEEN USED TO DETERMINE LITHOLOGY, BOTH WELL NUMBERS ARE LISTED

REFERENCES:

MODIFIED FROM U.S.G.S. PROFESSIONAL PAPER NO. 605–A USING AVAILABLE OHIO STATE WATER WELL RECORDS, SOWC WATER WELL DATA, DAMES & MOORE'S "FEED MATERIALS PRODUCTION CENTER GROUNDWATER STUDY, TASK C REPORT" FOR THE DOE (1985), AND IT (1986) RI/FS BORING LOGS.

303317–B532 (GW) (PGH)

FIGURE 17–4
WASTE PITS/LIME SLUDGE PONDS
GLACIAL OVERBURDEN
CROSS SECTION N–N'

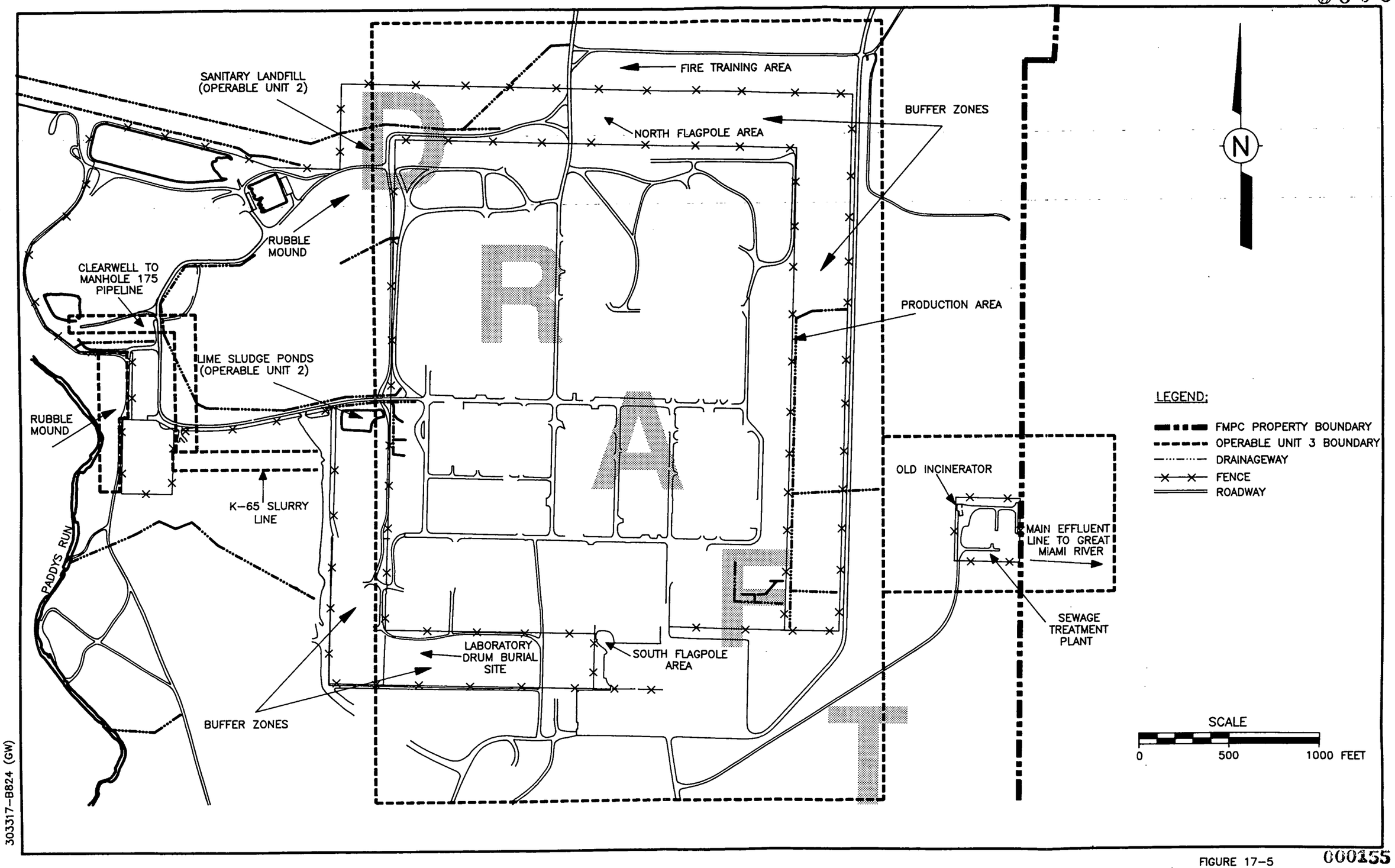
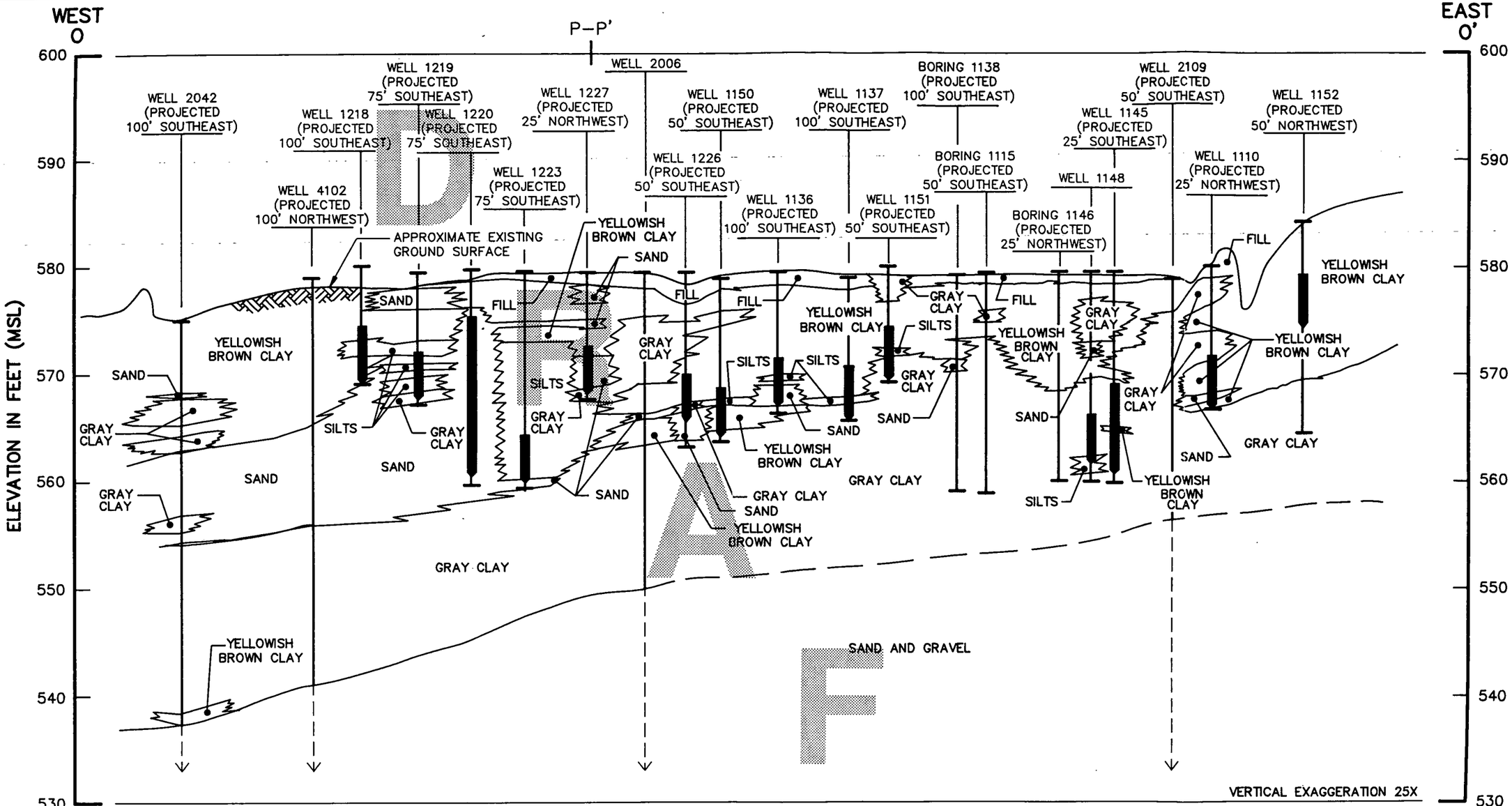


FIGURE 17-5

OPERABLE UNIT 3
SUSPECT AREA LOCATIONS



THE BORING LOGS AND RELATED INFORMATION DEPICT SUBSURFACE CONDITIONS ONLY AT THE SPECIFIC LOCATIONS AND DATES INDICATED. SOIL CONDITIONS AND WATER LEVELS AT OTHER LOCATIONS MAY DIFFER FROM CONDITIONS OCCURRING AT THESE BORING LOCATIONS. ALSO THE PASSAGE OF TIME MAY RESULT IN A CHANGE IN THE CONDITIONS AT THESE BORING LOCATIONS.

THE DEPTH AND THICKNESS OF THE SUBSURFACE STRATA INDICATED ON THE SECTIONS WERE GENERALIZED FROM AND INTERPOLATED BETWEEN THE TEST BORINGS. INFORMATION ON ACTUAL SUBSURFACE CONDITIONS EXISTS ONLY AT THE LOCATION OF THE TEST BORINGS AND IT IS POSSIBLE THAT SUBSURFACE CONDITIONS BETWEEN THE TEST BORINGS MAY VARY FROM THOSE INDICATED.

LEGEND:

INFERRED CONTACT

INDICATES THAT THE BORING EXTENDS BEYOND THE VERTICAL BOUNDS OF THE CROSS SECTION

SCREENED INTERVAL FOR 1000 SERIES WELL. WHERE A DEEPER WELL HAS BEEN USED TO DETERMINE LITHOLOGY, BOTH WELL NUMBERS ARE LISTED.

CROSS SECTION O–O'

(LOOKING NORTH)

VERTICAL SCALE

0 10 20 FEET

HORIZONTAL SCALE

0 250 500 FEET

NOTE:

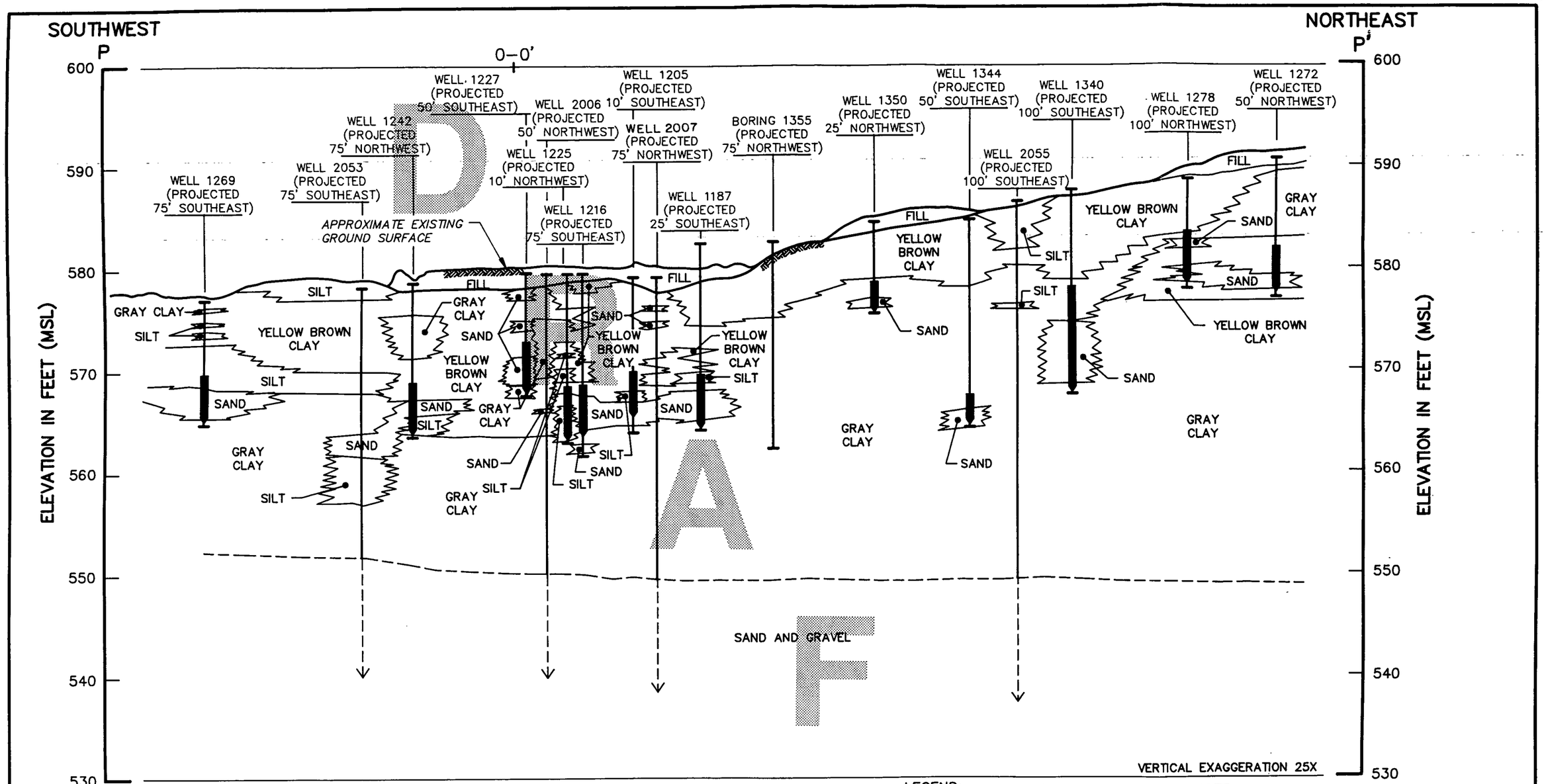
FOR LOCATION OF CROSS SECTION O–O', SEE FIGURE 17–2

REFERENCE:

MODIFIED FROM U.S.G.S. PROFESSIONAL PAPER NO. 605–A USING AVAILABLE OHIO STATE WATER WELL RECORDS, SOWC WATER WELL DATA, DAMES & MOORE'S "FEED MATERIALS PRODUCTION CENTER GROUNDWATER STUDY, TASK C REPORT" FOR THE DOE (1985), AND IT (1986) RI/FS BORING LOGS.

303317–B637 (GW) (PGH)

FIGURE 17–6
OPERABLE UNIT 3
GLACIAL OVERBURDEN
CROSS SECTION O–O'



THE BORING LOGS AND RELATED INFORMATION DEPICT SUBSURFACE CONDITIONS ONLY AT THE SPECIFIC LOCATIONS AND DATES INDICATED. SOIL CONDITIONS AND WATER LEVELS AT OTHER LOCATIONS MAY DIFFER FROM CONDITIONS OCCURRING AT THESE BORING LOCATIONS. ALSO THE PASSAGE OF TIME MAY RESULT IN A CHANGE IN THE CONDITIONS AT THESE BORING LOCATIONS.

THE DEPTH AND THICKNESS OF THE SUBSURFACE STRATA INDICATED ON THE SECTIONS WERE GENERALIZED FROM AND INTERPOLATED BETWEEN THE TEST BORINGS. INFORMATION ON ACTUAL SUBSURFACE CONDITIONS EXISTS ONLY AT THE LOCATION OF THE TEST BORINGS AND IT IS POSSIBLE THAT SUBSURFACE CONDITIONS BETWEEN THE TEST BORINGS MAY VARY FROM THOSE INDICATED.

CROSS SECTION P–P'

(LOOKING WEST)

VERTICAL SCALE

0 10 20 FEET

HORIZONTAL SCALE

0 250 500 FEET

LEGEND:

INFERRED CONTACT

INDICATES THAT THE BORING EXTENDS BEYOND THE VERTICAL BOUNDS OF THE CROSS SECTION

SCREENED INTERVAL FOR 1000 SERIES WELL. WHERE A DEEPER WELL HAS BEEN USED TO DETERMINE LITHOLOGY, BOTH WELL NUMBERS ARE LISTED

NOTE:

FOR LOCATION OF CROSS SECTION P–P', SEE FIGURE 17–2

REFERENCES:

MODIFIED FROM U.S.G.S. PROFESSIONAL PAPER NO. 605–A USING AVAILABLE OHIO STATE WATER WELL RECORDS, SOWC WATER WELL DATA, DAMES & MOORE'S "FEED MATERIALS PRODUCTION CENTER GROUNDWATER STUDY, TASK C REPORT" FOR THE DOE (1985), AND IT (1986) RI/FS BORING LOGS.

303317–B641 (GW) (PGH)

FIGURE 17–7

OPERABLE UNIT 3
GLACIAL OVERBURDEN
CROSS SECTION P–P'

Weathered zones typically have an enhanced bulk hydraulic conductivity with respect to unweathered zones due to the presence of joints and fractures.

In OU1, the distribution of weathered versus unweathered till suggests that the primary groundwater movement within the glacial overburden is within the upper half of the unit, except along the western edge, where the groundwater is relatively free to move throughout the section. The distribution of infiltrating surface water is likely to be as follows:

- Evapotranspiration over the entire operable unit
- Lateral westward movement and subsequent discharge along the east bank of Paddys Run
- Vertical infiltration through the glacial overburden and direct recharge of the Great Miami Aquifer, especially in the western portion of the area

A minor component of glacial overburden is silt of either glaciolacustrine or aeolian origin. At the FMPC, silts were deposited near the top of the stratigraphic section but have been subsequently eroded. They are not extensive across OU1, being found only at the surface in the northeastern portion of the area in Wells 1081, 1082, 2027, 3019, and 3084, and in the west in Well 1028 (Plate Q-1 and Figure 17-3). These deposits are of uncertain origin and may represent either glaciolacustrine silt or postglacial loess. These silts are jointed and weathered throughout their depth and, thus, have a substantial secondary permeability.

Within the glacial overburden, beds and stringers of glaciofluvial sands and gravels are pervasive. Some of the glaciofluvial beds are locally continuous and can be correlated between two or more nearby wells, as seen in Plate Q-1. A 7-foot-thick bed of sand and gravel in Well 2027 has been correlated to a 6-foot-thick bed of sand and silt in Well 1038. Similarly, interbeds in the upper parts of Wells 3019 and 2027 appear to be continuous. Wells 3011, 1078, 1079, 3019, 1076, 2027, and 1031 show one or more interbeds of silt, sand, and gravel at the base of the glacial overburden, but overlying the Great Miami Aquifer. This zone may represent a water-lain sequence deposited in front of the advancing glacier and is likely to be significantly more permeable than the overlying clayey glacial till.

Some of the glaciofluvial beds are recharged by infiltration through overlying weathered till. Unless these beds are laterally continuous, however, lateral groundwater movement would be constrained. Many of these perched zones are being monitored, and their lateral extent has been inferred from water-level recovery times during well-purging operations. Most OU1 wells in the glacial overburden recovered sufficiently within 24 hours to be sampled, but typically did not

recover fast enough to be sampled the same day as purging. The exceptions were Wells 1011, 1019, 1021, 1079, and 1083. The permeable zones monitored by these wells may be more extensive than those in the majority of OU1 wells. Wells 1011 and 1079 probably have interconnected water-bearing zones (Plate Q-1). Wells 1019, 1021, and 1083 are all in the eastern part of the operable unit south of Pits 4 and 6. The relatively rapid recharge of these wells, combined with significant total uranium concentrations (see Section 17.1.4), suggests that these wells may also be interconnected and receiving water from Pit 6.

Because most of the 1000 Series monitoring wells are completed in zones composed of glaciofluvial materials, slug tests in these wells reflect the high end of the range of hydraulic conductivities within the glacial overburden. Typical values of measured hydraulic conductivities were on the order of 1 ft/day, or slightly more than 10^{-4} cm/sec. Two exceptions are Wells 1025 and 1079. Well 1025 is completed in a zone of weathered clay with a trace of gravel. Well 1079 is also completed in weathered clay, with only the lower portion of the screened zone intersecting a sand and gravel bed. These wells, adjacent to Pit 5 (Figure 17-4), yielded hydraulic conductivity values of 7.1×10^{-3} ft/day (2.5×10^{-6} cm/s) and 5.1×10^{-2} ft/day (1.8×10^{-5} cm/s), respectively.

Groundwater within the glacial overburden is typically held within the weathered and/or fractured clays and silts and within the more permeable sand and gravel lenses and beds. Because these units do not yield significant amounts of water, they are considered to be discontinuous perched zones with very limited potential for lateral groundwater flow. Groundwater elevations within OU1 tend to be highest in the vicinity of Pits 4 and 6. There is a relatively low gradient across most of the area, although water levels drop off sharply toward Paddys Run in that immediate area. This follows the general topography of the area and implies at least limited flow through the glacial overburden at the western margin.

17.2 OPERABLE UNIT 2

17.2.1 Operable Unit Features

Operable Unit 2 (OU2) consists of the solid waste units at the FMPC. These include the Sanitary Landfill, the North and South Lime Sludge ponds, and the Inactive and Active Fly Ash areas/Southfield. A brief description and history of each follows.

Sanitary Landfill

The FMPC Sanitary Landfill, which was in operation from about 1954 to 1986, was used for the disposal of cafeteria wastes, rubbish, and other wastes from nonprocess areas. Materials that were

reportedly accepted at the landfill include nonburnable, nonradioactive sanitary wastes generated on site; nonradioactive, construction-related rubble (Weston 1987b); and double-bagged and bulk quantities of nonradioactive asbestos (DOE 1988). Radionuclide-contaminated materials such as construction rubble and soil once used to cover exposed wastes may also have been disposed of in the landfill (DOE 1988).

The Sanitary Landfill is located on a three-acre tract in the northeast corner of the Waste Storage Area. The facility is organized into 17 individual cells; five are full and out of service, and the remaining 12 were to be constructed after issuance of an OEPA permit. The five filled cells occupy approximately one acre of land and are covered by a two-foot-thick layer of compacted soil. Each of the five cells is estimated to be approximately 225 feet long by 25 feet wide by 10 feet deep (Weston 1987b). Waste material was also disposed of in an area adjacent to the five cells. Therefore, the total volume of waste believed to be contained in the Sanitary Landfill is estimated to be 16,000 to 18,000 cubic yards.

Surface drainage in the Sanitary Landfill area has been controlled by topography recontouring. The general drainage pattern within the landfill boundaries is to the north and west. The surface drainage flows to an apparently altered intermittent stream channel passing westward along the northern portion of the landfill. This channel eventually discharges into Paddys Run near the far western boundary of the FMPC.

Lime Sludge Ponds

The two Lime Sludge ponds are located near the western boundary of the Production Area, approximately midway between the north and south boundary of the Waste Storage Area. Spent lime alum sludge from the FMPC water treatment plant operations and boiler plant blowdown were disposed of in the Lime Sludge ponds (Weston 1987b). Typical constituents of the spent lime include aluminum, calcium, iron, magnesium, manganese, and potassium (Weston 1987b).

The North Lime Sludge Pond is an active, unlined pond approximately 200 feet long by 100 feet wide by 6 to 8 feet deep. Total volume is about 5,000 cubic yards, and the pond is approximately 90 percent full and partially covered with water (Weston 1987b). The water volume is conservatively estimated to be about 600,000 gallons.

The South Lime Sludge Pond is also an unlined pond that, until recently, had a dry surface with no standing water. The pond was reactivated recently by WMCO and again receives spent lime sludge. It is located directly south of the North Lime Sludge Pond. The South Pond is

approximately 200 feet long by 100 feet wide. The depth of the pond, as indicated by borehole log information (Weston 1988), is approximately 11.5 feet, with an estimated volume of 11,500 cubic yards. The South Pond remains defined as a topographical depression because of the elevated dikes.

Generally, the topography in the vicinity of the ponds slopes very gently to the west as a result of recontouring during construction of the ponds and the natural slope to the land. The ponds are bermed in a manner consistent with unlined surface impoundment construction practices of the preregulatory era. Common to both ponds is a central east-west oriented dike. Water entering the ponds includes moisture contained in the waste material delivered to the ponds and precipitation that falls within the diked perimeter of each pond.

Fly Ash Area/Southfield

This area comprises the Inactive Fly Ash Disposal Area, the Active Fly Ash pile, and Southfield. The Inactive Fly Ash Disposal Area is located approximately 2,000 feet southwest of the Production Area and adjacent to Paddys Run. The Active Fly Ash pile is located approximately 2,000 feet south of the Production Area and adjacent to the Storm Sewer Outfall Ditch. Southfield covers approximately 11 acres between the Active Fly Ash pile and the Inactive Fly Ash Disposal Area.

Bituminous fly ash and bottom ash from the coal-fired boiler plant were disposed of in the Inactive Fly Ash Disposal Area, and are currently being disposed of in the Active Fly Ash pile. A spectrochemical analysis performed on ash composite indicated that the major constituents in the ash were aluminum, calcium, iron, magnesium, silicon, and titanium (Bogar 1987). An analysis for uranium indicated concentrations ranging from < 5 ppm to 34 ppm in bottom ash, mechanical precipitator ash, and electrostatic precipitator ash (Bogar 1987).

The Inactive Fly Ash Disposal Area is estimated to contain 50,000 cubic yards of fly ash. Elevated levels of uranium were found during sampling activity performed in the CIS, so it is suspected that waste oils containing uranium were sprayed on the pile as a dust suppressant. Approximately 1,000 kg of uranium may be present in the Inactive Fly Ash Disposal Area. Building rubble such as concrete, gravel, asphalt, masonry, and steel rebar also were discarded at this site.

The Active Fly Ash pile contains approximately 38,000 cubic yards of fly ash. It has been reported that waste oils were periodically sprayed onto the Active Fly Ash pile as a means of dust control (DOE 1988a). Elevated levels of radiological contaminants have been found in surface samples.

Southfield was reportedly used as a burial site for construction rubble that contained low levels of radioactivity, including debris from the razing of the old administration building. Other wastes may have been deposited in this area as well, although supporting records are not available.

The primary surface water runoff patterns in the area are to the southwest and southeast. The north-south oriented Range Road serves as the drainage divide between the two patterns. Fill and grade operations in the central portion of Southfield have resulted in a gradual slope to the southwest, culminating in significantly steeper grades that indicate the extent of fill and grade activity. The northern portion of the Inactive Fly Ash Disposal Area was constructed over an old drainage leading to Paddys Run. The local runoff from the working face of the Active Fly Ash pile flows into a small drainage channel that empties into the Storm Sewer Outfall Ditch.

All three disposal areas are bounded by relatively steep slopes on the downgradient sides, below which is either Paddys Run, the Storm Sewer Outfall Ditch, or the flood plain of Paddys Run.

17.2.2 Glacial Overburden

Fence diagrams detailing the characteristics of the glacial overburden have been prepared for the Sanitary Landfill (Plate Q-2), the Lime Sludge ponds (Plate Q-3), and the Fly Ash Area/Southfield (Plate Q-4). Additional information is shown on Cross Section M-M' (Figure 17-3), which intersects the Sanitary Landfill, and Cross Section N-N' (Figure 17-4), which crosses the Lime Sludge ponds. The stratigraphic descriptions are based on lithology and coloration as observed from borehole samples. The units generally correlate with depositional units described by other researchers (Leggett 1976; Cravens 1987; Barari 1988; Brockman 1988; Hendry 1988).

The glacial overburden unconformably overlies the sands and gravels of the Great Miami Aquifer. The gray and brown clays and silts represent the Wisconsin glacial till. The variations in color indicated in Plates Q-2, Q-3, and Q-4 are due to differential weathering of the unit rather than depositional differences. Based on the descriptions found in the U.S. Army Corps of Engineers' boring program conducted during plant construction, it appears that silts represent either lacustrine deposits associated with a glacial lake or postglacial retreat loess deposits. Sand and gravel stringers and beds found within the glacial overburden are undifferentiated glaciofluvial outwash deposits.

The depth of weathering is variable across each of the areas within OU2. At the Sanitary Landfill (Plate Q-2), the weathered zone is 13 feet thick in the immediate vicinity of the landfill and thickens to the southwest to 24 feet at Well 1081. The weathered zone in the Lime Sludge Pond

area (Plate Q-3) averages 11.5 feet in thickness, with no systematic trends. In the immediate Fly Ash Area/Southfield (Plate Q-4), the glacial overburden has been largely removed through erosion by the Storm Sewer Outfall Ditch and Paddys Run. The weathered zone is typically about 10 feet thick outside of the erosion zone. In Well 2065, the weathered material extends to the top of the regional aquifer, while in Wells 1045 and 2049 the glacial overburden is composed entirely of weathered clay. In OU2, the distribution of weathered versus unweathered till suggests that the primary groundwater movement within the glacial overburden is within the upper half of the unit, except in the areas immediately adjacent to major surface drainages, where the groundwater is able to drain from the entire section.

In the area of the Sanitary Landfill (Plate Q-2), silts are seen to the southwest in Wells 1081, 1082, and 1083. The Lime Sludge ponds do not have any silt at the surface, but a bed appears intermittently, as seen in Wells 1210, 1229, and 2008. Compared to the rest of the FMPC, silt beds are pervasive in the Fly Ash Area/Southfield. They are thickest in Well 1048, thinning and laterally interfingering with clays toward Wells 1046 and 2065. These are likely to represent deposits from the formation of periodic glacial lakes or morainal debris.

Some of the glaciofluvial beds are locally continuous and can be correlated between two or more nearby wells, as seen in the Sanitary Landfill area (Plate Q-2, Wells 1035 and 1038), the Lime Sludge Pond area (Plate Q-3, Wells 1039, 1134, and 2042), and the Fly Ash Area/Southfield (Plate Q-3, Wells 1048, 1046, and 2065). These sand and gravel beds are potential pathways for contaminant migration within the otherwise low permeability clays and silts.

Well recovery rates for OU2 wells completed in the glacial overburden were observed to be generally good to excellent, allowing same-day sampling in about 60 percent of the wells. The remaining wells were either always dry, remained dry after pumping, or had poor recovery, necessitating next-day sampling.

In the Sanitary Landfill area, Wells 1035 and 1038 both demonstrated good recovery after purging, indicating that the sand layer in the two wells (Plate Q-2) had a significant lateral extent. Well 1037 did not recover, and purging notes indicated that grout was found in this well; thus, the filter pack may be clogged. Wells in the Lime Sludge Pond area all had good recovery rates. These are Wells 1039, 1041, and 1042, which are completed in a laterally extensive sand layer (Plate Q-3).

The Fly Ash Area/Southfield had the highest percentage of wells with poor or no recovery. The wells demonstrating good recovery were Wells 1016, 1046, and 1048. Well 1016 is actually completed in the Great Miami Aquifer. Wells 1046 and 1048 monitor a silt and sand sequence which can be correlated between the two wells (Plate Q-4) and is laterally extensive.

Slug test data available for Wells 1008 and 1041 in the Lime Sludge ponds area indicate that the hydraulic conductivities of the perched zones are about 0.34 ft/day (1.2×10^{-4} cm/s). The zones monitored by Wells 1008 and 1041 may be interconnected, but insufficient data exist to make a conclusive determination. The hydraulic conductivities in Wells 1046 and 1045 in the Fly Ash Area/Southfield ranged from 0.45 ft/day (1.6×10^{-4} cm/s) to 0.19 ft/day (6.8×10^{-5} cm/s).

17.3 OPERABLE UNIT 3

17.3.1 Operable Unit Features

Operable Unit 3 (OU3) generally encompasses the soils and perched groundwater underlying the Production Area. This operable unit also includes 10 suspect areas.

Production Area

The production of uranium metal products at the FMPC involved a series of chemical and metallurgical conversions that occurred in nine specialized plants within the Production Area to achieve a manageable technical framework for the OU3 RI/FS. The Production Area was separated into four distinct quadrants, as shown in Figures 7-1 and 7-2, which are:

- Northwest Quadrant

  Plant 1 and Drum Storage Pad. Principal activities included materials sampling, analysis, and storage.

- Northeast Quadrant

  Plant 9, Decontamination and Decommissioning Facility, Maintenance Building, and Metal Scrap Pile. Principal activities included special uranium products casting, thorium processing, decontamination and decommissioning of equipment, equipment maintenance, and metal scrap storage.

- Southwest Quadrant

  Plants 2/3, Plant 8, and General Sump. Processing activities included uranium digestion in the refinery, uranium recovery and oxidation, and waste/wastewater treatment.

Pilot Plant and Laboratory. Activities included uranium reduction, pilot-scale operations, sample testing, and analysis.

- Southeast Quadrant

Plants 4, 5, 6, and 7. Processing activities included metals production, fabrication, and machining.

Main electrical substation of the plant and the garage.

Suspect Areas

Areas classified as suspect areas either are suspected of having contamination due to past activities, or were identified as being contaminated through field testing within the FMPC boundary. These areas are: three rubble mounds, the K-65 slurry line, the main effluent line, the pipeline connecting the Clearwell to Manhole 175, the incinerator area, one area in the buffer zone, the fire training areas, and a small area around the flagpole (Figure 17-5).

The three rubble mounds describe areas of discarded construction materials and soils. The Clearwell to Manhole 175 pipeline is a pressurized process line which now carries surface water runoff that enters the Clearwell. The buffer zone involves an area where unexpected levels of total uranium were discovered in surface soils.

17.3.2 Glacial Overburden

Fence diagrams showing correlations and relationships between units within the glacial overburden have been developed for each of the four quadrants of the Production Area. The Northwest Quadrant is shown on Plate Q-5, the Northeast Quadrant on Plate Q-6. the Southwest Quadrant on Plate Q-7, and the Southeast Quadrant on Plate Q-8. In addition, detailed depositional characteristics are shown on Glacial Overburden Cross Sections O-O' (Figure 17-6) and P-P' (Figure 17-7), which transect the Production Area, at locations shown in Figure 17-2. The stratigraphic descriptions are based on lithology and coloration as observed in borehole samples. The units generally correlate with depositional units described by other researchers (Leggett 1976; Cravens 1987; Barari 1988; Brockman 1988; Hendry 1988).

The glacial overburden unconformably overlies the sands and gravels of the Great Miami Aquifer and ranges from about 23 to 40 feet thick beneath the Production Area, as seen on the cross sections (Figures 17-6 and 17-7). The gray and brown clays and silts represent the Wisconsin glacial till. The variations in color indicated in Figures 17-6 and 17-7 and Plates Q-5 through Q-8 are due to differential weathering of the unit rather than depositional differences. Based on the descriptions found in the U.S. Army Corps of Engineers' boring program conducted during plant

construction, it appears that near-surface brown silts represent either lacustrine deposits associated with glacial lake or postglacial retreat loess deposits. Sand and gravel stringers and beds found within the glacial overburden are undifferentiated glaciofluvial outwash deposits. In OU3, the depth of weathering in the glacial overburden is variable, typically ranging from 7 to 14 feet. There is no systematic variation, and the weathered thickness may be relatively constant over small areas. Weathered till is found only below unweathered till in the northwest-central portion of the Production Area (Plate Q-5: Borings, 1039, 1130, 1348, 1354, 1357, 1359, 1360, 1361, and 1363).

The most consistent region of the glacial overburden with respect to depth of weathering is the northeast quadrant of the Production Area (Plate Q-6). In the north-central portion of this quadrant, the glacial overburden is composed primarily of clay with relatively few sand or silt beds. Borings in this area showed the depth of weathering to be between 9 and 12 feet. In other portions of the FMPC, the weathered zone may be as thin as 2 or 3 feet or as thick as 20 feet.

A minor component of the glacial overburden is silt of either glaciolacustrine or aeolian origin. Surficial silts were only found in Borings 1341, 1338, and 2055, in the northwest quadrant (Plate Q-5). These silts are joined and weathered throughout their depth and have a substantial secondary permeability.

Silts within the glacial overburden are more common in the western half of OU3 (Plates Q-5 and Q-7) than in the eastern half (Plates Q-6 and Q-8). In the east, only two significant silt deposits were found in Borings 1112 and 1308, and are not laterally extensive. In the western half, the silts become thicker and more laterally extensive to the south. In this area, the silts are commonly associated with sand beds. The silt and sand layers may represent small glacial lakes dammed by recessional moraines which were overrun by later glacial advances.

Areas with laterally extensive sand beds are as follows:

- In the north-central part of the Production Area (Plates Q-5 and Q-6) is a sand bed from 9 to 13 feet thick in Borings 1287, 1336, and 1339.
- In the southwest portion of the Production Area (Plates Q-5 and Q-7), 2- to 3-foot-thick and 4- to 5-foot-thick sand beds are pervasive both at the surface and within the upper 15 feet of the glacial overburden
- In the eastern part of the Production Area (Plates Q-6 and Q-8), thin sand beds of limited extent are scattered throughout the section; at best, these can only be correlated between 2 to 4 wells

Six wells in or near the Production Area have well yield records (Wells 1010, 1020, 1053, 1054, 1055, and 1064). Of these, Wells 1054, 1055, and 1064 had relatively poor recovery. These three wells are all completed in clays or silts; there are no significant sand or gravel layers noted in those borings. Wells 1020 and 1053 are located in or near the southwestern quadrant (Plate Q-7). This area has many laterally extensive sand layers which could explain the rapid recharge observed in those wells.

Slug test data are available for two 1000 Series monitoring wells located near the Production Area. Well 1035 (immediately to the northwest of the Production Area) and Well 1041 (immediately to the west of the Production Area) are completed in fine-grained units within the glacial overburden. Well 1035 is screened in a zone of unweathered clay with some silt, but the sand pack extends into interbedded poorly sorted gravels. Well 1041 is completed in unweathered clay with some silt. These wells yielded hydraulic conductivity values of 0.071 ft/day (2.5×10^{-5} cm/s) and 0.31 ft/day (1.1×10^{-4} cm/s), respectively.

Groundwater elevations within the Production Area tend to be highest in the north and decrease to the south. There is a very low gradient across the area, approximately 0.8 percent. Discharge points for this perched groundwater are unknown, and depend largely on the overall continuation of sand and gravel beds within the glacial overburden.

17.4 OPERABLE UNIT 4

17.4.1 Operable Unit Features

Operable Unit 4 (OU4) consists of four large concrete silos. Built in 1951 and 1952, these waste storage silos are located south of the Waste Storage Area on the west side of the FMPC (Figure 7-1). The silos are 80 feet in diameter, constructed with floors of 4-inch-thick concrete, over an 8-inch layer of gravel containing an underdrain system of 2-inch-diameter slotted pipe which drains to a collection sump. Below the gravel is a 2-inch-thick layer of asphaltic concrete underlain by 18 inches of compacted clay. The walls are 26 feet high and are constructed of 8-inch-thick concrete with a 0.75-inch-thick gunite coating on the exterior. The domed roofs taper from 8 inches thick at the silo walls to 4 inches thick at the apex.

The silos were constructed to provide storage for the residues resulting from processing of Belgium Congo pitchblende ore to extract its uranium content. The term "K-65" refers specifically to various radium-bearing raffinate wastes generated in the processing of pitchblende and other

radium-bearing ores which were placed in Silos 1 and 2. Other wastes, however, are also contained in Silo 2.

Silo 3 was constructed in mid-1952 and was designed to receive dry materials only. Slurries from refinery operations were dewatered in an evaporator and spray calcined to produce a dry waste form for removal to Silo 3. The waste was blown under pressure into the silo. Silo 4 was also constructed in mid-1952; however, this silo was never used.

In 1964, gunite spalling on the exterior surfaces of the K-65 silos was repaired and a waterproofing membrane was applied. Subsequently, an earthen embankment was built to the top of the walls of Silos 1 and 2 to provide relief from tensile stress that had developed within the walls. This embankment also provides weather protection, a reduction of radon emissions, and increased shielding from penetrating radiation. The embankments were originally constructed on a slope of 1.5:1; however, the slope was subsequently modified to 3:1 in 1983 to reduce soil erosion (Grumski 1987a; Boback et al. 1987). Vents in the silos were sealed in 1979 (Carmargo 1986). In late 1985, the FMPC had a temporary dome, 9.1 meters (30 feet) in diameter, designed and installed to span across the deteriorated central portion of the concrete domes.

In 1987, an Interim Stabilization Project was implemented at the K-65 silos to comply with the CERCLA section of the FFCA (Grumski 1987a). Compliance required that DOE take immediate measures to control radioactive emissions from the FMPC. Work performed as part of the Interim Stabilization Project included:

- Implementation and periodic operation of a closed radon gas treatment system which removes radon from the air and recirculates the air back into the silos.
- Application of a layer of rigid polyurethane foam insulation to the exterior of the dome surfaces of Silos 1 and 2. The foam coating was designed to reduce weathering, temperature changes within the silos, and radon gas emissions.

Topographic relief of the OU4 area is moderately high, with variations of about 35 feet across the area. The original topography has been highly modified by silo and berm construction, the latter constituting approximately 30 feet of relief. Surface water drains from much of the silo area to Paddys Run.

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17.4.2 Glacial Overburden

Depositional characteristics of the glacial overburden specific to OU4 are shown on the Operable Unit 4 Glacial Overburden Fence Diagram (Plate Q-9). The stratigraphic descriptions are based on lithology and coloration as seen in borehole samples. The units generally correlate with depositional units described by other researchers (Leggett 1976; Cravens 1987; Barari 1988; Brockman 1988; Hendry 1988).

The glacial overburden unconformably overlies the sands and gravels of the Great Miami Aquifer and ranges from 20 to 35 feet thick in OU4. The gray and brown clays and silts represent the Wisconsin glacial till. The variations in color indicated in Plate Q-9 are due to differential weathering of the unit rather than depositional differences. Based on the descriptions found in the U.S. Army Corps of Engineers' boring program conducted during plant construction, it appears that near-surface brown silts represent either lacustrine deposits associated with glacial lake or postglacial retreat loess deposits. Sand and gravel stringers and beds found within the glacial overburden are undifferentiated glaciofluvial outwash deposits.

The depth of weathering in the glacial overburden varies across the silo area. The typical range is from 7 to 15 feet deep, with an exceptionally thin, 2-foot-thick section in Well 2034 at the south end and an exceptionally thick, 29-foot-thick section in Well 1031 to the northwest of OU4. The latter boring has interbedded unweathered clays with a 9-foot-thick unweathered silt and clay layer directly overlying the Great Miami Aquifer. In OU4, the distribution of weathered versus unweathered till suggests that the primary groundwater movement within the glacial overburden is within the upper half of the unit, except to the north of the silos, where the groundwater is relatively free to move throughout the section.

Silts are not extensive across OU4, and were found at the surface only in the western portion of the area in Wells 1029 and 1032 (Plate Q-9). These deposits are of uncertain origin and may represent either glaciolacustrine silt or postglacial loess. The silt in the upper few feet of Well 1032 is probably artificial fill, since this well is located within a suspected rubble mound. Silts within OU4 were observed in Wells 1031, 1032, 1033, and 2008. Only in the latter two wells is interconnection possible. These silts are probably proglacial lake deposits which were subsequently overrun during a readvance of the glacier.

Within the glacial overburden, beds and stringers of glaciofluvial sands and gravels are found in the southern and western portions of the area (Plate Q-9). A 3- to 4-foot-thick sand bed under the silos is relatively continuous and can be correlated between Wells 2008, 1032, 1033, and 2034.

Their lateral extent could be inferred from water-level recovery times during well-purging operations. The glaciofluvial sand beds underlying OU4 are recharged by infiltration through overlying weathered till. Whether the 3- to 4-foot-thick sand bed under the silos is laterally continuous to a discharge zone on Paddys Run or its tributary remains unknown. This perched zone is monitored by Wells 1032, 1033, and 1034.

Well 1032 exhibited poor yield, but that well only has a 2-foot screen. Wells 1033 and 1034 have shown moderately good recovery, indicating that the sand layer in which the wells are screened has some lateral continuity. The other wells for which purge records are available are Wells 1008, 1029, and 1072. The first two had slow recovery, and the last was dry; all three are completed in unweathered clay.

There are slug test data available for Well 1034, located immediately south of Silo 1. Well 1034 is completed in a layer consisting of clay, fine sand, and some silt and gravel. It yielded a hydraulic conductivity value of 0.071 ft/day (2.5×10^{-5} cm/s).

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PART IV. GROUNDWATER MODELING

Part IV of the FMPC groundwater report provides the formulation, testing, and results of regional site modeling, as well as the procedure for the long-term modeling related to each operable unit. Figure IV-1 presents a schematic of the overall structure of the FMPC groundwater studies documented in this report. The data obtained and interpreted via the literature search, field investigation program, and data analysis are used as input data into the groundwater models to provide spatial and temporal variation of regional and site hydrogeology and groundwater quality. Therefore, Part IV provides a link between most direct observation and projections.

Part IV has been divided into five chapters. The purpose and scope of modeling are discussed in Chapter 18.0. In Chapter 19.0, the design of the modeling study is presented. The groundwater flow and solute transport models are discussed in Chapters 20.0 and 21.0, respectively. The procedure for modeling the long-term migration potential pertinent to each operable unit is presented in Chapter 22.0.

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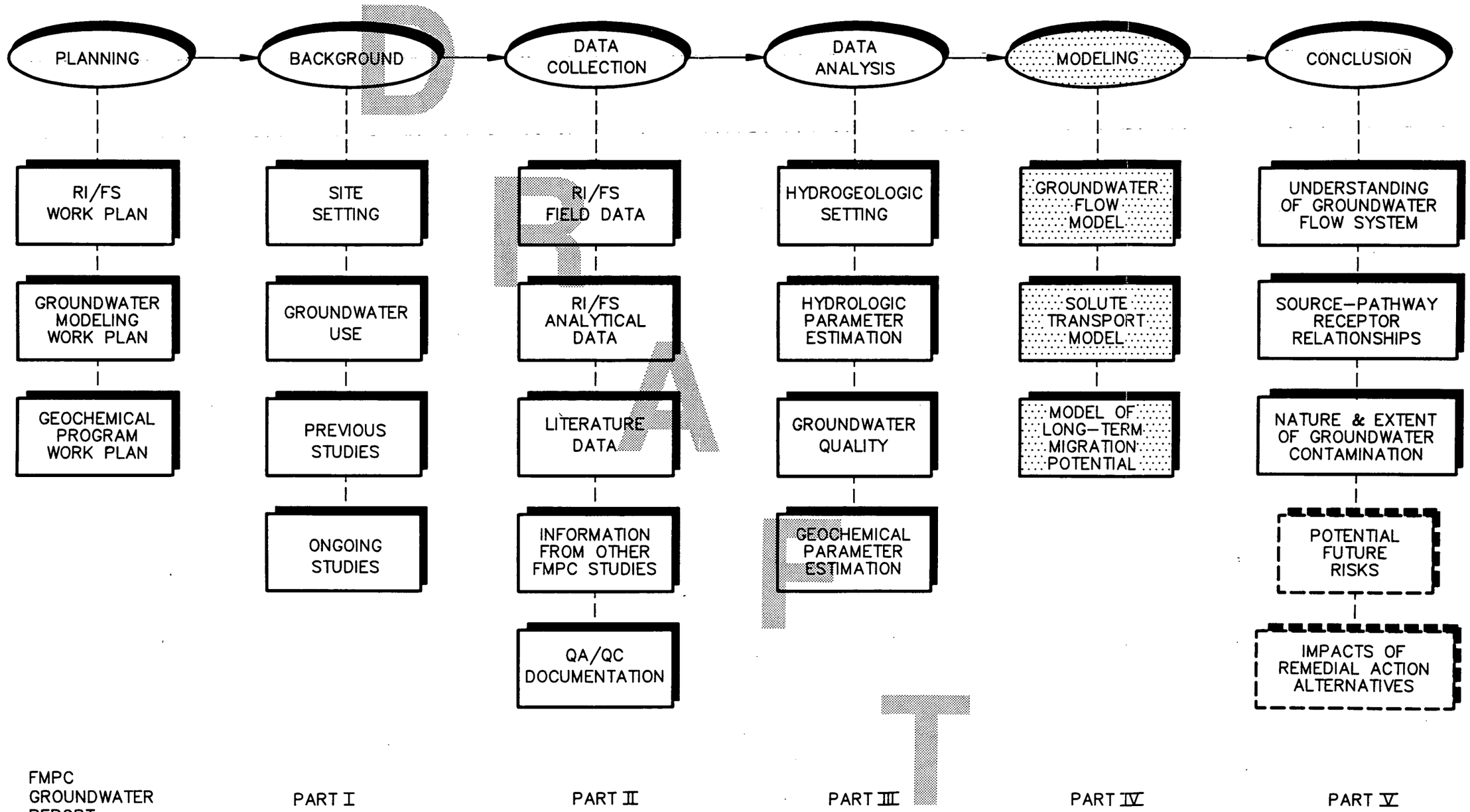

FIGURE IV-1

OVERVIEW OF GROUNDWATER MODELING COMPONENT

18.0 PURPOSE AND SCOPE

Groundwater modeling studies were initiated to support interim activities and different aspects of the RI/FS and associated risk assessment. The work plan covering these studies was developed by IT Corporation (1988a). This report describes the construction, calibration, and some initial applications of the finite-difference groundwater flow and solute transport models developed for use in the FMPC RI/FS. Model construction is the development of computer input files that constitute the model. Model calibration is the adjustment of parameters in the input file until the model is an adequate representation of the real system.

Two successive solute transport model calibrations are reported herein. The first calibration was developed before the results of the geochemical modeling were available. It was a provisional calibration that was adequate for use in supporting a removal action for mitigating contamination south of the FMPC pending final action under the RI/FS. The second calibration was completed when the results of the geochemical modeling were available. The second calibration produced a more refined model that benefited from the geochemical results and from additional uranium concentration data that had become available. Its degree of calibration is higher, and it will be used to support ongoing RI/FS activities. The solute transport model calibration was completed for total dissolved uranium only, since uranium is the principal contaminant of concern. The reasons for this designation are presented in Chapter 15.0.

The scope of the report is limited to a description of those modeling activities, data, and results necessary to establish the validity and applicability of the model to FMPC conditions. It excludes such technical details as descriptions of all minor data preprocessing and postprocessing computer programs and procedures, descriptions of numerous individual model calibration runs, and other routine modeling activities and technicalities. These details are recorded in project files and computer files stored on magnetic tapes.

18.1 CURRENT SITE CHARACTERIZATION

The hydrologic regime impacted by the FMPC activities encompasses a large area with complex geologic and hydrologic conditions. The development of the model was necessary to supplement direct field observations so that the combined informational base would be sufficient to support the understanding of the current situation in the RI, the projection of future conditions in the public health and environmental risk assessment, and the evaluation of the remedial action alternatives in the FS. The specific support provided to these activities is described in Sections 18.2 through 18.4.

The field observations, including groundwater monitoring, establish whether contamination is present at a specific location and to what level. Groundwater modeling provides a systematic means to interpolate between and extrapolate beyond the field observations of the contaminant concentration throughout the area of interest. In particular, the information from the modeling is used in conjunction with available field observations to:

- Estimate the location and value of the maximum uranium concentration
- Estimate the mass of the contaminant present in the aquifer
- Explain the occurrence of the field observations
- Indicate whether or not any field observations should be considered as anomalies
- Determine the need for additional data

The following hydrologic, geologic, and geochemical issues are addressed by the models described in this report:

- Groundwater flow rates and directions relative to the FMPC Waste Storage Area, waste discharge areas, and areas of postulated releases
- Recharge to the regional aquifer, including the interrelationship between the aquifer and the principal surface water courses
- Relationship between water quality data and the identified potential contaminant sources
- The relative importance of the identified sources in terms of uranium loading rates to the aquifer
- The effects of geochemical variability in the aquifer system as it relates to contaminant mobility

18.2 SUPPORT TO REMEDIAL INVESTIGATION

The purpose of the RI is to determine the nature and extent of any release, or threat thereof, of hazardous or radioactive substances, pollutants, or contaminants, and to gather all necessary data to support the FS. The RI is intended to satisfy the following specific objectives that are supported by the groundwater modeling:

- Identify and characterize any sources of radiological and chemical contamination
- Determine the nature and extent of radiological and chemical contaminants

- Identify the pathways and mechanisms for radiological and chemical constituent migration, and conduct public health risk assessments and environmental impact studies

- Develop, validate, and apply various site models to augment the current understanding of the site environment, and to predict future impacts with and without remedial actions

- Provide necessary information for the identification, evaluation, and selection of the most environmentally and economically acceptable alternatives in the FS

For the purposes of the RI, a principal use of the groundwater flow and solute transport models is to develop an understanding of the nature and extent of the contamination. Significant pathways associated with past and current releases are evaluated to refine the assessment of current risk posed by contaminated groundwater. A key application of the model is to augment the existing data base by providing a systematic means to fill in the gaps between the single-point observations. Prediction of the locations of the maximum concentration resulting from the integration of the contributions from multiple sources, as well as the approximate extent of coalesced contaminant plumes, represent the types of information that can be most effectively generated via the application of the models. It is these types of information that are critical to the evaluation of current and future risk.

As presented in Chapter 15.0, uranium has been designated as the principal contaminant of concern at this time and is the only contaminant selected for detailed modeling. If deemed necessary at a later date, other contaminants could be considered through an appropriate adjustment of parameters in the solute transport model.

18.3 SUPPORT TO RISK ASSESSMENT

The baseline risk assessment evaluates the no-action alternative for the FMPC considering both the radionuclides and indicator chemicals. The baseline risk assessment is comprised of an assessment of the exposure pathways and an evaluation of the risk posed by the intake of the contaminants of concern by hypothetical off-FMPC individuals or population groups.

An evaluation of the no-action alternative requires the prediction of future conditions and associated risks if all sources and pathways are unaltered from their current status. Since past and current field observations cannot yield such predictions directly, the results of the models become a critical input to the no-action assessment. The groundwater flow and solute transport models will be used

to predict whether concentrations in groundwater at critical receptor locations will increase and whether additional receptors will be impacted to a significant extent.

The initial condition for the no-action model is equated to the model output in the evaluation of current conditions, thus the effects of any past releases are accounted for by existing concentrations in the groundwater. Current and potential future releases are quantified as sources to the extent possible with available data. The specific application of the model to the risk assessments for different operable units is presented in Chapter 22.0.

18.4 SUPPORT TO FEASIBILITY STUDY

The objective of the FS is to develop and evaluate remedial action alternatives to protect public health, public welfare, and the environment from releases or threatened releases of hazardous or radioactive substances, pollutants, or contaminants at or from the FMPC. The groundwater flow and solute transport models are used to support the evaluation of the remedial action alternatives in the FS.

One application of the model in the FS involves using the groundwater flow and solute transport models to evaluate the technical feasibility of remedial actions, including evaluating the locations and pumping rates of extraction wells, and determining the areas that could serve as sources of water for any potable water replacement wells. The modification of pathways by depressing the groundwater levels through pumping to control groundwater flow can also be accounted for in the groundwater flow model. The resultant flow field, however, influences contaminant transport patterns; thus, impacts at receptor locations must be further examined by application of the solute transport model.

The second type of model application in the FS involves the solute transport model for the prediction of future conditions under various remedial action alternatives. Actions that are source control measures are simulated by eliminating current or future releases from the model input. However, since full source control is rarely realized in practice, the model is also used to assess the effects of incomplete source control and to evaluate various source reduction measures to determine the most cost-effective program. The temporal and spatial variations of uranium concentration resulting from the different remedial actions are predicted.

The groundwater flow model also finds use as a tool for the conceptual design phase of the FS. For example, the flow model can be used to determine the number, location, orientation, and pumping rates of wells to most cost-effectively capture a contaminant plume.

19.0 DESIGN OF MODELING STUDY

19.1 GROUNDWATER FLOW AND SOLUTE TRANSPORT MODEL

Groundwater modeling is a technique for producing a representation of a groundwater system that behaves in a manner similar to the real groundwater system. Many types of models are available. Mathematical models range from the solution of simple equations to very complex computer programs. In general, the more complex the model, the closer its behavior approaches that of the real system using higher-order numerical simulations. Therefore, more complex models tend to produce better estimates of the actual behavior of the system than simpler models since the physical systems and processes can be more completely represented. The selection of a groundwater model is guided by the complexity of the real aquifer system, the amount and quality of data available, and the degree of representativeness needed.

In the case of the FMPC, the system is complex, numerous data are available, and a good representation of the aquifer system is required due to the importance of the decisions that will be at least partially based on model results. The complexity of the aquifer system is caused by the following site conditions:

- Location of the site over a thick, highly-permeable, irregularly-shaped aquifer in a bedrock valley with low-permeability walls.
- Presence of horizontal and vertical variations in hydraulic conductivity.
- Presence of a clay interbed beneath the site that separates the aquifer into upper and lower parts, but does not extend far from the site.
- Presence of several potential contaminant source areas with different strengths and periods of release.
- Presence of water supply wells that serve potential receptors and draw water from various depths within the aquifer.
- The irregular course of the Great Miami River, which is a potential receptor location.
- A river and aquifer interrelationship that involves flow from the river to the groundwater near the collector wells and flow of water from the groundwater to the river in other reaches.
- Presence of the collector wells, which influence the groundwater flow over a large area.
- Presence of large changes in horizontal hydraulic gradients across the study area.

- Presence of significant vertical hydraulic gradients within the study area.
- Variation of areal recharge through different surficial materials composed of glacial overburden, flood plain, and alluvial deposits.
- Presence of periodic significant recharge to the aquifer from Paddys Run.

Chemical and radiological contaminants associated with the FMPC site have been introduced to the Great Miami Aquifer from one or more of the following:

- Leakage from the waste pits in the Waste Storage Area
- Discharges to and leakage from Paddys Run
- Discharges to and leakage from the Storm Sewer Outfall Ditch
- Waste or product spills in the Production Area as well as from "suspect areas" within the FMPC site
- Facilities in the Production Area used for the storage, containment, or transfer of radiological or chemical materials
- Seepage from Southfield, sanitary landfill, fly ash piles, and Lime Sludge ponds

Figure 2-1 shows the FMPC site and these possible source areas.

The computer code used to simulate the complex groundwater system at the site utilizes the finite-difference method in modeling technique. This computer modeling method involves setting up a grid which divides the subsurface of the modeled area into rectangular volumes, called cells. For flow modeling, a water budget that satisfies mass conservation is established for each cell. Through a series of complex differential (finite-difference) equations, it sets inflow equal to outflow plus the increase in water stored within the cell. Inflow and outflow include water flowing across the cell faces in response to different hydraulic heads in adjacent cells. Inflow also includes seepage from streams and seepage that results from rainfall and snowmelt. Outflow includes pumping of any water well drawing water from a cell and groundwater discharge into streams. Hydraulic head is the only quantity in the water budget equation that is not given a value in most cells; quantities related to seepage from streams, pumping, etc., are supplied. Consequently, there is one water budget equation and one unknown (hydraulic head) for a large number of cells in the model. A system of equations results which may be solved simultaneously to obtain the hydraulic head in each cell. Simply stated, finite-difference flow modeling calculates hydraulic heads throughout the system by solving a large set of simultaneous equations.

Finite-difference solute transport modeling utilizes the results of the flow modeling and sets up another set of finite-difference equations that describe the movement and conservation of dissolved matter in response to various factors. These factors include rate of supply of the matter at the source, movement of the matter with the groundwater, dispersion of the matter as it moves through the complex pore system of the aquifer, and adsorption of the matter onto the aquifer matrix in accordance with the concentration in the water. This set of equations is solved for solute concentration in each cell.

In practice, many of the parameters that have to be supplied to these equations, such as hydraulic conductivities, recharge rates, river leakage factors, solute source rates, dispersivities, and distribution coefficients are not accurately known and vary with location and time. Usually the hydraulic heads and the solute concentrations in some cells are known. Pumping rates may be known. Consequently, the only way to develop a model that approximates the known hydraulic heads and solute concentrations is to try different values for the parameters within an acceptable, physically meaningful range until the model produces hydraulic heads and solute concentrations that best match the known values across the area of interest. In the process, the model calculates hydraulic heads and solute concentrations in all cells throughout the model in accordance with basic physical laws. Since the model is a simplified numerical representation of the actual aquifer system, it is not practical for the model results to match the hydraulic heads and solute concentrations perfectly.

Calibration is the process of selecting parameters for successive trial computer runs until the field-measured hydraulic heads and solute concentrations are adequately approximated. The calibration process involves choosing parameters that are reasonable when the known hydrologic and geologic conditions of the area are considered. Calibration does not necessarily produce a unique solution to the problem of representing the actual system, because more than one combination of parameters may cause hydraulic heads and solute concentrations to be approximated equally well. However, it does produce a realistic solution for a complex problem. Once the modeler finds an acceptable solution, judgment must be used to decide whether the model calibration process should be continued in an effort to find a better solution. The objective of the calibration is the production of a model that is capable of satisfying the objectives of the modeling. In the present case, these objectives are the following:

- To characterize the groundwater system by supplying groundwater flow rates and directions, supplying rates of recharge via infiltration of precipitation and stream water, relating contamination to possible sources and receptors, supplying contaminant source loading rates, estimating uranium

concentrations between monitoring wells, and evaluating uranium mobility in the system.

- To evaluate the need for additional groundwater monitoring and field investigations.
- To estimate future concentrations at potential receptor locations under the no-action remedial alternative.
- To evaluate the feasibility of different remedial actions.
- To estimate temporal and spatial variations in uranium concentration that would result from different remedial actions.

19.1.1 Model Code Selection

For purposes of the RI/FS modeling, minimum requirements for the model code were determined to be:

- Three-dimensional modeling capability. This capability is necessary to account for possible vertical flow through the clay interbed at the FMPC site and to simulate the effects of vertical hydraulic gradients caused by pumping at depth.
- The capacity to quantitatively predict contaminant concentration at receptor locations as necessary to fully satisfy the requirements of the RI/FS. Therefore, only codes with options to model solute transport and associated attenuation/retardation processes were considered.
- Adequate verification of the code and previous applications under similar project settings.
- The availability of the modeling code and accompanying documentation.

Other criteria were applied to selecting the model code to refine the selection process with respect to the specific site conditions and study requirements. These criteria included:

- The capacity to model water table aquifer conditions.
- The capacity to model decay chains. Although the study of uranium does not require the consideration of daughter products, this would become a consideration if other radionuclides are found to be important.
- The ability to represent adsorption and decay processes so as to provide flexibility in the range of constituents that can be modeled.
- The option to consider density variations and temperature or concentration effects on fluid viscosity. Although not considered necessary at this time, this option could be beneficial to best simulate certain critical processes (e.g., leakage through the river bed).

- The convenience of model application based on features such as preprocessing and postprocessing capabilities, user documentation, mesh generation, solution method, restart capability, applicability to available computer systems, and user familiarity.

SWIFT III was determined to best satisfy the full set of selection criteria. Therefore, a decision was made to select SWIFT III as the code for use in the hydrogeologic investigation of the FMPC site-wide RI/FS (IT Corporation 1988b).

19.1.2 Model Code Verification

The SWIFT code was originally developed by Sandia National Laboratories for the Nuclear Regulatory Commission for use in the high-level nuclear waste isolation program. The first extension of the SWIFT code was made by Geotrans, Inc. (Geotrans), and was given the title SWIFT II.

Comparisons of results from the SWIFT II model code and analytical solutions appear in many documents. Results of the SWIFT II model applications have also been compared with data collected in numerous field studies. The comparisons provide evidence that the equations solved in the model are numerically correct and properly simulate the physical processes controlling groundwater system behavior. Applications of the code to actual sites have also appeared in several reports, including those appearing in Ward, et al. (1984), and Reeves, et al. (1986), which are summaries of the model verification process and field comparisons. Particularly noteworthy are previous applications of the SWIFT II code to the FMPC (Geotrans 1985) and to a nearby site with a similar hydrogeologic setting (Ward et al. 1987).

Modifications to the SWIFT II code were made by Geotrans in 1986. The modifications included changing the code from FORTRAN IV to FORTRAN 77, and making modifications in data input and output routines. The resulting model was termed the SWIFT III code.

Pursuant to the modifications, Geotrans and IT conducted verification work that tested the validity of several features of the SWIFT III code. The results of Geotrans' most recent model code verification work are presented in "SWIFT III Quality Assurance Benchmark Problem Execution Fiche" (Geotrans 1988). IT's verification program included comprehensive testing of those SWIFT III features of importance to the FMPC. Both the groundwater flow and solute transport components of the SWIFT III code were tested against analytical solutions and the results from other established numerical models for a wide range of hypothetical conditions. The results of the

IT verification program found no significant problems with the model. The IT code verification work is documented in a separate report (IT Corporation 1990).

19.1.3 Overview of Technical Approach

The products of the model development process, using the SWIFT III code, are a regional three-dimensional groundwater flow model and a local three-dimensional solute transport model. The process by which these models were constructed followed a progressive development that passed through simpler two-dimensional numerical and analytical models to arrive at the final, more complex models. The simpler models were used in the early stages of model development because of their relative ease of utilization. They had short computer execution times and required no significant waiting time for results of runs, and the computer input and output files were relatively short and easy to handle. Consequently, the simpler models were used both to develop a general understanding of model conditions and to generate initial estimates of input parameters for calibrating the more complex models. Also, the two-dimensional models provided a relatively easy way to obtain information on the sensitivity of model results to variations in the values of input parameters.

The SWIFT III numerical code was used in the modeling effort for this study. The notations 2DFLOW and 3DFLOW are used to represent the simulation of the two- and three-dimensional flow modeling utilizing the SWIFT III code. For the solute transport modeling effects, the notations 2DSOL and 3DSOL are used to represent the two- and three-dimensional modeling simulations.

The modeling progression is fully explained in Chapter 20.0 and is summarized as follows:

- A preliminary two-dimensional regional flow model, developed to support a previously completed hydrogeologic study of FMPC discharge to the Great Miami River (IT Corporation 1988b), was refined using subsequently collected field data to establish the initial two-dimensional flow model (2DFLOW). The finite-difference grid used in the previous study was not changed. (Hereafter, the previous study will be referred to as the Zone of Influence Study.)

- A three-dimensional flow model (3DFLOW) was produced by introducing five model layers, thereby representing the aquifer system as five distinct zones in the vertical direction. River stages and well pumping rates were imported from 2DFLOW. Hydraulic conductivities, boundary heads, recharge rates, and river leakage factors from 2DFLOW were used to provide initial estimates of the related parameters in 3DFLOW. (Recharge rates in this report refer to recharge due to infiltration of rainfall and snowmelt as well as seepage from Paddys Run, unless another source is specifically mentioned.)

- Two-dimensional analytical solute transport models were produced to roughly simulate and better understand the uranium plumes that are present in the groundwater system downgradient from the Waste Storage Area and within and downgradient from the southwest portion of the FMPC.

- A two-dimensional finite-difference solute transport model (2DSOL) was produced. It covered a smaller area than 2DFLOW and 3DFLOW and had much smaller cells in the finite-difference grid. Boundary heads, recharge rates, and hydraulic conductivities were imported from equivalent cells in 2DFLOW. An initial estimate for the uranium distribution coefficient was obtained from the two-dimensional analytical modeling.

- A three-dimensional finite-difference solute transport model (3DSOL) was produced. Boundary heads, recharge rates, and hydraulic conductivities were imported from equivalent cells in 3DFLOW. Initial estimates of contaminant source loading rates (rate of discharge of uranium to the aquifer) and a distribution coefficient for uranium were obtained from the calibrated 2DSOL model.

This model development was accompanied by a review of pertinent literature and reports of previous studies. The purpose of the review was to make the model consistent with available information on the site. Parameter values were maintained within reasonable ranges during the model calibration. SWIFT III code verification work (IT Corporation 1990) preceded and accompanied the model development work. Preliminary results of concurrent geochemical analyses and geochemical modeling studies were incorporated into the calibration of the solute transport model as they became available.

Radioactive decay is not considered in the present modeling. While uranium is radioactive and will decay to other radioisotopes and ultimately to stable lead, the half-lives of the Uranium-238, -235, and -234 isotopes are 4.9×10^9, 7.04×10^8, and 2.47×10^5 years, respectively. The uranium has been present in the regional aquifer less than 40 years, a time period that is inconsequential relative to the reported half-lives. Therefore, modeling of uranium migration can be conducted without considering radioactive decay, and it can be concluded that the total mass of uranium within the aquifer will not decrease significantly as a result of radioactive decay during any period of interest.

Only two models, 3DFLOW and 3DSOL, are being applied to the RI/FS and risk assessment. The analytical models and the two-dimensional flow and solute transport models were only prepared as parts of the procedure for constructing the three-dimensional models and facilitating sensitivity analyses. The 3DFLOW and 3DSOL models are better representations of the real physical system. Nevertheless, the results of the models should be treated as estimates. Consequently, the use of models should be confined to treating the results as estimates and taking appropriate action to accommodate any error in the estimates. Such action could include monitoring the groundwater

system around extraction wells to see if they are producing the expected results, monitoring uranium concentrations at receptor locations, and continuing the sampling of selected existing monitoring wells.

19.2 MODEL OF LONG-TERM MIGRATION POTENTIAL

Flow and solute transport models were used to project the long-term migration potential of waste constituents currently stored in the source operable units. This type of long-term predictive modeling is referred to as fate and transport modeling and is described in detail in Chapter 22.0. The development of a fate and transport model progresses from the development of a conceptual model based upon the depositional history and hydrogeology of a site to the formulation of a mathematical representation of the conceptual model. Groundwater and vadose zone systems are complex and require some degree of simplification to numerically represent these systems. An important aspect of modeling is the understanding of the assumptions and limitations of the analysis. In any application of the results of the fate and transport analysis, the assumptions, limitations, and uncertainties must be considered.

A combination of analytical models and a numerical model were selected for projecting the long-term fate and transport of constituents through the glacial overburden. Since the primary receptor in both cases is a user of groundwater from the Great Miami Aquifer, the basic requirement for the glacial overburden model was to project the time it would take a constituent to migrate from a source unit through the glacial overburden to the Great Miami Aquifer and the concentration of the constituent reaching the aquifer over time.

The two major processes important to groundwater contamination that can be expected at a waste site involve the leaching of solid contaminants and the percolation of the contaminated leachate to the underlying aquifer. Within the FMPC, spills and leaks of contaminated liquids were documented during active operations. However, under current conditions, the direct discharge of fluids is not likely and the leaching of the solid waste materials is the most likely continuing source of contamination. Solid material itself does not migrate through the porous medium. Therefore, it is necessary for a liquid such as precipitation, surface water run-on, or groundwater to contact the waste and leach a portion of the available constituents from the solid material and transport the resulting leachate to the aquifer. Groundwater contamination by this route depends on the recharge rate and the solubility of the solid constituents.

Contaminant migration from the waste source to the receptor involves flow through both an unsaturated zone (vadose zone) and saturated zones (regional aquifer; perched zones). Flow in

these zones is controlled by Darcy's Law which states that the volumetric flow is proportional to the product of the driving force, the material's ability to transmit water, and the cross-sectional area perpendicular to the flow direction. The driving force for the flow is the hydraulic gradient, which is the difference in the energy or head between two points divided by the distance between those points. In the vadose zone, however, the matrix is not saturated and the physical affinity of water to the particle surfaces and capillary pores greatly affects water movement. Therefore, Darcy's equation is modified to account for both hydraulic gradient and the matrix suction forces. Generally, the unsaturated hydraulic conductivity at residual moisture content is very small. Unsaturated hydraulic conductivity increases, gradually at first and then more rapidly, as the degree of saturation increases from the residual moisture content to the saturated moisture content. Because of this relationship, unsaturated hydraulic conductivity changes with the degree of saturation, but is usually a fraction of the saturated hydraulic conductivity.

The movement of contaminants in groundwater can be described by the gross fluid movement (advective flow) and dispersion. Dispersion can be further described by mechanical dispersion and diffusion. Mechanical dispersion in groundwater is primarily caused by varying pore sizes, varying path lengths, variation in velocity gradients across the pore space, and flow splitting around matrix particles. The first three factors are responsible for longitudinal dispersion, while flow splitting causes transverse mixing. Molecular diffusion, which will produce both longitudinal and transverse dispersion, is the major component of dispersion in fine-grained deposits such as till, but does not cause significant dilution of a plume.

Because of the above factors, the magnitude of longitudinal dispersion is much greater than transverse dispersion in groundwater systems. The dilution of a contaminant plume due to transverse dispersive mixing with clean water is, therefore, of little consequence. On the other hand, longitudinal dispersion can cause dilution of a contaminant plume as the contaminants mix with clean water ahead of and behind the plume. Such longitudinal dispersive mixing is most effective for short-term releases of a contaminant slug into an aquifer. For long-term continuing releases such as those being modeled in this case, only the leading edge of a long plume effectively mixes with clean water and the overall amount of plume dilution is relatively small.

In addition to dispersion, both attenuation and retardation should be considered in fate and transport models. Attenuation is the irreversible loss of the contaminants from the plume. While the mass of the contaminant is not changed, a certain percentage of the total contaminant will be removed from the solution and no longer be available for transport. Attenuation mechanisms can include hydrolysis, complexation, oxidation/reduction reactions, acid/base reactions, radioactive decay, and

chemical precipitation. For the FMPC, the attenuation process is accounted for by a combination of chemical modeling to determine a solubility-constrained leachate concentration along the potential flow path and the use of decay constants for the radionuclides.

Retardation consists of reversible reactions that slow the arrival of a constituent, but do not act as a sink for contaminants. Many types of reversible reactions can cause retardation in groundwater systems. However, two mechanisms, organic retardation and cationic retardation, are generally used for modeling. Organic retardation is the sorbing of hydrophobic compounds onto the organic materials in the matrix. Cationic retardation refers to charged ions being attracted to ionic particles within the matrix. In both cases, retardation is reversible, and, over time, contaminants can reenter the plume as soluble forms. This causes the development of a plume shape that has a long tail segment of decreasing contamination which will be present for extended time periods following the removal of the source. These absorption processes have a substantial potential to alter the migration rate of reactive solutes and, in situations with high partitioning coefficients, may be the dominant influence on the transport process.

The distribution coefficient (K_d) is the parameter used to derive a retardation factor (R_d), which in turn is used in the model to numerically describe the slowing down of the velocity of contaminant movement. The K_d formulation of the reaction term of the transport equation has numerous assumptions and uncertainties associated with it. Nevertheless, these are widely recognized and the formulation provides a practical means of incorporating the reaction process into transport models.

19.2.1 Model Code Selection

Models were selected based upon the need to account for these processes. In order to expedite the process, a two-stage evaluation was used. Initially, the constituents were evaluated using a screening model to determine if the constituents would even reach the Great Miami Aquifer within the time period of interest under conservative conditions. For those constituents that would reach the aquifer, a more detailed evaluation was then conducted.

ST1D and ODAST are both based on the solution originally developed by Ogata and Banks (1961) and can also be found in Freeze and Cherry (1979) and van Genuchten and Alves (1982). The solution was modified to account for retardation of contaminants, (ST1D) and retardation, source depletion, and source decay (ODAST). The computer code ODAST which was used to provide the final results calculates the normalized concentrations in a steady-state flow field from a source having varying concentrations. As outlined above, the source in the lower layers varied as a result

of the changing input from the previous layer. ST1D has been verified against STRIP1B (IT 1990) and GEOFLOW (IT 1986).

The selection of the analytical model was based upon the following factors:

- Based on the fine-grained nature of the deposit, transverse dispersion is limited; therefore, the one-dimensional solution matches the physical flow system.
- The computer code should provide flexibility to perform numerous iterations in a shorter time. One-dimensional computer codes such as ODAST satisfy this requirement.
- The method was consistent with approaches used for similar radionuclide assessment codes such as the flow portions of PRESTO (U.S. EPA, 1987), and other site studies (Echart et al. 1989).
- The basis of the solution is well documented and the code has been extensively verified.

The model code used for the analysis of contaminant transport in the regional aquifer i.e., Great Miami Aquifer, is SWIFT III. The SWIFT III code is a fully transient, three-dimensional, finite-difference model which solves the coupled equations for flow and transport in geologic media. The model has been applied at the site as discussed in earlier sections.

To estimate concentrations at the receptor using SWIFT III, a series of unit loading runs were made at varying retardation factors for a 500-year period. These runs assumed that the only contaminant loading to the aquifer was coming from the waste unit in question. From these runs, a ratio analysis that compared the concentration produced by the unit loading run with the loading estimated from the vadose zone modeling was used to project a concentration at the receptor.

19.2.2 Overview of Technical Approach

To account for the various components of contaminant hydrogeology, the following steps were taken to model the long-term migration potential of the constituents of concern at the individual operable units:

- Based on the site-specific geologic setting, a conceptual hydrogeologic model was developed.
- The hydrologic parameters determining fate and transport of the constituents were determined or estimated for each hydrostratigraphic unit within the conceptual model.

- The available information on the specific waste unit was reviewed to establish the characteristics of the waste.
- Geochemical modeling was completed to partially account for attenuation by determining the chemical speciation that would result from the reactions of infiltrating water with the waste materials and the matrix of the glacial overburden.
- Estimates of cationic retardation were completed based on partitioning coefficients that were selected from the literature.
- Estimates of organic carbon retardation were developed based upon the grain-size distributions and organic carbon present in the glacial overburden matrix.
- Flow modeling was completed to determine the effects of dispersion and retardation on the projected leachate concentration at the receptor locations under the selected scenarios.

A more-detailed discussion of the technical approach to modeling long-term contaminant release and migration is presented in Chapter 22.0.

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20.0 GROUNDWATER FLOW MODEL

20.1 PRELIMINARY TWO-DIMENSIONAL FLOW MODEL

The two-dimensional flow model (2DFLOW) was developed by refining the model that was used in the Zone of Influence Study (IT 1988b). The refinements were based on an increasing data base and resulted in some degree of modification to all features of the model except the grid and the stream stages applied to cells representing the Great Miami River. The refinements included adjusting well pumping rates and bedrock elevations to include values obtained subsequent to the Zone of Influence Study. A more refined calibration was also performed, which involved successive trial adjustments of hydraulic conductivity, recharge rates, river leakage factors, and boundary heads to produce simulated hydraulic heads that achieved a better fit to hydraulic heads measured in the actual system. The measured hydraulic heads were from water level data gathered in a comprehensive regional study in April 1986. An independent check of model performance was based on new water level data from May 1988. A sensitivity analysis was also performed. The construction, calibration, and sensitivity analyses of the preliminary flow model are described below.

20.1.1 Model Grid and Boundary Conditions

The model grid imported from the Zone of Influence Study is shown in Figure 20-1. The exterior boundary lines of the grid are located far enough from the site to include the collector wells, the Great Miami River, and important features of the bedrock valleys. The boundaries are considered to be outside the area of influence of any pumping wells that affect the flow regime within the study area. The Shandon Tributary is a bedrock valley that enters the model north of the FMPC and the Paddys Run Outlet is a bedrock valley south of the FMPC (Figure 11-1). Both valleys are included in the model. The groundwater conditions in the Shandon Tributary affect flow under the northern part of the FMPC, and the Paddys Run Outlet channels groundwater flowing from the FMPC southward toward the Great Miami River.

The model grid is oriented 30 degrees west of north to approximately align it with the axis of the main bedrock valley. This orientation also aligns the grid with the axes of the New Haven Trough and Shandon Tributary, which are roughly normal to the main valley. In the grid east-west direction, the grid extends from about two miles west of the FMPC to approximately 1/2 mile east of Ross. In the grid north-south direction, it extends from about 3/4 of a mile south of Shandon to about 1/2 mile south of New Baltimore. The exterior boundary of the grid is a rectangle, about 32,000 feet by 25,000 feet (6.1 by 4.7 miles). The grid covers an area of approximately 29 square miles.

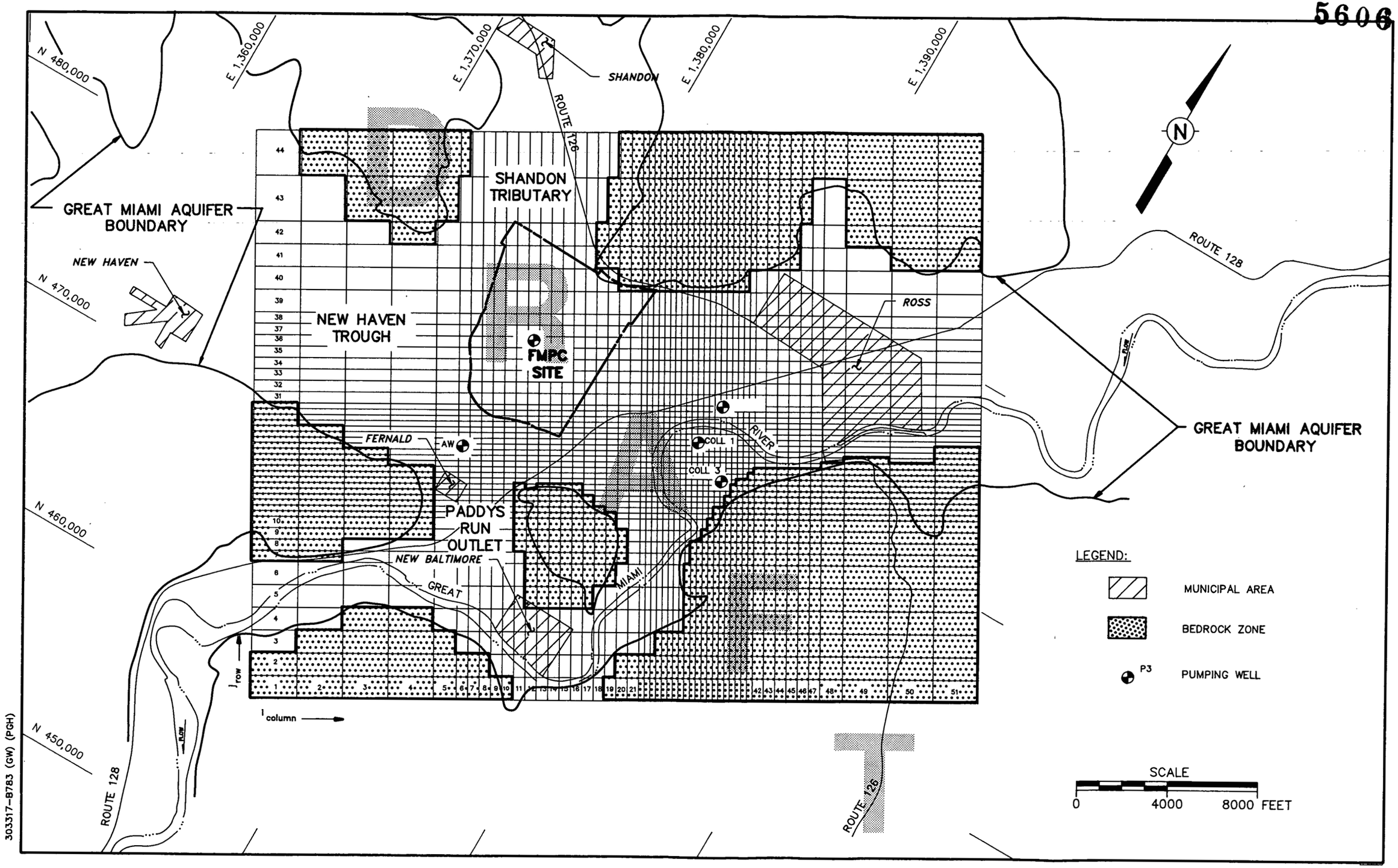

FIGURE 20–1
SWIFT III FLOW MODEL GRID
FOR RI/FS MODEL AREA

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The cell sizes in the grid vary from 250 feet by 250 feet in the vicinity of the collector wells to 2,000 feet by 2,000 feet at the northeastern and northwestern corners of the grid. This arrangement places relatively small cells where the hydraulic gradients are highest. Using large cells in peripheral parts of the model, where the cell size has little effect on the modeling, reduces the total number of cells that must be handled.

The flow system in the glacial overburden which overlies the Great Miami Aquifer across much of the study area was not included in the model. The overburden contains only localized flow systems and provides limited recharge to the Great Miami Aquifer by downward percolating water. The effects of the low permeability overburden are included in the model by using lower values for recharge where the low-permeability overburden is present than in areas where it is not present.

The bedrock underlying the regional aquifer has a much lower permeability than the Great Miami Aquifer. The bottom faces of the cells within the grid system were treated as a no-flow boundary. Vertical faces representing bedrock at the exterior model boundaries were also treated as no-flow boundaries. Constant-head boundary conditions were supplied at cell faces representing the Great Miami Aquifer at the exterior boundaries of the model grid. These aquifer boundary conditions were subject to change during the calibration process.

20.1.2 Input Data

Stratigraphic data and other information related to the model input were obtained from the reported results of previous investigations in the area (Fenneman 1916; Durrell 1961; Dove 1961; Spieker and Norris 1962; Spieker 1968a; Watkins and Spieker 1971; Leow 1985). Pertinent data and information were also obtained from boring logs of wells constructed for the RI.

A map of the elevation of the top of the bedrock that had been prepared for the Zone of Influence Study was refined by incorporating additional bedrock data from new wells constructed for the RI and some additional pre-existing bedrock data. Also, cells treated as bedrock in the model were redefined as those that contained less than 25 feet of saturated Great Miami Aquifer, which resulted in minor modification of the bedrock distribution in the model.

The assigned discharge rates for the Collector Wells 1 and 2 were 8.68 Mgd each for March and April, 1986, based on an approximate equal distribution of flow among the wells (Miami Conservancy District unpublished data). The collector well pumping records for the period from 1952 through 1989 indicate that an equal allocation of the discharge rates would also be realistic for average long-term conditions. Flow rates for the other production wells within the study area

were based on estimates obtained from well owners by WMCO. No production rate records were supplied. The combined discharge rate for the Albright & Wilson Co. wells (Well 3062) in 2DFLOW was initially estimated at 100 gpm. This rate was changed in subsequent models as new information became available. The rate for the FMPC production well P3 (also referred to as Well 4103) was 292 gpm, which is the average pumping rate for the period from March 26 through April 6, 1986 (WMCO 1986). Residential water wells were not included in the model, because their pumping rates are too small to significantly affect flow in the highly permeable Great Miami Aquifer. Indeed, even the pumping rates of the Albright & Wilson Co. wells and the FMPC production wells were found not to have a significant effect on the flow.

River stages input to the model were derived from a calculation of the water surface profile of the Great Miami River under average flow conditions using HEC-2, a surface water flow code. These stages were calculated for the Zone of Influence Study. The input for the HEC-2 program included flow regime, discharge for average flow conditions, cross-sectional geometry, starting water surface elevation, reach lengths, and energy loss coefficients. All input data for the Great Miami River were obtained from the U.S. Army Corps of Engineers, Louisville District (1987). Stages for each river cell chosen to represent the influence of the river were calculated by interpolation between river cross sections used in the HEC-2 modeling. The river cells are shown in Figure 20-2.

A steady-state groundwater flow option of SWIFT III was used for the modeling. This option is considered adequate since short-term fluctuations in configuration of the hydraulic head distribution in the aquifer are not representative of the long-term behavior of the aquifer, and the long-term behavior is of principal interest in the application of the model. Such short-term fluctuations are caused by temporary changes in well pumping rates, recharge rates, and stream stage. Monthly contoured maps of hydraulic head in the groundwater system in the FMPC area show little change in the general configuration of groundwater elevations for the period of observation. Three such maps are shown in Figures 12-1 through 12-3.

During monthly monitoring from January 1989 through July 1989, a seasonal rise in hydraulic head was identified near Paddys Run for the period March to June 1989. This rise is shown in Figures K-13 through K-19 in Appendix K and in the hydrographs in Appendix N. An analysis of hydrographs and monthly groundwater elevation maps over an extended period indicates that this rise in water levels, which is due to recharge along Paddys Run, is an annual occurrence with varying duration up to five or six months. An investigation of the effect of this seasonal change in potentiometric surface configuration shows that it causes very little deflection of the normal

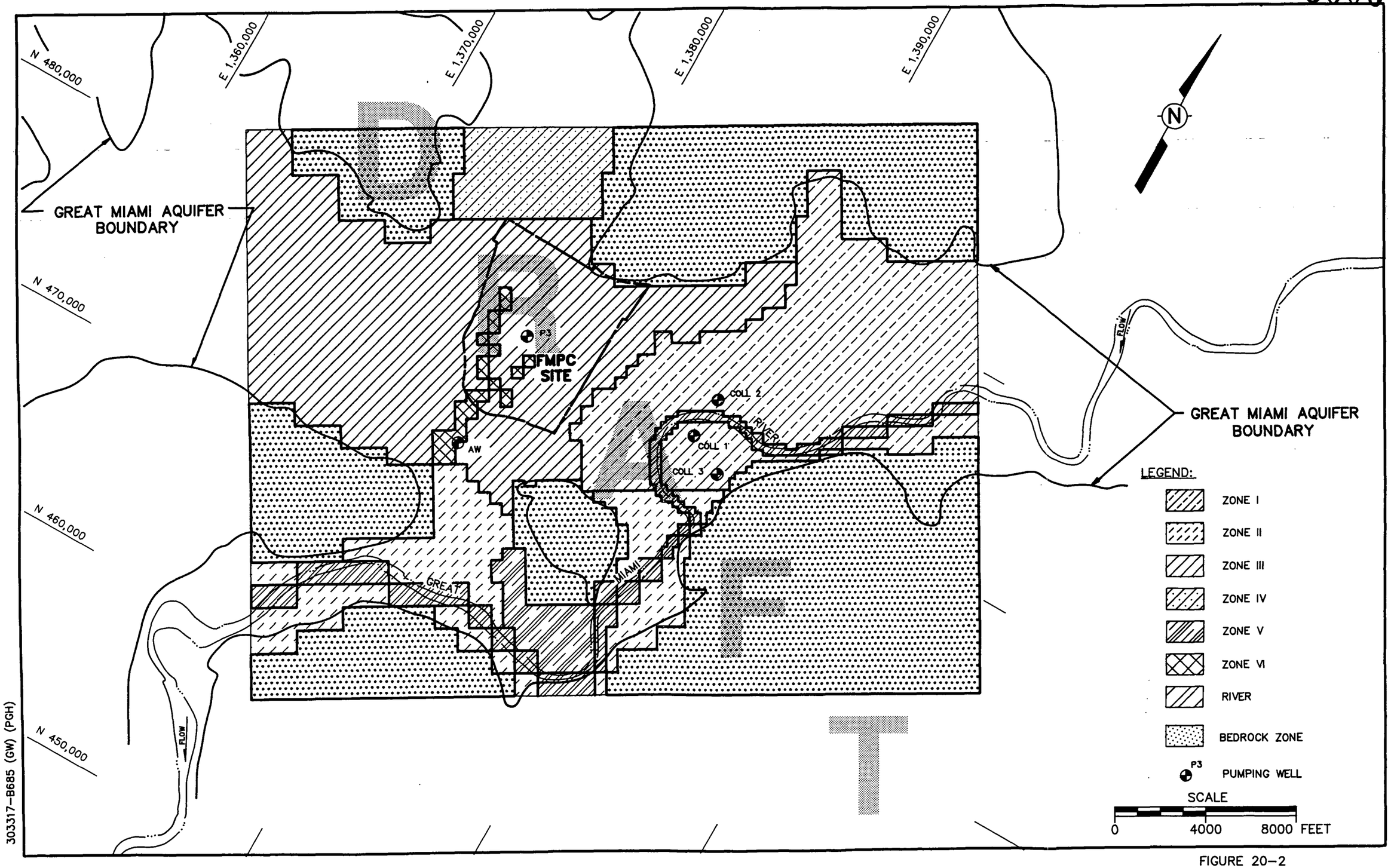

FIGURE 20-2
REFINED 2-DIMENSIONAL FLOW MODEL
MODEL ZONES

long-term movement of a solute that is subject to retardation processes in groundwater. This investigation involved plotting the track of a particle moving over a succession of one-month periods at the average rates of flow indicated by the monthly groundwater elevation contour maps, with the retardation factor set at 12. The resultant tracks are shown in Figures 12-10 and 12-11. The water particle deviates only slightly from the long-term path movement as a result of the transient rise in hydraulic head under Paddys Run. The observed lack of a large effect by transient phenomena is a result of the slow movement of solutes in groundwater.

20.1.3 Model Calibration

The purpose of model calibration is to produce a realistic representation of the groundwater system that behaves in a manner similar to the real system. The 2DFLOW model calibration started with the same values for hydraulic conductivity, recharge rate, river leakage factor, and boundary heads that were used in the final production runs of the Zone of Influence Study model. Subsequently, the parameters were changed through successive trials to produce an improved match between computed hydraulic heads and hydraulic heads observed in monitoring wells that represent conditions in the Great Miami Aquifer. Successive trials involved comparing computed and observed hydraulic heads at each well and checking the magnitude, distribution, and sign of residuals; i.e., observed head minus computed head. These residual values were analyzed with respect to reasonable changes in hydrological and geological parameters. The input was then revised for the succeeding run. The results of the succeeding simulation were then analyzed, and the process was repeated until model results adequately matched the field data in accordance with prescribed statistical criteria.

Evaluating the adequacy of the match was aided by the use of GEOSTAT, a statistical program for examining the goodness of fit of groundwater modeling data (IT Corporation 1987). GEOSTAT yields nine statistical measures. Five of these measures were used as statistical criteria for flow model calibration. The criteria were stated prior to beginning the modeling (IT Corporation 1988a), and were satisfied by the calibrated model. Furthermore, within the constraints of the criteria, the acceptability of the calibration was also based on the judgment of engineers and geologists working on the project.

A brief summary of the statistics used as predetermined criteria for flow model calibration follows:

- Mean Residual: A negative value indicates that computed hydraulic heads are, on the average, greater than observed hydraulic heads. A positive value has the opposite meaning. The value of this parameter must be reasonably close to zero, otherwise bias exists in the computed hydraulic heads. The predetermined criterion was ± 0.5 foot.

- Mean of the Absolute Values of the Residuals. This parameter is a measure of the average difference between the observed and computed values without regard to the sign of these deviations. The optimal value is zero. The predetermined criterion was ±2 feet.

- Residual Standard Deviation: This is a measure of the degree of variance of the residuals from their mean. The optimal value is zero. The greater the value, the poorer the fit. The predetermined criterion was 3 feet.

- Regression Coefficient Between Measured and Computed Values: This is a measure of the tendency of computed hydraulic heads to increase as corresponding measured hydraulic heads increase. The optimal value is one, indicating a 1:1 correspondence between computed and measured values. The predetermined criterion was 0.95.

- Nearest Neighbor Autocorrelation, Unit Normal Deviate: This value is used to test for significant spatial clustering of positive or negative residuals. Significant spatial clustering is indicated by a value greater than 1.645 (IT Corporation 1987). This value was used for the predetermined criterion.

Other statistics were inspected for evidence of poor fit. They included mean observed head, mean computed head, coefficient of skewness of the residuals, error variance of the residuals, correlation coefficient between computed and observed heads, and correlation and regression coefficients between residuals and computed heads.

The performance of the model was also subjectively evaluated by examining the map of hydraulic head and flow vectors, an example of which is presented in Figure 20-3. The groundwater elevation contours shown in Figure 20-3 and all other potentiometric contour maps of computer model results were produced by SURFER, Version 4 (Golden Software, Inc. 1989), using Kriging. All of the contouring options of SURFER were tested, and Kriging produced the most accurate contours for these particular maps. The vectors shown in Figure 20-3 were produced by postprocessing programs written by IT to prepare data output from SWIFT III for AutoCad vector plots. Each vector points in the direction of horizontal groundwater movement at that location, and its length is proportional to the horizontal component of the average velocity. Horizontal velocities beneath the FMPC are so low compared to velocities in the Paddys Run Outlet and near the collector wells that they appear as dots on the map.

In Figure 20-3, the flow directions and the hydraulic head distribution around the collector wells are shown to be reasonable. Flow is directed toward the wells and accelerates as it approaches the wells. The channeling of flow in the Paddys Run Outlet results in a relatively high hydraulic gradient and large flow vectors. Differences between this map and the hand contoured map in Figure 12-2 are a result of the smoothing effect of allowing model heads at the observation wells to

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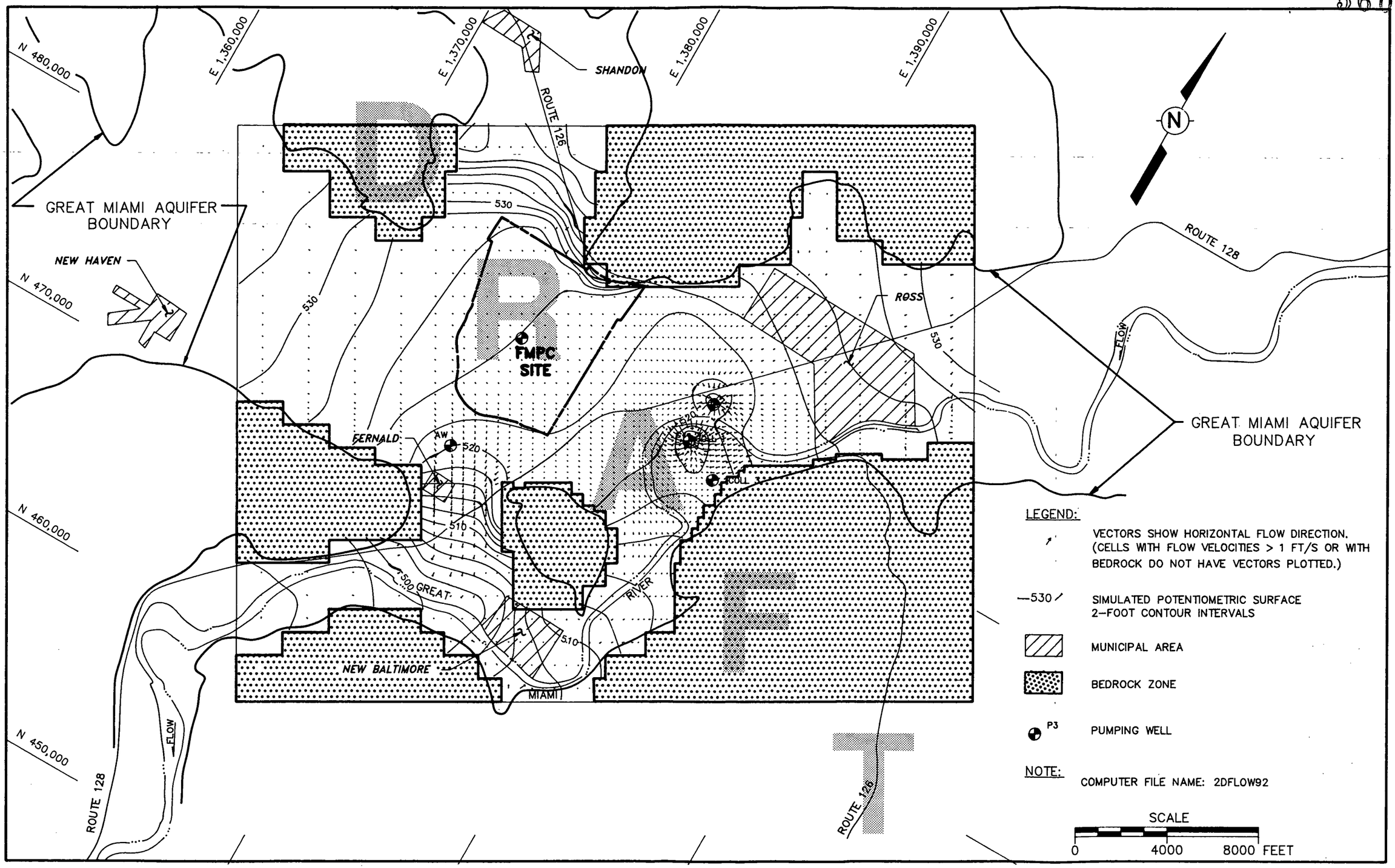

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FIGURE 20-3
REFINED 2-DIMENSIONAL FLOW MODEL
CALIBRATED RUN
FLOW VECTORS AND POTENTIOMETRIC SURFACE

deviate from the field measurements as long as the deviations are within the guidelines discussed above. Local and temporary anomalies that are not representative of steady-state conditions are also smoothed out of the model results.

The performance of the model was also checked by calculating a water balance for a large area around the collector wells. This balance was performed for the 1986 calibrated run described below. The area included in the balance covered the entire width of the bedrock valley from the eastern boundary of the FMPC to the western edge of Ross. The calculated inflows and outflows are reported in Table 20-1. The inflow/outflow ratio is 1.0002, or within 0.02 percent of a perfect flow balance. This result provides a successful check on the performance of 2DFLOW.

Model Calibration for April 1986 Data

The primary reason for using the Apri 1986 data for model calibration was that a large number of water level measurements over an extensive area were made at that time. The degree of representativeness of these water level data is indicated by the hydrograph shown in Figure 12-4. This hydrograph is for Well 02E, located next to Highway 128 about a half mile north of Willey Road, over 2,000 feet from the Great Miami River and over 4,000 feet from the nearest collector well. It is far enough from the river and the collector wells that neither would likely dominate the water level fluctuations and cause the well to be nonrepresentative of the general area. Inspection of the hydrographs shows that a median value for the water level prior to the 1987-1988 drought is about 520.5 feet. The water level at this well during the period of measurement (March 27 through April 11, 1986) was about 1.5 feet above the medial level. This 1.5-foot difference is small and should have very little effect on the application of the calibrated model to average conditions in the aquifer. Water levels measured in the drought of 1987 and 1988 are not representative water levels. As such, the May 1988 water levels provide an excellent data base for an independent test of model validation.

Initial values for hydraulic conductivity supplied to the model at the start of the calibration runs were differentiated according to hydrogeologic zones described by Spieker (1968a). The zones for the present model were numbered as shown in Figure 20-2. Zone I is an area where the Great Miami Aquifer is not overlain by clayey overburden. It is located in the eastern part of the model around Ross. Zone II is also an area where clayey deposits do not overlie the Great Miami Aquifer, located in this case along the Great Miami River where it bends southward in the vicinity of New Baltimore. Zone III is the upland area around the FMPC and west of the FMPC, where the Great Miami Aquifer is covered by glacial overburden. Zone IV is in the Shandon Tributary and Zone V is an area near New Baltimore. The Great Miami Aquifer contains an abnormal

TABLE 20-1

WATER BALANCE IN THE VICINITY OF THE SOUTHWESTERN OHIO WATER COMPANY COLLECTOR WELLS FOR THE TWO-DIMENSIONAL FLOW MODEL

Inflow Locations	Inflow Rates (Cubic Feet Per Day)	(Mgd)
West Boundary, I - 21, J = 10-39	181,358	1.36
East Boundary, I = 50, J = 22-40	292,509	2.19
North Boundary, I = 46-49, J = 41	26,886	0.20
Great Miami River Leakage	1,788,381	13.38
Recharge	308,227	2.30
Total Inflow	2,597,361	19.43

Outflow Locations	Outflow Rates (Cubic Feet Per Day)	(Mgd)
South Boundary, I = 21-32, J = 10	101,818	0.76
Collector Wells	2,317,307	17.33
Discharge to Great Miami River	117,603	0.88
Total Outflow	2,536,728	18.97

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number of clay interbeds in Zones IV and V. Zone VI is where Paddys Run has eroded through the clayey glacial deposits to the Great Miami Aquifer. The initial value for hydraulic conductivity assigned to Zones I, II, and IV was 400 ft/day (1.4×10^{-1} cm/s). A hydraulic conductivity of 300 ft/day (1.1×10^{-1} cm/s) was assigned to Zones III and VI, and 150 ft/day (5.3×10^{-2} cm/s) was assigned to Zone IV. These values were based on a review of available data on aquifer transmissivity and stratigraphy, as modified by model calibration in the Zone of Influence Study. Bedrock was assigned a hydraulic conductivity of 0.003 ft/day (1.0×10^{-6} cm/s), which is in the lower part of the range for limestone and above the range for shale given by Freeze and Cherry (1979, p. 29).

Initial values for infiltration were applied to these same zones. Recharge in Zones I, II, VI and the river cells was assigned a value of 14 inches per year (in/yr); and recharge in the remainder of the area, where the aquifer is overlain by glacial overburden, was assigned a value of 6 in/yr. These values came from the Zone of Influence Study. The value of 14 in/yr for the alluvial area is generally consistent with the value of 12 in/yr estimated by Dove (1961) and the slightly high monitoring well water levels mentioned earlier in this section.

The initial value assigned to the river leakage factor was 0.35 day^{-1}. The same value was assigned to all river cells. This value was imported from the Zone of Influence Study. Its initial value in that study was set at about three orders of magnitude below the aquifer hydraulic conductivity.

Initial values for hydraulic heads at the exterior boundaries of the model grid were imported from the Zone of Influence Study, where they were estimated by extrapolating hydraulic heads from the hand-contoured groundwater elevation map of April 1986. This method of estimating the boundary heads avoided introducing sharp changes in hydraulic gradient between the boundary and the nearest monitoring well. The external boundaries of the model in the bedrock areas were treated as no-flow boundaries in the regional models.

Initial parameter values imported from the Zone of Influence Study were reviewed and refined by successive trials during the model calibration runs. The degree of calibration achieved by these runs is indicated in Figure 20-4, which shows the magnitude and distribution of the residuals. The aim of the calibration was to produce small residuals, without producing clusters of positive or negative values. Further reduction of the residuals would not have increased the representativeness of the model significantly. The predetermined statistical criteria previously described in this section were also satisfied with minor exceptions, as shown in Table 20-2.

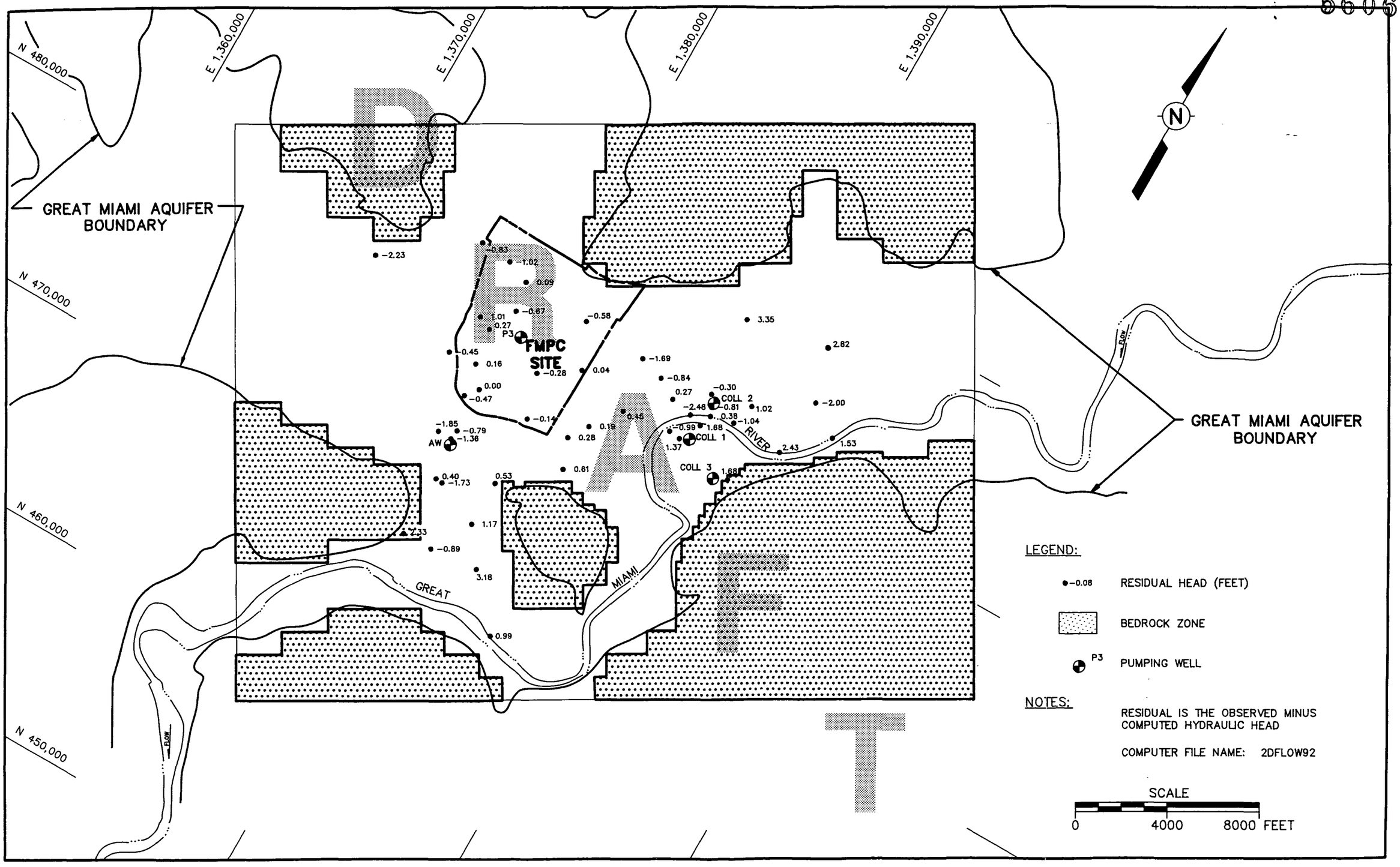

FIGURE 20-4

REFINED 2-DIMENSIONAL FLOW MODEL
RESIDUAL HEADS FOR CALIBRATED RUN
APRIL 1986 DATA

5606

TABLE 20-2

STATISTICAL TESTS FOR FLOW MODEL CALIBRATION

Statistical Measure	Predetermined Criteria	2-D Flow 1986 (2DFLOW)	2-D Flow 1988 (2DFLOW)	3-D Flow 1986 (3DFLOW)
Mean Residual (Observed Minus Computed Head) (ft)	$\pm$ 0.5	0.083	0.088	0.326
Mean of The Absolute Residuals (ft)	$\leq$ 2	1.080	0.805	1.083
Standard Deviation of Differences (ft)	$\leq$ 3	0.20	0.138	1.365
Regression Coefficient Between Measured and Computed Values	1 $\pm$ 0.05	1.02	0.89	0.937
Nearest Neighbor Autocorrelation Unit Normal Deviate	< 1.90	0.636	3.493	0.673

20.1.4 Model Results

Model Results for April 1986 Data

The distribution of hydraulic conductivities resulting from the model calibration is shown in Figure 20-5. The hydraulic conductivities in the Shandon area and in an area near the New Baltimore Outlet south of the FMPC were reduced to 30 ft/day (1.0×10^{-2} cm/s). These areas are known to contain more clay interbeds in the aquifer, so this reduction not only benefited the calibration but is defendable in relation to observed conditions.

Infiltration rates are shown in Figure 20-6. Changes were not needed for calibration. The river leakage factor was increased to 0.5 day^{-1} for all river cells. This change was needed to improve the calibration near the collector wells. Some hydraulic heads at external boundaries of the model were modified to provide a better calibration. Ranges of boundary heads in the Great Miami Aquifer are shown in Figure 20-6.

The simulated groundwater flow system that resulted from the model calibration is shown in Figure 20-3. Features of the flow system include the following:

- Groundwater flowing beneath the FMPC comes from the Shandon Tributary and the Dry Fork of the Whitewater River. Groundwater leaving the northern part of the FMPC flows eastward toward the collector wells.
- Groundwater leaving the southwestern corner of the FMPC moves toward the Great Miami River through the Paddys Run Outlet, where a high hydraulic gradient and high velocities result from the constriction of the flow within the bedrock trough.
- Other groundwater flow from the FMPC moves toward the Great Miami River south of the collector wells.
- Potentiometric contours swing around the collector wells, indicating that they have a pronounced effect on the flow system.
- The effects of the FMPC water well and the Albright & Wilson Co. wells on the potentiometric contours are small and are not discernible on this map.
- A high hydraulic gradient is indicated in the Shandon Tributary to compensate for the low hydraulic conductivity where the aquifer material is relatively high in clay content as compared to the main valley.

Model Results for May 1988 Data

The model was also tested against the May 1988 water level data, which includes a large number of water level measurements made during a severe drought. The use of these data provided a test of model performance when hydrologic input is significantly different from that of the calibration

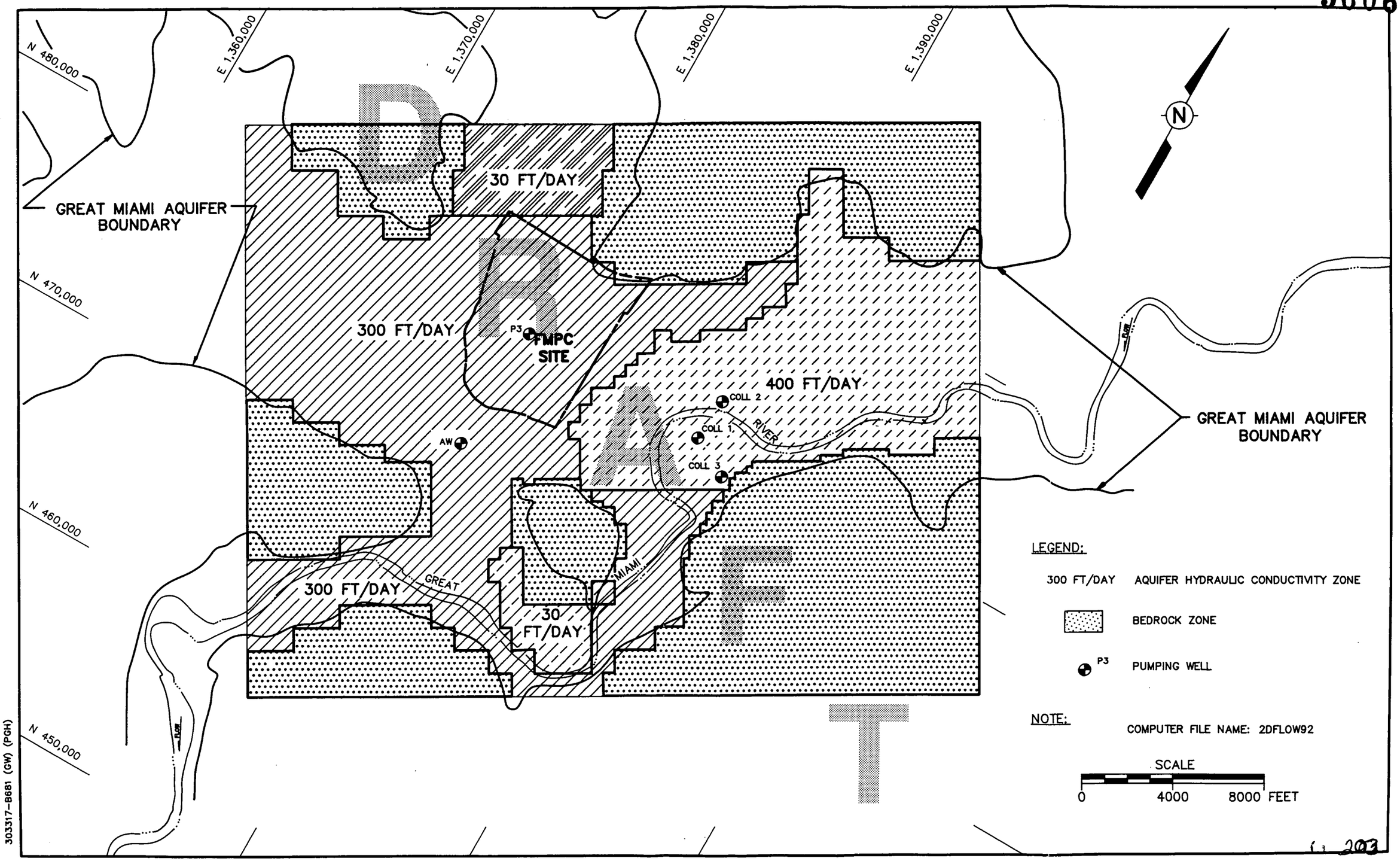

FIGURE 20-5
REFINED 2-DIMENSIONAL FLOW MODEL
HYDRAULIC CONDUCTIVITY ZONES

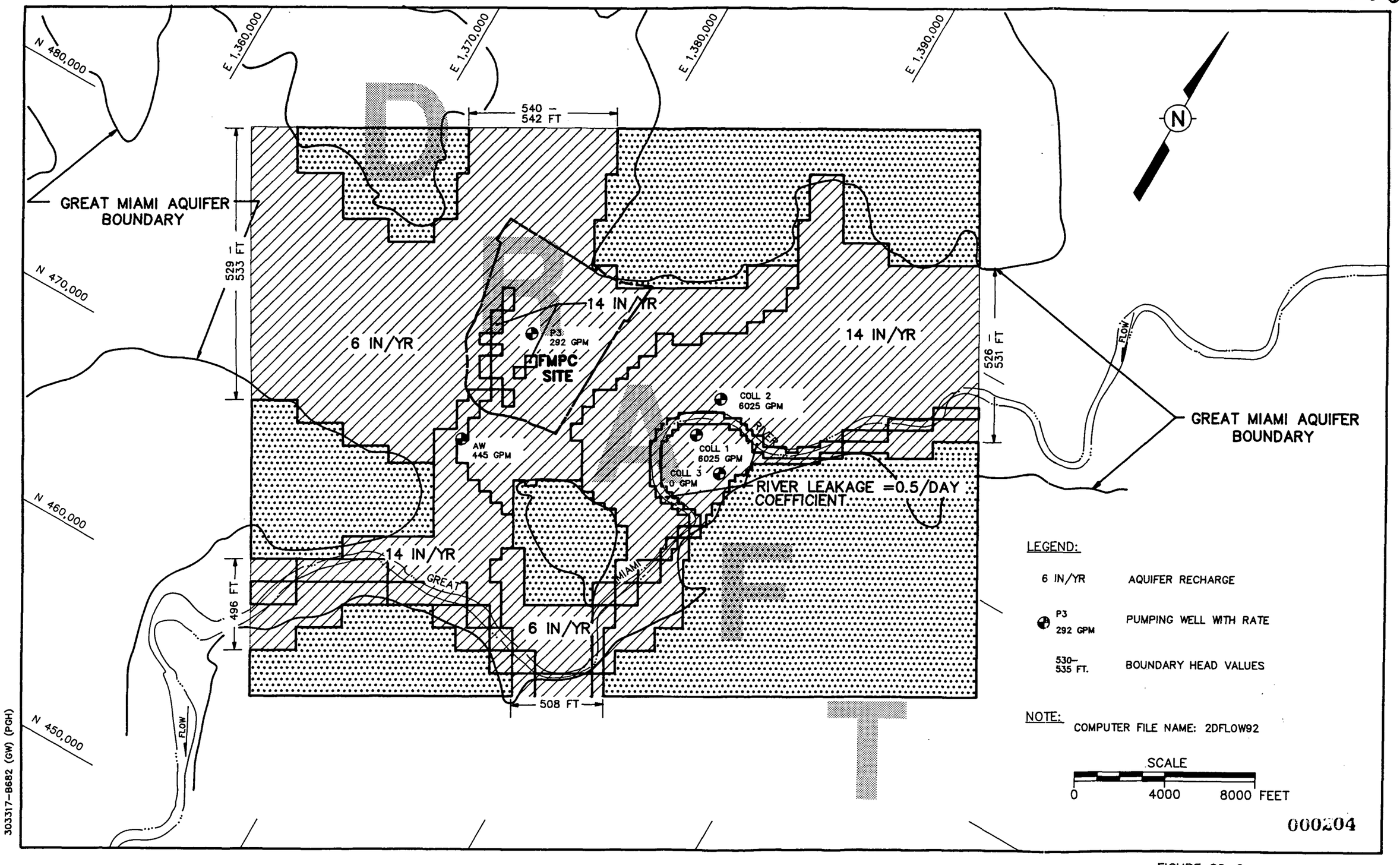

FIGURE 20–6

REFINED 2–DIMENSIONAL FLOW MODEL RECHARGE ZONES, PUMPING RATES, AND BOUNDARY HEADS

conditions. A large amount of water level data were available for this month. The hydrograph for Well 02E shown in Figure 12-4 demonstrates that the groundwater level was abnormally low compared to water levels in the preceding years. These low water levels are clearly not representative of long-term conditions.

The model was successful in matching the 1988 drought data without changing the values of hydraulic conductivities or any other basic parameters. Matching the drought conditions only involved reducing infiltration rates, the Great Miami River stage, and external boundary heads. Infiltration rates were reduced to 10 in/yr where they had been 14 in/yr and to 4 in/yr where they had been 6 in/yr. The Great Miami River stage was reduced 3.6 feet in accordance with the actual difference in river stage between April 1986 and May 1988; and external boundary heads were reduced 3.5 feet in accordance with the average reduction in water levels in monitoring wells compared with the April 1986 data. The pumping rates of the collector wells were also changed to conform to reported pumping rates. The combined pumping rate of the collector wells (1 and 2) was increased from 12,052 gpm (17.35 Mgd) in the 1986 period to 12,844 gpm (18.55 Mgd) in the 1988 period. The change was an increase of 832 gpm (1.2 Mgd), or about 7 percent.

The degree of calibration is indicated by the statistical results presented in Table 20-2. The criteria related to the size of the residuals are satisfied. Other results indicate clumping of positive and negative residuals. The clumping is considered acceptable in this case because the residuals are small, and further refinement of the parameter values is not necessary for verification of the model. This model of drought conditions will not be used in future applications.

These results show that the model is capable of adequately simulating two different sets of hydrologic conditions, while leaving parameters related to unchanging geologic conditions constant. This verification step, which demonstrated model performance under conditions significantly different than those used in the model calibration, provides additional grounds for confidence in model applications that involve changing hydrologic input.

20.1.5 Sensitivity Analysis

The sensitivity analysis applied to 2DFLOW investigated the effects of varying the input values of significant parameters. These parameters included hydraulic conductivity, recharge rate, and river leakage factor. The purpose of the sensitivity analysis was to determine the effect of parameter

uncertainty on the representativeness of the calibrated model. Consequently, the degree to which the parameters were varied was based on the degree of uncertainty associated with each parameter.

The results of the sensitivity analysis applied to the model of 1986 conditions are shown in Table 20-3. They are summarized as follows:

- Variation of hydraulic conductivity within the range of likely regional values has only a minor effect on model calibration. Consequently, even though varying hydraulic conductivity may be used to refine model calibration, further changes in hydraulic conductivity would not result in a significantly better representation of the groundwater system.

- Variation of recharge rates within the range of likely values also has only a minor effect on model calibration. Consequently, further changes in the rates would not result in a significantly better representation of the groundwater system.

- The range of likely values of river leakage factor is difficult to establish from published information. Hydraulic heads calculated by the model are shown to be very sensitive to this factor when the value assigned is low. In addition, because model results are not very sensitive to variation of other parameters within reasonable ranges, there is little freedom in the selection of the river leakage factor. As a result, the model itself can be used to establish a reasonable range of leakage values since values outside of this range produce meaningless results. The sensitivity of this particular model to the river leakage factor is caused by the effect of the collector wells, which obtain much of their water by induced infiltration from the river. When the induced infiltration is reduced by decreasing the leakage factor, drawdown due to pumping the wells increases greatly. Since the wells have been operating for more than 35 years, the river bed clogging should not increase significantly in the future. Therefore, the river leakage factor considered to best represent the real system based on model results is not expected to change.

The results of the sensitivity analysis applied to the model of 1988 conditions are shown in Table 20-4. These results merely confirm that the model is sensitive to river leakage factor. It is also noteworthy that the model is sensitive to hydraulic heads assigned to the Great Miami Aquifer at external boundaries. Consequently, if the model is used to simulate drought conditions or other extreme events, the external boundary heads must be changed in addition to recharge rates and river stage. In this regard, it should be noted that any differences between the assigned hydraulic heads at the boundaries and what may actually occur in the prototype (i.e., real system) have little effect on calibration results in those parts of the model area of principal concern in subsequent applications. These parts are the FMPC area and the areas hydraulically downgradient from the FMPC, extending southward from the FMPC to the Great Miami River and eastward from the FMPC to the collector wells and the Great Miami River.

TABLE 20-3

RESULTS OF THE SENSITIVITY ANALYSIS
FOR THE TWO-DIMENSIONAL FLOW MODEL
BASED ON APRIL 1986 DATA
UTILIZING RUN 2DFLOW

Sensitivity Run Number	Parameter	2DFLOW Value	Sensitivity Run Value	Sensitivity Run[a] Statistics: Mean Residual (ft)	Sensitivity Run[a] Statistics: Mean Absolute Residual (ft)
1	Hydraulic Conductivity Zone I (ft/day)	400	300	0.174	1.175
2	Hydraulic Conductivity Zone I (ft/day)	400	600	0.023	1.099
3	Hydraulic Conductivity Zone III (ft/day)	300	200	0.466	1.127
4	Hydraulic Conductivity Zone VI (ft/day)	300	200	-0.447	1.36
5	River Bed Leakage Factor (day^{-1})	0.5	0.05	3.694	3.839
6	River Bed Leakage Factor (day^{-1})	0.5	0.25	0.745	1.240
7	River Bed Leakage Factor (day^{-1})	0.5	1.0	-0.284	1.186
8	River Bed Leakage Factor (day^{-1})	0.5	5.0	-0.638	1.416
9	River Bed Leakage Factor (day^{-1})	0.5	50.0	-0.729	1.494

See footnotes at end of table.

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TABLE 20-3
(continued)

Sensitivity Run Number	Parameter	2DFLOW Value	Sensitivity Run Value	Sensitivity Run[a] Statistics: Mean Residual (ft)	Sensitivity Run[a] Statistics: Mean Absolute Residual (ft)
10	Recharge Rate Zone I (in/yr)	14	20		
10	Recharge Rate Zone III (in/yr)	6	8		
10	Recharge Rate Zone IV (in/yr)	6	8		
10	Recharge Rate Zone V (in/yr)	6	8		
10	Recharge Rate Zone VI (in/yr)	14	20		
10				-0.434	1.301
11	Recharge Rate Zone I (in/yr)	14	10		
11	Recharge Rate Zone III (in/yr)	6	4		
11	Recharge Rate Zone IV (in/yr)	6	4		
11	Recharge Rate Zone V (in/yr)	6	4		
11	Recharge Rate Zone VI (in/yr)	14	10		
11				0.534	1.117
		2DFLOW STATISTICS:		0.100	1.077

[a] Mean residual is the mean of the differences between observed heads and calculated heads. Mean absolute residual is the mean of the absolute differences between observed heads and calculated heads.

TABLE 20-4

RESULTS OF THE SENSITIVITY ANALYSIS FOR THE TWO-DIMENSIONAL FLOW MODEL BASED ON MAY 1988 DATA UTILIZING RUN 2DFLOW

Sensitivity Run Number	Parameter	2DFLOW Value	Sensitivity Run Value	Sensitivity Run[a] Statistics: Mean Residual (ft)	Sensitivity Run[a] Statistics: Mean Absolute Residual (ft)
1	River Bed Leakage Factor	0.5	0.25	0.851	0.949
2	River Bed Leakage Factor	0.5	0.375	0.357	0.759
3	River Bed Leakage Factor	0.5	0.625	-0.082	0.872
4	River Bed Leakage Factor	0.5	0.750	-0.200	0.923
5	River Bed Leakage Factor	0.5	1.0	-0.354	1.005
		2DFLOW STATISTICS:		0.088	0.805

[a]Mean Residual is the mean of the differences between observed heads and calculated heads. Mean Absolute Residual is the mean of the absolute differences between observed heads and calculated heads.

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20.2 THREE-DIMENSIONAL FLOW MODEL

20.2.1 Extension of Input Data

The three-dimensional flow model (3DFLOW) was developed by using the 2DFLOW grid and separating the vertical extent of the aquifer into five distinct layers. The layers were designed in accordance with the following guidelines:

- Simulate the hydrologic effects of a clay interbed that is present beneath much of the FMPC property.
- Provide sufficient layers for realistic three-dimensional simulation of contaminant plumes originating from sources at the ground surface.
- Avoid causing pumping wells and monitoring wells to be screened in more than one layer. While the SWIFT III code can handle pumping wells that have screened intervals in more than one layer, model construction is slightly easier and particle tracking (Section 20.2.5.1) works better if each well is screened in one layer. Calibration is also simpler if each monitoring well is assigned to a single layer.

The layers were developed through the use of 12 intersecting cross sections, described in Section 11.5, taking care to maintain consistency across adjacent sections. The result was layers of fairly consistent thickness and elevation throughout the model, but with some variation in layer thicknesses and elevations resulting from the above guidelines. The elevations of the base of each layer were then interpolated between cross sections and entered into the model input data.

Figure 20-7 shows the locations of Cross Sections Q-Q' and R-R'. The layers created for the model are shown in Figures 20-8 and 20-9. The top layer, Layer 1, represents the upper part of the Great Miami Aquifer that contains the water table. Layer 3 represents the clay interbed that is present beneath the FMPC. Where the clay is not present, Layer 3 is treated as sand and gravel material of the Great Miami Aquifer and assigned an appropriate hydraulic conductivity. Layer 2 represents the lower part of the upper aquifer, which is immediately above the clay interbed. Layers 4 and 5 divide the lower aquifer into two parts to realistically simulate the part of the aquifer serving the pumping wells, and to provide for better simulation of vertical contaminant dispersion.

The layout of grid rows and columns used in 2DFLOW (Figure 20-1) was also assigned to 3DFLOW. Values for the Great Miami River stage, water well pumping rates, boundary heads, elevation of the bedrock surface, and hydraulic conductivity of the bedrock were imported from the calibrated 2DFLOW model. The combined pumping rate for the Albright & Wilson Co. wells was

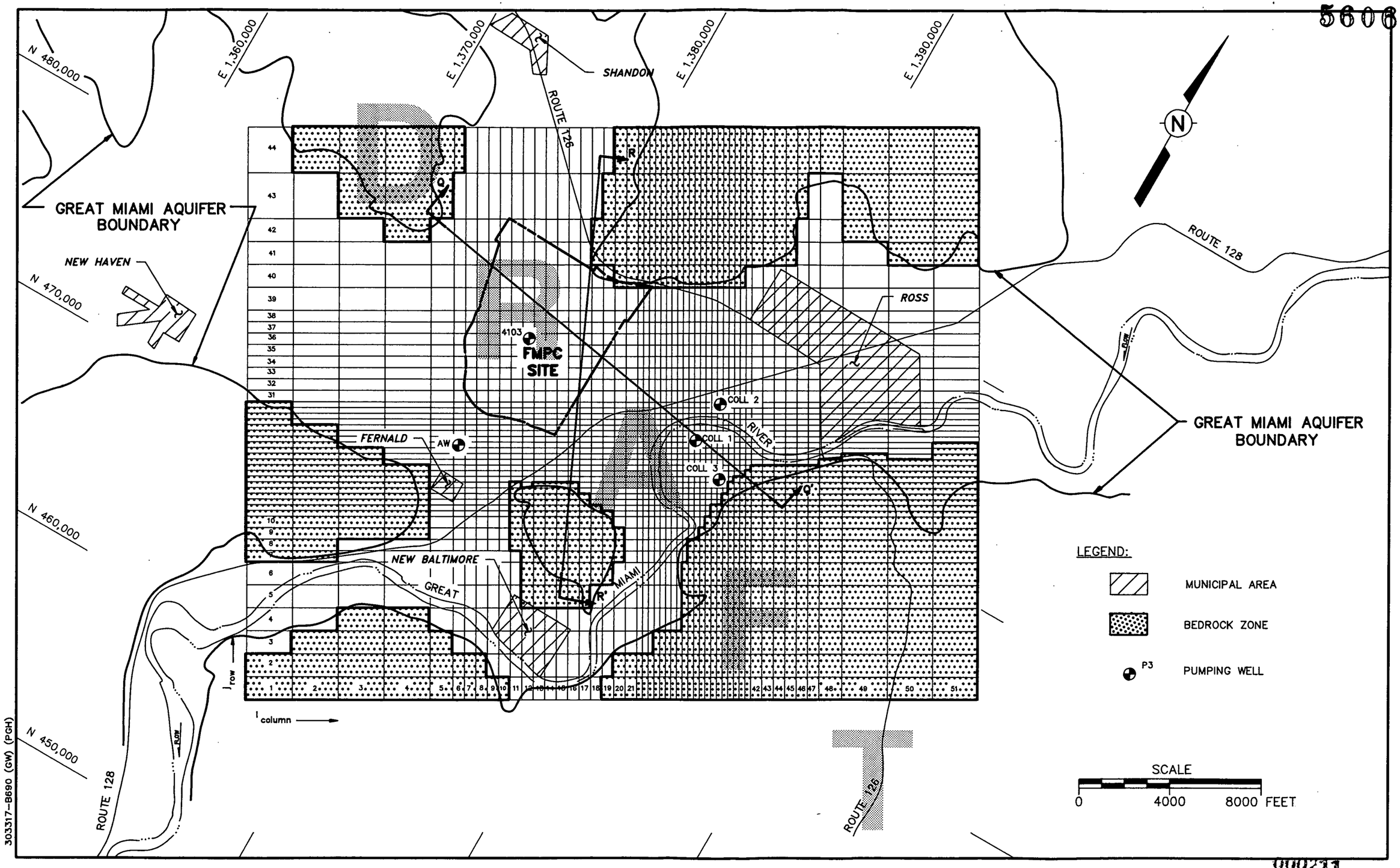
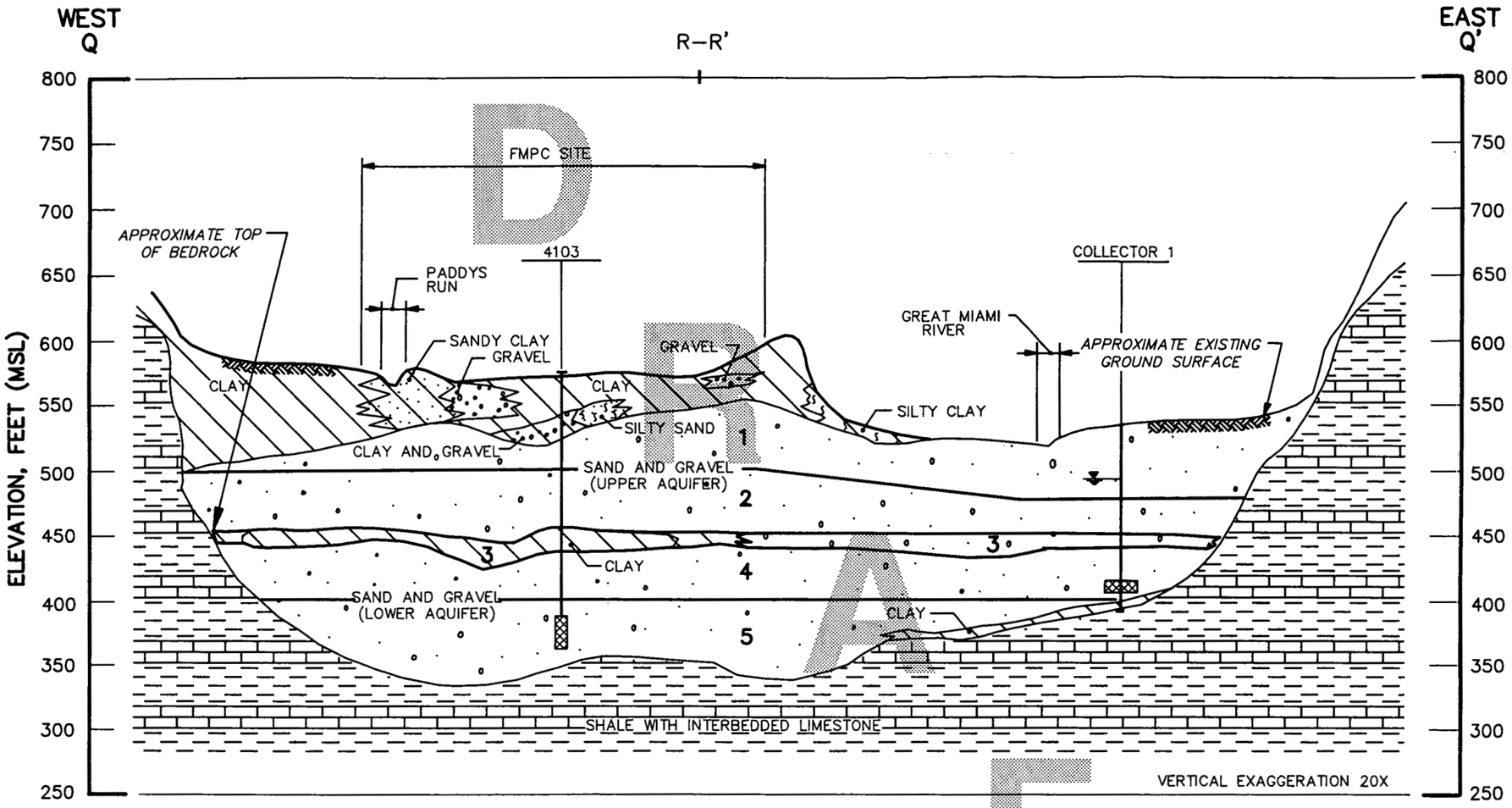


FIGURE 20-7
SWIFT III MODEL GRID
FOR RI/FS MODEL AREA
WITH CROSS SECTION LOCATIONS



THE BORING LOGS AND RELATED INFORMATION DEPICT SUBSURFACE CONDITIONS ONLY AT THE SPECIFIC LOCATIONS AND DATES INDICATED. SOIL CONDITIONS AND WATER LEVELS AT OTHER LOCATIONS MAY DIFFER FROM CONDITIONS OCCURRING AT THESE BORING LOCATIONS. ALSO THE PASSAGE OF TIME MAY RESULT IN A CHANGE IN THE CONDITIONS AT THESE BORING LOCATIONS.

THE DEPTH AND THICKNESS OF THE SUBSURFACE STRATA INDICATED ON THE SECTIONS WERE GENERALIZED FROM AND INTERPOLATED BETWEEN THE TEST BORINGS. INFORMATION ON ACTUAL SUBSURFACE CONDITIONS EXISTS ONLY AT THE LOCATION OF THE TEST BORINGS AND IT IS POSSIBLE THAT SUBSURFACE CONDITIONS BETWEEN THE TEST BORINGS MAY VARY FROM THOSE INDICATED.

VERTICAL SCALE

0 100 200 FEET

HORIZONTAL SCALE

0 2000 4000 FEET

LEGEND:

WATER LEVELS (MEASURED 3/27/86 THROUGH 4/11/86)

2 HYDROGEOLOGIC LAYER USED IN SWIFT III MODELS

OPEN OR SCREENED INTERVAL

NOTE:

FOR LOCATION OF CROSS SECTION Q–Q', SEE FIGURE 20–7

REFERENCES:

MODIFIED FROM U.S.G.S. PROFESSIONAL PAPER NO. 605–A USING AVAILABLE OHIO STATE WATER WELL RECORDS, SOWC WATER WELL DATA, DAMES & MOORE'S "FEED MATERIALS PRODUCTION CENTER GROUNDWATER STUDY, TASK C REPORT" FOR THE DOE (1985), AND IT (1986) RI/FS BORING LOGS.

CROSS SECTION Q–Q'

(LOOKING NORTH)

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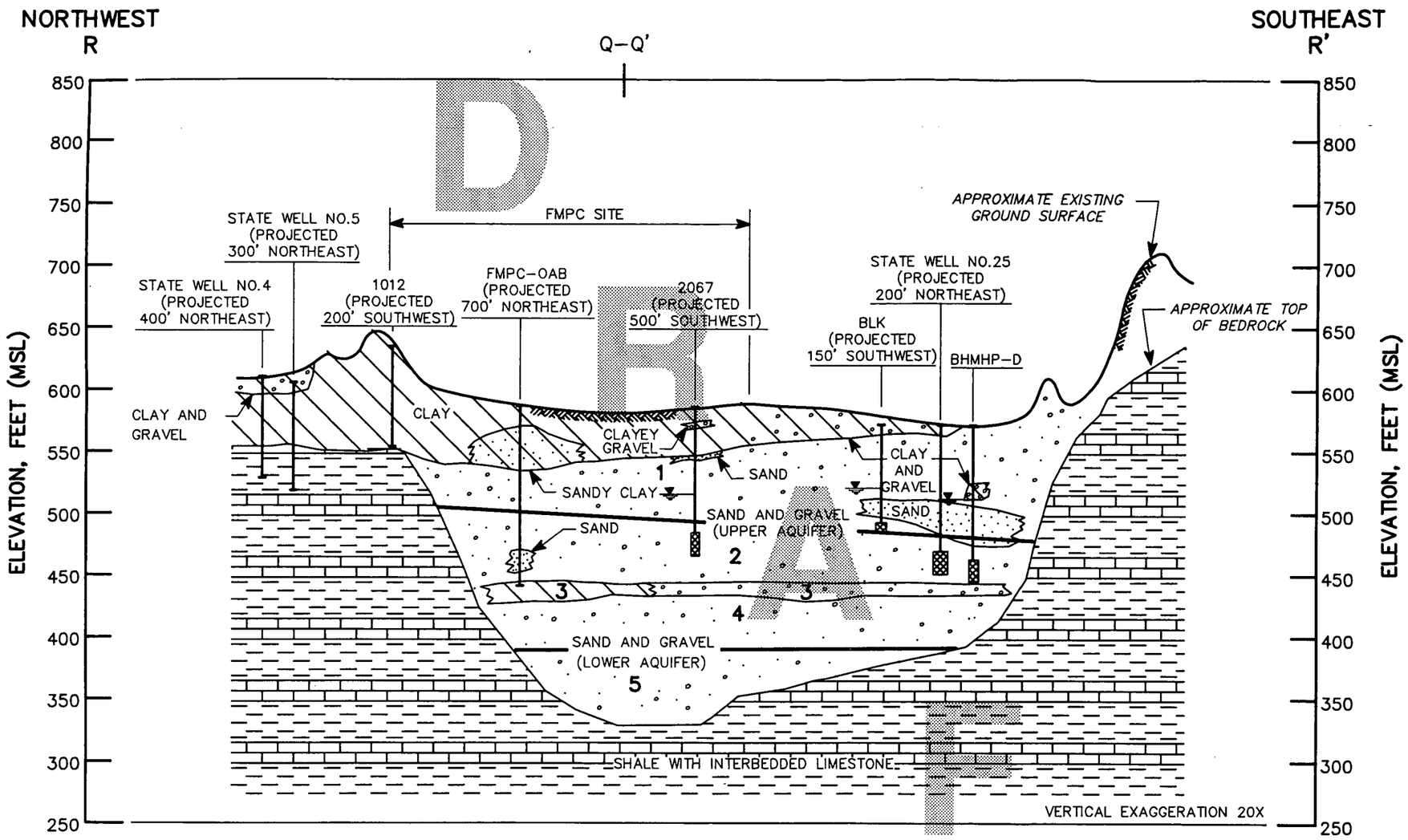
303317–B544 (GW) (PGH)

FIGURE 20–8
GEOLOGIC CROSS SECTION Q–Q'
SHOWING MODEL LAYERS



THE BORING LOGS AND RELATED INFORMATION DEPICT SUBSURFACE CONDITIONS ONLY AT THE SPECIFIC LOCATIONS AND DATES INDICATED. SOIL CONDITIONS AND WATER LEVELS AT OTHER LOCATIONS MAY DIFFER FROM CONDITIONS OCCURRING AT THESE BORING LOCATIONS. ALSO THE PASSAGE OF TIME MAY RESULT IN A CHANGE IN THE CONDITIONS AT THESE BORING LOCATIONS.

THE DEPTH AND THICKNESS OF THE SUBSURFACE STRATA INDICATED ON THE SECTIONS WERE GENERALIZED FROM AND INTERPOLATED BETWEEN THE TEST BORINGS. INFORMATION ON ACTUAL SUBSURFACE CONDITIONS EXISTS ONLY AT THE LOCATION OF THE TEST BORINGS AND IT IS POSSIBLE THAT SUBSURFACE CONDITIONS BETWEEN THE TEST BORINGS MAY VARY FROM THOSE INDICATED.

VERTICAL SCALE

0 100 200 FEET

HORIZONTAL SCALE

0 2000 4000 FEET

LEGEND:

WATER LEVELS (MEASURED 3/27/86 THROUGH 4/11/86)

2 HYDROGEOLOGIC LAYER USED IN SWIFT III MODELS

OPEN OR SCREENED INTERVAL

NOTE:

FOR LOCATION OF CROSS SECTION R–R', SEE FIGURE 20–7

REFERENCES:

MODIFIED FROM U.S.G.S. PROFESSIONAL PAPER NO. 605–A USING AVAILABLE OHIO STATE WATER WELL RECORDS, SOWC WATER WELL DATA, DAMES & MOORE'S "FEED MATERIALS PRODUCTION CENTER GROUNDWATER STUDY, TASK C REPORT" FOR THE DOE (1985), AND IT (1986) RI/FS BORING LOGS.

CROSS SECTION R–R'

(LOOKING NORTHEAST)

000213

303317–B542 (GW) (PGH)

FIGURE 20–9
GEOLOGIC CROSS SECTION R–R'
SHOWING MODEL LAYERS

changed from 100 gpm to 445 gpm to 225 gpm during the course of the modeling as new information was received. The value used for final calibration was 225 gpm. Initial values for horizontal hydraulic conductivity, recharge rates, and river leakage factor were also derived from 2DFLOW; however, hydraulic conductivity and recharge rates were subsequently changed, as necessary, during 3DFLOW calibration.

The clay interbed was given an initial horizontal hydraulic conductivity of 0.0003 ft/day (1×10^{-7} cm/s) and a vertical hydraulic conductivity of 0.00003 ft/day (1×10^{-8} cm/s). The value for horizontal hydraulic conductivity was estimated from published information on the range of hydraulic conductivity values for clay (Freeze and Cherry 1979, p. 29; Fetter 1980, p. 75; Todd 1963, p. 53). The initial ratio of vertical to horizontal hydraulic conductivity was assumed to be 1/10 throughout the model. This value resulted in adequate simulation of the vertical hydraulic gradient across the interbed. This ratio was based on typical published values and was subject to calibration and sensitivity analysis, as described below.

20.2.2 Model Calibration

Calibration of 3DFLOW was performed against the same April 1986 data that were used for 2DFLOW. The calibration procedure was also similar to the procedure followed for 2DFLOW. Each observed monitoring well water level was compared to the computed hydraulic head in the appropriate cell in the layer that represented the depth of the well or screened interval; that is, residuals were assigned to their locations in the appropriate layer of the model. Well completion data on Wells 3001 and 3126 were not available to ascertain the well depth or screened interval. The completion depths of these wells were assumed, therefore, to be at the midpoint between the base of the aquifer and the water table. The statistical criteria used to test for the adequacy of calibration were the same as those used for 2DFLOW, and the judgment of the engineers and geologists working on the project was still applied.

The distribution of hydraulic conductivities upon calibration of the 3DFLOW model are shown for the five layers in Figures 20-10 through 20-14. The figures show the increase in hydraulic conductivity and the decrease in width of the bedrock valley with depth.

In general, hydraulic conductivities were increased compared to those in 2DFLOW. While there is no direct way to determine why the increase was necessary for improved model performance, the features of 3DFLOW that would tend to affect hydraulic conductivity are (1) the effect of the clay interbed, which inhibited the efforts of recharge in the FMPC area and movement of water coming from the Shandon Tributary; and (2) hydraulic head losses due to vertical flow components that are

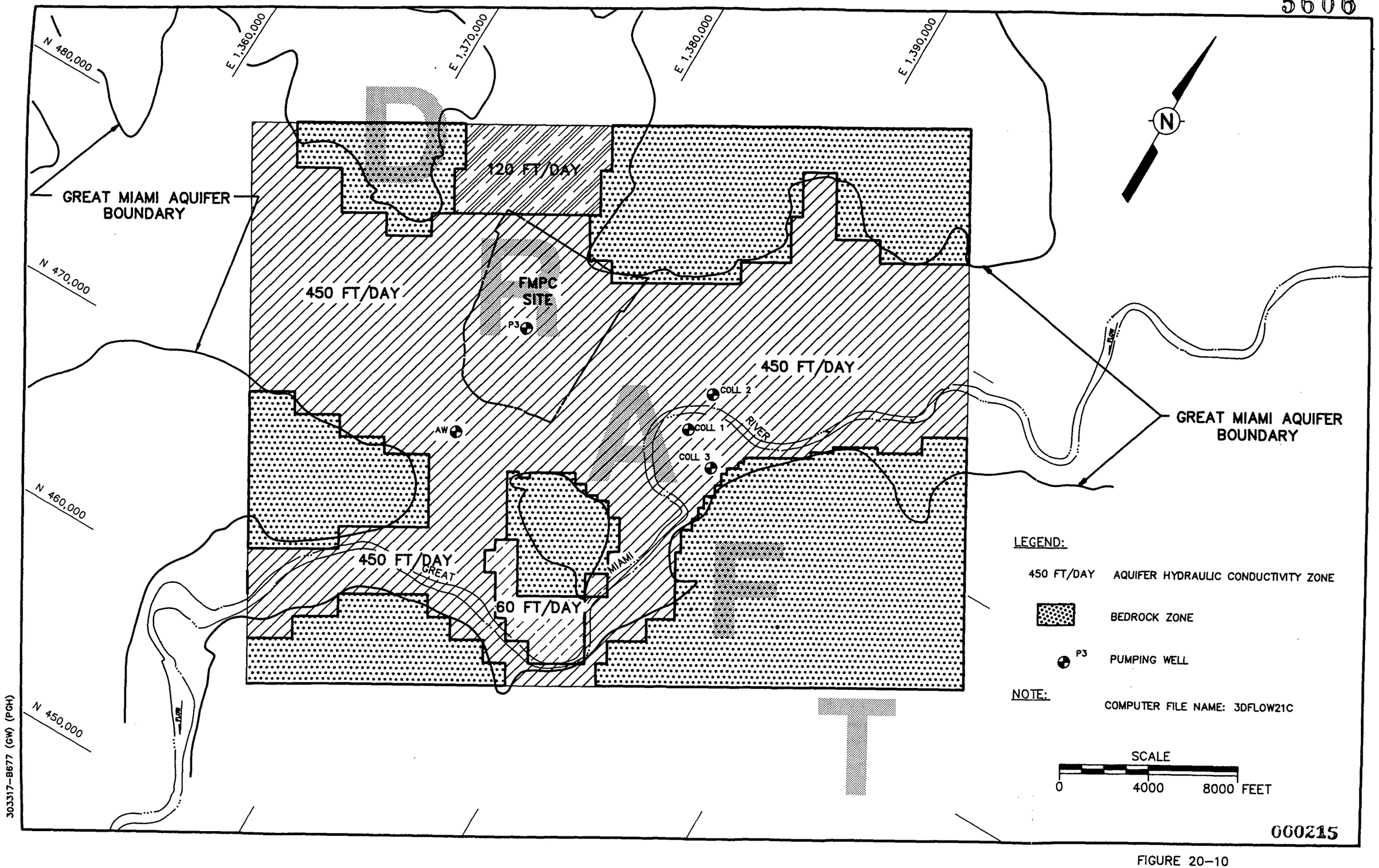


FIGURE 20–10
3–DIMENSIONAL FLOW MODEL
HYDRAULIC CONDUCTIVITY ZONES
LAYER 1

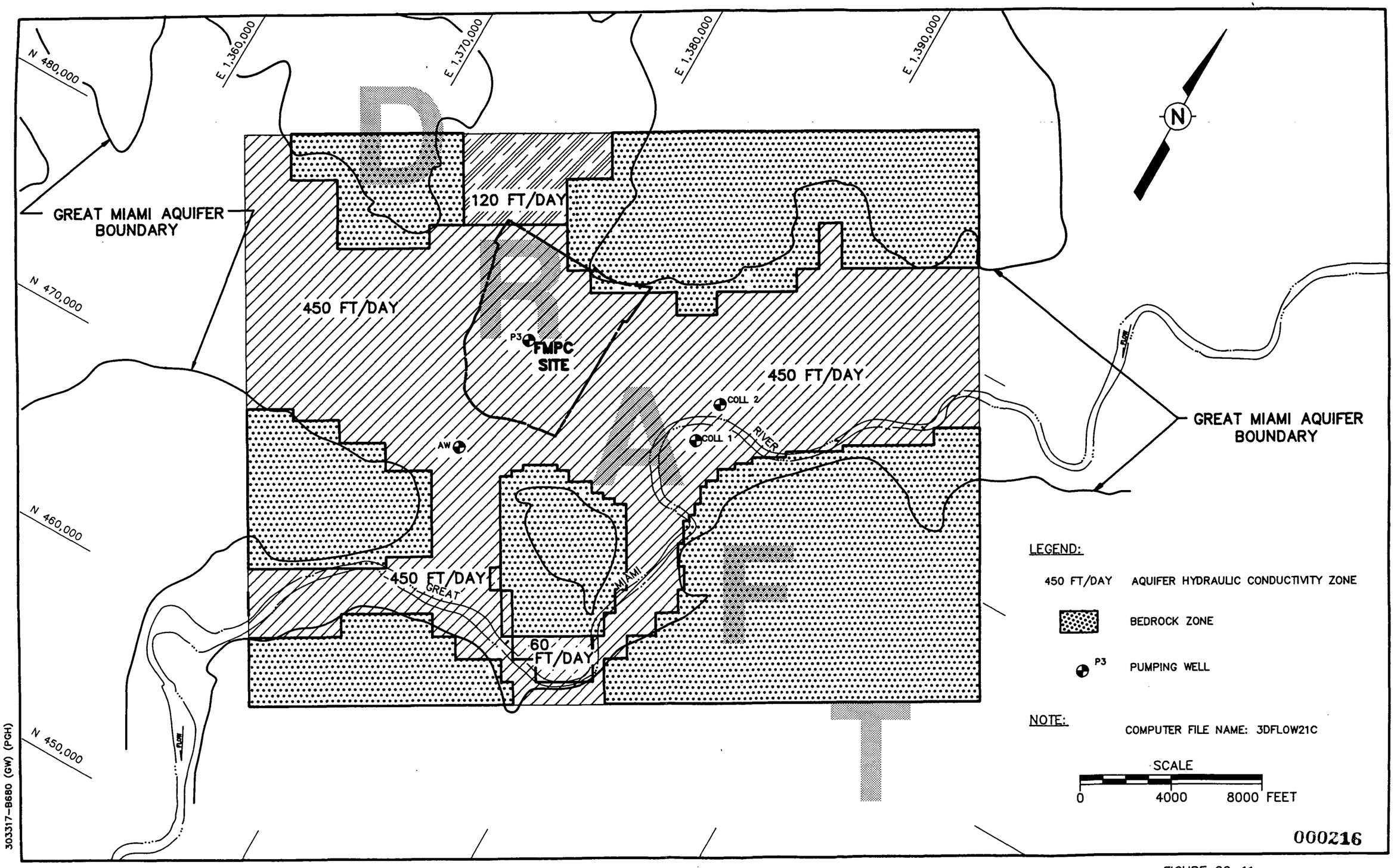


FIGURE 20–11
3–DIMENSIONAL FLOW MODEL
HYDRAULIC CONDUCTIVITY ZONES
LAYER 2

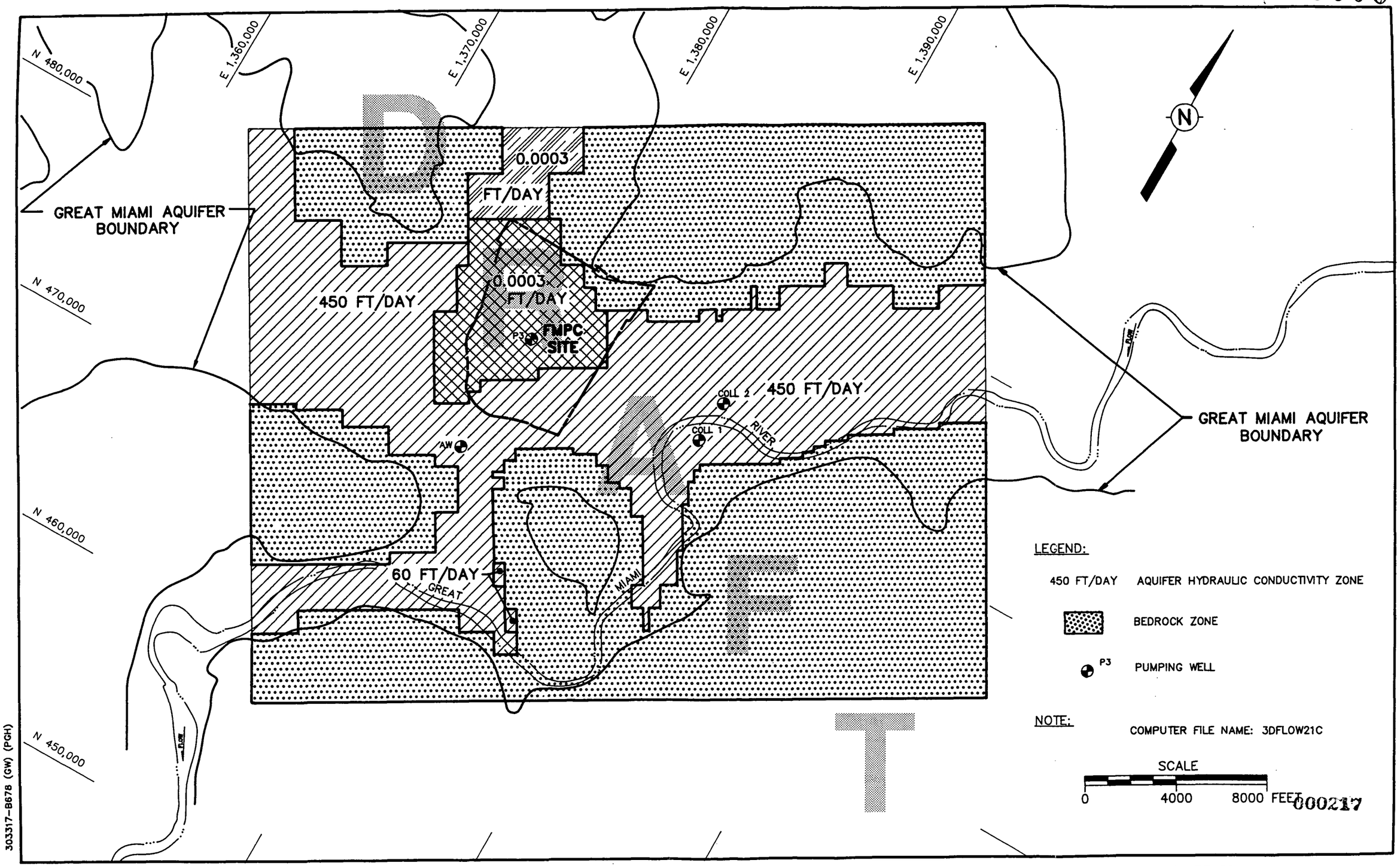


FIGURE 20-12
3-DIMENSIONAL FLOW MODEL
HYDRAULIC CONDUCTIVITY ZONES
LAYER 3

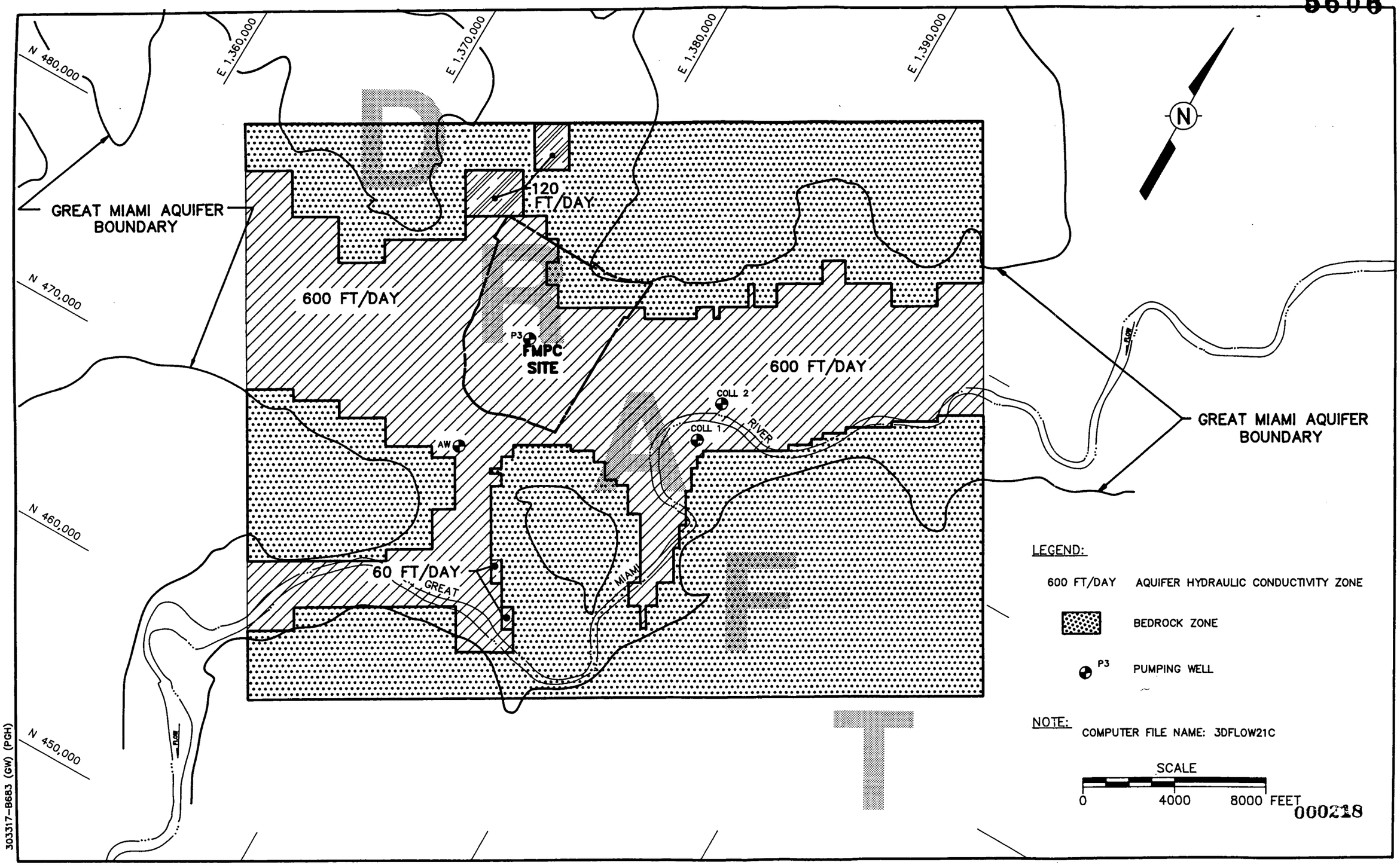


FIGURE 20–13

3–DIMENSIONAL FLOW MODEL
HYDRAULIC CONDUCTIVITY ZONES
LAYER 4

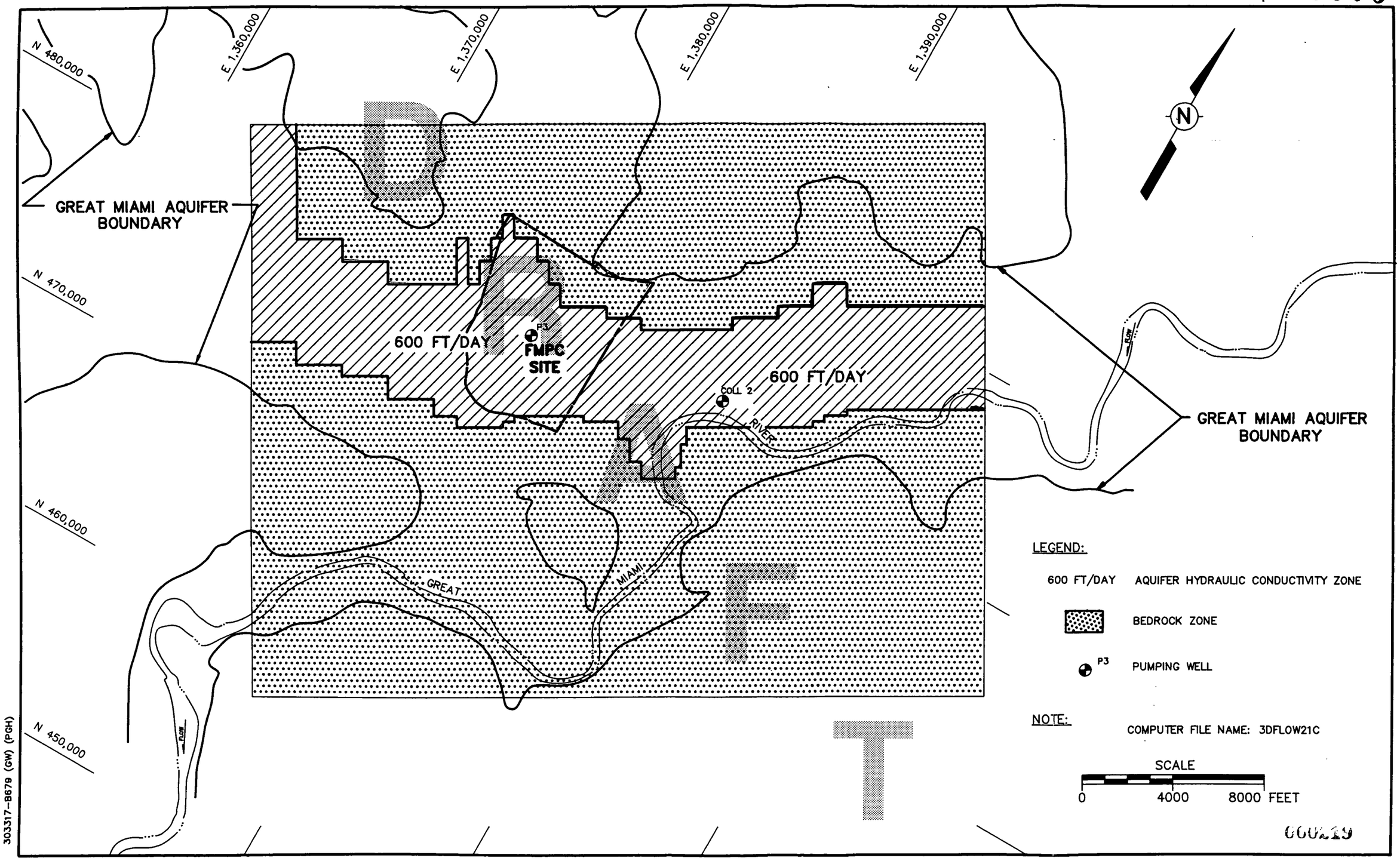


FIGURE 20–14
3–DIMENSIONAL FLOW MODEL
HYDRAULIC CONDUCTIVITY ZONES
LAYER 5

not represented in 2DFLOW. Hydraulic conductivities were increased more in the lower layers than in the upper layers. This redistribution helped avoid clustering of negative residuals at the FMPC site, which indicated that calculated hydraulic heads were too high. The distribution also helped to increase the vertical hydraulic gradient across the clay interbed and decrease the hydraulic head in the lower aquifer. The latter condition resulted in a better hydraulic connection between the lower aquifer and the collector wells and other downgradient discharge locations. Changes in hydraulic conductivities in the upper part of the Great Miami Aquifer produced small modifications in the hydraulic gradient, thereby effecting a better match between calculated and observed water levels within the FMPC site.

Recharge rates were also changed to help remove clustering of negative residuals across the FMPC site. The recharge rate over part of the site was reduced to 2 in/yr so that less water would have to drain from the aquifer above the clay interbed to maintain the hydraulic head values near observed levels. This reduction in simulated recharge rate is consistent with the industrial use of the site. Factors reducing recharge include water carried away from the site by storm sewers draining roofed and paved areas, precipitation falling on lined ponds, runoff and evaporation from roads, and increased runoff and evaporation from soils compacted by the industrial activities at the site. The recharge rates used in the calibrated model are shown in Figure 20-15.

Some small refinements were made where the north boundary of the model intersects the Shandon Tributary. Boundary heads were lowered by 0.5 foot and the bedrock configuration was modified to produce a more realistic valley cross section. These changes were in conjunction with changes in hydraulic conductivity in the tributary to refine the calibration in the northern part of the FMPC. The recharge heads elsewhere in the model remained at the same as those in 2DFLOW. The river leakage factor remained at 0.5 day^{-1}, unchanged from 2DFLOW.

The residual heads achieved by the calibration runs are indicated in Figures 20-16 through 20-18, which show the magnitude and distribution of the residuals. Further reduction of the residuals was not warranted. The predetermined statistical criteria described in Section 20.1.3 were satisfied, as shown in Table 20-2.

In addition, a water balance was calculated for the same area covered by the water balance for 2DFLOW. The calculated inflows and outflows are shown in Table 20-5. The inflow/outflow ratio is 0.9998, again within 0.02 percent of a perfect balance. This result provides a successful check on the accuracy of 3DFLOW in satisfying the conservation of mass across the model area.

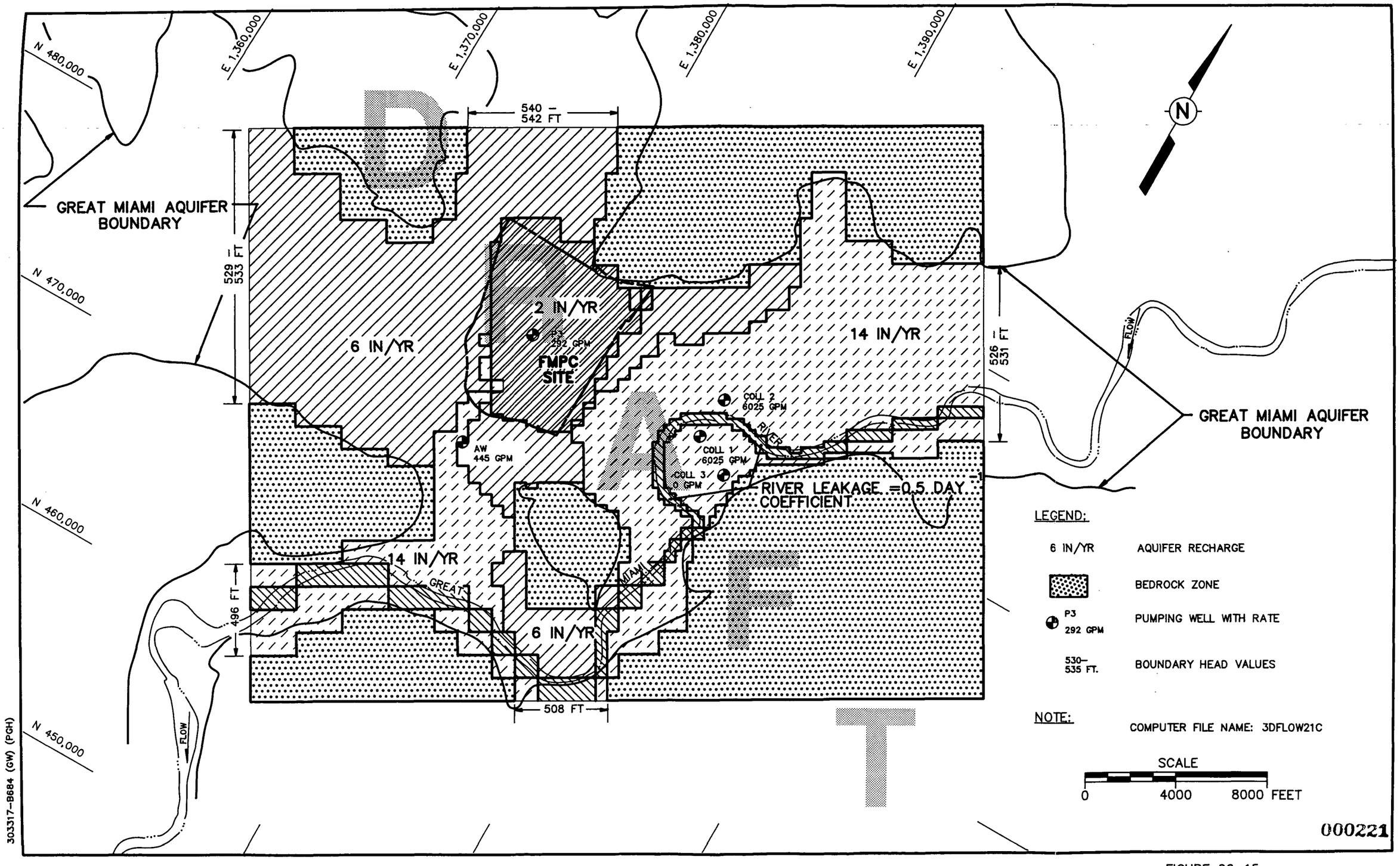


FIGURE 20–15

3–DIMENSIONAL FLOW MODEL RECHARGE ZONES, PUMPING RATES, AND BOUNDARY HEADS

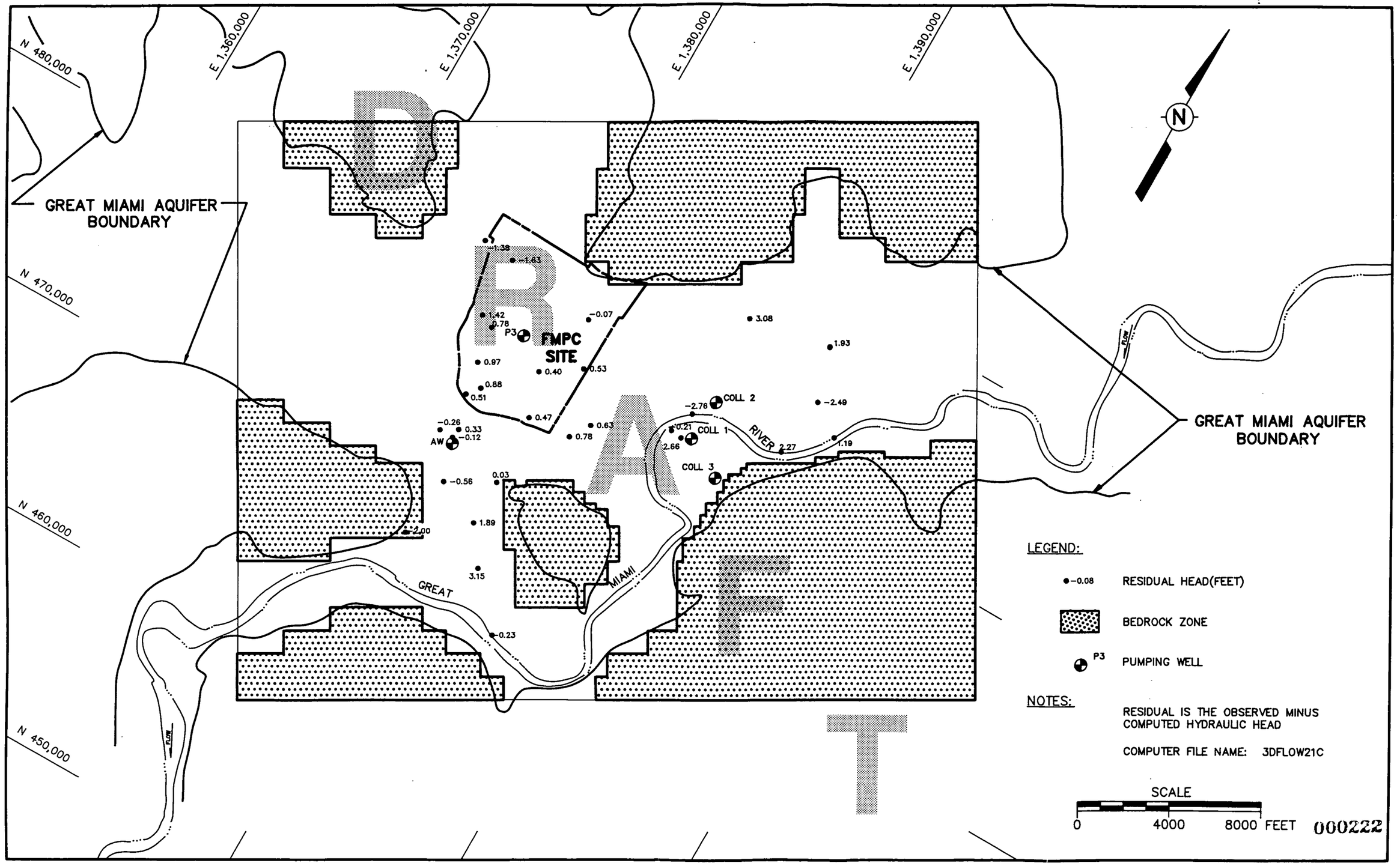


FIGURE 20-16
3-DIMENSIONAL FLOW MODEL
RESIDUAL HEADS FOR CALIBRATED RUN
APRIL 1986 DATA
LAYER 1

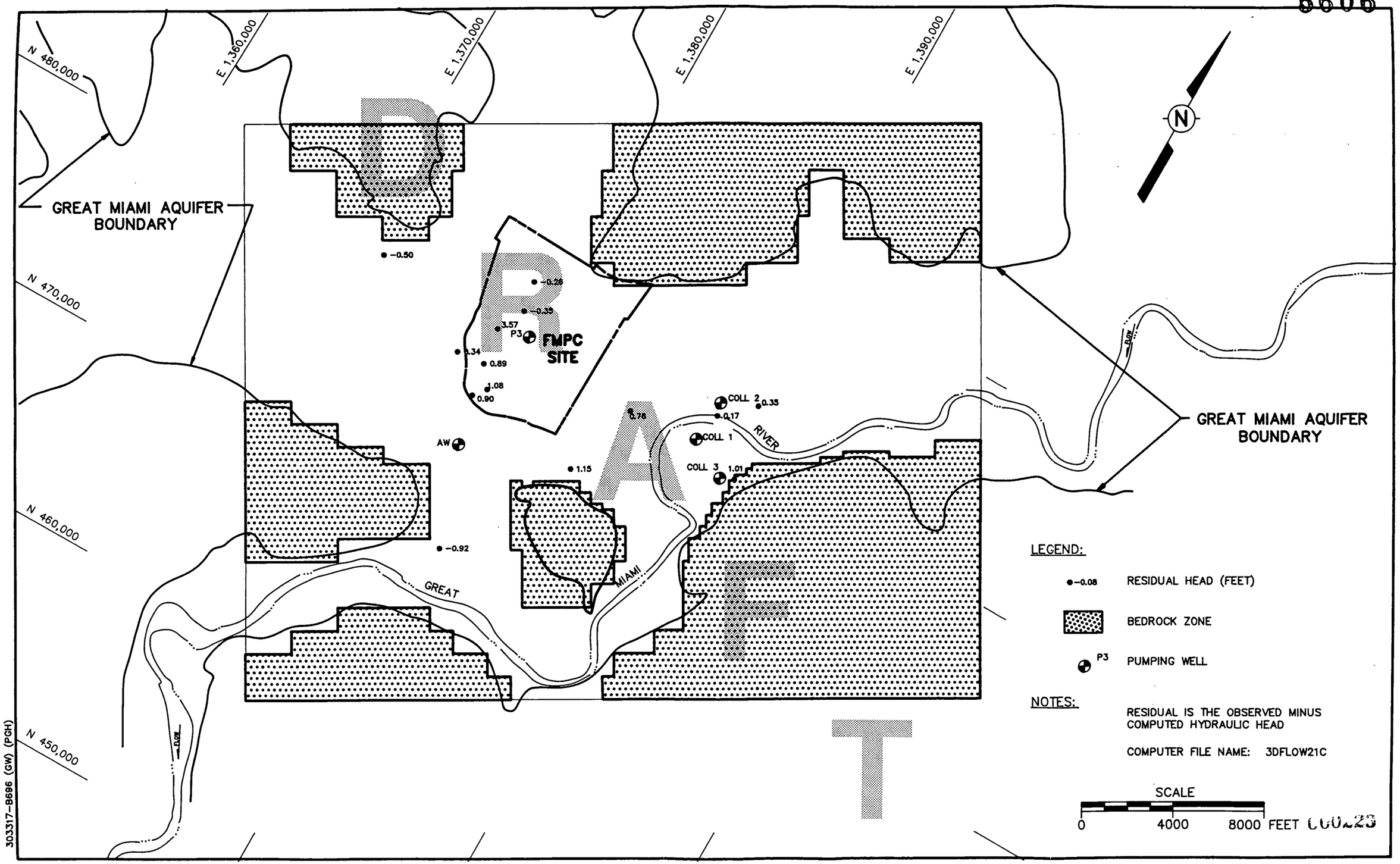


FIGURE 20–17
3–DIMENSIONAL FLOW MODEL
RESIDUAL HEADS FOR CALIBRATED RUN
APRIL 1986 DATA
LAYER 2

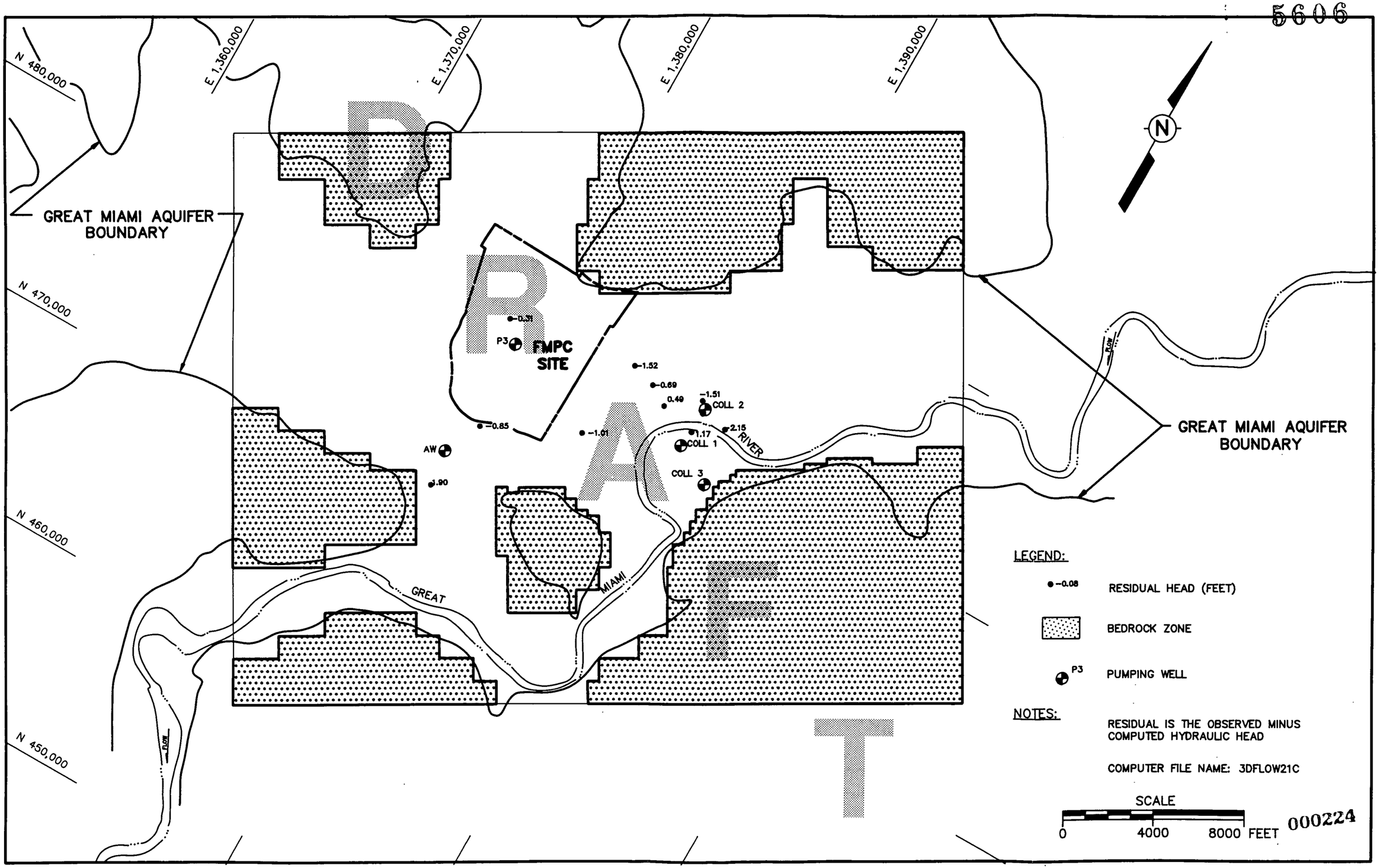


FIGURE 20-18
3-DIMENSIONAL FLOW MODEL
RESIDUAL HEADS FOR CALIBRATED RUN
APRIL 1986 DATA
LAYERS 3,4,5

TABLE 20-5

WATER BALANCE
IN THE VICINITY
OF THE SOUTHWESTERN OHIO WATER COMPANY COLLECTOR WELLS
FOR THE THREE-DIMENSIONAL FLOW MODEL

Inflow Locations	Inflow Rates (Cubic Feet Per Day)	(Mgd)
West Boundary, I = 21, J = 10-39	235,128	1.76
East Boundary, I = 50, J = 22-40	327,720	2.45
North Boundary, I = 46-49, J = 41	26,671	0.20
Great Miami River Leakage	1,693,089	12.66
Recharge	308,227	2.31
Total Inflow	2,590,835	19.38

Outflow Locations	Outflow Rates (Cubic Feet Per Day)	(Mgd)
South Boundary, I = 21-32, J = 10	101,327	0.76
Collector Wells	2,317,307	17.33
Discharge to Great Miami River	166,676	1.25
Total Outflow	2,585,310	19.34

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The model results were also compared to field data by checking computed versus observed vertical hydraulic gradients. Hydraulic heads measured at different depths in monitoring well clusters were compared to calculated heads at those depths. Seven suitable well clusters were available for this comparison. One was in the area of the clay interbed at the FMPC site. This cluster had a well completed above the clay layer at the depth of model Layer 2 and another well completed below the clay layer at the depth of Layer 5. The head difference measured in the field was 0.65 foot. The corresponding computed head difference was 1.04 feet. Given the greater complexity of the prototype system compared to model system, this result indicates that vertical gradients are being adequately represented. The other clusters were outside of the area of the clay interbed. For these clusters, measured head differences were very low because the vertical hydraulic conductivity of the aquifer is high and a large vertical gradient is not required to maintain vertical flow components. The maximum measured vertical head difference in these clusters was 0.38 foot. The computed head difference at that location was zero. This result may be due to the presence of some minor clay bed that would be impractical to simulate. Comparisons between observed and computed head differences were better at other clusters. In summary, the model is adequately representing vertical head differences, and further calibration to match local influences on vertical hydraulic gradient is not warranted.

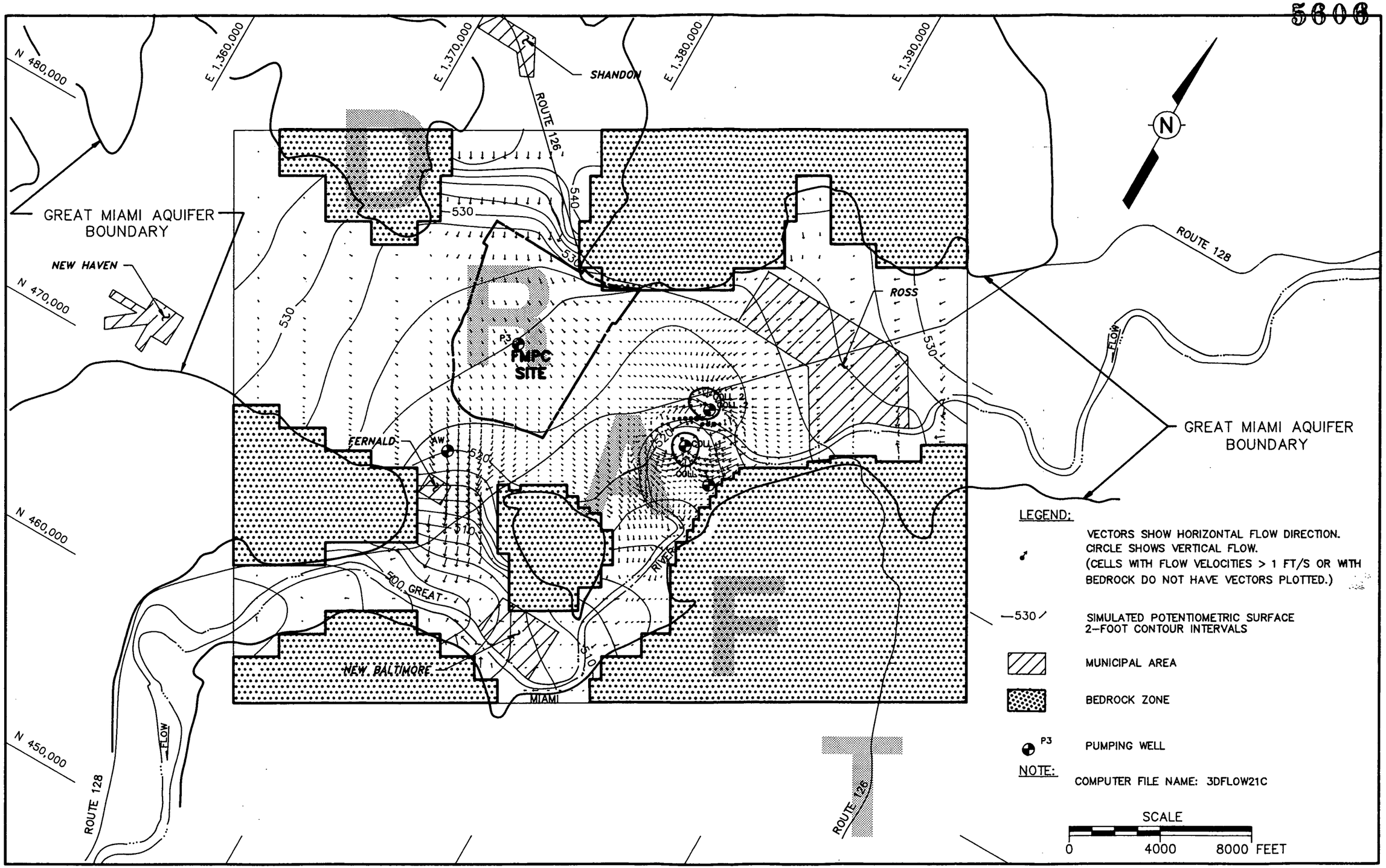
20.2.3 Model Results

The simulated groundwater flow system that resulted from the 3DFLOW model calibration is shown for Layers 1 through 5 in Figures 20-19 through 20-23, respectively. Some large vectors were automatically deleted from the vector maps to improve their appearance and legibility. Such anomalies occur in areas near the collector wells and in the Paddys Run Outlet. The circles on the tails of the vectors indicate the vertical velocity component. The sizes of the circles are proportional to the rates of vertical movement. Over much of the area, the vertical velocities are so low that the circles are not discernible. Features of the simulated flow system include the following:

- Most of the groundwater in the Great Miami Aquifer above the clay interbed at the FMPC comes from the Shandon Tributary, while the groundwater in the Great Miami Aquifer below the interbed comes from the main bedrock valley to the west.

- Groundwater flow directions above and below the clay interbed beneath the FMPC are different. For example, the direction of the horizontal component of flow in a representative cell in the Waste Storage Area differs by about 60 degrees between Layer 1 and Layer 4, which are above and below the clay interbed, respectively. Further south, the difference decreases, but it is as great as 20 degrees under much of the FMPC.

303317-B691 (GW) /01 100% BLUE



303317-B691 (GW) /01 (PGH)

FIGURE 20-19
3-DIMENSIONAL FLOW MODEL
CALIBRAED RUN, LAYER 1
FLOW VECTORS AND POTENTIOMETRIC SURFACE

303317-B698 (GW) 100% BLUE

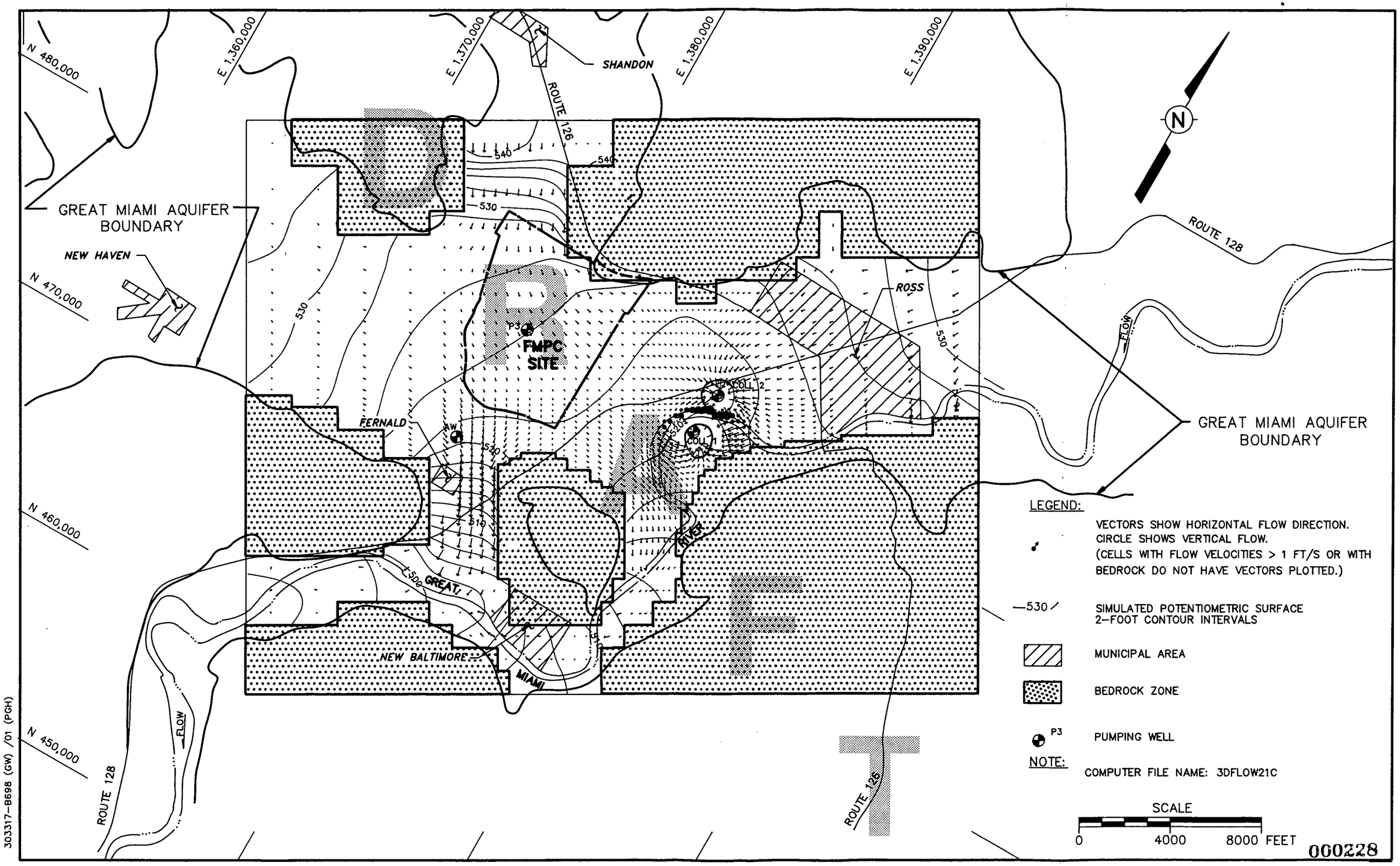


FIGURE 20-20
3-DIMENSIONAL FLOW MODEL
CALIBRAED RUN, LAYER 2
FLOW VECTORS AND POTENTIOMETRIC SURFACE

303317-B698 (GW) /01 (PGH)

303317-B693 (GW) /01 100% BLUE

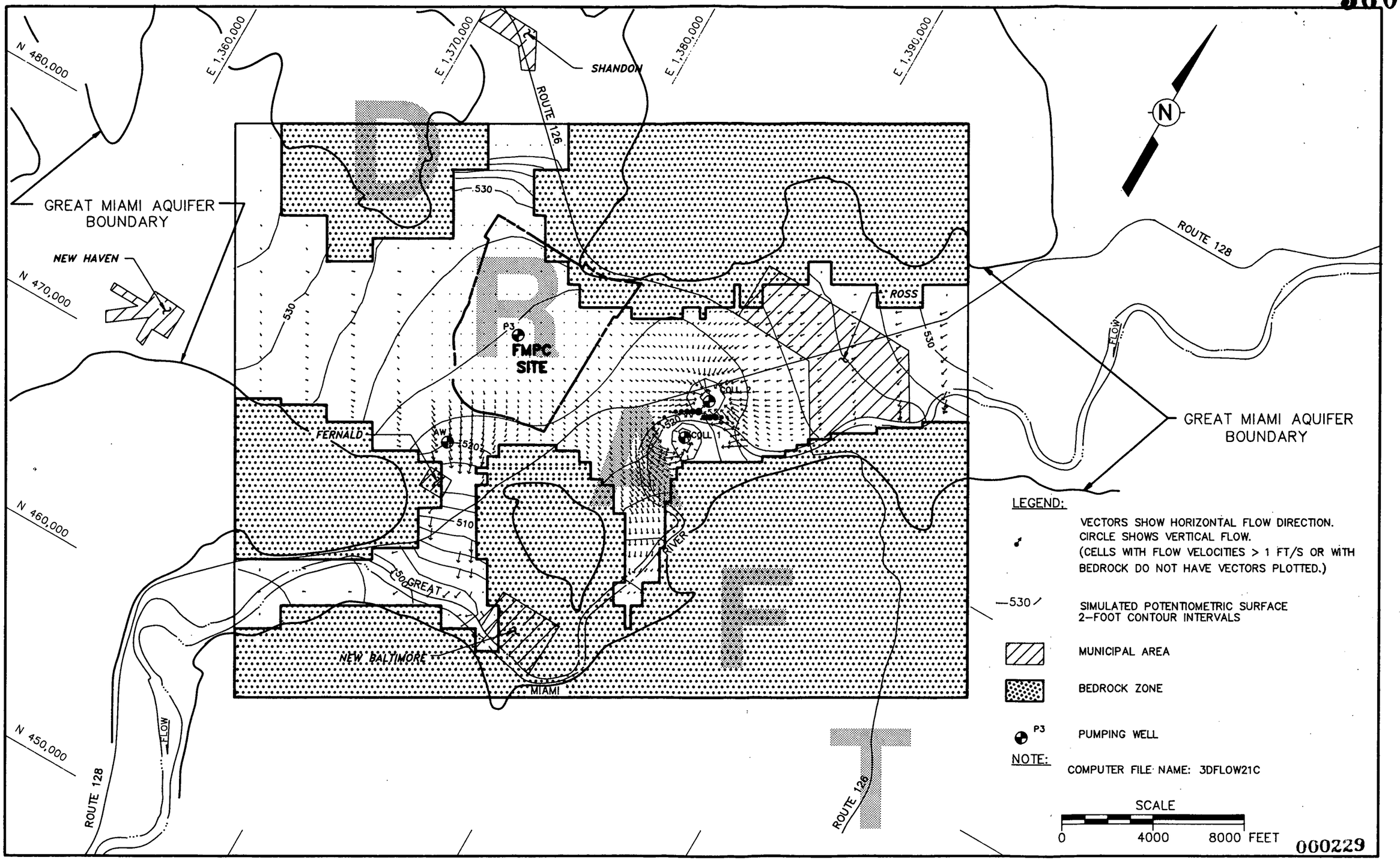
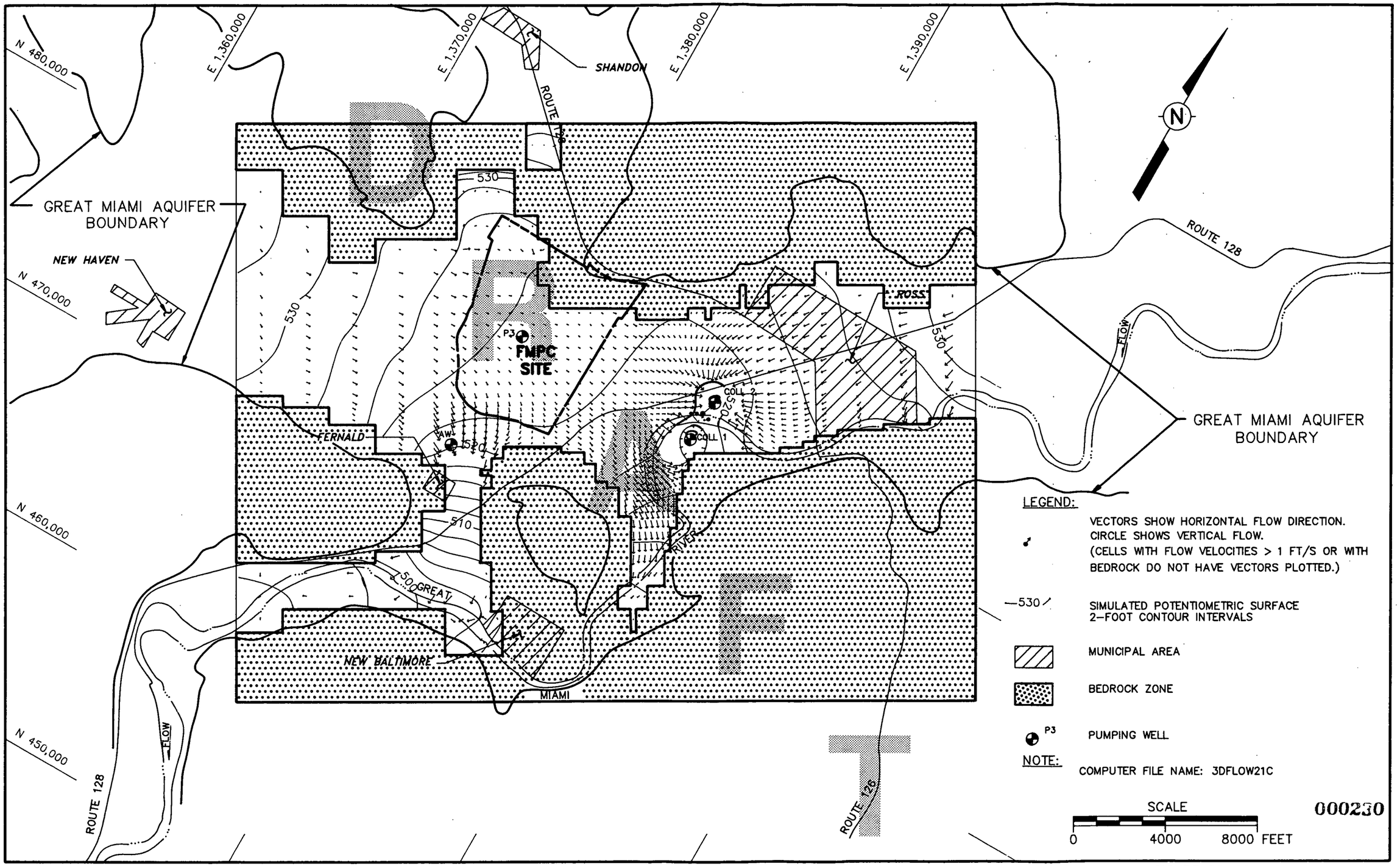


FIGURE 20-21
3-DIMENSIONAL FLOW MODEL
CALIBRAED RUN, LAYER 3
FLOW VECTORS AND POTENTIOMETRIC SURFACE

303317-B693 (GW) /01 (PGH)

303317-B692 (GW) /01 100% BLUE



303317-B692 (GW) /01 (PGH)

FIGURE 20-22
3-DIMENSIONAL FLOW MODEL
CALIBRAED RUN, LAYER 4
FLOW VECTORS AND POTENTIOMETRIC SURFACE

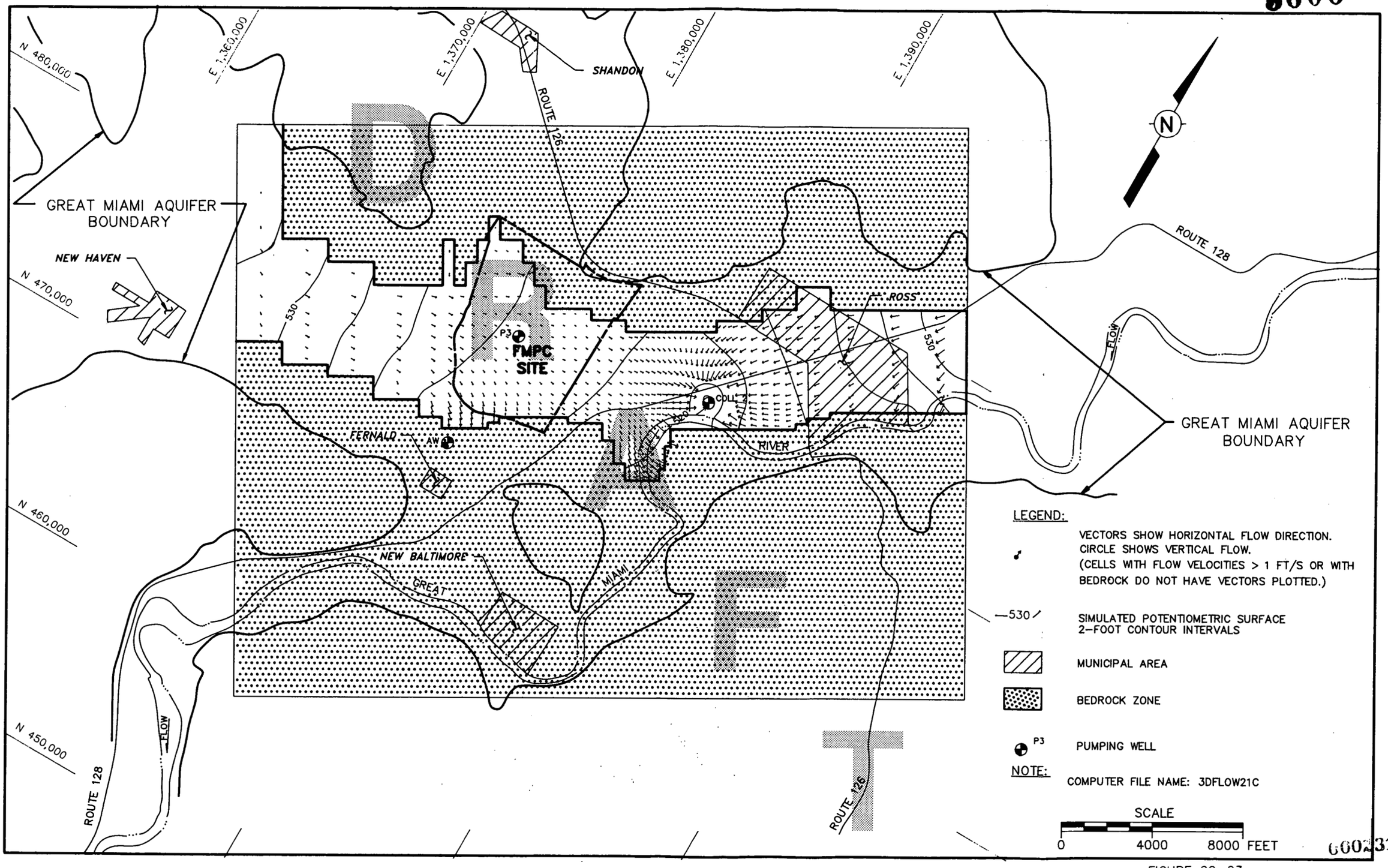


FIGURE 20-23
3-DIMENSIONAL FLOW MODEL
CALIBRATED RUN, LAYER 5
FLOW VECTORS ANMD POTENTIOMETRIC SURFACE

- The horizontal hydraulic gradient above the clay interbed is slightly higher than the horizontal gradient below the interbed.

- The hydraulic head difference across the clay interbed ranges to more than a foot, with the hydraulic head above the interbed being greater than the head below (i.e., downward flow component).

The difference in flow directions above and below the clay interbed has a noteworthy consequence when compared to the vertically integrated results of the 2DFLOW model. That is, the flow direction in Layer 1 in the Waste Storage Area is found to be about 35 degrees different from the corresponding flow direction in 2DFLOW. Furthermore, consideration of vertical variants results in a simulated source-receptor relationship in 3DFLOW that is different from that of 2DFLOW. The 2DFLOW model shows virtually all of the groundwater from beneath the Waste Storage Area moving toward the collector wells; but 3DFLOW shows much of the water from beneath the Waste Storage Area moving toward the Great Miami River downstream from the collector wells. The results of the 3DFLOW model in the FMPC area are more representative of the natural system than the 2DFLOW results. Other features of 3DFLOW are similar to those of 2DFLOW.

20.2.4 Sensitivity Analysis

Sensitivity analysis for 3DFLOW was performed on two interim calibrated runs that were available prior to completion of the calibration. This sensitivity analysis not only provided information on the effect of parameter changes on model results, but also provided information pertinent to planning further calibration efforts for the purpose of producing minor improvements that make the calibration even better. The results of the sensitivity analysis applied to 3DFLOW are shown in Tables 20-6 and 20-7. They may be summarized as follows:

- A large increase in hydraulic conductivity in the lower aquifer (Layers 4 and 5) resulted in both a greater variation of values of residuals and undesirable clustering. This indicates that a large increase in hydraulic conductivity in the lower part of the aquifer would cause the degree of calibration (i.e., the model performance) to begin to decline.

- Variation of the vertical hydraulic conductivity of the clay interbed by an order of magnitude had no significant effect on the overall model calibration. Indeed, increasing the vertical hydraulic conductivity to 0.4 ft/day (1.4×10^{-4} cm/s)had no significant effect on the overall calibration.

- Decreasing the ratio of vertical to horizontal hydraulic conductivity by an order of magnitude had a small adverse effect on model calibration. Setting the vertical hydraulic conductivity equal to horizontal hydraulic conductivity had negligible effect on the degree of calibration.

- Variation of recharge rates within the area of the clay interbed beneath the FMPC site also had negligible effect on overall model calibration. However,

TABLE 20-6

RESULTS OF THE SENSITIVITY ANALYSIS
FOR THE THREE-DIMENSIONAL FLOW MODEL
UTILIZING RUN 3DFLOW

Sensitivity Run Number	Parameter	3DFLOW Value	Sensitivity Run Value	Sensitivity Run[a] Statistics: Mean Residual (ft)	Sensitivity Run[a] Statistics: Mean Absolute Residual (ft)
1	Hydraulic Conductivity in Layers 4 and 5 (Excluding the Zone at the FMPC) (ft/day)	500 600	1150 1530	0.08	1.33
2	River Bed Leakage Factor	0.50	0.35	0.29	1.11
3	Recharge Rates in the Northwest Corner of the FMPC (in/yr)	6	4	0.05	1.08
4	Recharge Rates across the FMPC (in/yr)	6	2	0.23	1.06
5	Boundary Condition along North Boundary (ft)	542	541.5	0.21	1.06
		3DFLOW STATISTICS		0.04	1.08

[a]Mean Residual is the mean of the differences between observed head and calculated heads. Mean Absolute Residual is the mean of the absolute differences between observed heads and calculated heads.

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TABLE 20-7

RESULTS OF THE SENSITIVITY ANALYSIS
FOR THE THREE-DIMENSIONAL FLOW MODEL
UTILIZING RUN 3DFLOW

Sensitivity Run Number	Parameter	3DFLOW Value	Sensitivity Run Value	Sensitivity Run[a] Statistics: Mean Residual (ft)	Sensitivity Run[a] Statistics: Mean Absolute Residual (ft)
6	Vertical Hydraulic Conductivity Through Clay Interbed (ft/day)	0.00003	0.0003	0.36	1.06
7	Vertical Hydraulic Conductivity Through Clay Interbed (ft/day)	0.00003	0.003	0.38	1.05
8	Ratio of Vertical to Horizontal Hydraulic Conductivity (k_v/k_h)	1/10	1/100	1.03	1.45
9	Ratio of Vertical to Horizontal Hydraulic Conductivity (k_v/k_h)	1/10	1	0.19	1.04
		3DFLOW STATISTICS:		0.36	1.06

[a]Mean Residual is the mean of the differences between observed heads and calculated heads. Mean Absolute Residual is the mean of the absolute differences between observed heads and calculated heads.

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it did result in improved model performance near the FMPC, which is an important part of the model.

- One test of sensitivity to river leakage factor was performed, and the result was similar to the results obtained in 2DFLOW. That is, the model is sensitive to reduction of the river leakage factor.

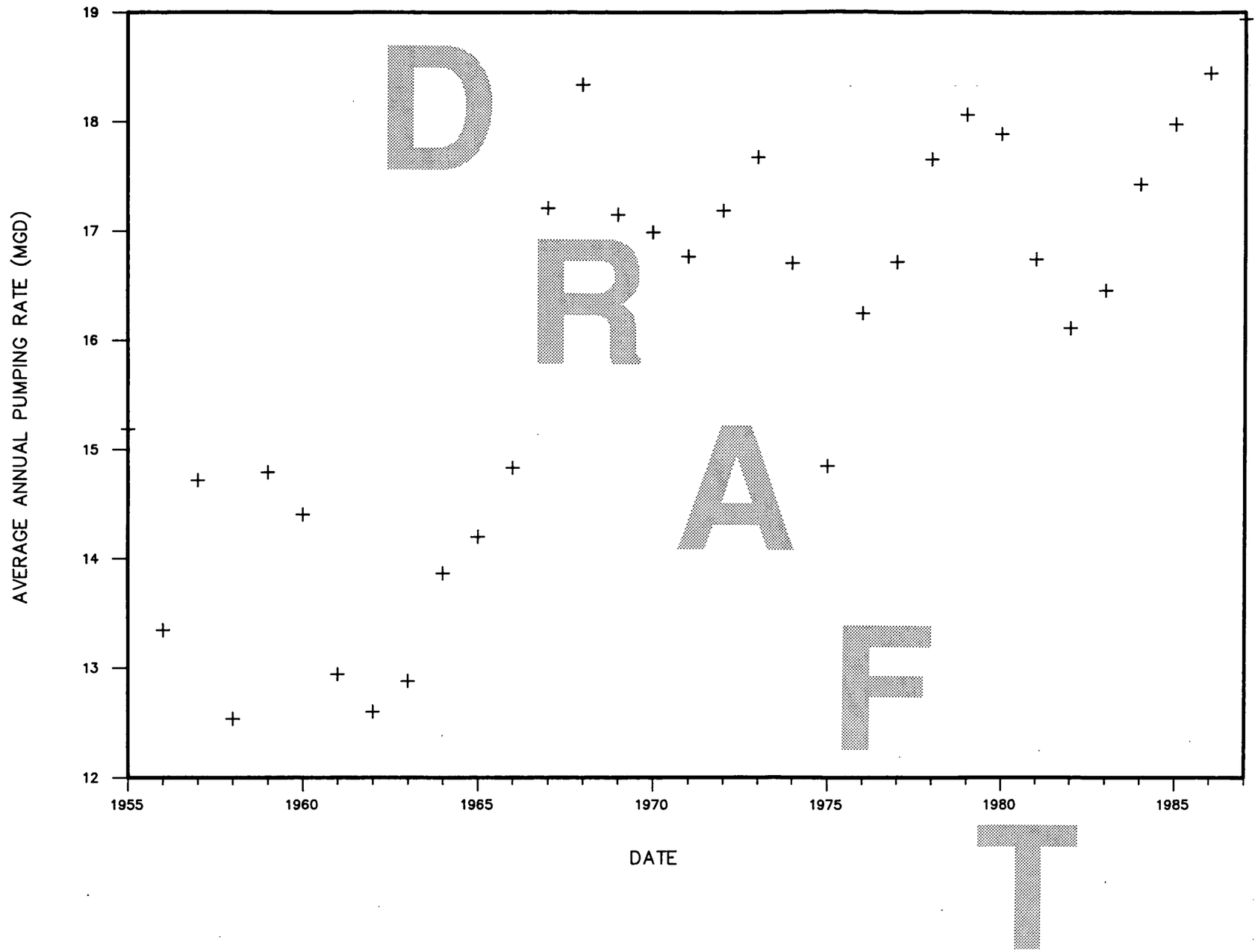
The results of increasing and decreasing the pumping rates of the collector wells by 25 percent are discussed below in connection with initial model application. The mean residual and the mean absolute residual that resulted from increasing the pumping rate were 0.73 and 1.19 feet, respectively. The mean residual and the mean absolute residual that resulted from decreasing the pumping rate were -0.12 and 1.08 feet, respectively. The historical records indicate no significant trend of increasing or decreasing pumping rate versus time since 1967 (Figure 20-24). Therefore, there is no reason to expect that there will be future changes in the pumping rates of these wells that would have a significant effect on any projections of contaminant movement based on the present models. Even if the pumping rates of the wells should increase by 25 percent, the effect on flow velocities would be less than 20 percent beyond a distance of 750 feet from a collector well.

20.2.5 Initial Application

The initial application of the groundwater flow model had as its objective the refinement of the understanding of the site conditions. The results are described below. Other applicants will be described in future RI/FS reports.

20.2.5.1 Particle Tracking

Steady-state flow paths away from source location areas of the FMPC are shown by the particle tracking map presented in Figure 20-25. Particle tracking was performed using the STLINE computer code (Geotrans 1987). Particle tracking is a technique for determining and depicting the three-dimensional movement of groundwater in a finite-difference flow model. In the present investigation, it involved processing output from 3DFLOW and computing successive positions of particles moving in the direction of flow and at the average linear velocity (Freeze and Cherry, 1979, p. 71) of water in the Great Miami Aquifer. The average linear velocity describes the velocity in the aquifer pores and varies along the flow path. The horizontal traces of the particles are shown by lines that depict particle paths over 5-year time increments. Vertical movement of the particles is shown by changing path symbols when the particle moves to a different layer. Rate of movement is proportional to the distance between successive five-year particle locations. The rate of horizontal movement between successive particle locations may be calculated by dividing the distance traveled by the travel time. The effects of vertical velocity components on particles

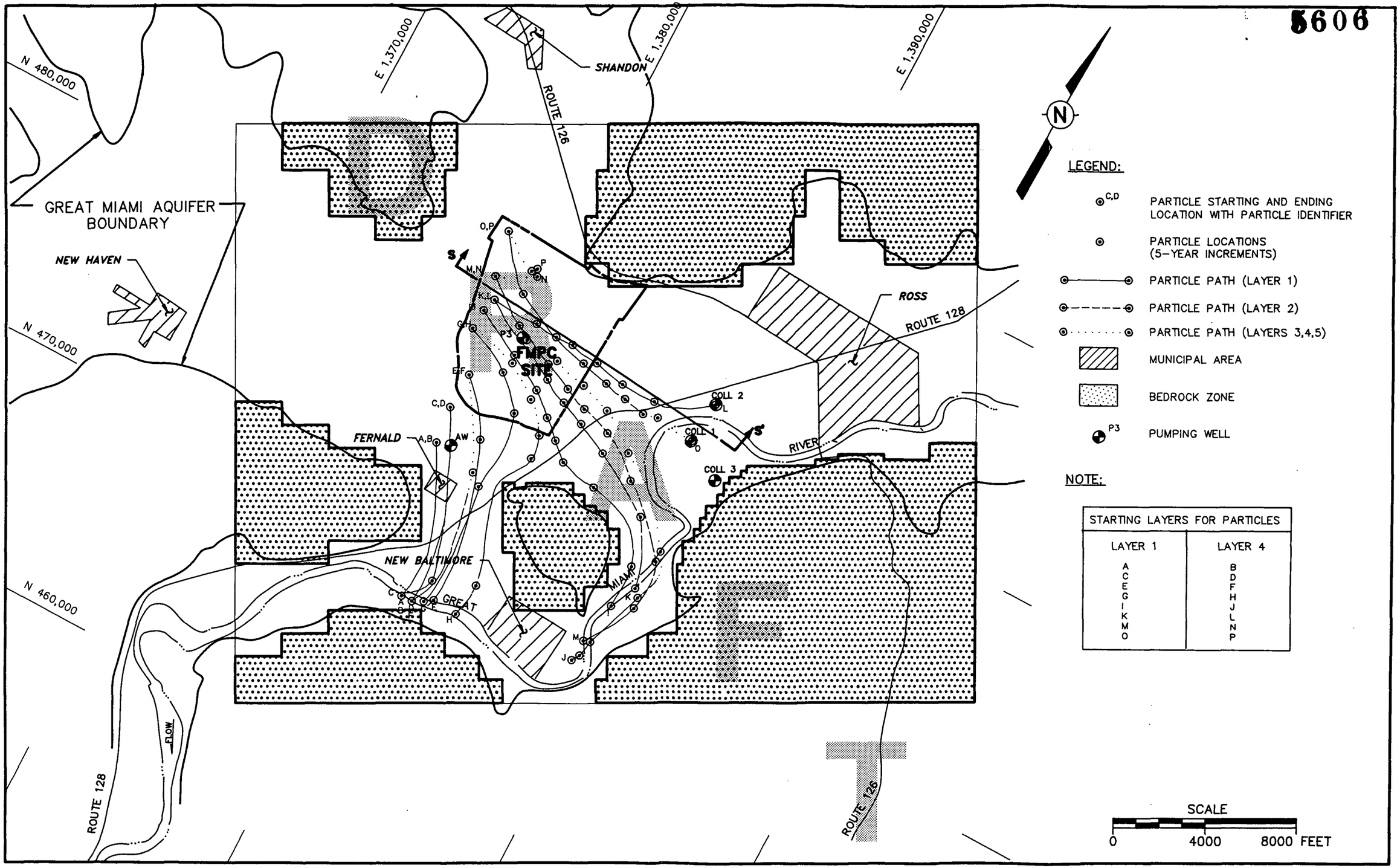


LEGEND:

+ COMBINED AVERAGE ANNUAL PUMPING RATE

303317-B752 (GW) (PGH)

FIGURE 20-24
COLLECTOR WELLS
COMBINED AVERAGE ANNUAL PUMPING RATE



303317-B687 (GW) (PGH.)

FIGURE 20-25
3-DIMENSIONAL FLOW MODEL PARTICLE TRACKING

moving toward the collector wells are shown in Figure 20-26. Rate and direction of flow is also shown for individual cells by the vectors included in the potentiometric surface maps (Figures 20-19 through 20-23).

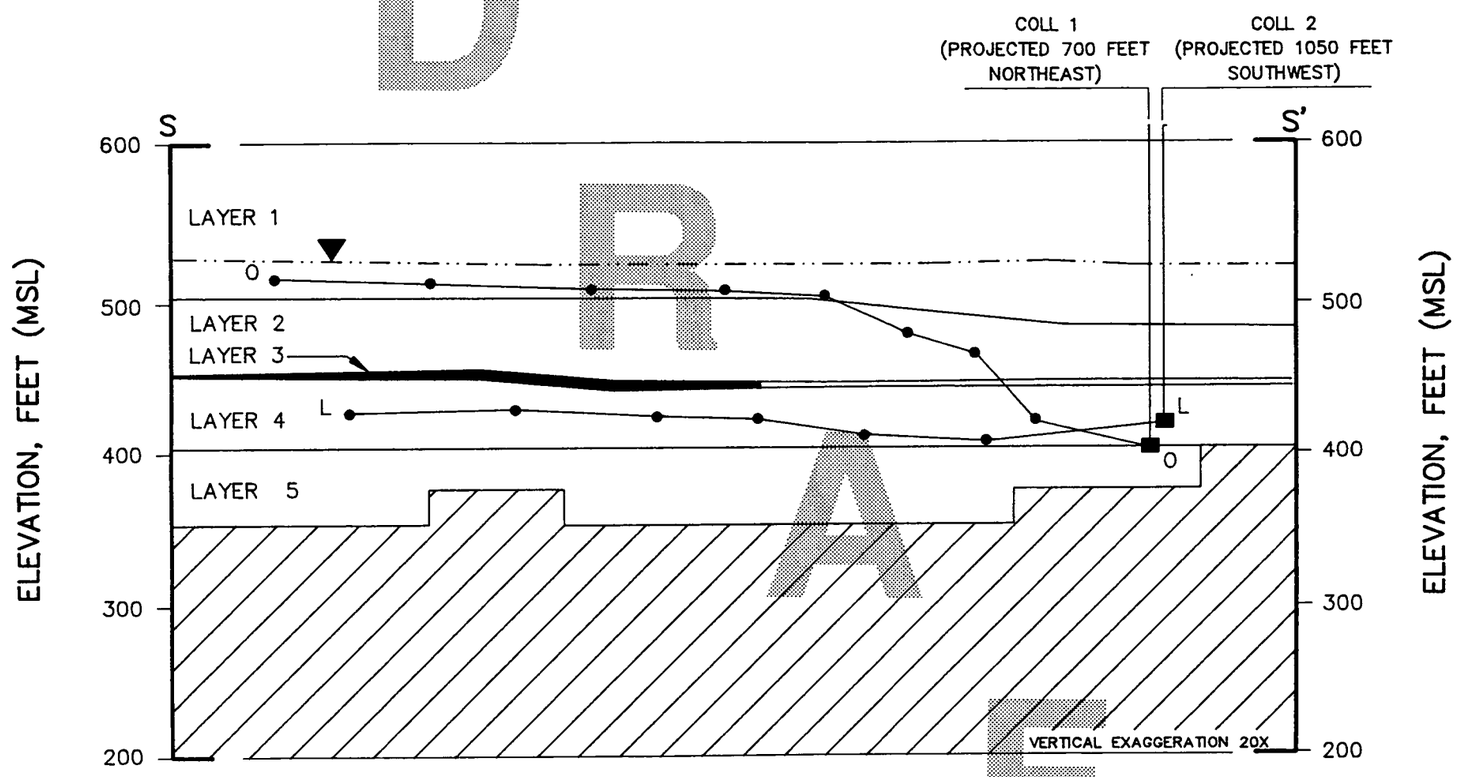
The particle tracking shown in Figure 20-25 and the vectors shown in Figures 20-19 through 20-23 indicate that the pumping rates of the Albright & Wilson Co. wells and the FMPC production well are too low to have any significant effect on the rate and direction of groundwater flow in the Great Miami Aquifer. Reasonable variation of their pumping rates should not produce any significant changes in rate and direction of groundwater flow.

Sensitivity runs made by increasing the pumping rates of the collector wells by 25 percent in 3DFLOW resulted in an additional foot of drawdown in Layer 1 at distances as great as 760 feet from Collector Well No. 2. Darcy velocities increased by about 20 percent at this distance. An additional 0.1 foot of drawdown was produced at distances as great as 7,250 feet from Collector Well No. 2. Darcy velocities increased about 13 percent at this distance. The general configuration of the potentiometric surfaces in the model layers did not change.

Sensitivity runs made by decreasing the pumping rates of the collector wells by 28 percent (which represents the lowest pumping rate recorded) resulted in a one-foot reduction of drawdown at a distance as great as 1,425 feet from Collector Well No. 2. Darcy velocities decreased by about 14 percent at this distance. A reduction of 0.1 foot in the drawdown occurred at distances as great as 8,200 feet. Darcy velocities decreased by about 12 percent at this distance. The general configuration of the potentiometric surface in the model layers did not change.

20.2.5.2 Water Balances at Major Pumping Wells

A water balance was established for the collector wells by using particle tracking to identify the sources of water that enters the wells, in combination with model output to calculate the rate that water is supplied from each source. This water balance differs from the balance reported in Section 20.2.2 in that it includes only water that enters the wells, including all water that flows into and out of a specified region of the model. The water balance at the wells is not as accurate as the regional water balance because source areas derived from particle tracking are not as precisely defined as source areas specified at the boundaries of a region.



CROSS SECTION S–S'
(LOOKING NORTHEAST)

LEGEND:

- BEDROCK
- SIMULATED POTENTIOMETRIC SURFACE
- O PARTICLE STARTING AND ENDING LOCATION WITH PARTICLE IDENTIFIER
- PARTICLE LOCATION (5–YEAR INCREMENTS)
- OPEN INTERVAL
- PARTICLE PATH

NOTE:
PARTICLE TRACKS HAVE BEEN PROJECTED INTO SECTION AND ARE PLOTTED AT FIVE–YEAR INCREMENTS

FOR LOCATION OF CROSS SECTION S–S', SEE FIGURE 20–25

VERTICAL SCALE
0 100 200 FEET

HORIZONTAL SCALE
0 2000 4000 FEET

303317–B541 (GW) (PGH)

FIGURE 20–26
3–DIMENSIONAL FLOW MODEL
CROSS SECTION S–S'
PARTICLE TRACKING

The water balance results are presented in Table 20-8. This balance shows that about 3.3 percent of the water pumped from the collector wells comes from beneath the FMPC site. About half of this amount comes from the part of the aquifer above the clay interbed. The main source (64.4 percent) of water for the collector wells is the Great Miami River. The particle tracking indicated that the travel time for groundwater from the Waste Storage Area to the collector wells is about 35 years. If the retardation factor applied to uranium transport is 12, which corresponds to the distribution coefficient selected for use in the solute transport model (Chapter 21.0), then the travel time for uranium from the Waste Storage Area to the collector wells would be 420 years.

Particle tracking shows that particles that enter Albright & Wilson Co. wells come from the Great Miami Aquifer to the northwest. None of the water is drawn from Paddys Run. The combined production rate from the wells is estimated at only 225 gpm, and particle tracking with and without the wells operating shows that the wells have no important effect on flow rates and flow directions in the Great Miami Aquifer.

Particle tracking in the vicinity of the FMPC production well yields similar results. The simulation of the well at a pumping rate of 292 gpm produces effects that are too small to deflect particle tracks downward through the clay interbed. Model results indicate that virtually all of the water entering the FMPC well comes from the Great Miami Aquifer to the northwest of the FMPC site. The ultimate source of the water is upgradient recharge to the northwest of the well.

20.2.5.3 Interaction with the Great Miami River

Model results show that the Great Miami River is a gaining stream receiving groundwater along most of its course through the model area. However, near the collector pumping wells, the river loses water at a rate of 13 Mgd and provides 64 percent of the water pumped from the wells. Particle tracking similar to that shown in Figure 20-26 shows that the reach of the Great Miami River that would receive water from beneath the FMPC gains water at a rate of about 637,000 ft^3/day (4.8 Mgd). This amount is about 32 percent of the total discharge to the total reach of the Great Miami River included in the modeled area.

When the pumping rates of the collector wells increased by 25 percent, induced infiltration from the Great Miami River increased 27 percent and extended an additional 500 feet upstream. When the pumping rates of the collector wells were decreased by 28 percent, induced infiltration from the Great Miami River decreased 31 percent.

TABLE 20-8

ORIGIN OF WATER DISCHARGED BY THE SOUTHWESTERN OHIO WATER COMPANY COLLECTOR WELLS

	Sources (ft^3/day)					
Model Layer	FMPC Effluent	Upgradient Groundwater	River Recharge	Infiltration From Precipitation	Total (cubic feet per day)	Total (Mgd)
1	16,816	175,770	1,729,597	303,965	2,226,148	16.65
2	32,613	81,049	N/A	N/A	113,662	0.85
3	586	5,754	N/A	N/A	6,340	0.05
4	27,331	157,877	N/A	N/A	185,208	1.39
5	18,462	134,168	N/A	N/A	152,630	1.14
TOTAL:	95,808 (3.6%)	554,618 (20.7%)	1,729,597 (64.4%)	303,965 (11.3%)	2,683,988 (100.0%)	20.08

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When the pumping rate of the collector wells is increased by 25 percent, the Great Miami River loses water at the rate of 16.5 Mgd by induced infiltration. When the pumping rate of the collector wells is decreased by 28 percent, the Great Miami River loses water at the rate of 9 Mgd by induced infiltration.

20.2.5.4 Area of Influence of the FMPC Production Well

Particle tracking showed that water entering the FMPC production well comes from the Great Miami Aquifer northwest of the site. Any influence on the aquifer above the clay interbed is too small to be detected. Particles introduced into the particle tracking routine immediately above the interbed and radially around the well were not drawn downward into the interbed. They simply moved southeastward and showed no deflection toward the well. The model assumes that the well itself is sealed through the clay.

The pumping well has such an inconsequential effect on the groundwater system that reasonable increases or decreases in the pumping rate will have no significant effect on the configuration of the potentiometric surface or on seepage through the clay interbed. Simulated seepage through the clay interbed is very small. The gradient across the interbed at the well is 0.00056 ft/ft. Since the vertical hydraulic conductivity has been assigned a value of 0.00003 ft/day (1.0584×10^{-8} cm/s), the flow across the interbed is calculated to be 1.7×10^{-8} $ft^3/day/ft^2$ into the cell that contains the well. This amount of water is negligible.

20.2.5.5 Effects of Varying River Elevation and Recharge

Changes in river elevation and recharge are necessarily transient conditions of relatively short duration. As discussed above, in connection with using the steady-state option, such transient changes have no significant effect on the long-term behavior of the groundwater system being simulated by the model.

21.0 SOLUTE TRANSPORT MODEL

21.1 ANALYTICAL SOLUTE TRANSPORT MODELING

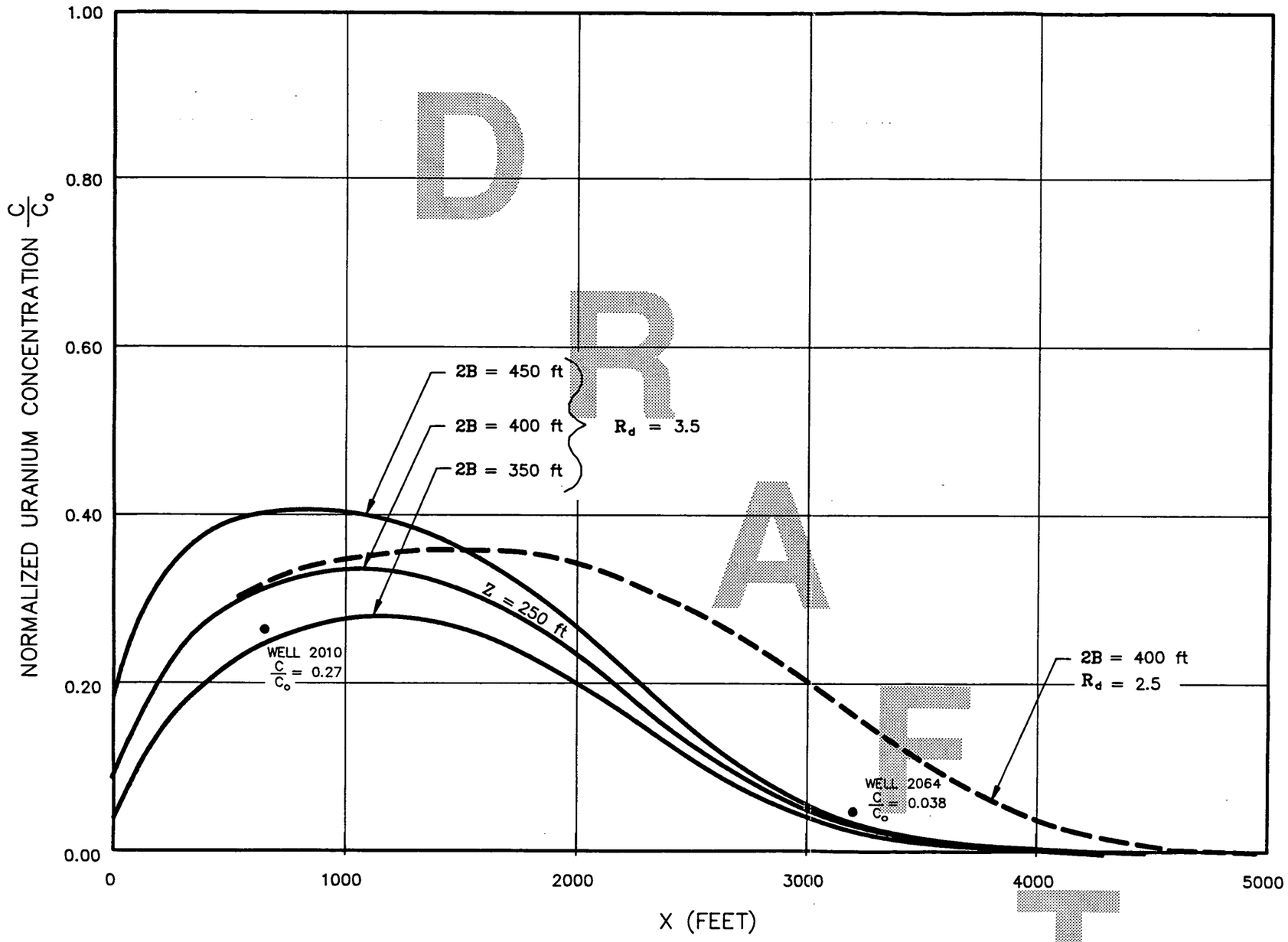
The development of the numerical solute transport model was preceded by the development of an analytical solute transport model to provide a preliminary evaluation of parameters for the numerical modeling. Analytical solute transport modeling was applied to two areas: (1) the Waste Storage Area; (2) the South Plume area, which contains elevated uranium concentrations on and south of the FMPC. The analytical models only treat solute transport under simplified conditions. Due to the need to achieve closed form mathematical solutions, analytical models can only calculate concentrations for a physical system that is a highly simplified representation of the real system. Consequently, they are only used as an efficient transition to the more complex finite-difference models. The degree of calibration of the analytical models is adequate only for initial parameter estimates for use in the two-dimensional finite-difference solute transport model.

21.1.1 Approach

For the purposes of this analytical modeling, the flow directions of the models were aligned with the primary direction of groundwater flow in each area where the modeling was applied. Because the distance of observation wells away from the axis of the natural plume is a significant factor in determining uranium concentrations at the wells, two-dimensional analytical models had to be used. The two-dimensional analytical modeling was based on the following assumptions:

- Groundwater flow is uniform
- Effective porosity, longitudinal and transverse dispersivities, and retardation factor are constant
- The aquifer is infinite in all directions (i.e., no boundary effects)
- A line source of uranium is perpendicular to the direction of groundwater flow
- The source is continuous through the thickness of the aquifer
- The contaminant concentrations are initially zero
- Molecular diffusion is negligible

The models were developed using the STRIP1B and STRIP1FBCG computer codes. These codes were developed by IT and have been extensively verified with the use of IT's finite element groundwater flow and solute transport code, GEOFLOW (IT Corporation 1986), and the Princeton



LEGEND:

C_o = 78μg/L
U = 0.64 ft/day
α_L = 100 ft
α_T = 10 ft
● OBSERVED NORMALIZED CONCENTRATION

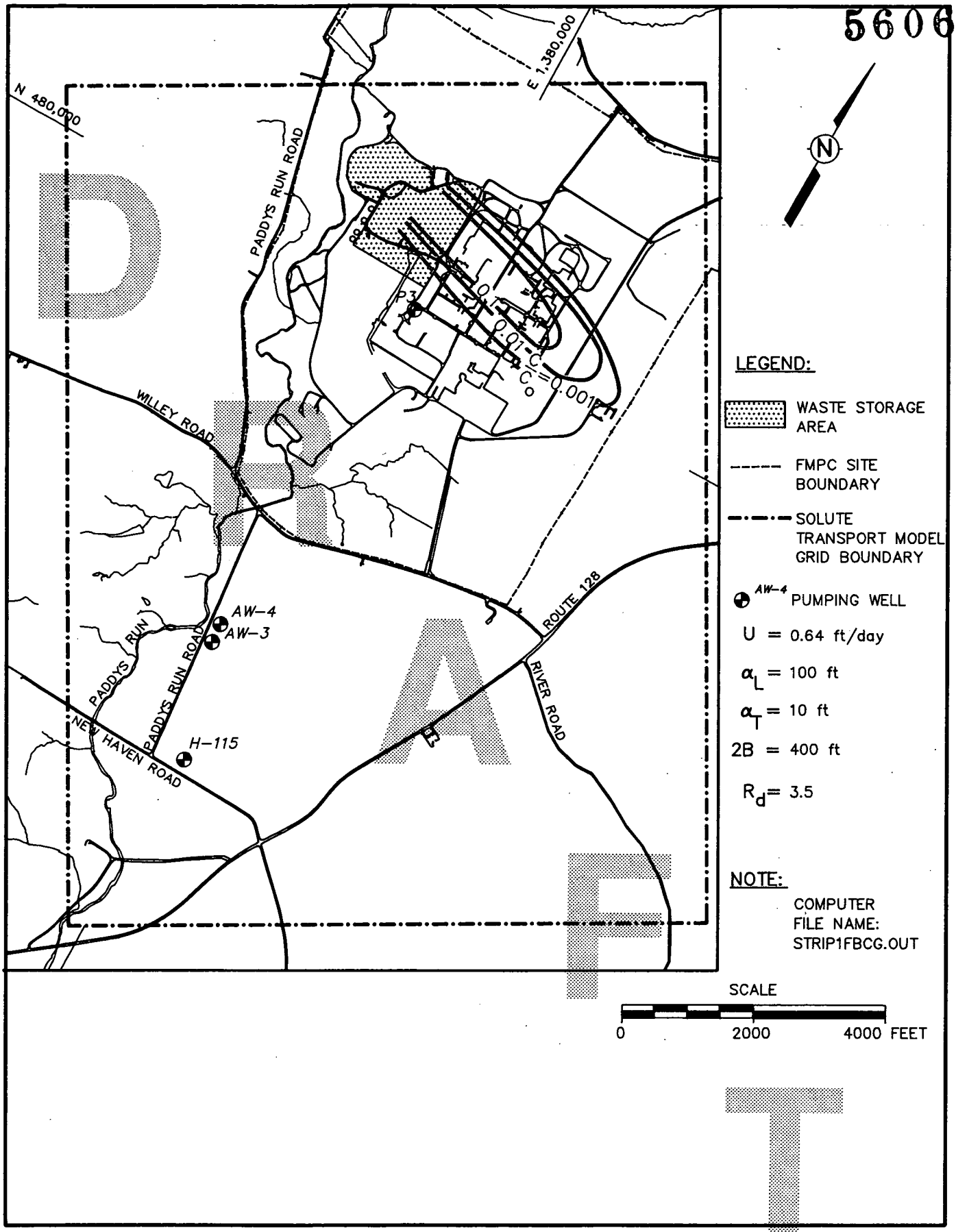
NOTE:

COMPUTER FILE NAME: STRIP1FBCG.OUT

303317-B281 (GW) (PGH)

000244

FIGURE 21-1
ANALYTICAL SOLUTE TRANSPORT MODEL
OBSERVED AND PREDICTED NORMALIZED
URANIUM CONCENTRATIONS
FOR NORTH PLUME AREA
VARYING SOURCE WIDTHS



303317-A148 (GW) /01 (PGH)

FIGURE 21-2
ANALYTICAL SOLUTE TRANSPORT MODEL
PREDICTED NORMALIZED URANIUM CONCENTRATION
DISTRIBUTION FOR WASTE STORAGE AREA
IN 1988

Transport Code (PTC) developed at Princeton University (Babu 1987). The verifications, as well as the theoretical bases for STRIP1B and STRIP1FBCG, are published in the technical literature (Batu 1989; Batu and van Genuchten 1990).

21.1.2 Results: Waste Storage Area

The principal direction of flow and the average hydraulic gradients were determined from hand-contoured potentiometric surface maps that were available at the time of the analytical modeling. Using Darcy's Law, preliminary estimate of hydraulic conductivity of 400 ft/day (0.14 cm/s) was used to calculate an average groundwater velocity of 0.64 ft/day. The longitudinal dispersivity selected was 100 feet, and the transverse dispersivity was taken to be 10 feet. The use of 100 feet for longitudinal dispersivity was based on information on scale-dependent dispersivity in the scientific literature. Walton 1985 (Figure 2.16) presents a graph of mean travel distance versus longitudinal dispersivity. If the mean travel distance in the plume area is 2,500 feet, Walton's graph yields a longitudinal dispersivity of a little over 100 feet. The transverse dispersivity was held constant at 10 feet, because the ratio 1/10 for longitudinal to transverse dispersivity is commonly reported in the scientific literature (Walton 1985 p. 38).

A line source was placed at the Waste Storage Area. The starting time of the uranium release was assumed to be 1955, and the measured concentrations corresponded to 1988 RI/FS sampling data. Therefore, a total simulation period of 33 years was used for the model. An appropriate source width perpendicular to the groundwater flow field was unknown, and the retardation factor was also unknown. Successive trials were used to calibrate the model by varying the source width and the retardation factor. The calibration targets were the wells listed in Table 21-1. The calibrated result was a source width of 400 feet and a retardation factor of 3.5. (As will be discussed later in this chapter, the results of subsequent geochemical modeling and solute transport model calibration increased the retardation factor to 12.)

Figure 21-1 shows the relationship between retardation factor, source width, and the degree of calibration relative to two wells that are located about 250 feet from the axis of the simulated plume. Figure 21-2 shows the normalized concentration in the simulated plume at 33 years. Table 21-1 shows the comparison between measured and calculated concentrations at wells in the vicinity of the plume. The calculated concentrations are based on a source concentration of uranium of 78 ug/L, which was a preliminary value measured at Well 2010 at the point of entry into the regional aquifer. The value of 78 ug/L was the highest previously measured at Well 2010.

TABLE 21-1

COMPARISON
OF THE MEASURED AND PREDICTED CONCENTRATIONS
FOR SHALLOW WELLS IN THE WASTE PIT AREA

Well No.	Measured Concentration[a] (µg/L)	Predicted Concentration[b] (µg/L)
2042	4	< 0.08
2037	4	< 0.08
2013	< 1	< 0.08
2051	3	< 0.08
2068	3, < 1	< 0.08
2064	3	2.6
2010	21	24.4

[a]Based on RI/FS data that have not been validated.

[b]For 400-foot source width.

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21.1.3 Results: South Plume Area

A representative groundwater velocity of 1.33 ft/day was calculated for the southern plume area using the method described above for the Waste Storage Area. The value of 3.5 for the retardation factor determined for the Waste Storage Area was used for the southern plume area because the same aquifer is involved. Values for dispersivity were also kept the same. The source location was assumed to be the Storm Sewer Outfall Ditch. The source concentration and source width were unknown. These quantities were estimated as described below.

As can be seen in Figure 21-3, which was developed from reported estimates of discharge to the Storm Sewer Outfall Ditch (Table 21-2), the mass of uranium released to the Storm Sewer Outfall Ditch between 1961 and 1984 may be represented by the following formula:

$$M(t) = M_o e^{\gamma t} \qquad (1)$$

where M(t) is the uranium mass (kg), t is time (days), and M_o and γ (day^{-1}) are parameters. Two fitted curves are shown in Figure 21-3. The selected fit is the case in which M_o equals 1200 kg and γ equals 0.0004 day^{-1}. Therefore, Equation 1 takes the form:

$$M(t) = 1200e^{-0.0004t} \qquad (2)$$

Equation 2 corresponds to the period of 1961 to 1984 and describes the reduction in the rate at which uranium was discharged to the Storm Sewer Outfall Ditch. With the assumption that the source concentration decreases in the same form as Equation 2, the following expression can be written:

$$C_o(x = o, z, t) = C_m e^{\gamma t} \qquad -B<Z<B \qquad (3)$$

in which C_o is the source concentration, C_m is the maximum concentration corresponding to the situation in 1961, and B is half the length of the line source. Equation 3 is the source condition used in the STRIP1B computer program.

Successive trials using different source widths were used to find a plume configuration that caused roughly equal source concentrations to be back-calculated from measured concentrations at Wells 2060 and 2061. The result was a source width (2B) of 1,000 feet, and an initial concentration (C_o) corresponding to 1961 of about 4,500 ug/L. Figure 21-4 shows the normalized concentration in the plume at the simulation period of 33 years.

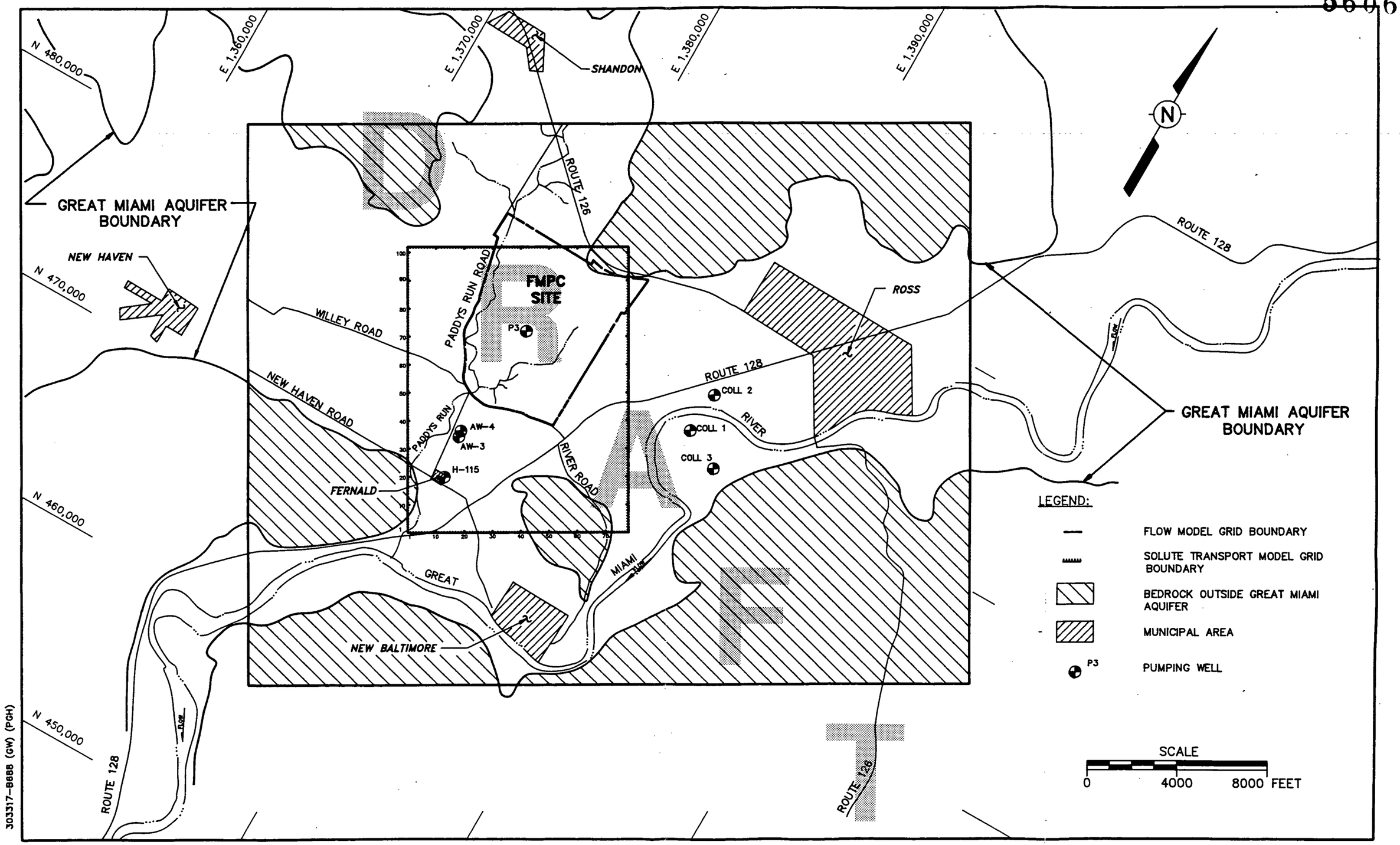

FIGURE 21-5
GROUNDWATER MODELING GRID BOUNDARIES

TABLE 21-2

ESTIMATED URANIUM LOSS TO STORM SEWER SYSTEM AND STORM SEWER OUTFALL DITCH

Year	Recorded Annual Uranium Loss to Storm Sewer System (lbs)	Recorded Annual Loss to Storm Sewer Outfall Ditch (lbs)	Assumed Annual Loss to Storm Sewer Outfall Ditch[a] (lbs)	(kg)	Source
1956	1,800	--	373	169	b
1957	2,600	--	538	244	b
1958	5,400	--	1,118	507	b
1959	6,300	1,400	1,400	635	b
1960	11,200	2,000	2,000	907	b
1961	11,800	2,600	2,600	1,179	b
1962	14,760	--	2,624	1,190	c
1963	12,180	--	2,580	1,170	d
1964	9,600	--	2,470	1,120	c
1965	7,070	--	1,464	664	c
1966	9,490[e]	2,400[f]	1,863	845	c
1967	4,190	--	867	393	c
1968	4,080	--	845	383	c
1969	5,568	--	1,153	523	c
1970	2,976	--	622	282	c
1971	4,184	--	867	393	c
1972	2,584	--	536	243	c
1973	2,605	--	540	245	c
1974	1,752	--	395[g]	179	c
1975	1,584[h]	--	725[i]	329	c
1976	--	98.5	194	88	j
1977	--	157.6	320	145	j
1978	--	46.0	95	43	j
1979	--	75.5	154	70	j

See footnotes at end of table.

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TABLE 21-2
(continued)

Year	Recorded Annual Uranium Loss to Storm Sewer System (lbs)	Recorded Annual Loss to Storm Sewer Outfall Ditch (lbs)	Assumed Annual Loss to Storm Sewer Outfall Ditch[a] (lbs)	(kg)	Source
1980	--	16.4	33	15	j
1981	--	4.3	9	4	j
1982	--	42.7	24	11	j
1983	--	98.5	97	44	j
1984	--	124.8	123	56	j
1985	--	85.4	85.4		k
		TOTAL	26,629	12,076	

[a]The assumed annual uranium loss to the Storm Sewer Outfall Ditch is calculated, if not recorded, as 21.85 percent of the annual loss to the Storm Sewer System. The 21.85 percent figure is the average ratio of the recorded annual loss to the Storm Sewer Outfall Ditch to the recorded loss to the Storm Sewer System for the years 1959, 1960, 1961, and 1966.

[b]NLO Report of FMPC Ground Contamination Study Committee, September 30, 1962.

[c]NLO Aquifer Contamination Control Reports, dated January 21, 1965 to October 1, 1975.

[d]Data for 1963 are not available. Value presented is the average of 1962 and 1964 data.

[e]Value presented is calculated based on a loss of 8,700 pounds for the period January 1 to November 30, 1966.

[f]Value presented is calculated based on a loss of 2,200 pounds for the period January 1 to November 30, 1966.

[g]For January-November 1974, uranium loss to the Storm Sewer Outfall Ditch was 21.85 percent of the loss to the Storm Sewer System. For December 1974, the uranium loss to the Storm Sewer Outfall Ditch was 100 percent of the loss to the Storm Sewer System. Value presented is calculated accordingly.

[h]Value presented is calculated based on a loss of 132 pounds per month for the first four months of 1975.

[i]For January-June 1975, uranium loss to the Storm Sewer Outfall Ditch was 100 percent of the loss to the Storm Sewer System. For July-December 1975, the uranium loss to the Storm Sewer Outfall Ditch was 21.85 percent of the loss to the Storm Sewer System. Value presented is calculated accordingly.

TABLE 21-2
(continued)

[j]NLO FMPC Environmental Monitoring Report for 1976-1984. Values presented are converted from Ci to pounds by 1 Ci = 3283.5 pounds.

[k]WMCO FMPC Environmental Monitoring Report for 1985. Value presented is converted from Ci to pounds by 1 Ci = 3283.5 pounds.

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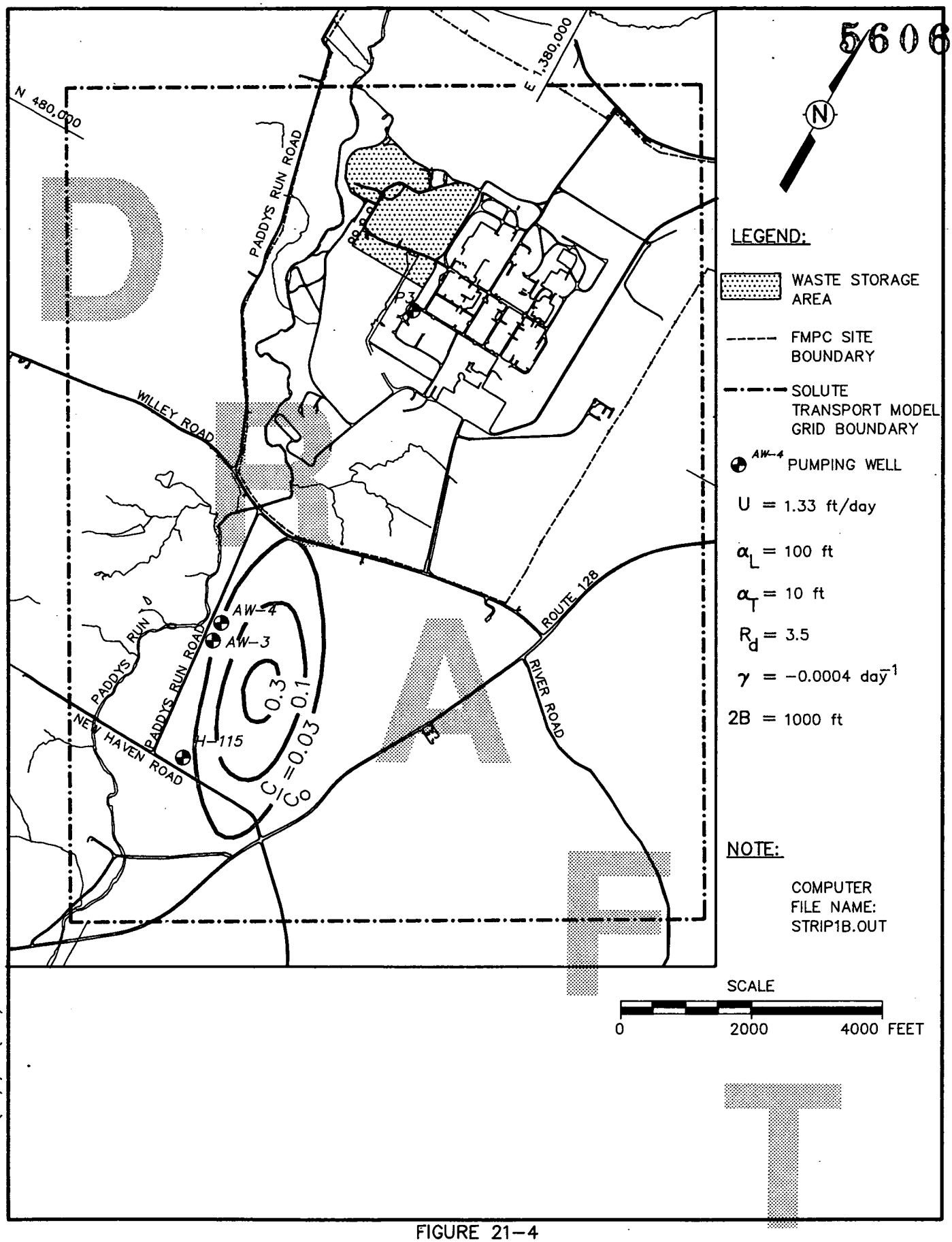

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FIGURE 21-4
ANALYTICAL SOLUTE TRANSPORT MODEL
PREDICTED NORMALIZED URANIUM CONCENTRATION
DISTRIBUTION FOR SOUTHERN PLUME AREA
IN 1988

21.2 TWO-DIMENSIONAL SOLUTE TRANSPORT MODEL

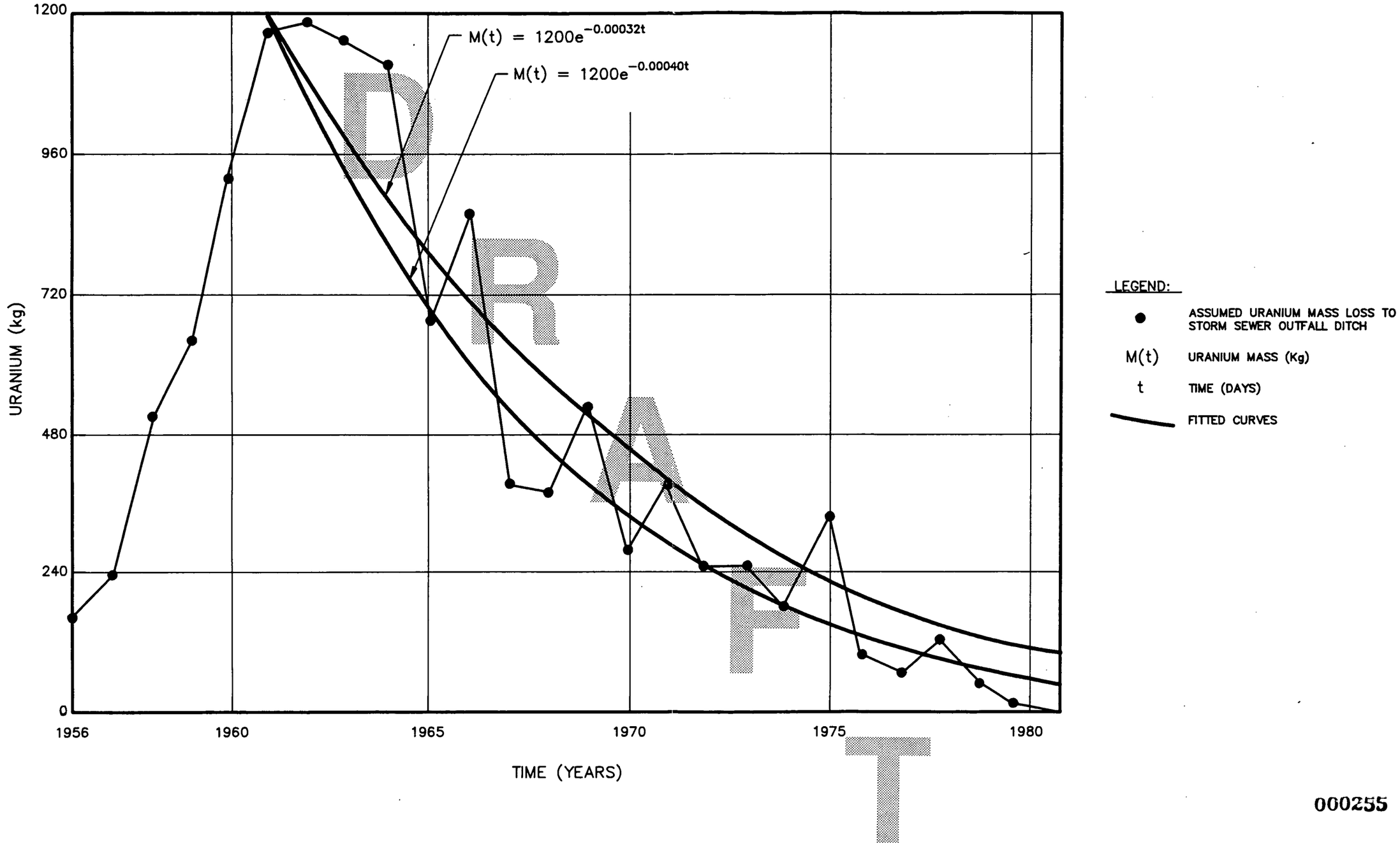
The purpose of the two-dimensional solute transport model (2DSOL) was to provide a transitional step toward the three-dimensional finite-difference solute transport model (3DSOL). This transitional step was included to promote efficiency in the development of 3DSOL, which is more complex and required more intensive use of labor and computer resources. Estimates of model parameters were made by calibration of 2DSOL, and then these estimates were supplied as initial values in the three-dimensional model. Application of the 2DSOL model also provided valuable insight into the processes affecting uranium concentration so that key geochemical studies could be planned and initiated.

21.2.1 Model Grid and Boundary Conditions

The grid for the solute transport model covers a smaller area around the FMPC site than the flow model grid (Figure 21-5). Such a reduction not only promotes efficient modeling, but can be justified by the fact that uranium migration occurs at a much slower rate than groundwater flow due to various retardation processes. The model area was reduced to the area that would be expected to maximize the value of information gained relative to the level of modeling effort. To this end, the grid area was made as small as possible while including all monitoring wells that would be used in the calibration with observed uranium concentrations. This procedure allowed for simultaneous calibration of all contaminant plume areas, which is important in case plumes from more than one source area coalesce. The resulting model grid is shown in Figure 21-6.

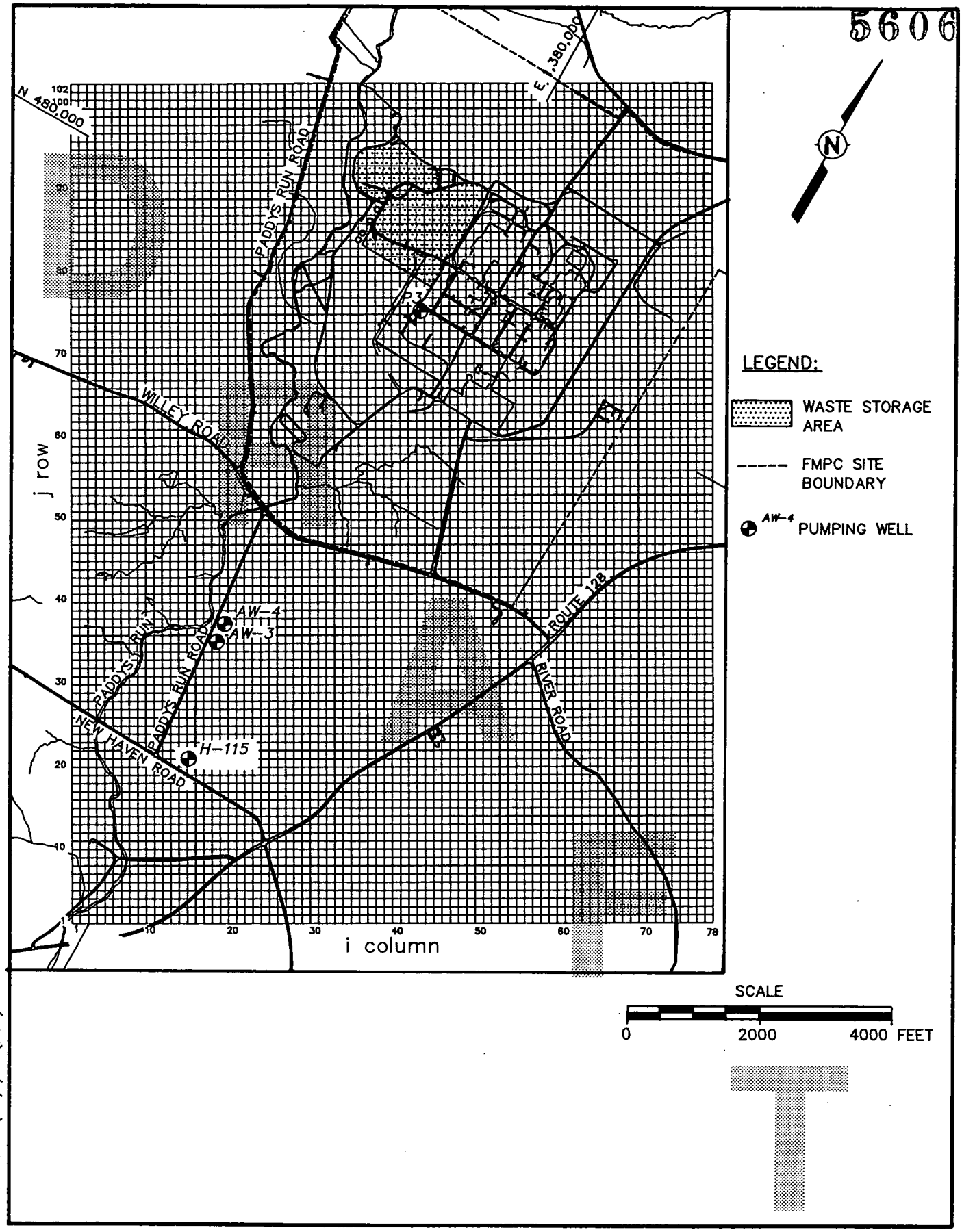
The cell size also had to be reduced to ensure numerical stability. It was chosen to conform to guidelines presented by Reeves et al. (1986). The critical criterion was that cell dimensions be no more than twice the longitudinal dispersivity. This is the overshoot/undershoot (see Glossary) criterion for finite-difference equations that are centered in time and central in space (CIT-CIS). The CIT-CIS difference scheme is provided to reduce numerical dispersion in model solutions. The chosen cell size was 125 feet. This cell size had several advantages:

- It allowed for some deviation from the original estimate of 100 feet for longitudinal dispersivity. The guideline for overshoot/undershoot would remain satisfied for longitudinal dispersivities as low as 62.5 feet. It should be noted that the overshoot/undershoot criterion is only a guideline, and it may be possible to use smaller dispersivities without introducing significant convergence problems.



303317-B280 (GW) (PGH)

FIGURE 21-3
ANALYTICAL SOLUTE TRANSPORT MODEL
TIME VERSUS URANIUM MASS LOSS
AT STORM SEWER OUTFALL DITCH AREA
WITH FITTED CURVES



303317-A109 (GW) /01 (PGH)

FIGURE 21-6
SOLUTE TRANSPORT MODEL GRID

- It allowed the grid to cover the desired area without introducing so many cells that computer run times and the size of input and output files would become intolerable.

- All flow model cell sizes were multiples of 125 feet; thus, each flow model cell corresponded exactly to four or more solute transport model cells. This condition facilitated importing pressures from the flow model to the corresponding boundary cells of the solute transport model, and it facilitated importation of bedrock elevation from one model to the other. It also facilitated comparison of pressures in all of the cells of the two models to verify that the flow system in the solute transport model was successfully imported from the flow model.

The 125-foot-by-125-foot cell size was used throughout the model. While using larger cell sizes outside of the area of primary interest would allow the model to be spatially extended, there was no way to substantiate whether such large cells would cause overshoot prior to running the model. If overshoot did occur, it would cause errors to be propagated throughout the model. Since setting up a model of this size uses a large amount of labor and computer resources, the reduced risk associated with smaller peripheral cells seemed to outweigh the advantage of covering a larger area.

A constant head external boundary condition was used in the solute transport model. For this condition, SWIFT III sets a convective flux boundary condition on radionuclide transport, which was the transport option used in the transport modeling. The output contains a contaminant mass-balance that includes the cumulative flux across the external boundary.

21.2.2 Initial Conditions

The solute transport is modeled as a transient process using steady-state flow conditions. Initial uranium concentrations throughout the aquifer are set at the default value of zero, which is reasonably close to the average background value of about 0.4 ug/L. Uranium concentrations reported for observation wells that are not located downgradient from the FMPC are given in Table 21-3. These wells are located west of the FMPC, north of the FMPC, and in the vicinity of the village of Ross. The values 2.7 and 1.7 ug/L test as statistical outliers using the methods described by Grubbs (1969). When these are rejected, the highest remaining concentration is 1.1 ug/L. Values reported as less than 1 ug/L are not informative and were not included in any statistical calculations. Other nondetection limits in the table were included in the calculations at the limiting value, which would tend to cause the representative background concentration to be overestimated.

TABLE 21-3
BACKGROUND URANIUM CONCENTRATIONS

Well Number	Uranium Concentration, µg/L (through March 1989)
2026	< 1, 0.3, < 1, < 1
2036	< 1, 0.3, 0.3, < 1
2056	< 0.03, 0.18, < 1, < 0.1, < 1, 2.7
2057	0.30, < 0.25, < 1, 0.3, 0.1, 0.2
2105	< 0.07, < 0.04, < 1, < 1, < 0.1, < 0.1
2121	< 1, 0.9, 0.6, < 1
2123	0.30, < 1, 0.4, 0.3, < 1
3063	< 1, 0.4, < 1, 0.6
3099	< 1, 0.5, 0.6, < 1
3100	< 1, 0.5, 0.6, 1.1
4023	< 1, 0.6, 0.4, 1.7

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21.2.3 Input Data

Hydraulic conductivities, bedrock elevations, recharge rates, and pumping rates were imported from 2DFLOW. Since one cell in the flow model corresponds to four or more cells in the solute transport model, simple computer programs were written to create relevant parts of the solute transport model input files from the flow model input files. The smaller cell size allowed the Albright & Wilson Co. wells to be represented individually in different cells and the 225 gpm combined pumping rate was divided equally by assigning 112.5 gpm for each well. The flow model calibrated to the April 1986 data was used because this model was a better representation of average conditions of the groundwater system. As mentioned in Section 20.1.4, the model calibrated to the May 1988 data represents abnormal drought conditions.

The time increment initially used was 180 days. This increment satisfies the guideline for overshoot and undershoot at groundwater velocities up to about 0.5 foot per day. The sensitivity of the model to the time increment was tested by lowering it to 60 days. The result was that the distribution pattern of calculated uranium concentrations did not change, the mass balance ratio improved from 0.92 to 0.99, and computer execution time was greatly increased. The small increase in accuracy was not sufficient to justify the extra computing time in routine calibration runs. Many routine computer runs with the time increment set at 180 days had mass balances closer to 1. Shorter time increments would be recommended only for model applications where early-time transient results are important. Trials also showed that a convergence criterion (maximum difference in concentration at any cell between successive iterations) of 1 ug/L was adequate for calibration runs.

21.2.4 Model Calibration

Calibration of 2DSOL involved successive trials in which uranium source loading rates and the distribution coefficient were modified within acceptable ranges to approach a match between observed and calculated uranium concentrations. No direct information on the rate of seepage of uranium to the aquifer was available. Consequently, source loading rates were made part of the calibration of the model. The procedure to generate the initial estimates of source loading involved the following steps:

- Calculating an initial estimate of the total amount of uranium present in the plumes.
- Reviewing available documents containing information related to possible source locations and uranium discharge rates.
- Choosing source areas based on the above review.

- Estimating the proportion of the uranium in the plume south of the Storm Sewer Outfall Ditch, and assigning that uranium to the Storm Sewer Outfall Ditch source area and Paddys Run immediately downstream from the Storm Sewer Outfall Ditch.

- Assigning the remaining uranium to other potential source areas, as follows:

  - Dividing the discharge time into periods when uranium discharge to the aquifer was probably significantly different from preceding and succeeding periods.

  - Prorating the amount of uranium from the various sources to the time periods.

Calculation of initial estimates of the amount of uranium in the plumes was generated from an isoconcentration plot of observed uranium data. The volume of the aquifer was estimated by use of planimetry. A porosity of 0.25 was used; and the distribution coefficient was taken as 0.005 ft^3/lb, which corresponds to the retardation factor of 3.5 selected for use in the preliminary analytical solute transport modeling. The distribution coefficient is a representation of the partitioning of the uranium between the soluble form and that adsorbed onto the solid matrix. (See Freeze and Cherry 1979, for a detailed definition.) An estimate of the uranium historically removed by pumping the Albright & Wilson Co. wells was also made and added to the estimated amount in the plume to obtain an initial estimate of the total amount of uranium to be introduced into the aquifer by the various source areas. This initial estimate was 4,749 pounds. The estimate was only for initiating calibration and was expected to be superseded by values generated during model calibration. Although these initial estimates could be very inaccurate due to a lack of direct information, it was felt that starting with logically derived estimates would tend to yield better results after calibration than starting calibration with completely arbitrary estimates.

The documents reviewed to obtain information related to source locations and loading rates included Dames and Moore (1985); Eye (1961); Facemire (1959); Geotrans (1985); Hartsock (1960); Spieker and Norris (1962); Thatcher and Phoenix (1987); U.S. Energy Research and Development Administration (ERDA) (1977); and WMCO (1987). The potential source areas chosen for inclusion in the model were the Storm Sewer Outfall Ditch area (including the fly ash piles and Southfield), Paddys Run downstream from the Storm Sewer Outfall Ditch, Paddys Run between the Storm Sewer Outfall Ditch and the Waste Storage Area, the Waste Pit Area, the Production Area, the silos area, the biodenitrification surge lagoon area, the Lime Sludge Pond area, and the abandoned incinerator area at the sewage treatment plant. The biodenitrification surge lagoon and the Lime Sludge Pond areas were assigned zero discharge rates in the initial run. No record of

radiological materials being placed in the Lime Sludge ponds was found, and there is no significant radioactivity in the Lime Sludge ponds (Thatcher and Phoenix 1987). The biodenitrification surge lagoon has a double liner system that should prevent significant loss of uranium to the underlying glacial overburden, and no significant amount of uranium would be expected to pass through the overburden and reach the Great Miami Aquifer within the short period of operation. Figure 21-7 shows the locations of the source cells used in the models.

The proportion of uranium in the plume south of the Storm Sewer Outfall Ditch was estimated from the isoconcentration plot. The assignment of loading cells for the uranium in this plume area to the Storm Sewer Outfall Ditch and the reach of Paddys Run just below the Storm Sewer Outfall Ditch was based on groundwater vector maps from 3DFLOW, which showed water moving from these areas toward the observed location of the plume.

The period of loading was divided into four source loading periods in the model. These periods correspond to periods of time when uranium-bearing water discharged to the Storm Sewer Outfall Ditch and Paddys Run was estimated to be significantly different from preceding and/or succeeding periods. The time periods were established by using estimated amounts of uranium discharged to the Storm Sewer Outfall Ditch as presented in Table 21-2. The first period is 1952 through 1958 and represents the early years of plant operation, when uranium discharge rates to the Storm Sewer Outfall Ditch were relatively low and only two of the waste pits had been constructed. The second period is 1959 through 1966, an eight year period when the rate of uranium discharge to the Storm Sewer Outfall Ditch was greater than during any other period and two more waste pits were put into operation, including Waste Pit 3. The third period is 1967 through 1975, a nine-year period when discharges to the Storm Sewer Outfall Ditch were moderate during the preceding period and Waste Pit 3 was closed. The fourth period is 1976 through 1988, a 13-year period of lowest total discharges to the Storm Sewer Outfall Ditch. Retention basins were constructed at the Storm Sewer Outfall Ditch near the end of this period. These periods were for initial calibration runs and were changed as required for calibration of the model.

Initially, the percentages of the total uranium discharged to the Great Miami Aquifer during the source loading periods were 8, 60, 27, and 5 percent for Periods 1, 2, 3, and 4, respectively. These amounts were later superseded by calibration of the solute transport models. They were only used as reasonable starting values to begin the calibration runs.

Uranium data collected from monitoring wells beginning in 1985 and continuing through August 1989 were used for calibrating the model. These data were from FMPC environmental monitoring

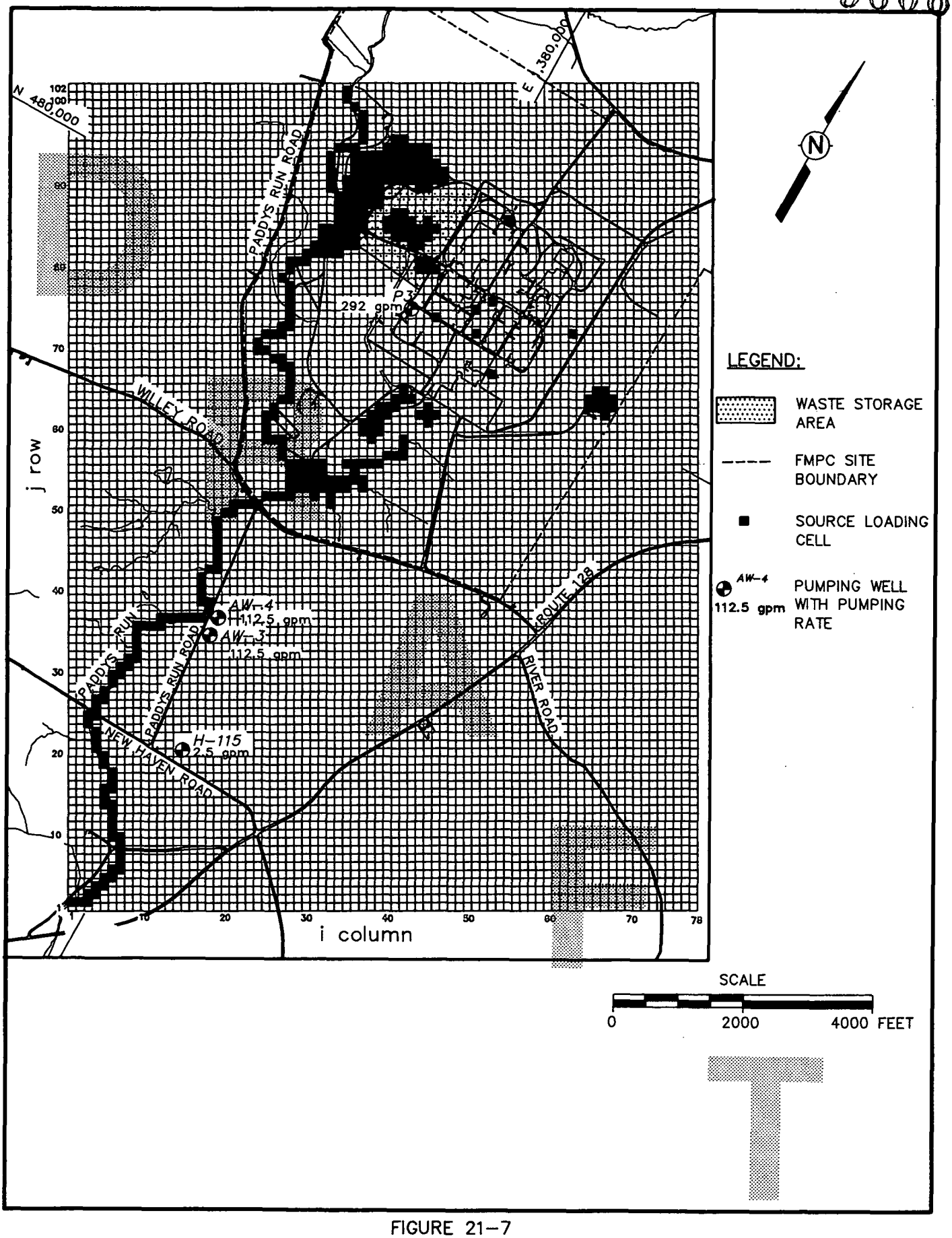

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FIGURE 21-7
SOLUTE TRANSPORT MODEL
SOURCE LOADING LOCATIONS AND
PUMPING WELL RATES

data from NLO and WMCO, RCRA Sampling Rounds Nos. 1 through 5 (Dames and Moore 1986a, 1986b, 1987a, 1987b, 1987c), IT data (IT Corporation 1986b), RI/FS data, and data from Albright & Wilson Co. (1986).

No general trend in uranium concentration variation with time was found in these data. That is, the observed concentration in each well was sufficiently stable that any general, area-wide, temporal trend that may exist was too small to separate from the scatter of the data. Consequently, the data at each well were treated as random sampling from a normal distribution, and calibration was based on this type of data. The scatter of the data at each well is thought to be due to short-term variations in recharge and source loading rates, and geochemical variability of the aquifer system combined with minor temporal variations in flow direction and rate.

The objective of the calibration was to reduce the difference between the uranium concentration calculated by the model at each well and the mean concentration of the actual population. The population means for the wells are not known; however, for wells where many concentrations have been measured, the sample means were assumed to be close to the population means. For wells where only a few concentrations have been measured, the sample means may be very different from the population means. Consequently, an adequately calibrated model might show large residuals at wells where the number of measurements are small, and some large residuals were allowed under such conditions in the 2DSOL model.

The criteria for calibration were: (1) to avoid bias indicated by too many residuals having the same sign; (2) to avoid clustering of residuals of the same sign; and (3) to avoid creating residuals that were too high to be acceptable representations of actual field conditions. No attempt was made to closely match each mean concentration because this would involve matching random variation rather than the underlying representative concentration. Furthermore, the lateral concentration gradients in the South Plume are very high. Consequently, a slight difference between natural and simulated isoconcentration lines can cause a large difference between measured and simulated concentrations at a point, even though the overall concentration distributions within the plumes are very similar.

During the 2DSOL calibration, the longitudinal dispersivity was held constant at 100 feet and the transverse dispersivity was held constant at 10 feet. The use of 100 feet for longitudinal dispersivity and 10 feet for transverse dispersivity was based on information on dispersivity in the scientific literature, as described in Section 21.1.2. Higher ratios (1/5, for example) are also reported, but since SWIFT III does not differentiate between lateral and vertical dispersivity and

vertical dispersivity tends to be low, the value 1/10 was considered to be most representative. The use of scale-dependent dispersivity values was necessary in this model because it is not possible to map or simulate the minor spatial variations in hydraulic conductivity which are thought to cause scale-dependency.

The calibration included trials with distribution coefficients of 0.005, 0.026, 0.0063, and 0.010 ft^3/lb corresponding to retardation factors of 3.5, 14, 4.13, and 6. Uranium source loading rates were also varied. The best calibration obtained without changing the longitudinal dispersivity from 100 feet was obtained using a distribution coefficient of 0.010 ft^3/lb; i.e., a retardation factor of 6. This calibration was considered sufficient for the purposes of 2DSOL; further refinement was postponed until the calibration of 3DSOL. A map of the plume produced by this transitional model is presented in Figure 21-8. This map shows the concentrations calculated at the end of the last time period (1978-1988) used in the model.

Experience with this model showed it to be most sensitive to the value of the uranium distribution coefficient and to the amount and location of source loading. The results of the trials showed that increasing the distribution coefficient shrinks the area of the plume and changes the concentrations in the plume if other factors are held constant. Increasing source loadings at all source cells increases the area of the plume and the concentrations if other factors are held constant. Consequently, various combinations of the source loading and the distribution coefficient can yield similar plumes. However, when the dispersion coefficient is held constant, the distribution coefficient controls the concentration gradients in the plume and the choice of parameters that yield a good calibration is limited.

21.3 THREE-DIMENSIONAL SOLUTE TRANSPORT MODEL

The purpose of the three-dimensional solute transport model (3DSOL) was to provide a useful representation of the transport of uranium in the groundwater system in the vicinity of the FMPC. Once achieved, the model could then be used to predict the future migration of uranium under a no-action condition and the effects of various remedial alternatives for dealing with the uranium contamination. The 3DSOL model is a better representation of the groundwater system than 2DSOL for the following reasons:

- It allows source loading to be applied to the upper layer of cells only, and provides a better representation of the effects of adding the contaminant at the water table where it will spread downward as well as laterally as it moves away from the sources. Consequently, it allows the concentrations near the bottom of the simulated aquifer to be smaller than concentrations at the top, which is the known condition in the natural aquifer.

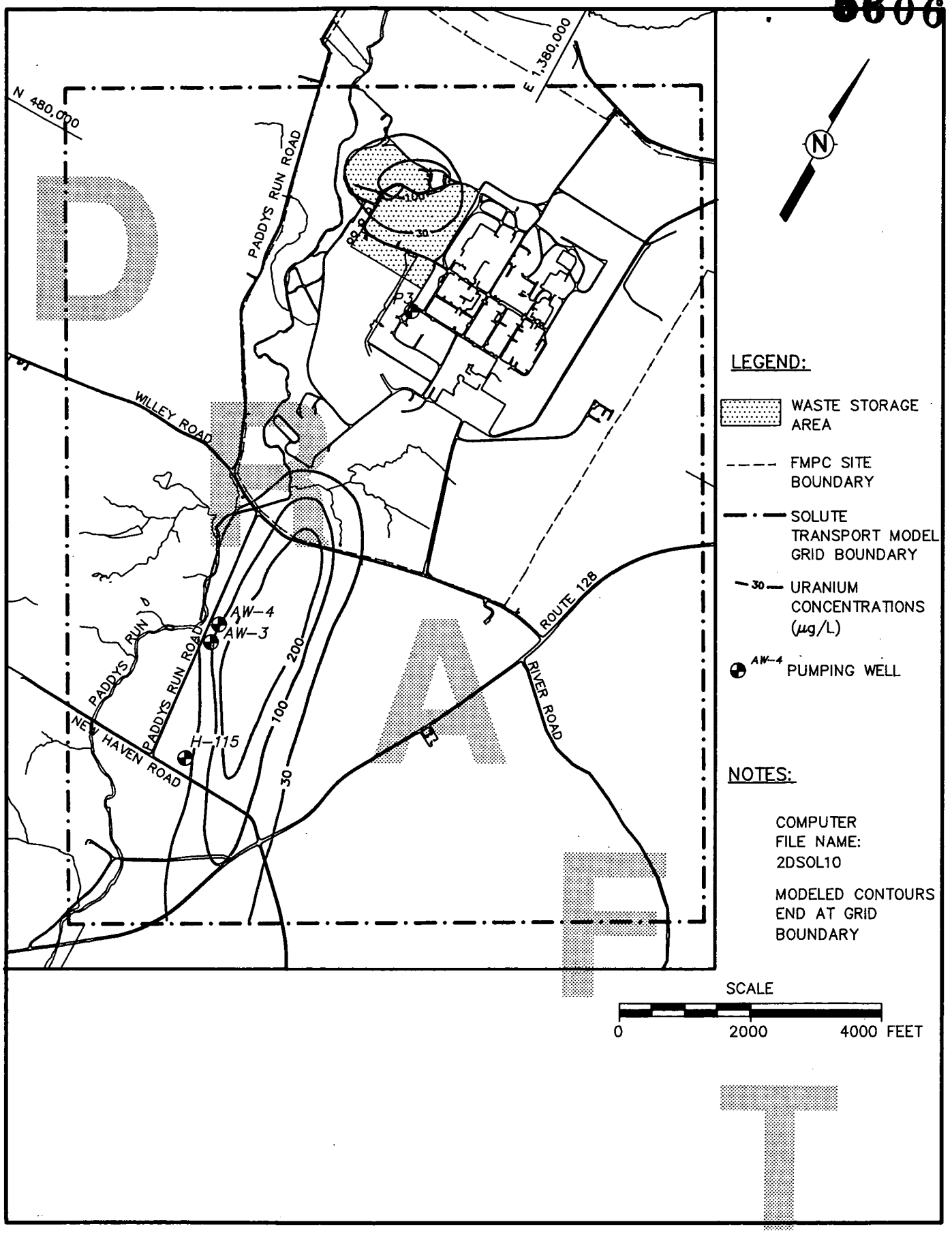

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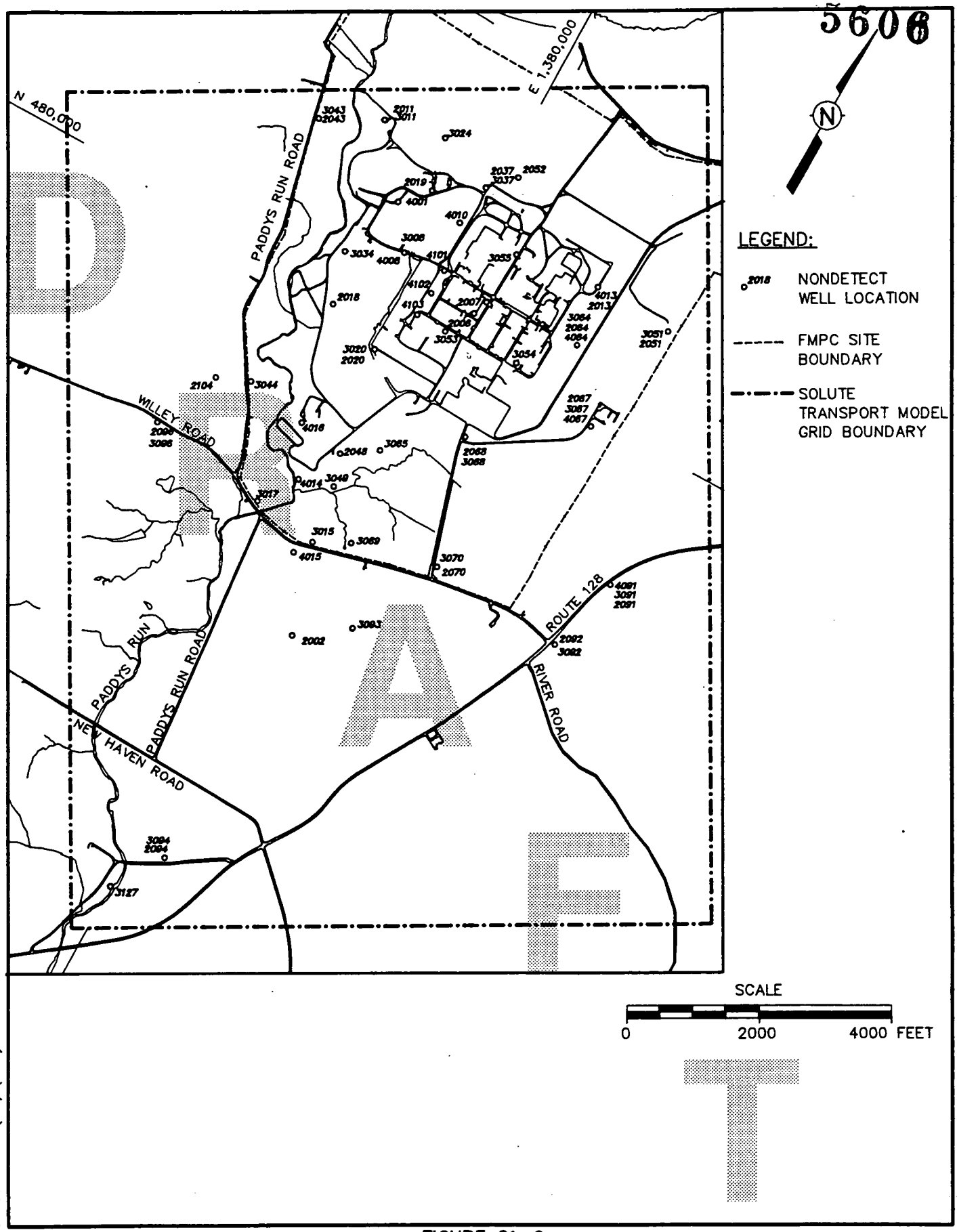
FIGURE 21-8
2-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
PRESENT CONDITIONS



303317-A177 (GW) (PGH)

FIGURE 21-9
3-DIMENSIONAL
SOLUTE TRANSPORT MODEL
NONDETECT WELL LOCATIONS

- It contains an explicit representation of the clay interbed that is present beneath the FMPC site. The bed impedes the downward movement of the contaminant, and it affects groundwater flow directions.

- It provides better estimates of source loading rates. In the two-dimensional model, concentrations had to be uniform with depth in the aquifer; whereas, in the three-dimensional model, concentrations decrease with depth. Most of the monitoring wells used for calibration are in the upper part of the aquifer. Consequently, the calibrated two-dimensional model contains more uranium in any vertical column of aquifer than the calibrated three-dimensional model; and the simulated plume in the two-dimensional model contains more uranium than the plume in the equivalent three-dimensional model. Since the mass of uranium in the plume has to balance with the mass of uranium entering the plume at the sources, the source loading rates in the two-dimensional model tend to be too high.

- It provides a better tool for examining the feasibility of removing uranium by using extraction wells than 2DSOL, because it will not tend to exaggerate the amount of uranium in the aquifer, as noted above. It also provides a way to estimate the depth required for extraction wells because it can simulate the effectiveness of partially penetrating wells in pulling water up from the lower layers.

21.3.1 Model Grid and Boundary Conditions

3DSOL used the same grid as 2DSOL, except that five layers were added in the vertical direction. These layers are the same as those of 3DFLOW and were imported from corresponding cells, of 3DFLOW. The resulting model contained 39,780 cells, which caused the importation of data from the preceding models to be complex. The computer files to be generated were very large; therefore, data manipulation and transfer had to be done by developing computer programs to produce various parts of the 3DSOL input file. Solute transport execution times of 3 to 12 hours on a Prime EXL325 computer were required for each calibration run.

Boundary heads were imported directly from 3DFLOW. The head at the boundary face of each boundary cell in the solute transport model was taken directly from the cell in the flow model that contained the boundary. This was done for each layer so that the boundary head in the transport model varied with depth in accordance with the results of the flow model. Again, the constant head boundary condition resulted in a convective flux boundary condition for radionuclide transport. Initial uranium concentrations throughout the aquifer were set at zero, as in 2DSOL.

21.3.2 Input Data

Hydraulic conductivities, bedrock elevations, recharge rates, and pumping rates were imported from 3DFLOW. However, bedrock elevations were changed near Paddys Run along the western edge of the Paddys Run Outlet. The purpose of this change was to more accurately simulate the effects of

Paddys Run. In the previous models, Paddys Run was simulated as crossing bedrock cells in this area, even though it is actually on thin alluvium. The latter condition could not be more realistically simulated until accommodated by the small cell size and layering in 3DSOL. The smaller cell size permitted simulating thin alluvium on a gradually sloping bedrock surface beneath Paddys Run. After this change was made, a special computer program was used to compare hydraulic heads calculated in 3DSOL with the corresponding heads in 3DFLOW to demonstrate that simulated flow was not significantly affected in the model, especially in the plume areas.

Along with the change in bedrock around Paddys Run, the recharge rates along Paddys Run were also refined. In transforming the input file from the flow model grid with large cell sizes to the solute transport model grid with smaller cell sizes, much of the area occupied by Paddys Run in the floor model was lost when the stream was more accurately depicted by the smaller cells. To compensate for this, the rate of recharge to the smaller stream cells was increased until the total volume of recharge to Paddys Run matched that used in the flow model. This resulted in increasing the recharge rate from 14 in/yr to 32 in/yr to compensate for the loss in area in Paddys Run. The hydraulic heads in the model were checked again after this change was made.

Well No. H-115 (Rutgers-Nease chemical plant well) was added to the model and assigned a pumping rate of 2.5 gpm. Again, the hydraulic heads in the model were checked to verify that the new well caused no significant change.

The time increment used in the transient solute transport modeling was maintained at 180 days.

21.3.3 Model Calibration

Calibration of 3DSOL involved successive trials in which uranium source rates, distribution coefficient, longitudinal dispersivity, and transverse dispersivity were varied. It was found that decreasing dispersivity increased concentration gradients by decreasing the spread of the plume, and the effect was somewhat similar to the effect of increasing the distribution coefficient. This condition allowed an acceptable calibration to be achieved through a range of distribution coefficients and dispersivity values by finding a balance between source loading rates, distribution coefficient, and dispersivities. Consequently, additional information became needed to find the most representative combination of these factors. During early stages of the modeling, additional information became available from preliminary results of geochemical modeling. During later stages of the solute transport modeling, the geochemical modeling had been completed and final results were available (IT Corporation 1989). This progressive change in the amount of geochemical data available resulted in the distribution coefficient being progressively increased to

0.016, which corresponds to a retardation factor of 9, and then to 0.022 ft^3/lb, which corresponds to a retardation factor of 12. A preliminarily calibrated model, based on a distribution coefficient of 0.016 ft^3/lb, was used for the determination of locations of additional monitoring wells required to characterize the contaminant plume, and for the evaluation of an interim removal action to mitigate contaminant migration pending final action under the RI/FS. More recent RI/FS work is using the final, more-refined calibration that was accomplished with a retardation factor of 12.

Preliminary Calibration

The preliminary calibration was based on early geochemical results, which indicated that the dissolved uranium is in complexes with neutral and negative charges. Such charges imply low adsorption and low distribution coefficients. Note that Walton (1985, p. 50) estimates retardation factors close to 1 for chlorides and nitrates and less than 10 for arsenic, which are present in groundwater as negative ions. Consequently, a representative model would have a low distribution coefficient while maintaining a reasonable dispersivity. Information in the literature suggests that a scale-dependent dispersivity of less than 40 feet would be marginal in a system as large as the one treated in the present model. For example, Walton (1985) shows representative longitudinal dispersivities for areal models of alluvial sediments and glacial deposits to be between 39 and 200 feet. The strategy, therefore, for obtaining a representative model for the FMPC was to use a low distribution coefficient that would allow calibration with a longitudinal dispersivity greater than 40 feet.

The calibration process was begun by importing initial source loading rates, distribution coefficient, and dispersivities from the transitional 2DSOL model. The monitoring well data used for 2DSOL calibration were also used for 3DSOL, and additional new data were used as they become available. The most recent data used for calibration were collected in August 1989. For calibration, only wells from which all samples contained uranium above the detection limit were included in calculations of nondetect means and other statistics. Wells that yielded any samples with nondetected uranium were treated as locations where calculated concentrations should be minimized, without attempting to actually match a mean concentration. Excluding these wells from calculations eliminated the problem of what concentration value to assign to such samples. Figure 21-9 shows these "nondetect" wells. For the purposes of calibration, the measured concentration data for each well with detectable concentrations were tested for outliers following the methods described by Grubbs (1969). Concentrations that tested as outliers at the 2.5 percent significance level were not included in the calculation of means used for model calibration. The purpose of eliminating the outliers was to make the model more representative by not including unlikely data in the

calibration. The statistical procedure provides an objective method of identifying the unlikely data. The rejected data are listed in Table 21-4. The wells used in the calibration are shown in Figure 21-10. In the calibration, the observed concentrations were compared with the concentrations calculated for the model layer in which the well was completed.

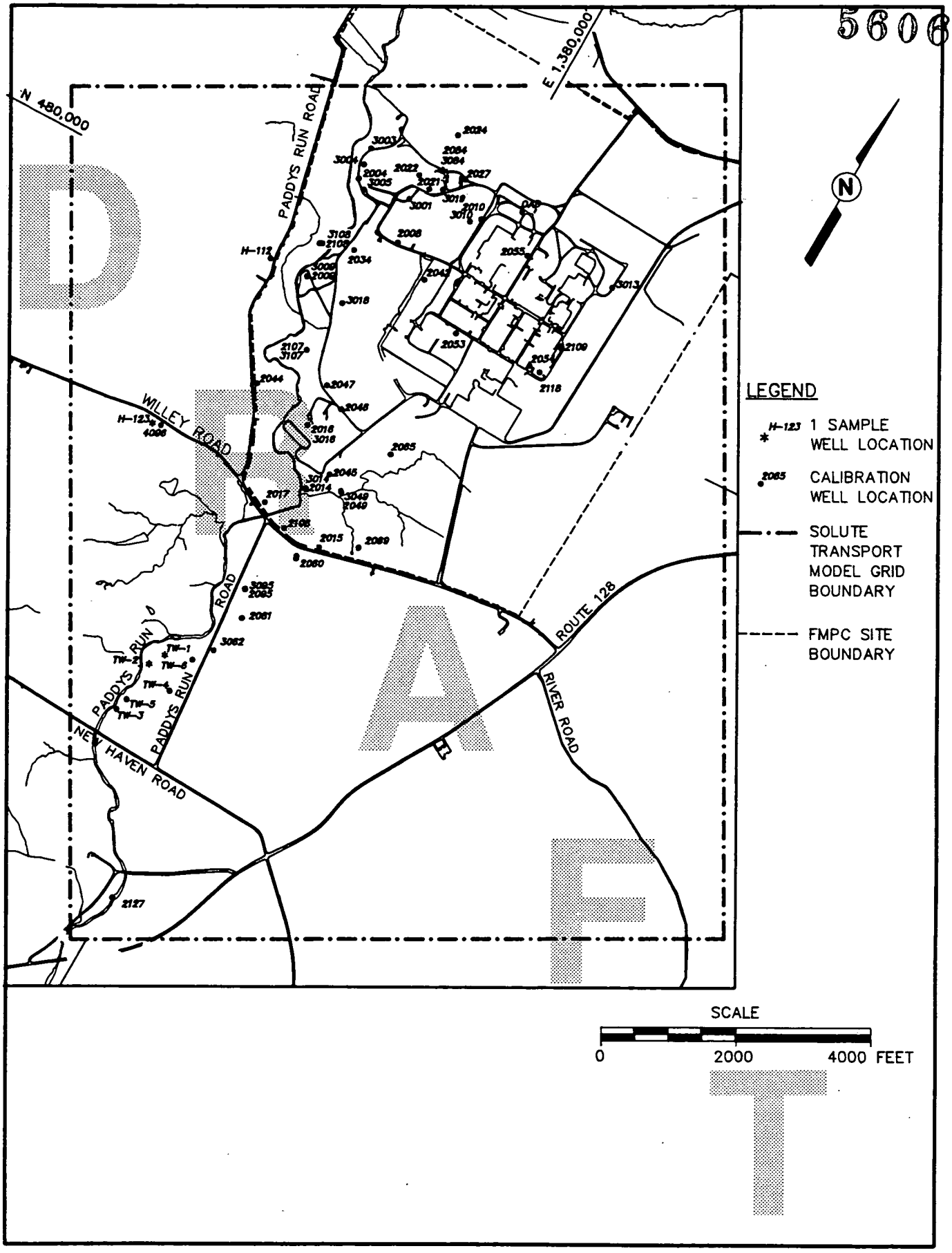
Attempts to calibrate the model using a distribution coefficient of zero (i.e., where there is no adsorption) were not successful. A simulated plume that matched field data could not be produced because the simulated plumes proved to be too broad and too long even with longitudinal dispersivity as low as 30 feet. Furthermore, most uranium exited the south boundary of the model in all runs. This plume behavior is not in accordance with monitoring well data. A residential well located south of the intersection of Route 128 and New Haven Road, in the direction of movement of the plume, yielded a uranium concentration below a detection level of 1 ug/L, which is in the range of background levels. If a plume had extended to this area, the elevated concentration at this residential well would still be present because the source is still present. Consequently, the evidence indicates that the plume has never extended to this area. Runs attempted with longitudinal dispersivity set lower than 30 feet (15 and 10 feet) diverged. The iterative procedure used to solve the simultaneous equations does not approach the solution. Another observation for the zero distribution coefficient case was that when the higher concentration values in the simulated plume were made to approximate observed values, the highest concentrations in successively deeper layers shift southward. No evidence for such shift exists in the observed concentration profile. The evidence from numerous computer runs using a distribution coefficient of zero indicates that an acceptable calibration cannot be reached, and the solute transport model supplies convincing evidence that significant adsorption of uranium occurs in the natural system.

As noted in Section 21.2.1, any longitudinal dispersivity less than 62.5 feet violates the guideline on grid size needed to avoid overshoot and undershoot. All runs with longitudinal dispersivity less than 62.5 produced some negative concentrations in low concentration cells, interpreted as undershoot. This undershoot appeared to have only a minor effect on the plume concentrations, because the plumes produced at slightly higher dispersivities with no negative concentrations were very similar to the plumes containing evidence of undershoot. Experience with these runs suggests that significant undershoot is easily identified in this model by the appearance of negative concentrations. The observation that undershoot is identifiable in the results has also been made by at least one other modeler (Frind and Palmer 1980).

TABLE 21-4

OUTLIERS ELIMINATED FROM URANIUM CONCENTRATIONS
DATA USED FOR CALIBRATION

Well Number	Cell I	Cell J	Cell K	Initial Number of Observed Concentrations	Initial Observed Concentrations Minimum	Maximum	Mean	Concentrations Rejected as Outliers
EMR-13	31	40	1	47	0.3	1.9	0.60	1.9, 1.3
EMR-14	59	38	1	48	0.34	1.7	1.10	0.34
EMR-16	49	57	1	47	0.40	2.8	0.7	2.8, 1.5, 1.4
EMR-20	19	15	4	30	0.1	0.8	0.3	0.8
EMR-21	24	17	2	48	0.2	6.1	0.6	6.1
EMR-22	22	62	1	44	0.6	2.3	1.1	2.3
2014	29	55	1	9	17.0	85.0	34.2	85.0
2015	30	48	1	9	154	401	201.2	401
2044	23	67	1	4	1	33.0	9.3	33.0
2060	28	47	1	56	105	591	248	591
2061	21	39	1	58	208	476	290	476
2027	48	92	1	4	6.0	7.0	6.3	7.0
3001	41	89	2	4	13.0	15.0	14.5	13.0
3005	36	90	2	4	3.0	10.0	4.8	10.0
3013	65	78	2	6	4.0	490	86.8	490
3062	18	35	2	47	31.0	110	51.2	110



303317-A176 (GW) (PGH)

FIGURE 21-10
3-DIMENSIONAL
SOLUTE TRANSPORT MODEL
CALIBRATION WELL LOCATIONS

Runs with the distribution coefficient set at 0.002 and 0.004 ft^3/lb (retardation factors of 2 and 3, respectively) also resulted in most of the uranium loaded at the source cells exiting the boundary of the model by the end of the 37-year simulation period. That is, the main body of the plume passed the boundary. Again, the background uranium concentration recorded in Well 5382 indicates the plume has never extended beyond its present terminus. Therefore, the evidence shows that a distribution coefficient of 0.004 ft^3/lb or less will not produce a representative model. Conversely, simulation runs at distribution coefficients at 0.010 ft^3/lb (retardation factor = 6) and higher result in less than 20 percent of the input uranium flowing out of the system. This amount of adsorption causes a plume to form in the South Plume area that resembles the observed plume.

Attempts to calibrate the model using a distribution coefficient of 0.010 ft^3/lb began by importing source loadings from the two-dimensional solute transport model, which was roughly calibrated with this distribution coefficient. Runs with longitudinal dispersivity of 100 feet and transverse dispersivity of 10 feet resulted in a plume with too much spread and was centered too far southeast. These results were judged to indicate the distribution coefficient was too low, and calibration moved on to a coefficient of 0.016 ft^3/lb that was consistent with interim findings of the geochemical program.

Calibration runs with the distribution coefficient of 0.016 ft^3/lb (retardation factor = 9) began with the longitudinal dispersivity set at the preferred value of 100 feet and the transverse dispersivity set at 10 feet. The plume produced by using 100 feet for the longitudinal dispersivity had too much spread; however, it was close enough to matching field concentrations that runs were made with lower dispersivity values in an attempt to refine the calibration without requiring a higher distribution coefficient. An acceptable calibration was accomplished. The final calibrated run had a longitudinal dispersivity of 50 feet and a transverse dispersivity of 1 foot. This reduction in dispersivity was needed to make the concentration gradients high enough (that is, make the shape of the plume sharp enough) to match the high concentrations near the axis of the plume, while matching very low concentrations at the edge of the plume a short distance east of the axis. The 50/1 dispersivity ratio was needed to increase the sharpness of the plume and to get a decrease in concentration with depth near the source areas, since the model uses transverse dispersivity for vertical dispersivity. This calibrated model is called 3DSOL9.

A comparison of the mean observed concentrations at 41 individual wells and the calculated concentrations in the model cells containing the wells is presented in Table 21-5. As mentioned above, the mean of observed concentrations at a well where only a few concentrations have been measured may not be an accurate estimate of the most representative value for that well;

TABLE 21-5

INLIER CONCENTRATIONS
CALCULATED BY THE THREE-DIMENSIONAL FINITE-DIFFERENCE
SOLUTE TRANSPORT MODEL
RETARDATION FACTOR 9

Well No.	Cell I	Cell J	Cell K	Number of Observed Concentrations	Observed Concentrations (µg/L) Minimum	Maximum	Mean	Calculated Concentrations (µg/l)
EMR-13	40	42	1	45	0.3	1	0.55	0.1
EMR-16	49	57	1	44	0.4	1.1	0.59	0.1
2004	34	83	1	4	7	18	11	0.4
2008	40	84	1	4	5	22	15.3	11.7
2009	29	80	1	4	17	24	20.5	18.5
2010	48	87	1	4	2	21	12.3	16
2014	29	55	1	8	17	35	27.1	45.1
2015	30	48	1	8	154	200	176	168
2016	29	62	1	8	12.0	22.0	18.1	29.1
2022	42	92	1	4	4	10	6.3	19.4
2024	47	97	1	4	2	6	4.5	0.1
2034	34	83	1	4	15	24	20.5	21.5
2042	43	80	1	4	2	4	2.8	6.9
2044	23	67	1	3	1	2	1.33	0.3
2045	31	56	1	3	278	341	301	19.7
2046	10	32	1	3	232	850	464	23.3
2047	11	32	1	3	9	15	11.3	19.2
2049	33	54	1	6	2.5	175	78.1	14.6
2054	55	69	1	2	21	23	22	1.5
2060	28	47	1	55	105	410	243	144
2061	21	39	1	57	208	378	287	180
2065	39	59	1	5	7	11.5	9.2	0.7
2069	35	38	1	5	6	14.8	11.6	5.3
2084	45	93	1	4	16	21	18.3	14.4
2106	26	49	1	2	16	61	38.5	75.4

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TABLE 21-5
(continued)

Well No.	Cell I	Cell J	Cell K	Number of Observed Concentrations	Observed Concentrations (µg/L) Minimum	Maximum	Mean	Calculated Concentrations (µg/L)
2107	28	71	1	2	32	37	34.5	0.1
2109	59	71	1	2	2	34	18	0.7
2118	57	68	1	2	9	16	12.5	1.1
2127	5	5	1	3	6	37	19	0.4
3003	36	95	2	3	3	4	3.3	0.1
3009	29	80	2	4	2	4	2.8	1.4
3010	49	87	2	4	15	20	17	4
3019	45	90	2	4	20	45	32	4.4
3014	29	54	2	9	19	38	26.7	13.5
3016	29	62	2	8	7	11	8.8	6.5
3062	18	35	2	46	31	76	49.9	42.5
3084	45	93	2	4	23	218	90.5	0.9
3095	21	42	2	4	4	13	7	32.3
4096	11	62	5	4	0.5	2	0.98	0.1
H-112	24	82	4	2	0.43	0.48	0.46	0.1
TW-3	7	28	2	3	1.1	2.1	1.53	0.1
TW-5	8	29	1	4	0.45	0.85	0.58	0.1

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consequently, such means were not matched more closely than their representativeness warranted. Instead, the model calibration emphasized (1) avoidance of excessive clustering of positive or negative residuals (observed mean minus calculated concentration), and (2) keeping the absolute values of the residuals reasonably low.

The clustering criterion is examined by calculating the unit normal deviate from a modification of Moran's I (IT Corporation 1987). A value greater than 1.645 indicates a nonrandom distribution of residuals at the 0.05 level of significance. The optimal value is zero. The value calculated for the calibrated run was 0.144.

The reasonably low absolute values of residuals corresponding to the second calibration criterion may be examined by using a statistical procedure to determine whether the calculated concentration at an observation well differs by an improbable amount from the mean concentration observed at the well. The statistical testing procedure used for this purpose followed methods described by Grubbs (1969). This method of testing goodness of fit is more informative than simply measuring deviations from means because it includes uncertainty related to the representativeness of the observed concentrations at a given well. It allows less deviation from means of large samples and/or samples with little variation in values, and greater deviation from means based on only a few samples and/or means based on samples with a large variation in values. As is the case when simply comparing computed concentrations with observed means, judgment must still be used in interpreting the results with regard to goodness of fit. However, the statistical procedure provides information upon which to base that judgment.

Calculated concentrations that tested as being improbable in the real system at a significance level of 2.5 percent are listed in Table 21-6 along with the observed mean, the number of measured concentrations, and the range of the observed concentrations. These results show that only one calculated concentration within the main plume was significantly different from the observed mean for the well in that cell (Well 2095).

The remaining 15 wells with calculated concentrations that exhibit statistically significant deviations from the observed means (Table 21-6) are all located peripherally to the main plume. Since these are all cases where both the observed and calculated concentrations are very low, the actual difference between observed and calculated concentrations is also low. The effect on the overall fit of the model is negligible, and it is not practical to attempt to match these data more closely. Such additional calibration might be very difficult for the following reasons:

TABLE 21-6

OUTLIER CONCENTRATIONS CALCULATED BY THE THREE-DIMENSIONAL FINITE-DIFFERENCE SOLUTE TRANSPORT MODEL RETARDATION FACTOR 9

Well Number	Cell I	Cell J	Cell K	Number of Observed Concentrations	Observed Concentrations (µg/L) Minimum	Maximum	Mean	Calculated Concentrations (µg/L)
EMR-11	22	53	1	46	0.9	1.8	1.46	0.1
EMR-14	59	38	1	47	0.7	1.7	1.09	0.1
EMR-20	19	15	4	30	0.1	0.8	0.34	7.6
EMR-21	24	17	2	47	0.2	0.7	0.43	3.6
EMR-22	22	62	1	43	0.6	1.6	1.07	0.1
EMR-29	17	46	4	12	1.7	2.2	1.9	0.1
EMR-30	18	66	4	19	0.4	0.8	0.6	0.1
2027	48	92	1	4	6	6	6	5.3
2095	22	43	1	4	146	202	173	349
3001	41	89	2	3	15	15	15	3.6
3004	35	93	2	4	10	13	11.3	0.1
3005	36	90	2	3	3	3	3	0.9
3013	65	78	2	3	4	4	4	0.1
3108	30	84	2	2	31	33	32	0.1
TW-4	13	31	1	3	1.71	1.92	1.82	0.1
TW-6	15	34	4	4	1.08	1.18	1.13	0.1

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- Concentration gradients on the flanks of the southern plume are high, and the finite-difference approximation may be causing a slight undershooting of the real values where the concentrations are approaching zero near the edge of the main plume.

- Some of the statistically significant deviations may be caused by rounding off the concentrations reported for the samples. Some observed concentrations are reported as one-digit integers, which tends to hide the true variation of the concentration and cause the statistical procedure to underestimate the range of likely concentrations.

- Wells 2096, 2108, 3108, 2044, 3044, 2104, 2017, 3017, and 2127 located west of Paddys Run yielded observed concentrations that were above background. The wells are at nine different locations. The highest concentration found in any of them was 37 ug/L in Well 2108. These elevated concentrations are thought to be caused by temporary seasonal reversals in hydraulic gradient near Paddys Run. Such reversals could intermittently carry small amounts of uranium a short distance westward from Paddys Run, but they cannot be simulated by the steady-state flow model.

It is noteworthy in Table 21-6 that Wells 2045 and 2046 have calculated concentrations considerably below their mean observed concentrations. The calculated concentrations for these wells do not test as outliers due the small number of observed concentrations and the large range in the observed concentrations. The source of uranium in the wells remains uncertain and continues to be investigated in the RI/FS.

Final Calibration

The RI/FS geochemical investigation (IT Corporation 1989) determined that many uranium distribution coefficients calculated from results of an iron- and manganese-leaching procedure were higher than the distribution coefficient 0.016 ft^3/lb used in the preliminary solute transport modeling. Consequently, use of a higher distribution coefficient in the model would be consistent with the final results of the geochemical investigation, although not required by it. This geochemical finding allowed the distribution coefficient to be increased to 0.022 ft/lb (retardation factor 12), so that the longitudinal dispersivity could be increased back to the preferred value of 100 feet.

Values of the distribution coefficient greater than $0.022 ft^3/lb$ were not considered because the occurrence of uranium in neutral and negatively charged complexes indicates a low value should be preferred, as previously discussed in this section. Other evidence for a low distribution coefficient is that Well 2060 has yielded high uranium concentrations since February 1982. Particle tracking indicates that an unretarded particle would require over two years to move from the nearest source (Paddys Run) to this well. Since the FMPC had only been operating for 30 years by 1982, the

concentrations at this well imply a retardation factor less than 15. The available evidence points to a retardation factor greater than 9 and less than 15. The value of 12 is, therefore, a preferred value in the middle of the acceptable range.

Calibration at a distribution coefficient of 0.022 ft^3/lb (retardation factor of 12) was more refined than that at 0.016 ft^3/lb. As mentioned in Section 21.2.3, the calibration aimed at reducing the difference between the uranium concentration calculated by the model at each well and the mean concentration of the actual population. Consequently, an appropriate goal for calibration is to attempt to put as many calculated concentrations as possible within the 95 percent confidence interval of the true mean. Calculation of confidence intervals for the mean is described in Mood and Graybill (1963, pp. 251-253). This test is a more restrictive one than the outlier test used in the preliminary calibration. Consequently, its application to the final calibration resulted in a more refined model. Hereafter, the 95 percent confidence interval for the true mean concentration at a well will be called the "range."

The result of the calibration was that all of the calculated concentrations in Layer 1 were brought into range except the concentration at Well 2061 and the concentrations at a few wells upgradient from any known sources. These upgradient wells are 2108, 2024, and H-112. Several concentrations at wells screened in lower layers also could not be brought into range. The results are presented in Table 21-7. Figures 21-11 and 21-12 show the deviations of calculated concentrations from the mean of measured concentrations at the wells. Concentrations that are out of range are underlined. Uranium data from domestic wells with unknown screen depths have not been included in this analysis. However, they have been inspected to verify that they contain no concentrations that are inconsistent with the solute transport modeling results reported herein.

Since the uranium concentration decreases with depth, it is more important to match concentrations in Layer 1 than in lower layers, because this layer contains a large proportion of the total mass of uranium in the ground. The degree of matching obtained required detailed study of particle tracks connecting sources to monitoring wells and very refined adjustment of source loadings. Such adjustments had to bring concentrations at one well into range while not causing the calculated concentration at another well near the same track to fall outside of the range. The time of the source loading as well as its location had to be considered.

The concentration at Well 2061 could not be brought into range because the measured concentrations at that well are higher than the concentrations at Well 2095, which is upgradient on the same particle track. The wells are only about 375 feet apart. The concentration at Well 2061

TABLE 21-7

OBSERVED CONCENTRATIONS
95% CONFIDENCE INTERVALS FOR THE MEANS,
AND CALCULATED CONCENTRATIONS
FOR THE THREE-DIMENSIONAL FINITE-DIFFERENCE
SOLUTE TRANSPORT MODEL
RETARDATION FACTOR 12

Well No.	Cell I	Cell J	Cell K	Number of Observed Concentrations	Observed Concentrations (µg/L) Minimum	Maximum	Mean	95% Confidence Intervals for the Means (µg/L) Lower End	Upper End	Calculated Concentrations (µg/L)
2002	30	35	1	2	1	2	1.5	0	7.9	7.8
2004	35	92	1	23	5.8	10	7.6	7.1	8.1	8.0
2008	40	84	1	4	5	21.5	15.9	3.7	28.1	6.1
2009	29	80	1	4	19	27	23	17.6	28.4	21.4
2010	48	87	1	4	2	21	12.3	0	28.4	2.9
2014	29	55	1	12	17	125	49.1	23.4	77.1	64.3
2015	30	48	1	10	132	200	174	159	189	160.5
2016	29	62	1	10	12	30	20.4	15.7	23.2	20.9
2017	24	53	1	8	2.6	5.1	3.7	3.0	4.4	32.0
2021	43	90	1	10	1	21.1	6.7	1.8	11.6	7.1
2022	42	92	1	7	3.2	6	4.3	3.4	5.2	4.8
2024	47	97	1	4	2	6	4.4	1.7	7.1	0.1
2027	48	92	1	3	6	6	6	6.0	6.0	3.9
2034	34	83	1	4	15	24	20.5	13.6	27.4	17.1
2042	43	80	1	4	1.5	4	2.6	0.9	4.4	1.3
2044	23	67	1	2	1	1	1	1.0	1.0	1.7
2045	31	56	1	4	278	462	341	249	312	275.4

TABLE 21-7
(continued)

Well No.	Cell I	Cell J	Cell K	Number of Observed Concentrations	Observed Concentrations (µg/L) Minimum	Maximum	Mean	95% Confidence Intervals for the Means (µg/L) Lower End	Upper End	Calculated Concentrations (µg/L)
2046	32	64	1	4	232	907	575	0	1,302	21.2
2047	32	68	1	4	9	15	12.0	3.4	19.3	7.6
2049	33	54	1	7	2.5	437	129	0	163	57.9
2053	47	72	1	2	1	6	3.5	0	35.3	23.3
2054	55	69	1	2	21	23	22	9.3	34.7	15.8
2055	55	82	1	2	3	4	3.5	0	9.9	0.2
2060	28	47	1	92	105	410	246	235	256	236.0
2061	21	39	1	93	208	578	335	318	352	173.4
2065	39	59	1	7	8	13	10.3	8.7	11.8	11.3
2069	35	48	1	5	6	14.8	11.6	7.5	15.7	10.9
2084	45	93	1	4	16	21	18.3	14.7	21.8	15.7
2095	21	42	1	5	87	201.5	156.1	137	210	191.6
2106	26	49	1	3	16	89	55.3	0	324	209.1
2107	28	71	1	2	9	14	11.5	0	43.3	9.7
2108	30	84	1	2	32	37	34.5	2.7	66.3	1.6
2109	59	71	1	2	2	35.5	18.8	0	232	5.2
2118	57	68	1	2	9	16	12.5	0	57.0	6.5
2127	5	5	1	3	6	37	19	0	59.0	15.3
3001	41	89	2	23	7.5	15	11.0	9.9	12.1	4.7
3003	36	95	2	21	2.1	4.3	3.0	2.7	3.3	2.4

TABLE 21-7
(continued)

Well No.	Cell I	Cell J	Cell K	Number of Observed Concentrations	Observed Concentrations (µg/L) Minimum	Maximum	Mean	95% Confidence Intervals for the Means (µg/L) Lower End	Upper End	Calculated Concentrations (µg/L)
3004	35	93	2	4	10	13	11.3	9.3	13.3	3.2
3005	36	90	2	24	3	6.3	4.3	3.9	4.7	4.1
3009	29	80	2	24	0.8	4	1.8	1.5	2.2	5.8
3010	49	87	2	22	13.1	29.2	20.9	18.7	23.1	1.3
3013	65	78	2	10	1.4	21	11.4	6.0	16.8	0.1
3014	29	54	2	10	19	38	27.5	22.3	31.0	43.9
3016	29	62	2	9	7	13.4	9.3	7.7	9.9	19.2
3018	33	77	2	9	0.85	2.9	1.8	1.3	2.2	4.8
3019	45	90	2	9	19	67	33.4	21.6	45.2	3.7
3062	18	35	2	79	31	78	53.0	50.7	55.3	53.4
3084	45	93	2	4	14	218	87.5	0	237	3.7
3095	21	42	2	5	4	13	7.4	0.5	13.5	123.6
3107	28	71	2	2	1	2	1.5	0	7.9	4.5
3108	30	84	2	2	31	33	32	19.3	44.7	0.7
3125	18	38	2	2	70.9	88.1	79.5	0	189	52.3
4096	11	62	5	4	0.5	2	1.0	0	2.1	0.0
EMR-11	22	53	1	74	0.8	2	1.4	1.3	1.4	17.9
EMR-13	40	42	1	75	0.3	1	0.6	0.6	0.7	0.0
EMR-14	59	38	1	82	0.6	1.8	1.2	1.1	1.2	0.0

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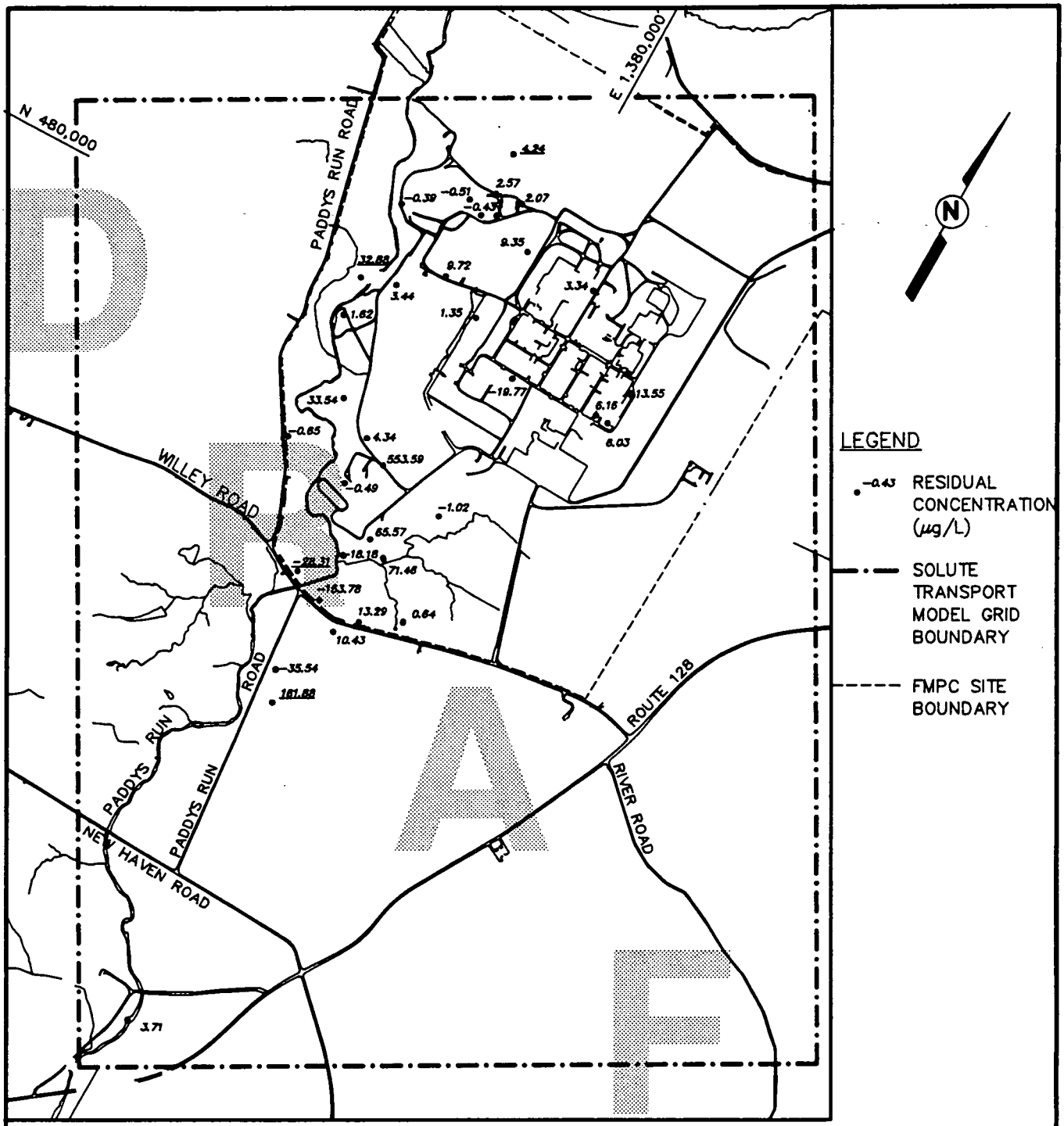
TAB[illegible]1-7
(continued)

Well No.	Cell I	Cell J	Cell K	Number of Observed Concentrations	Observed Concentrations (µg/L) Minimum	Maximum	Mean	95% Confidence Intervals for the Means (µg/L) Lower End	Upper End	Calculated Concentrations (µg/L)
EMR-16	57	27	2	78	0.3	1.5	0.7	0.7	0.8	0.0
EMR-20	19	15	4	53	0.09	0.6	0.2	0.2	0.3	9.4
EMR-21	24	17	4	79	0.2	0.8	0.4	0.4	0.5	13.0
EMR-22	22	62	1	52	0.6	1.7	1.1	1.0	1.1	0.4
EMR-30	18	66	4	19	0.4	0.8	0.6	0.5	0.6	0.0
H-112	24	82	1	2	0.43	0.48	0.5	0.1	0.8	0.0
TW-3	7	28	4	3	1.1	2.14	1.5	0.2	2.9	0.0
TW-4	13	31	1	3	1.71	1.92	1.8	1.6	2.1	1.6
TW-5	8	29	1	4	0.45	0.85	0.6	0.3	0.9	0.8
TW-6	15	34	4	4	1.08	1.18	1.1	1.1	1.2	6.2
OAB	58	87	3	9	0.7	4.8	1.7	1.0	3.0	0.0



NOTES:

RESIDUAL IS THE OBSERVED MINUS COMPUTED URANIUM CONCENTRATION

UNDERLINED VALUES ARE OUTSIDE OF 95% CONFIDENCE RANGE

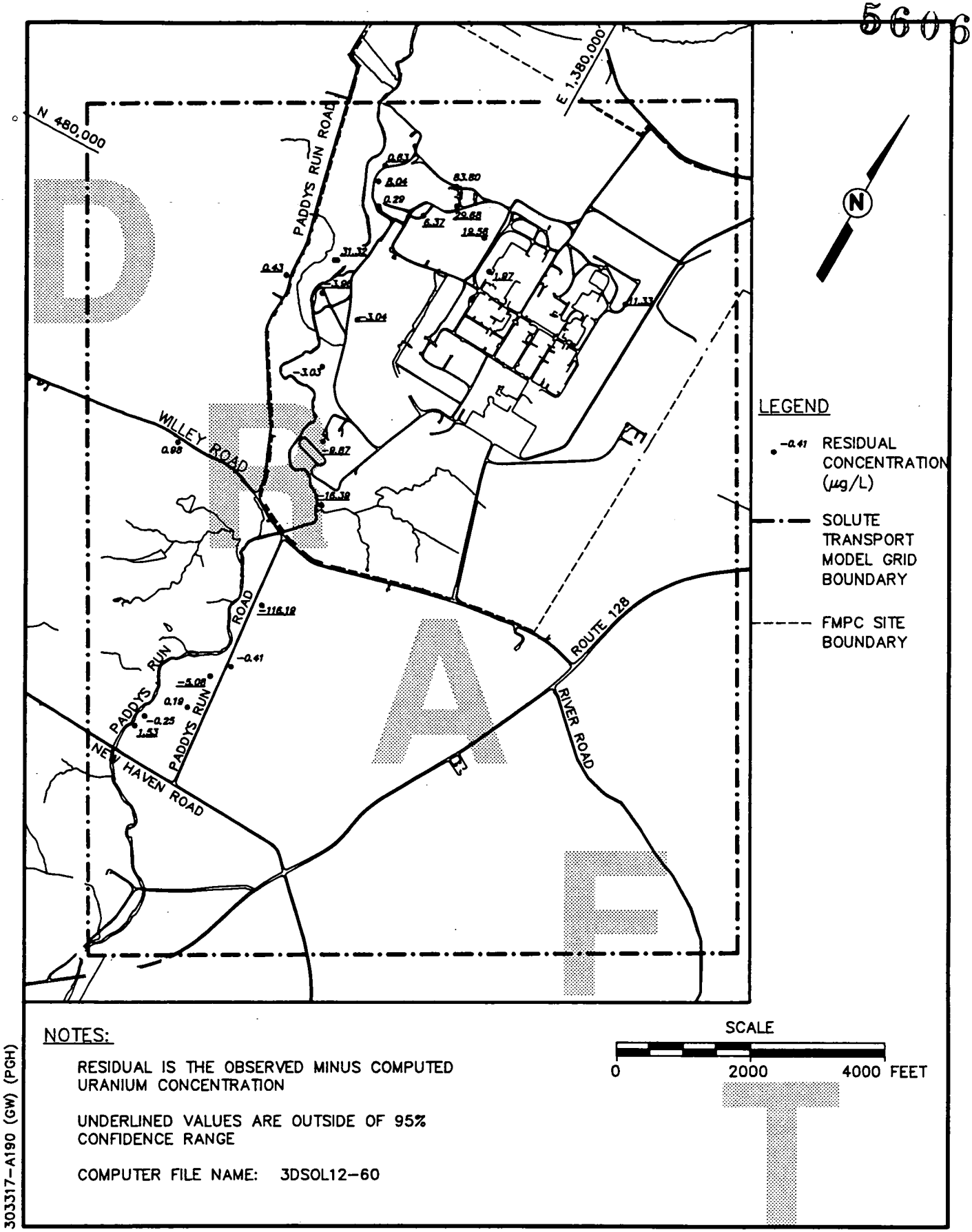
COMPUTER FILE NAME: 3DSOL12-60

SCALE

0 2000 4000 FEET

303317-A191 (GW) (PGH)

FIGURE 21-11
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
RESIDUAL CONCENTRATIONS
FOR CALIBRATED RUN
LAYER 1
RETARDATION FACTOR 12



NOTES:

RESIDUAL IS THE OBSERVED MINUS COMPUTED URANIUM CONCENTRATION

UNDERLINED VALUES ARE OUTSIDE OF 95% CONFIDENCE RANGE

COMPUTER FILE NAME: 3DSOL12-60

303317-A190 (GW) (PGH)

FIGURE 21-12
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
RESIDUAL CONCENTRATIONS
FOR CALIBRATED RUN
LAYERS 2 THROUGH 5
RETARDATION FACTOR 12

can be brought into range while keeping Well 2095 in range by taking advantage of lateral dispersion from Paddys Run downstream from Well 2095. However, source rates at cells upgradient from Well 3062 (Albright & Wilson Co. well) must be increased and the concentration at Well 3062 would increase too much. Breaking the source periods into smaller increments and varying source loadings to get a downgradient increase in concentration was not attempted, because the wells are close together and longitudinal dispersion would probably be too great to allow the development of a sufficient increase in concentration between the two wells. While this possibility remains untested, it is more likely that the anomalous concentration in Well 2061 is the result of preferential fingering of the contaminant front through the porous matrix in the manner described by Freeze and Cherry (1979, p. 399). It is not practical to attempt a detailed simulation of such fingering in a finite-difference model.

Wells 2108 and H112 are upgradient from Paddys Run. Minor westward movement of uranium from Paddys Run can occur due to the temporary reversal of groundwater flow direction caused by water table mounding under the stream during heavy runoff periods. However, the steady-state flow model cannot simulate such water movement. Since the calculated concentrations at these wells are less than 2 ug/L below the ranges, the representativeness of the model is not affected.

Well 2024 is upgradient from any known contaminant source, so it is not possible to raise the calculated concentration at that location. Again, the calculated concentration is less than 2 ug/L below the range, and the representativeness of the model is not affected.

The calculated concentration at Well 2017 is too high. This well is on the upgradient side of Paddys Run. The lack of fit here is related to the finite-difference simulation of source cells along Paddys Run. A source cell in the model is closer to the well than the stream actually is. The calculated concentration is 28 ug/L out of range and does not affect the representativeness of the model.

Several concentrations for wells screened in Layers 2 through 4 are out of range. Calculated concentrations tend to be too low in the northern part of the area and too high in the southern part of the area. A few concentrations are in range in both areas. All of the concentrations are less than 17 ug/L out of range, except the deviation at Well 3095 where the calculated concentration is 110 ug/L above the range. These deviations are interpreted to be caused primarily by irregular vertical fingering in the contaminant plume. Since wells that are in range are scattered north-south through the model in Layers 2 or lower, the model is adequately simulating vertical dispersion even though the precision of the simulated vertical dispersion is not great enough to get all of the

calculated concentrations in range. Since the wells monitoring the different layers are in clusters at the same location, transverse dispersivity would have to be varied throughout the model to bring the concentrations in the lower layers into range without causing the corresponding concentrations in Layer 1 to go out of range.

21.3.4 Sensitivity Analysis

Results of sensitivity analyses for the models described in previous sections also apply to 3DSOL. No new parameters were introduced into 3DSOL. However, since varying the dispersivities may have a different effect in 3DSOL than they did in 2DSOL, they were subjected to sensitivity analysis. The model with the distribution coefficient set at 0.016 (3DSOL9) was used for the analysis. The results are shown in Table 21-8. Increasing dispersivities caused the plume to spread and decreasing dispersivities had the opposite effect. This property of the model was instrumental in model calibration, as mentioned above.

The effect of increasing the hydraulic conductivity of the clay interbed beneath the FMPC was also checked. It was found to have virtually no effect on model calibration (Table 21-8). The change in the calculated vertical head difference across the clay was from 0.90 to 0.05 feet at the location of Wells 2008 and 3008 in 3DSOL9. The decrease in vertical hydraulic gradient was associated with a decrease in hydraulic head above the clay interbed and an increase below it.

The effect of varying transverse dispersivity was performed as a sensitivity analysis for final calibration of 3DSOL at a distribution coefficient of 0.022 ft^3/lb (retardation factor 12), while the source loading rates in the model were held constant. Decreasing transverse dispersivity from 10 feet to 5 feet without adjusting other parameters was found to significantly affect the calibrated three-dimensional solute transport model. Although it did not cause the plume to spread horizontally, it resulted in higher concentration gradients in the center part of the south plume, since the vertical dispersivity was designed to vary consistent with transverse dispersivity in this model. As a result, the calculated concentrations increased significantly in Layer 1 and decreased moderately at a few wells in Layer 2. This variation toward a lower transverse dispersivity caused the concentrations in most of the wells in Layer 1 to fall outside the range. The representativeness of the model was, therefore, adversely affected. Nevertheless, calibration could be accomplished using a transverse dispersivity of 5 feet if a very refined adjustment of source loading rates in time as well as location is made in the model. A further decrease of transverse dispersivity from 10 feet to 2 feet, with all other parameters remaining unchanged, was also tested. The plume resulting from this variation spread wider and generated a steeper concentration gradient in the center of the South Plume. Calculated concentrations at some major wells in Layer 1 were raised to relatively

TABLE 21-8

RESULTS OF THE SENSITIVITY ANALYSIS
FOR THE THREE-DIMENSIONAL FINITE-DIFFERENCE
SOLUTE TRANSPORT MODEL
RETARDATION FACTOR 9

Sensitivity Run Number	Parameter	3DSOL Value	Sensitivity Run Value	Sensitivity Run[a] Statistics: Mean Residual (µg/l)	Sensitivity Run[a] Statistics: Mean Absolute Residual (µg/l)
1	Longitudinal Dispersivity Transverse Dispersivity	50 ft 1 ft	50 ft 5 ft	25.31	30.36
2	Longitudinal Dispersivity Transverse Dispersivity	50 ft 1 ft	50 ft 25 ft	29.67	32.23
3	Longitudinal Dispersivity Transverse Dispersivity	50 ft 1 ft	300 ft 6 ft	27.29	33.34
4	Clay Interbed Horizontal Conductivity Clay Interbed Vertical Conductivity	0.0003 ft/day 0.00003 ft/day	0.4 ft/day 0.4 ft/day	17.68	28.16
		3DSOL STATISTICS:		17.85	28.16

[a]Mean Residual is the mean of the differences between observed concentrations and calculated concentrations. Mean Absolute Residual is the mean of the absolute difference between observed concentrations and calculated concentrations.

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high values, while the calculated concentrations at most of the wells in the lower layers were reduced significantly. The large deviations from the means of measured concentrations in this runmade this model much less representative. These results were judged to indicate that the transverse dispersivity of 2 feet was too low to accomplish the calibration.

Changes in flow model parameters in the flow modeling sensitivity analysis did not significantly affect the flow field within the solute transport model area. Consequently, the sensitivity of the solute transport model to these parameters was not investigated. Experience with the present model suggests that the calibrated three-dimensional solute transport model would not be significantly affected by variations in most of the parameters.

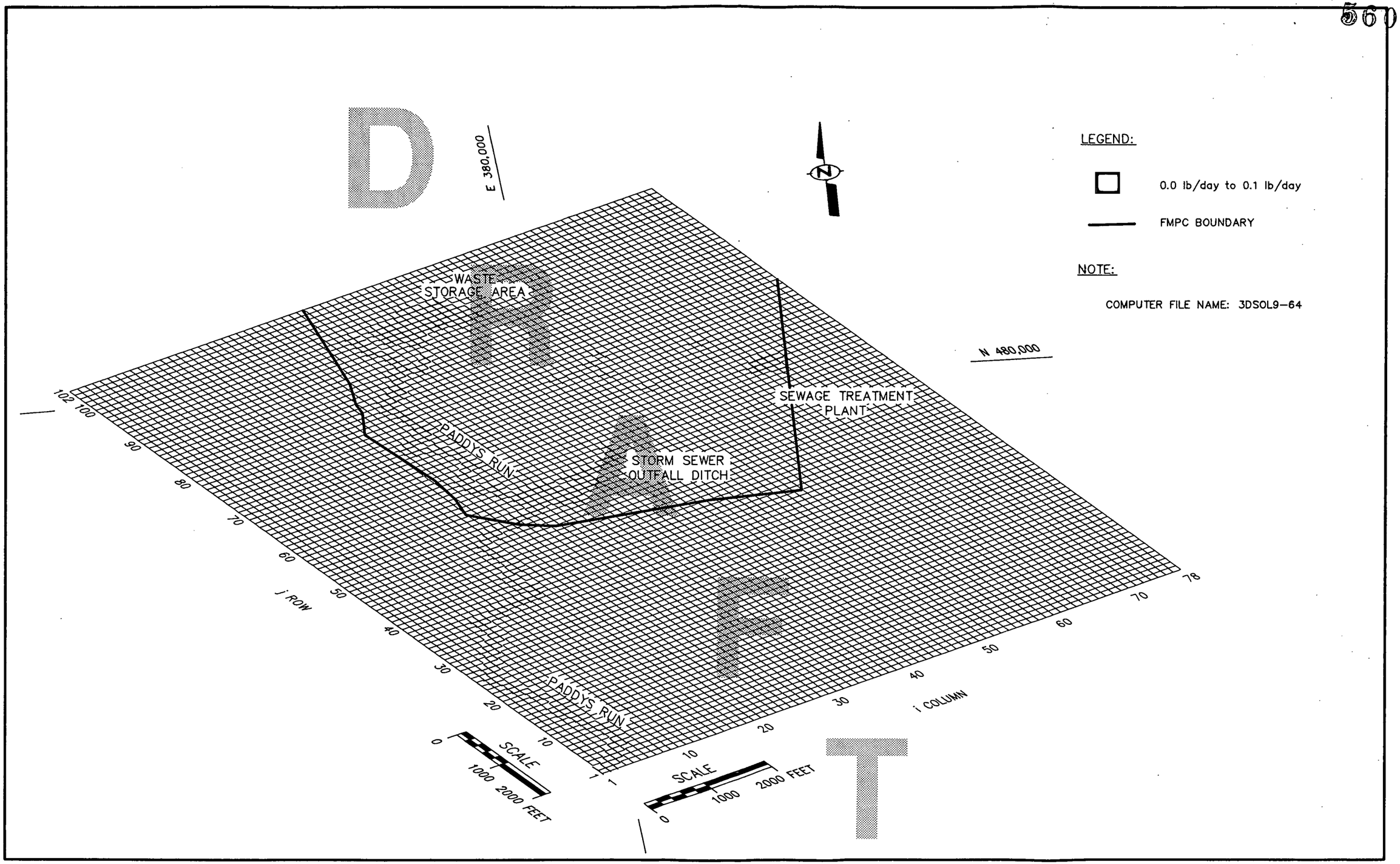
21.3.5 Model Results

The solute transport model supplies convincing evidence that significant adsorption occurs within the aquifer system underlying the FMPC. Over eighty percent of the estimated mass of uranium lost to the groundwater system is still located near the FMPC site within the boundaries of the local solute transport model grid.

The solute transport model is sensitive to the values of the dispersivities and distribution coefficient and to the amount and location of source loading. Balancing these three parameters can yield similar plumes. However, when the dispersion coefficient is held constant, the distribution coefficient controls the concentration gradients in the plume and the choice of parameters that yield a good calibration is limited.

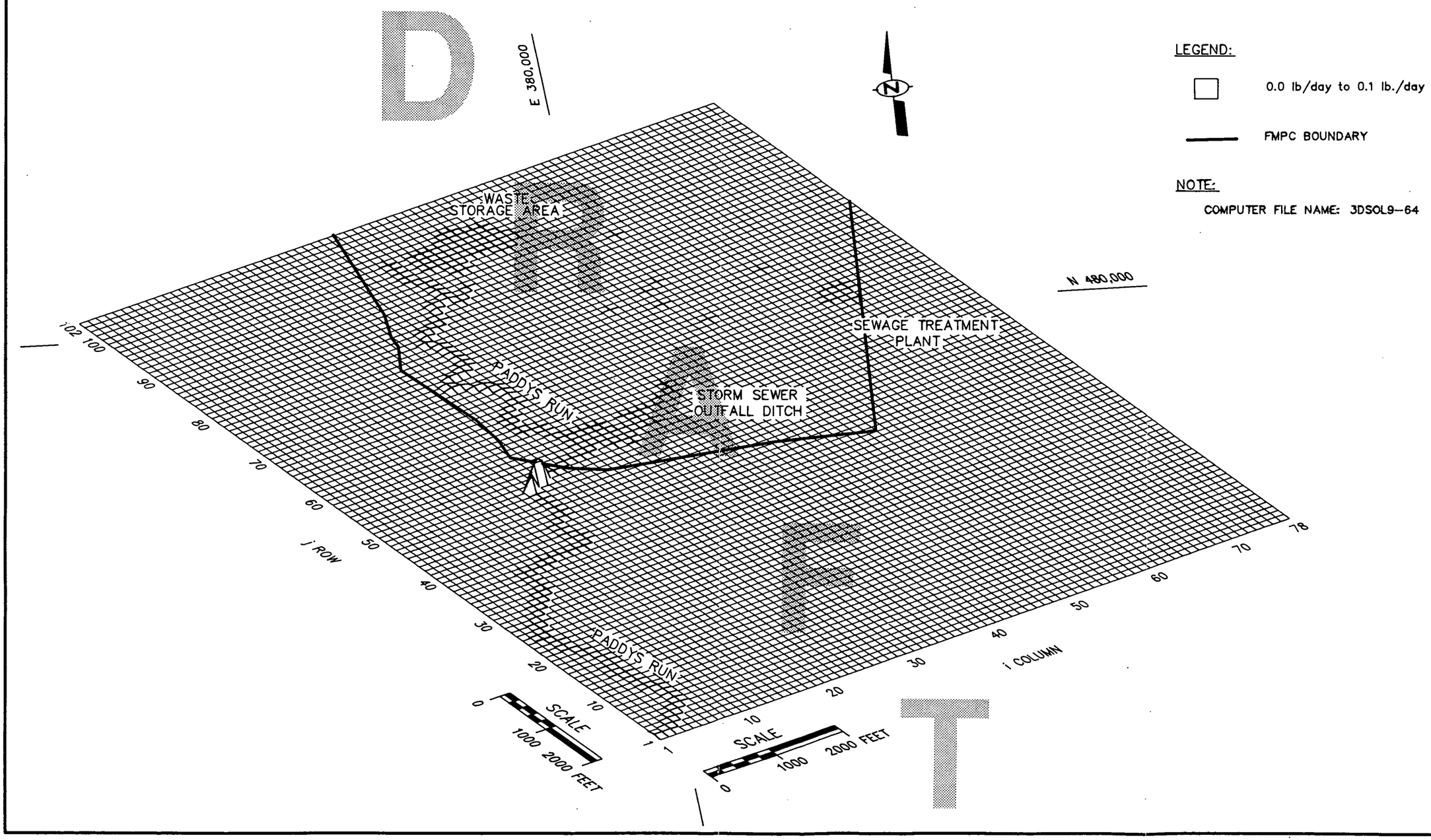
The model was successfully calibrated with two different distribution coefficients, 0.016 ft^3/lb and 0.022 ft^3/lb (retardation factors 9 and 12, respectively). The calibration at 0.022 ft^3/lb was more refined. Nevertheless, this result demonstrates that calibration can be accomplished using different distribution coefficients, if compensating adjustments are made in dispersivities and source loading rates. Longitudinal and transverse dispersivities of 50 feet and 1 foot, respectively, were used to calibrate with a distribution coefficient of 0.016 ft^3/lb, but these could be increased to the preferred values of 100 feet and 10 feet for calibration with a dispersivity of 0.022 ft^3/lb. The source loading rates for each period that resulted from the preliminary calibration with a distribution coefficient of 0.016 ft^3/lb are shown in Figures 21-13 through 21-16, and are summarized in terms of loading rates and total mass loadings in Tables 21-9 and 21-10. The source loading rates that resulted from the final calibration with a distribution coefficient of 0.022 ft^3/lb are shown in Figures 21-17 through 21-21, and are summarized in Tables 21-11 and 21-12. As shown in

303317-B715 /01 (GW) 100% RED



303317-B715 /01 (GW) (PGH)

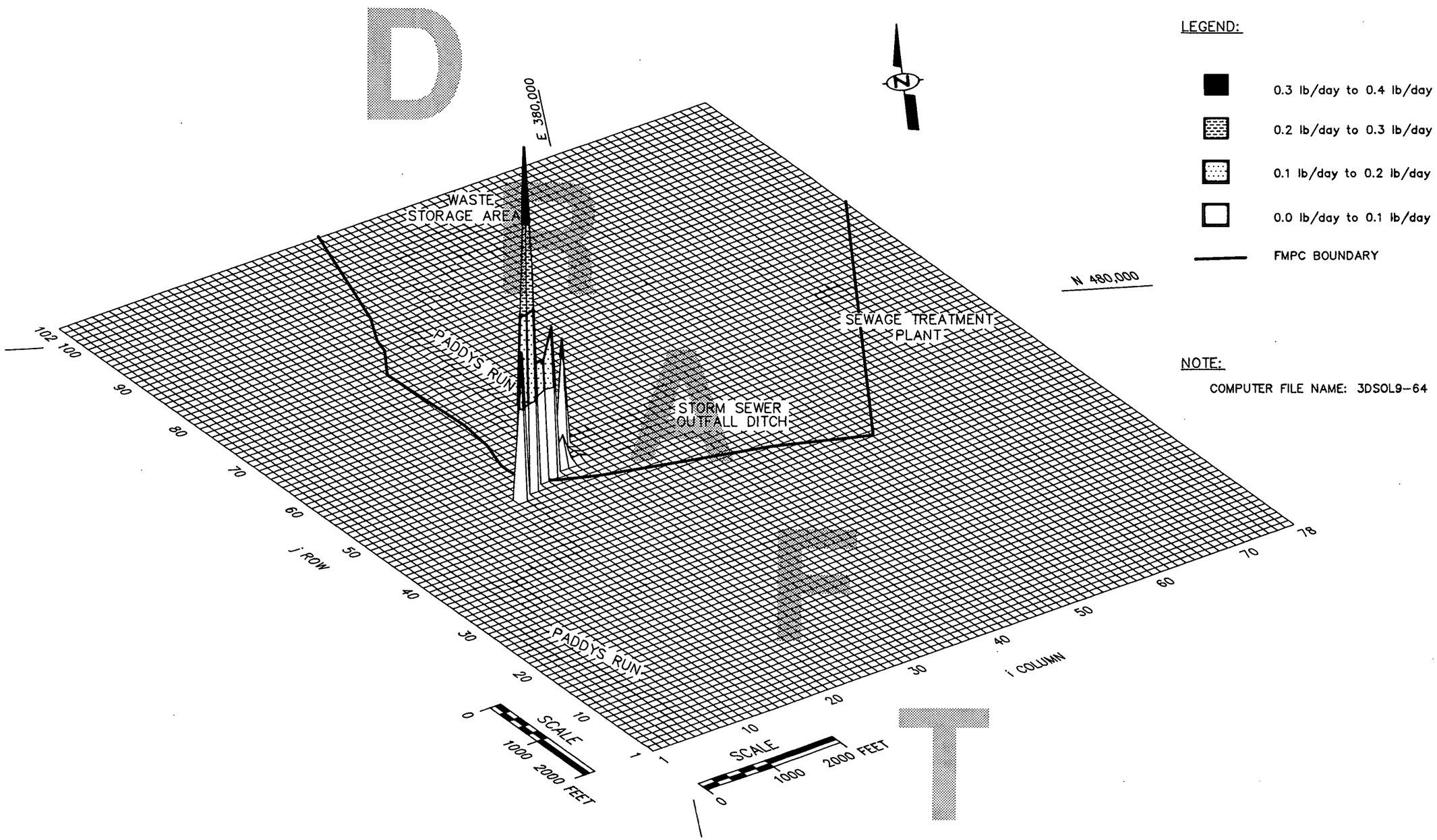
FIGURE 21-13
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SOURCE LOADING RATES, SOURCE PERIOD 1
RETARDATION FACTOR 9



303317-B711 /01 (GW) (PGH)

FIGURE 21-14
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SOURCE LOADING RATES, SOURCE PERIOD 2
RETARDATION FACTOR 9

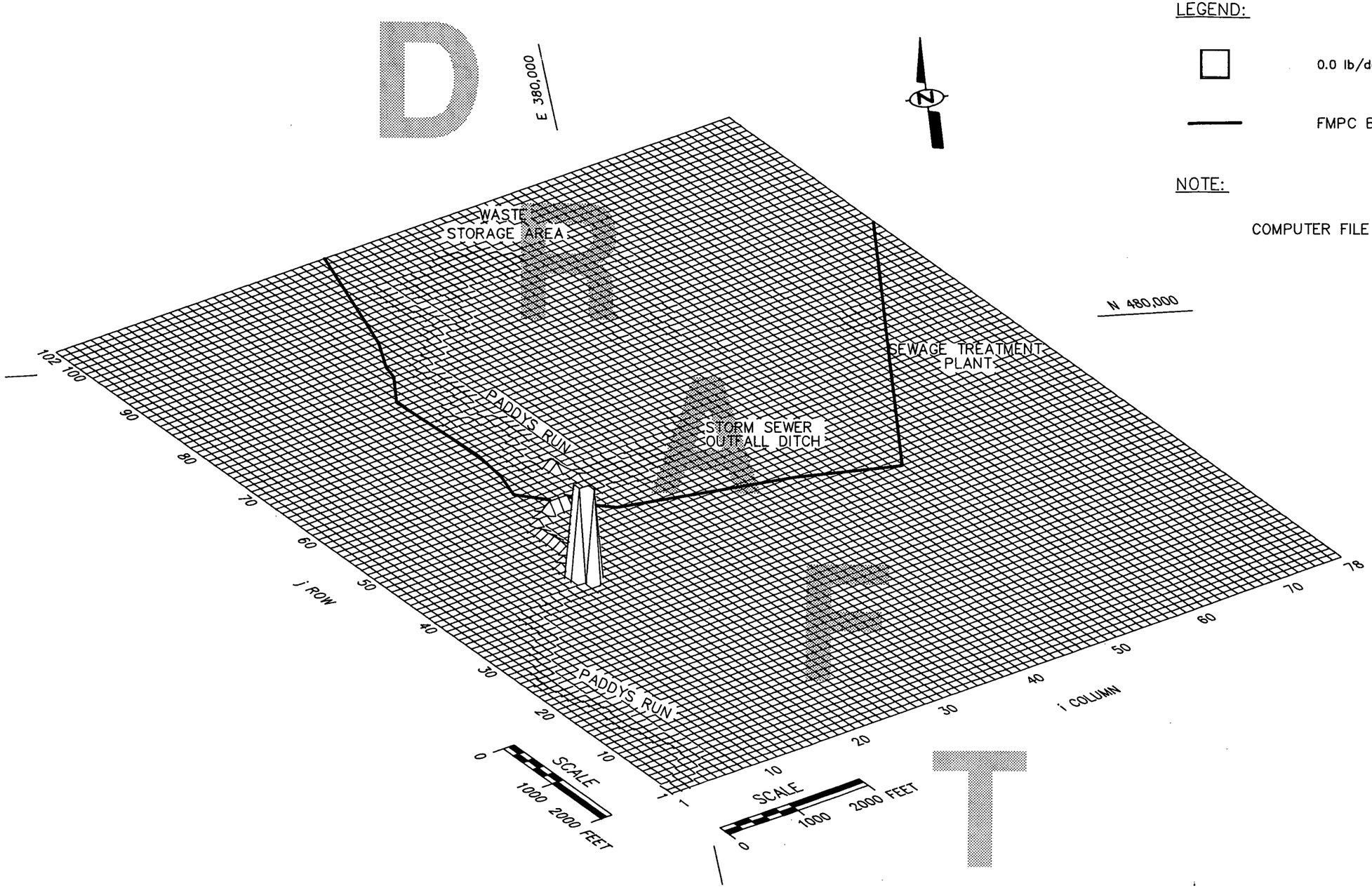
303317-B716 /01 (GW) 100% RED



303317-B716 /01 (GW) (PGH)

FIGURE 21-15
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SOURCE LOADING RATES, SOURCE PERIOD 3
RETARDATION FACTOR 8

303317-B712 (GW) /01 100% RED



LEGEND:

0.0 lb/day to 0.1 lb/day

FMPC BOUNDARY

NOTE:

COMPUTER FILE NAME: 3DSOL9-64

303317-B712 /01 (GW) (PGH)

FIGURE 21-16
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SOURCE LOADING RATES, SOURCE PERIOD 4
RETARDATION FACTOR 9

TABLE 21-9

URANIUM LOADING RATES FOR SOURCE AREAS
IN THE THREE-DIMENSIONAL FINITE-DIFFERENCE
SOLUTE TRANSPORT MODEL
RETARDATION FACTOR 9, LONGITUDINAL DISPERSIVITY 50 FEET,
AND TRANSVERSE DISPERSIVITY 1 FOOT

	Uranium Source Loading Rates (lb/day)			
Source Loading Area	Period 1 1952-1958	Period 2 1959-1966	Period 3 1967-1975	Period 4 1976-1988
Paddys Run:				
North of Waste Storage Area	0.0000286	0.000373	0.000372	0.000372
Waste Storage Area to Storm Sewer Outfall Ditch	0.00319	0.0602	0.0559	0.0559
Storm Sewer Outfall Ditch to Albright & Wilson Co. Wells	0.000428	0.0875	1.68	0.332
South of Albright & Wilson Co. Wells	0.00292	0.0192	0.00769	0.00101
Waste Storage Area	0.00262	0.0209	0.0209	0.0209
Waste Storage Silo Area	0.000197	0.000197	0.000197	0.000197
Biodenitrification Lagoon	0.0000000	0.0000000	0.0000000	0.0000000
Sludge Ponds	0.0000000	0.0000000	0.0000000	0.0000000
Production Area	0.00178	0.00178	0.00178	0.00178
Sewage Treatment Area	0.000593	0.000593	0.000593	0.000593
Storm Sewer Outfall Ditch	0.000521	0.00342	0.001960	0.000252
TOTAL	0.0123	0.194	1.77	0.413

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TABLE 21-10

TOTAL URANIUM LOADINGS FOR SOURCE AREAS IN THE THREE-DIMENSIONAL FINITE-DIFFERENCE SOLUTE TRANSPORT MODEL RETARDATION FACTOR 9, LONGITUDINAL DISPERSIVITY 50 FEET, AND TRANSVERSE DISPERSIVITY 1 FOOT

	Total Uranium Source Loadings (lb)				
Source Loading Area	Period 1 1952-1958	Period 2 1959-1966	Period 3 1967-1975	Period 4 1976-1988	Total
Paddys Run:					
North of Waste Storage Area	0.073	1.09	12.20	1.77	15.10
Waste Storage Area to Storm Sewer Outfall Ditch	8.15	176.00	184.00	265.00	633.00
Storm Sewer Outfall Ditch to Albright & Wilson Co. Wells	1.09	256.00	5510.00	1580.00	7347.00
South of Albright & Wilson Co. Wells	7.47	56.00	25.30	4.81	93.60
aste Storage Area	6.69	61.20	68.80	99.40	236.00
Waste Storage Silo Area	0.504	0.576	0.648	0.937	2.66
Biodenitrification Lagoon	0.00	0.00	0.00	0.00	0.00
Sludge Ponds	0.00	0.00	0.00	0.00	0.00
Production Area	4.54	5.19	5.84	8.43	24.00
Sewage Treatment Area	1.51	1.73	1.95	2.81	8.00
Storm Sewer Outfall Ditch	1.33	10.00	6.45	1.20	19.00
TOTAL	31.40	568.00	5815.00	1964.00	8378.00

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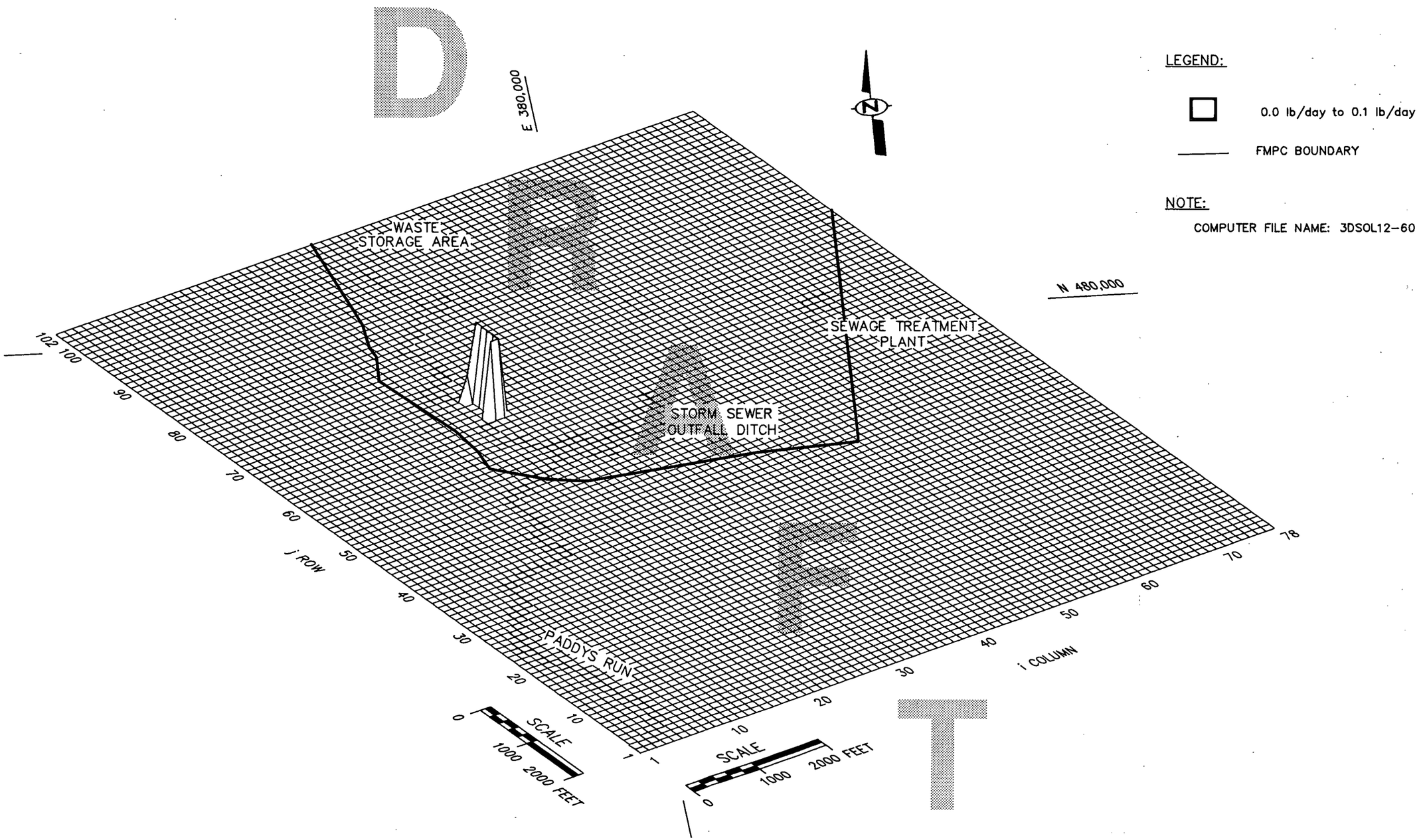
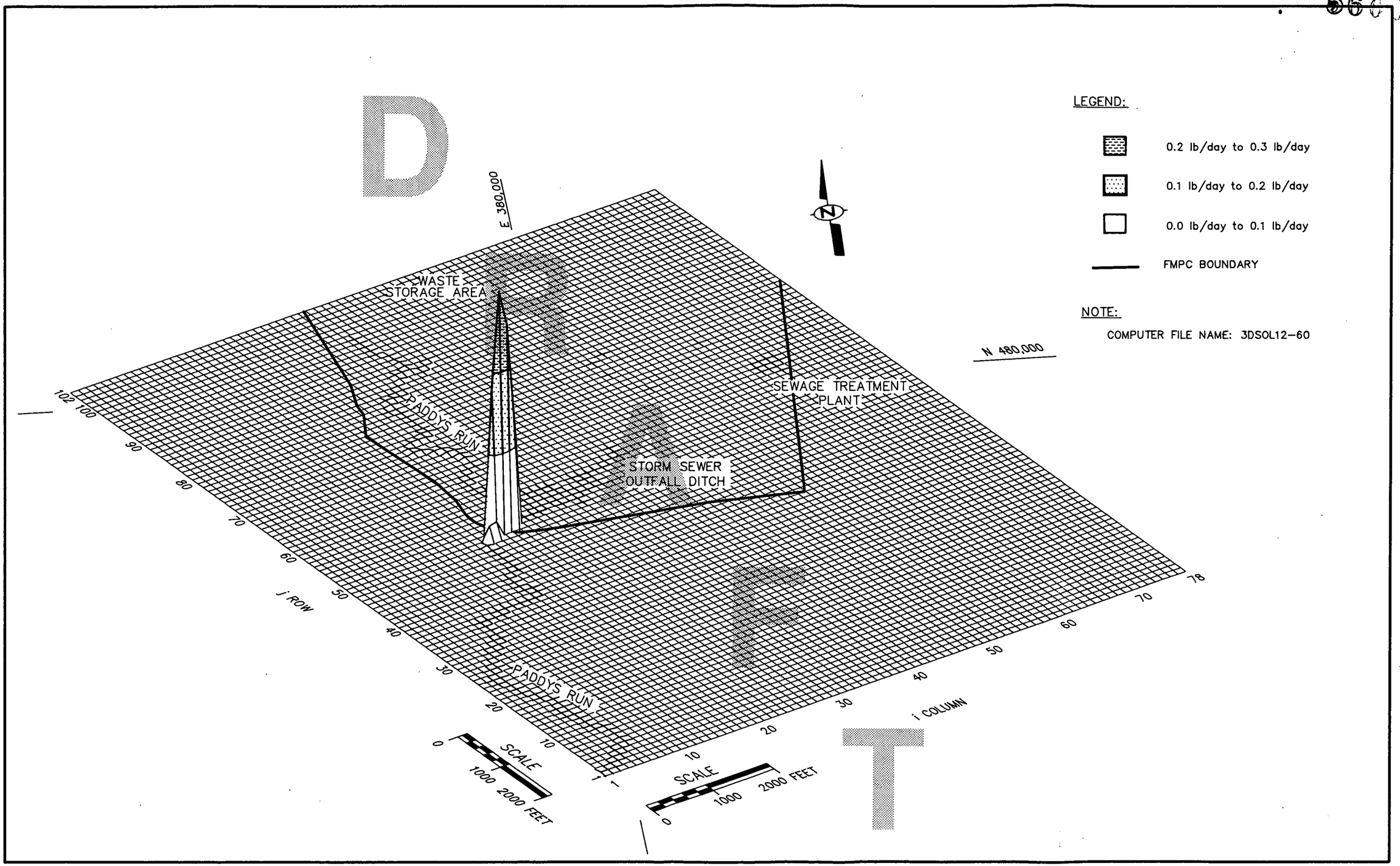
303317-B717 /01 (GW) 100% RED



303317-B717 /01 (GW) (PGH)

FIGURE 21-17
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SOURCE LOADING RATES, SOURCE PERIOD 21
RETARDATION FACTOR 12

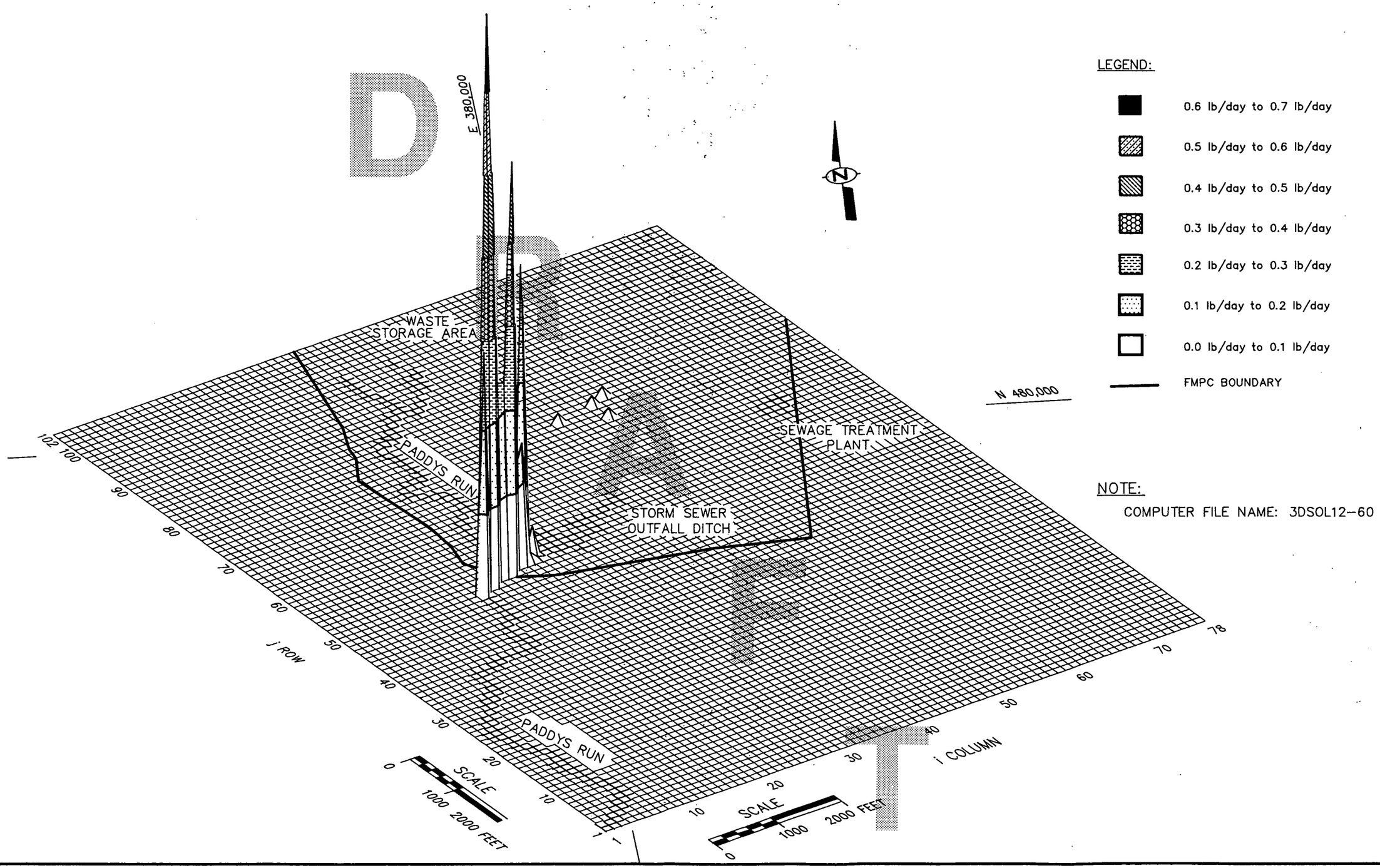
303317-B720 /01 (GW) 100% RED



303317-B720 /01 (GW) (PGH)

FIGURE 21-18
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SOURCE LOADING RATES, SOURCE PERIOD 2
RETARDATION FACTOR 12

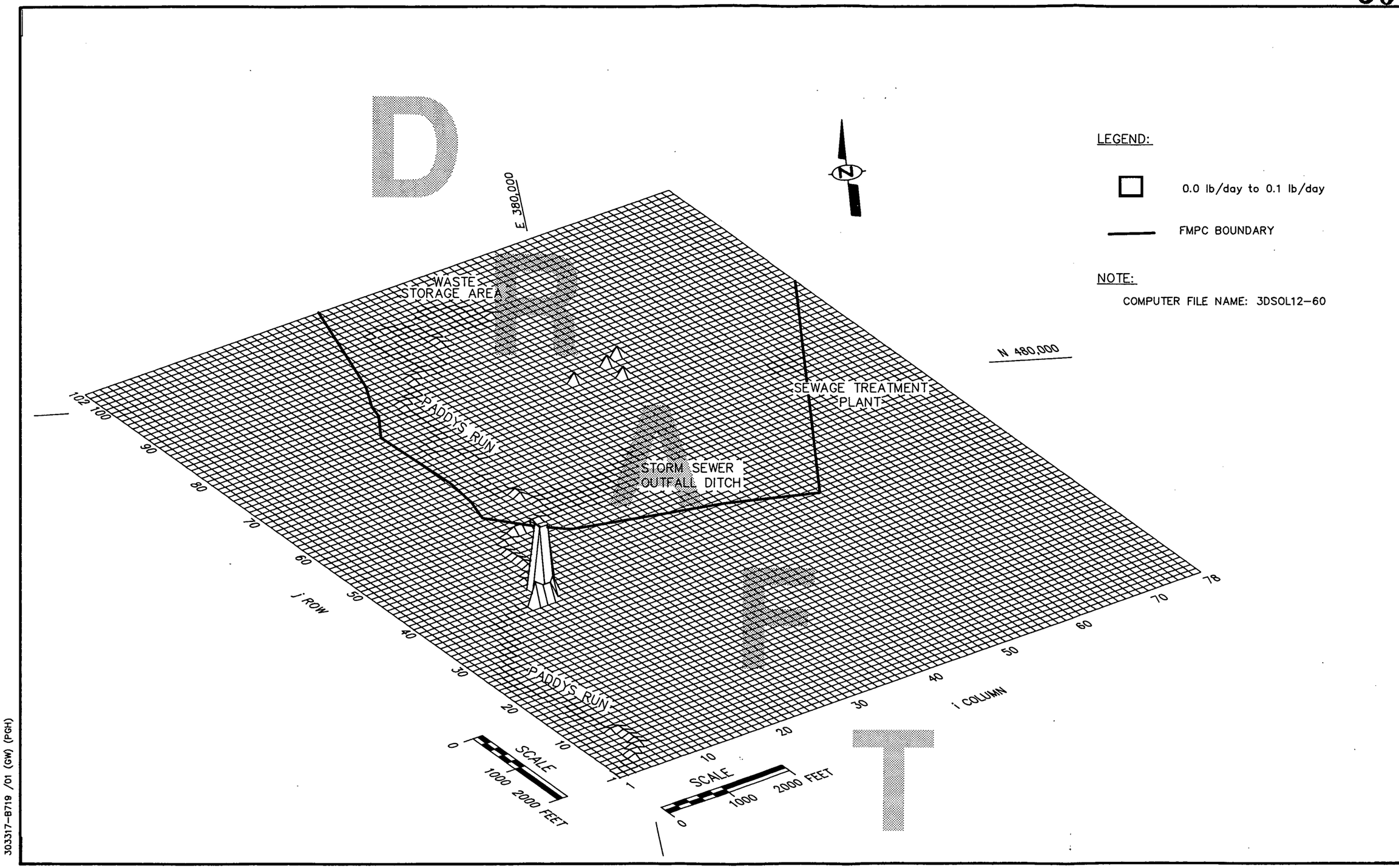
303317-B721 /01 100% RED



303317-B721 /01 (GW) (PGH)

FIGURE 21-19
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SOURCE LOADING RATES, SOURCE PERIOD 3
RETARDATION FACTOR 12

303317-B719 /01 (GW) 100% RED



303317-B719 /01 (GW) (PGH)

FIGURE 21-20
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SOURCE LOADING RATES, SOURCE PERIOD 4
RETARDATION FACTOR 12

TABLE 21-11

URANIUM LOADING RATES FOR SOURCE AREAS
IN THE THREE-DIMENSIONAL FINITE-DIFFERENCE
SOLUTE TRANSPORT MODEL
RETARDATION FACTOR 12, LONGITUDINAL DISPERSIVITY 100 FEET,
AND TRANSVERSE DISPERSIVITY 10 FEET

	URANIUM SOURCE LOADING RATES (lb/day)					
	Period 1	Period 2	Period 3	Period 4	Period 5	Total
Source Loading Area	1952-1958	1959-1966	1969-1975	1976-1988	1989-1989	
Paddys Run:						
North of Waste Storage Area	0.0000286	0.000373	0.000372	0.00464	0.00618	0.01159
Waste Storage Area to Storm Sewer Outfall Ditch	0.546	0.0492	0.0515	0.0486	0.0478	0.7431
Storm Sewer Outfall Ditch to Albright-Wilson Co. Wells	0.000353	0.915	0.0357	0.331	0.0662	1.34825
South of Albright-Wilson Co. Wells	0.00292	0.0192	0.00769	0.111	0.0409	0.18171
Waste Storage Area	0.0	0.0	0.00291	0.0242	0.0231	0.05021
Waste Storage Silo Area	0.000197	0.000197	0.0014	0.002	0.000957	0.00475
Biodenitrification Lagoon	0.0	0.0	0.0	0.0	0.0	0.0
Sludge Ponds	0.0	0.0	0.0	0.0	0.0	0.0
Production Area	0.00178	0.00178	0.0488	0.0488	0.0488	0.14996
Sewage Treatment Area	0.000593	0.000593	0.000593	0.000593	0.000593	0.00297
Storm Sewer Outfall Ditch	0.0000052	0.0000034	0.0000593	0.000299	0.0831	0.08347
TOTAL	0.552	0.986	0.149	0.571	0.318	2.58

000300

5606

TABLE 21-12

TOTAL URANIUM LOADING FOR SOURCE AREAS IN THE THREE-DIMENSIONAL FINITE-DIFFERENCE SOLUTE TRANSPORT MODEL RETARDATION FACTOR 12, LONGITUDINAL DISPERSIVITY 100 FEET, AND TRANSVERSE DISPERSIVITY 10 FEET

	Uranium Source Loadings (lb)					
Source Loading Area	Period 1 1952-1958	Period 2 1959-1966	Period 3 1967-1975	Period 4 1976-1988	Period 5 1989-1989	Total
Paddys Run						
North of Waste Storage Area	0.07	1.09	1.22	22.02	2.26	26.66
Waste Storage Area to Storm Sewer Outfall Ditch	1395	143.7	169.2	230.6	17.45	1956
Storm Sewer Outfall Ditch to Albright & Wilson Co. Wells	0.90	2672	117.3	1571	24.16	4385
South of Albright & Wilson Co. Wells	7.46	56.1	25.3	526.7	14.93	630.4
Waste Storage Area	0.00	0.00	9.56	114.8	8.43	132.8
Waste Storage Silo Area	0.50	0.58	4.60	9.49	0.35	15.52
Biodenitrification Lagoon	0.00	0.00	0.00	0.00	0.00	0.00
Sludge Ponds	0.00	0.00	0.00	0.00	0.00	0.00
Production Area	4.55	5.20	160.3	231.6	17.81	419.4
Sewage Treatment Area	1.52	1.73	1.95	2.81	0.22	8.23
Storm Sewer Outfall Ditch	0.01	0.01	0.19	1.42	30.33	31.96
Total	1410	2880	490	2710	116	7606

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Figures 21-11 and 21-12, a fifth period was added to represent the year 1989 in the final calibration of the model to update the uranium discharge into Paddys Run. This also allowed a closer match for the high concentration in Well 2045 which is just east of Paddys Run. These source loading rates differ for several reasons:

- Source loading must be increased when the distribution coefficient is increased to effect a calibration, providing all other parameters remain unchanged. Concentrations of dissolved uranium in a calibrated plume must be roughly the same regardless of the distribution coefficient, so more uranium must be supplied at the source cells to accommodate increased adsorption at higher distribution coefficients.

- Some source loadings may shift backward in time (i.e., earlier releases) at higher distribution coefficients because the retardation effect must be accommodated in the calibration.

- Some source loading cells may shift closer to affected monitoring wells at higher distribution coefficients because the retardation effect must be accommodated in the calibration.

These source loading rates do not represent a unique solution to the problem of estimating the historical loss of uranium to groundwater at the site. However, they are constrained by the following considerations:

- Concentrations in the model must be consistent with concentrations measured in the field. Therefore, calibration of the concentration values in the model results in plumes that contain the best estimate of the amount of uranium in the groundwater system, including that which is adsorbed.

- The total source loading is equal to the sum of dissolved and adsorbed uranium in the simulated plume, plus uranium that has crossed the model boundaries, and uranium that has been removed from the system by water wells. Therefore, the total source loading will be fairly accurate for a given distribution coefficient.

- The distribution of simulated source loading in time and space is severely limited by the constraints of calibration. Uranium must be introduced via source cells at the proper place at the proper time to have the desired effect on the calculated concentration at a monitoring well site.

Consequently, freedom in choosing location and time variation in source loading conditions was found to be limited if the distribution coefficient is held constant. The evidence that most loss of uranium to the groundwater system was in the vicinity of the confluence of the Storm Sewer Outfall Ditch and Paddys Run is very strong. The corollary is that the loss of uranium from the Waste Storage Area and other areas of the FMPC site to the regional aquifer has been relatively small, unless such losses involve discharge to Paddys Run that would then be accounted for under

the assumed principal source in Paddys Run. The source loading rates also indicate that uranium discharge to the Great Miami Aquifer was much greater in the past than it is now.

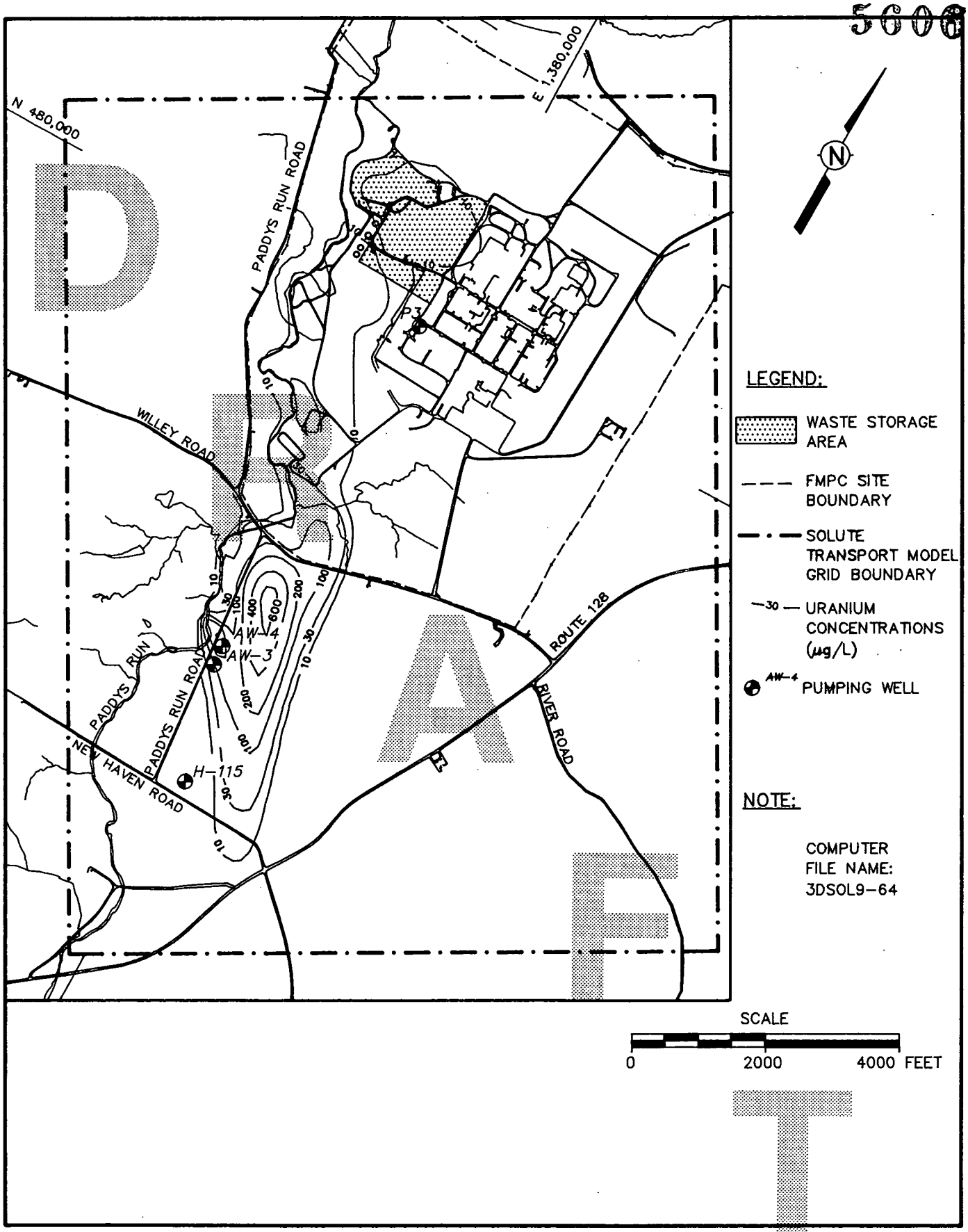
The plume resulting from the preliminary calibration at distribution coefficient 0.016 is shown for the respective vertical layers in Figures 21-22 through 21-26. The plume resulting from the final calibration at distribution coefficient 0.022 ft^3/lb is shown in Figures 21-27 through 21-31. As additional monitoring wells are installed in the future, it may be necessary to make minor modifications in this model to accommodate those data. The magnitude and location of the peak concentrations in the plume may not be exactly as shown by the model. Some unknown sources may exist, and they could be added to the model if they were discovered by field investigation.

21.3.5.1 No Action Alternative Analysis

The final calibrated run (3DSOL) was used to simulate the future spread of the plume in the case where no remedial action is taken. Source loading rates were unchanged from those given in Table 21-11 for the fifth (i.e., most recent) source period and the run was extended to include 70 years into the future. The concentrations that result after 70 years are shown in Figures 21-32 through 21-36. The 30 ug/L contour extends about 400 feet farther southeast in the northern plume area, due to a continuing supply of uranium seeping downward to the aquifer from the glacial overburden. The plume in the southern area was reduced, due to a reduction in the supply of uranium from Paddys Run and the Storm Sewer Outfall Ditch relative to the historic loading rates. This reduction is related to the construction of the storm water retention basins.

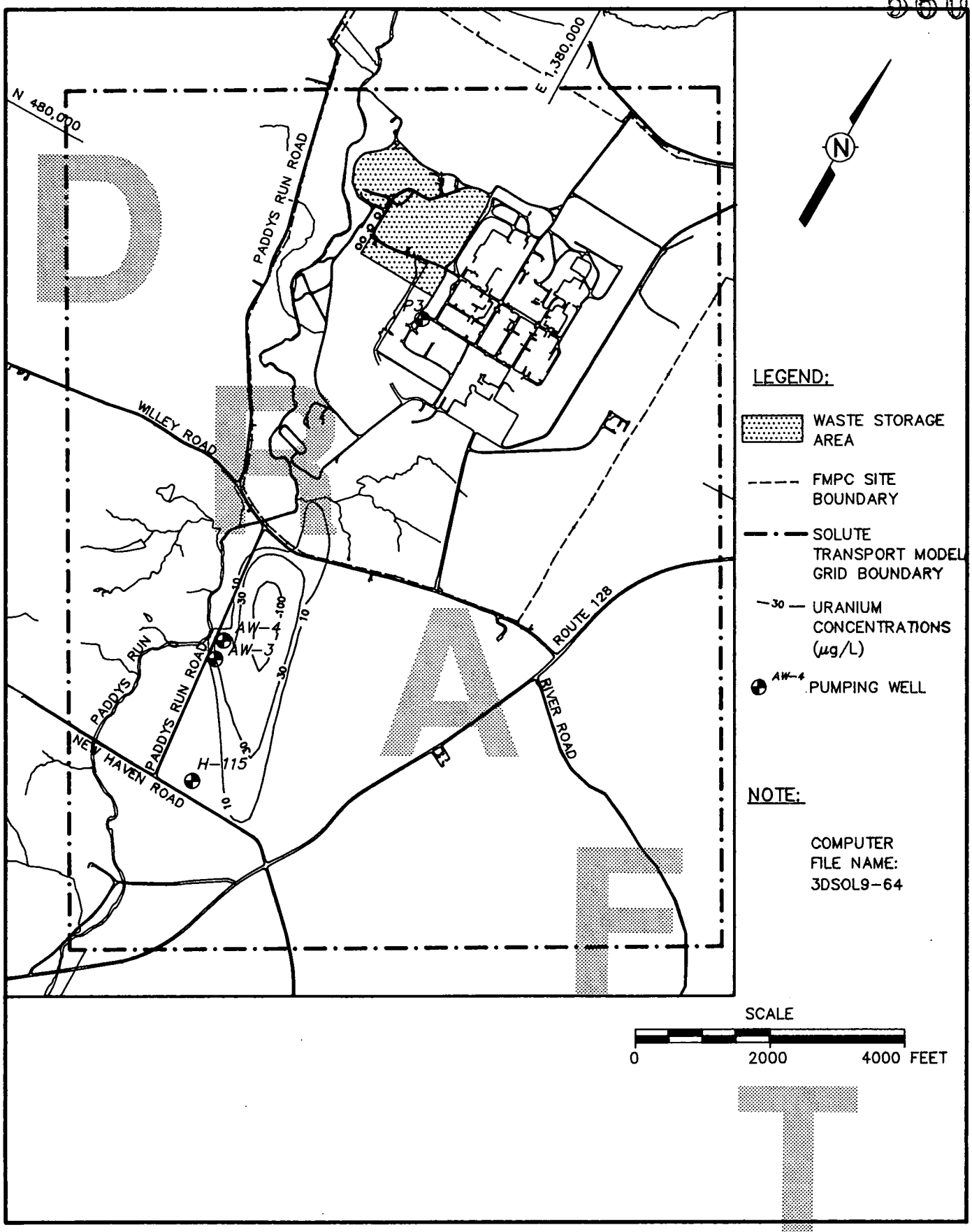
21.3.5.2 Loading to the Great Miami River

The final calibrated run was also used to estimate the rate of discharge of uranium into the Great Miami River. This simulation involved the assumption of plug flow between the southern boundary of the model and the river. The rate of movement of uranium across the face of each cell along the south boundary of the model was calculated from model output of concentration and velocity. Particle tracking results were used to estimate the retarded travel time between each cell and the river. Uranium inflow to the river was then calculated by using the retarded travel time to convert flow rate across the boundary to flow rate into the river. The results shown in Figure 21-37 are for the case where uranium source rates from the fifth source period are extended 70 years into the future.



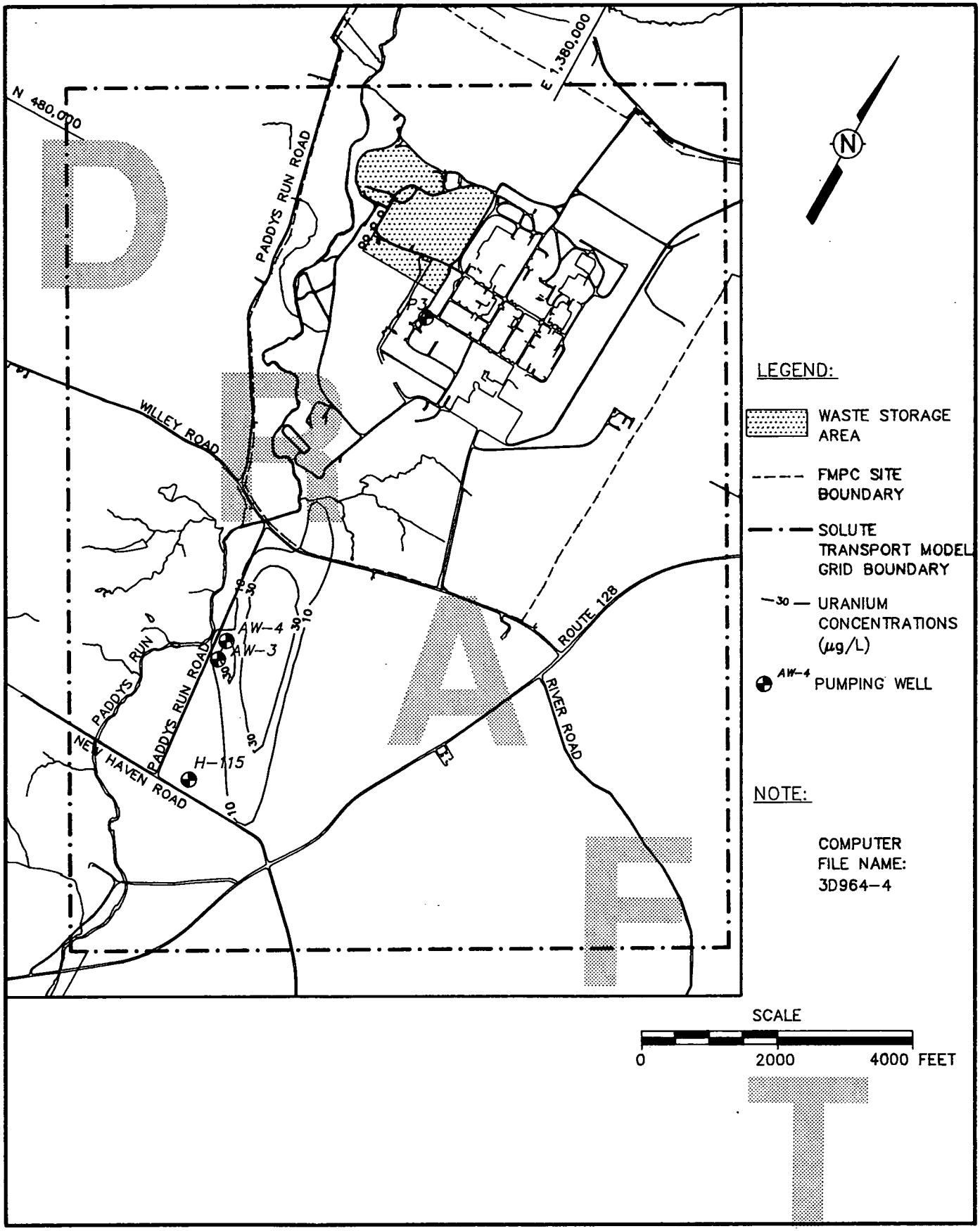
303317-A154 (GW) /01 (PGH)

FIGURE 21-22
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
PRESENT CONDITIONS
LAYER 1
RETARDATION FACTOR 9



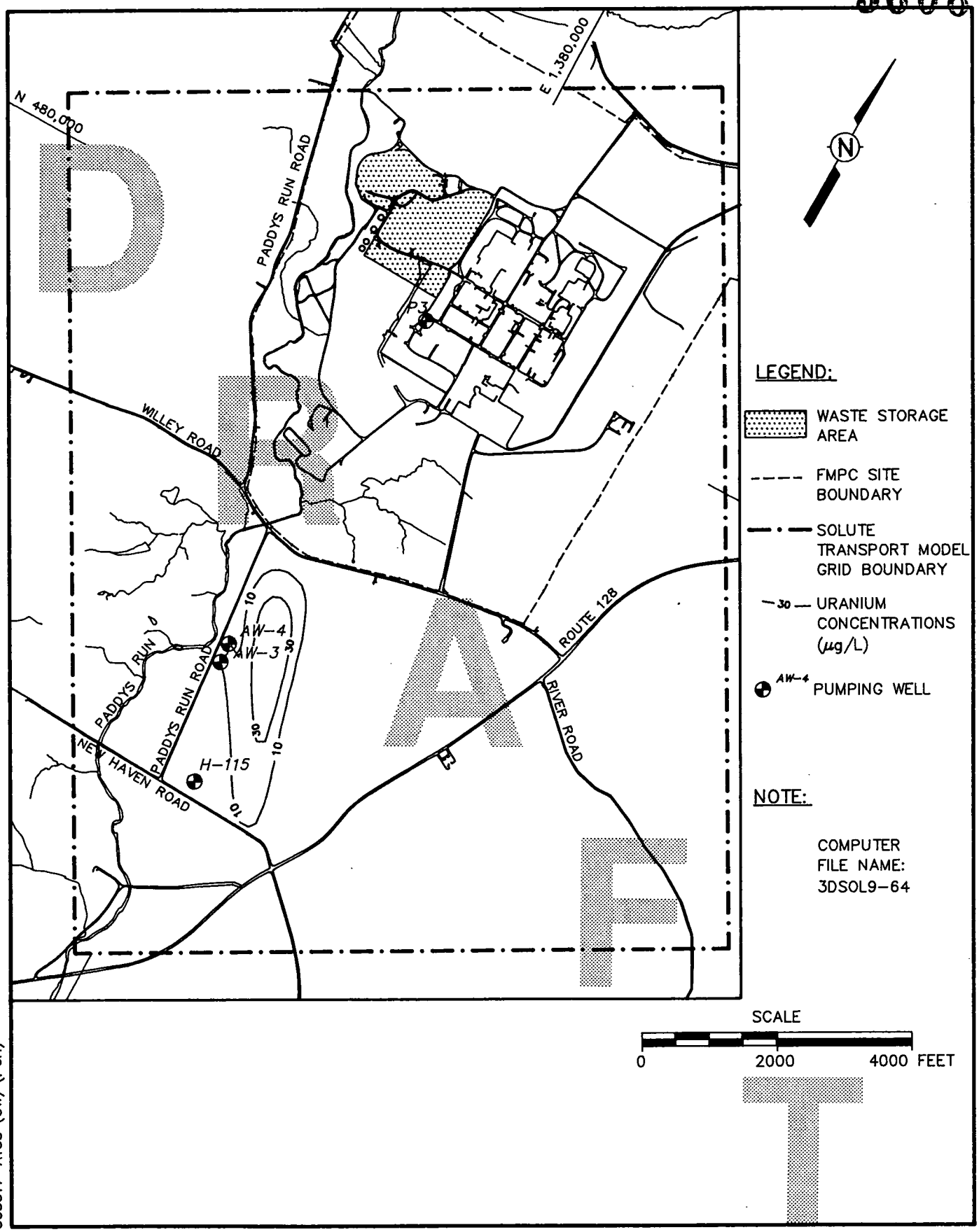
303317-A158 (GW) /01 (PGH)

FIGURE 21-23
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
PRESENT CONDITIONS
LAYER 2
RETARDATION FACTOR 9



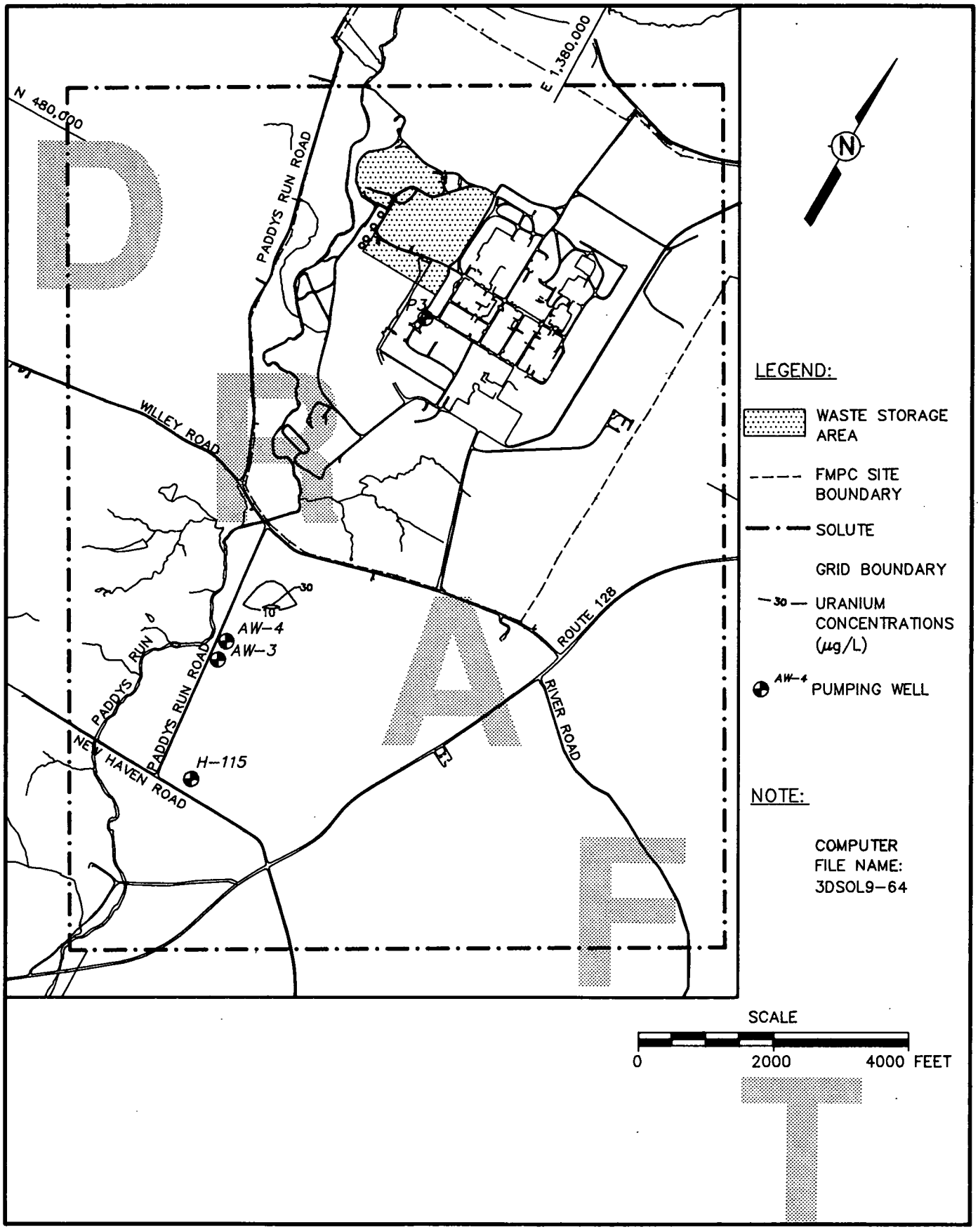
303317-A156 (GW) /01 (PGH)

FIGURE 21-24
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
PRESENT CONDITIONS
LAYER 3
RETARDATION FACTOR 9



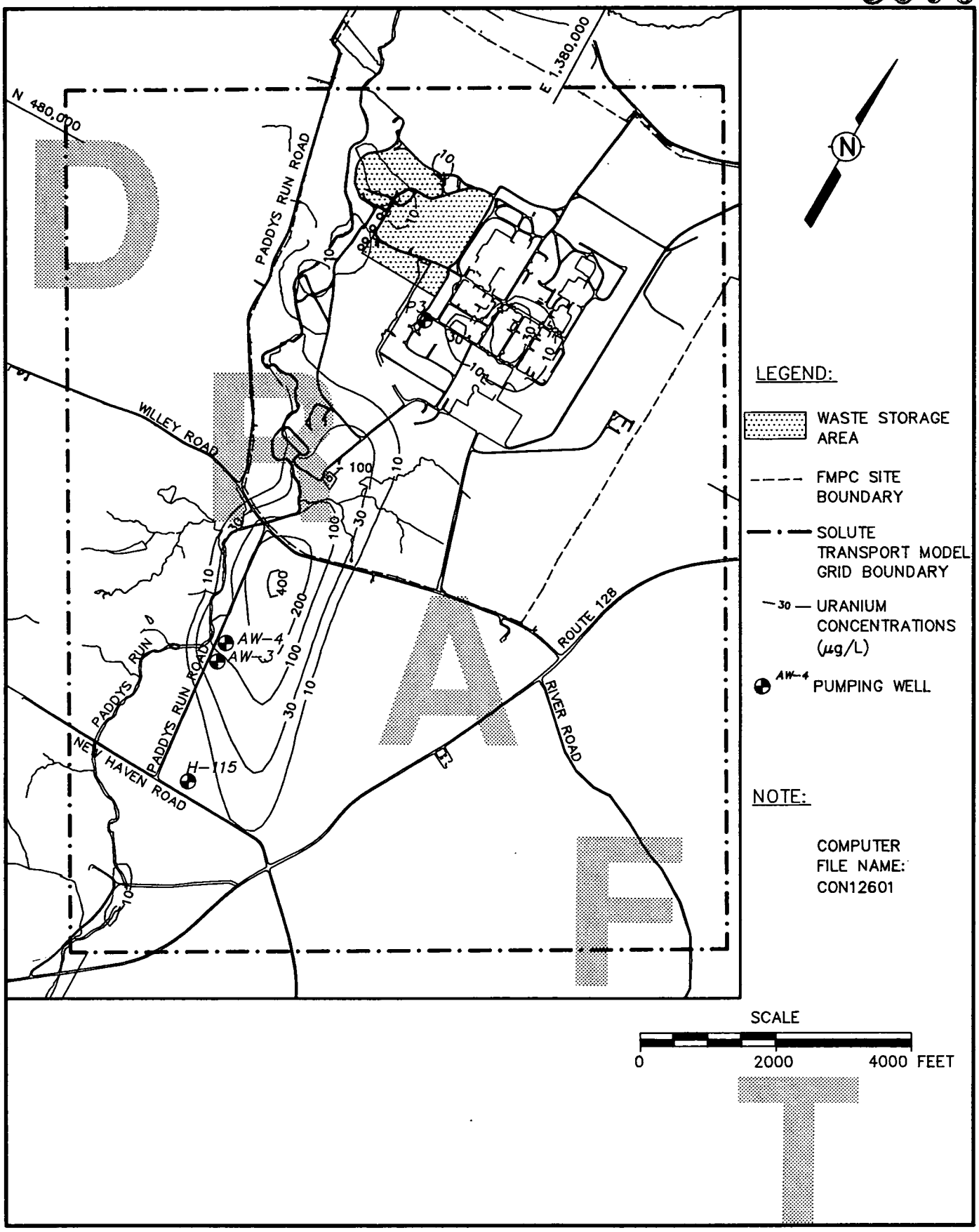
303317-A155 (GW) (PGH)

FIGURE 21-25
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
PRESENT CONDITIONS
LAYER 4
RETARDATION FACTOR 9



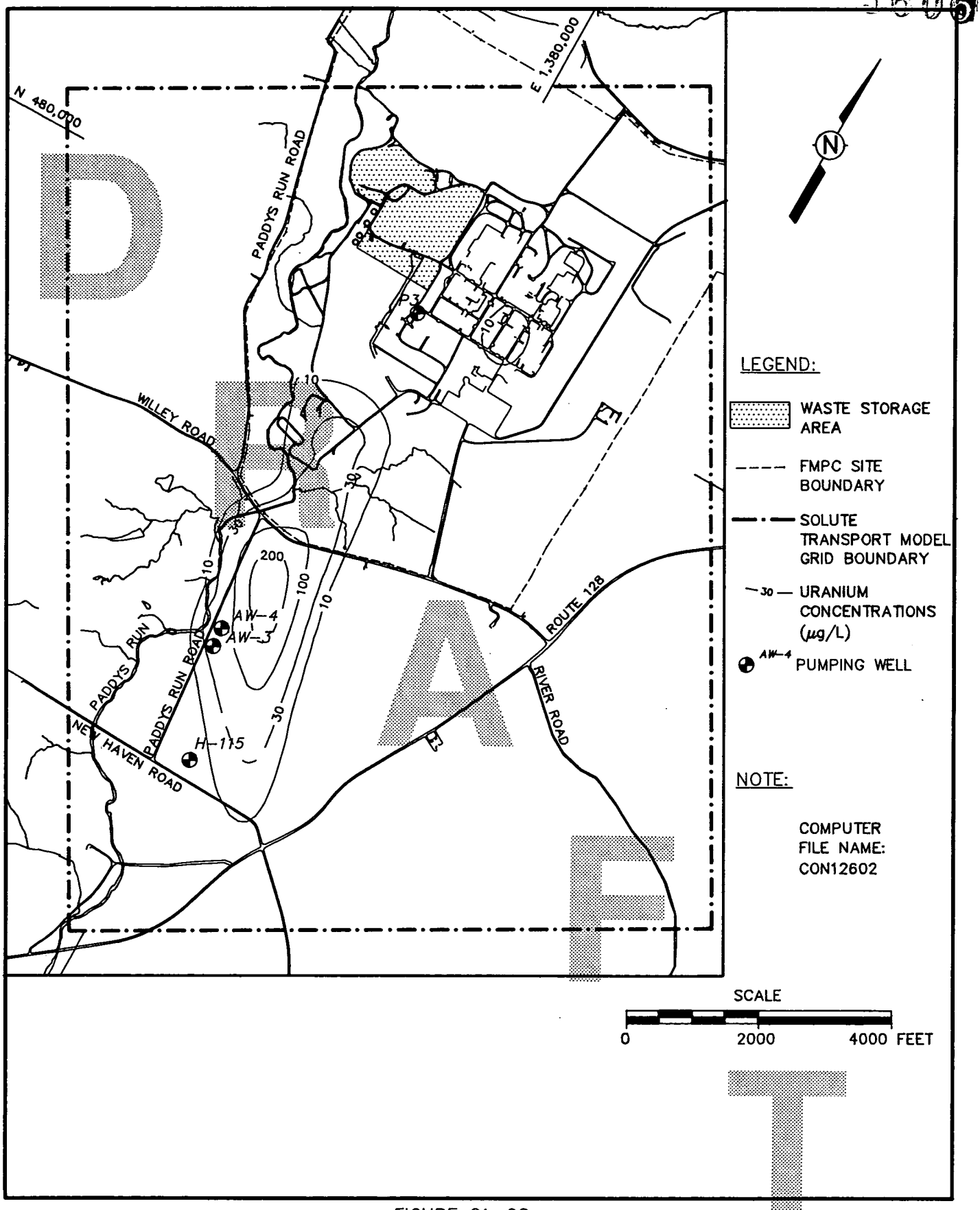
303317-A157 (GW) /01 (PGH)

FIGURE 21-26
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
PRESENT CONDITIONS
LAYER 5
RETARDATION FACTOR 9



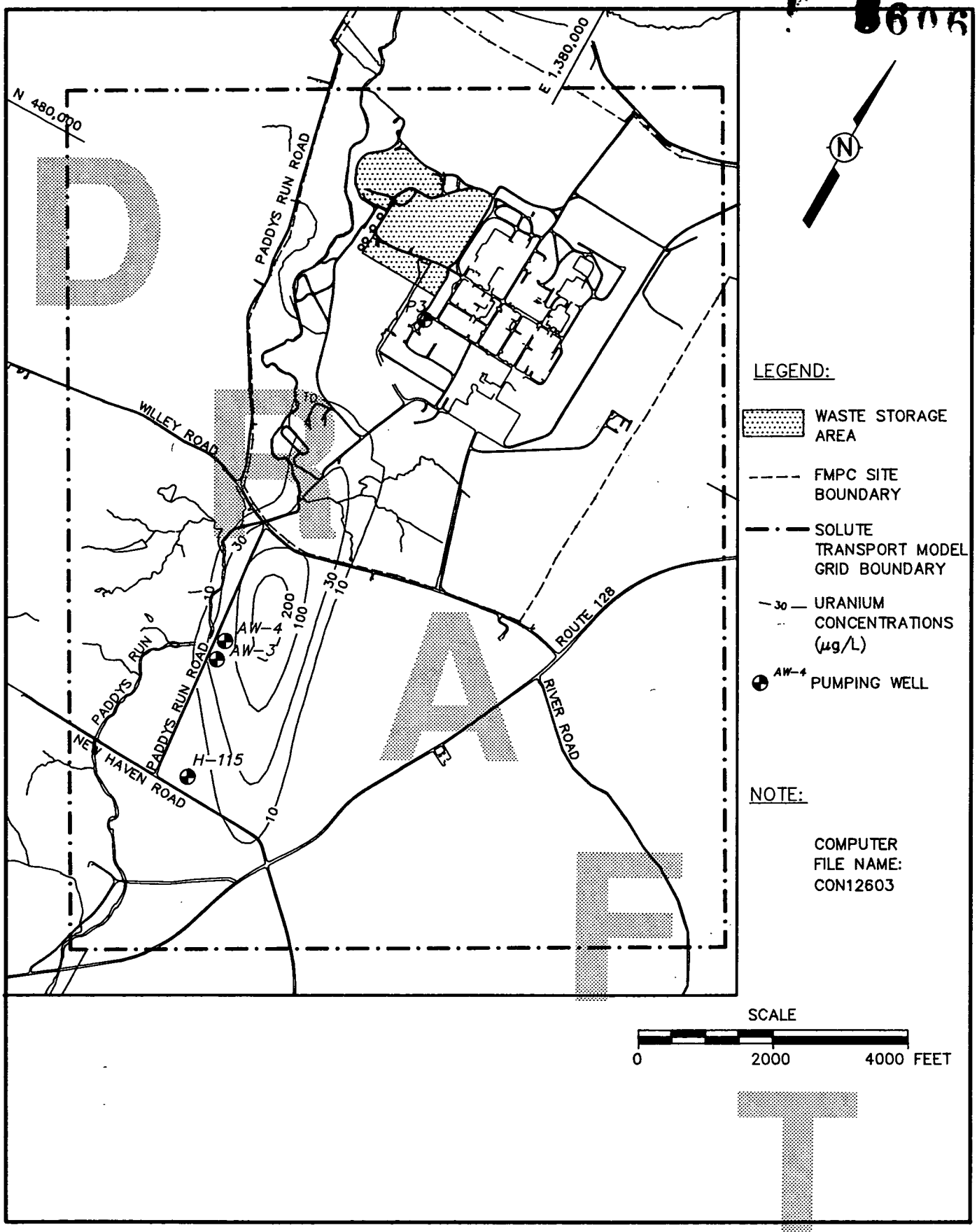
303317-A161 (GW) /01 (PGH)

FIGURE 21-27
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
PRESENT CONDITIONS
LAYER 1
RETARDATION FACTOR 12



303317-A169 (GW) /01 (PGH)

FIGURE 21-28
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
PRESENT CONDITIONS
LAYER 2
RETARDATION FACTOR 12



303317-A164 (GW) /01 (PGH)

FIGURE 21-29
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
PRESENT CONDITIONS
LAYER 3
RETARDATION FACTOR 12

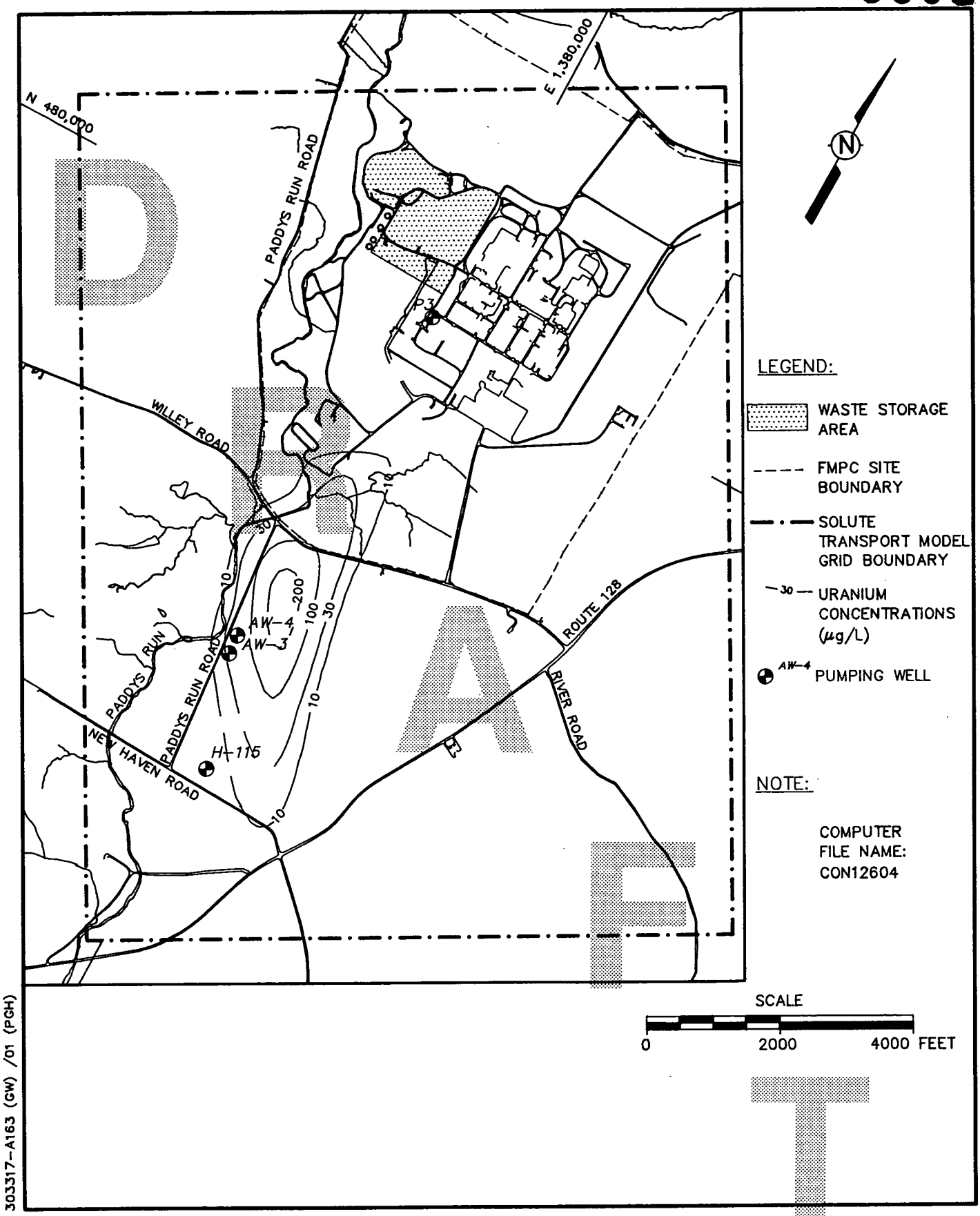
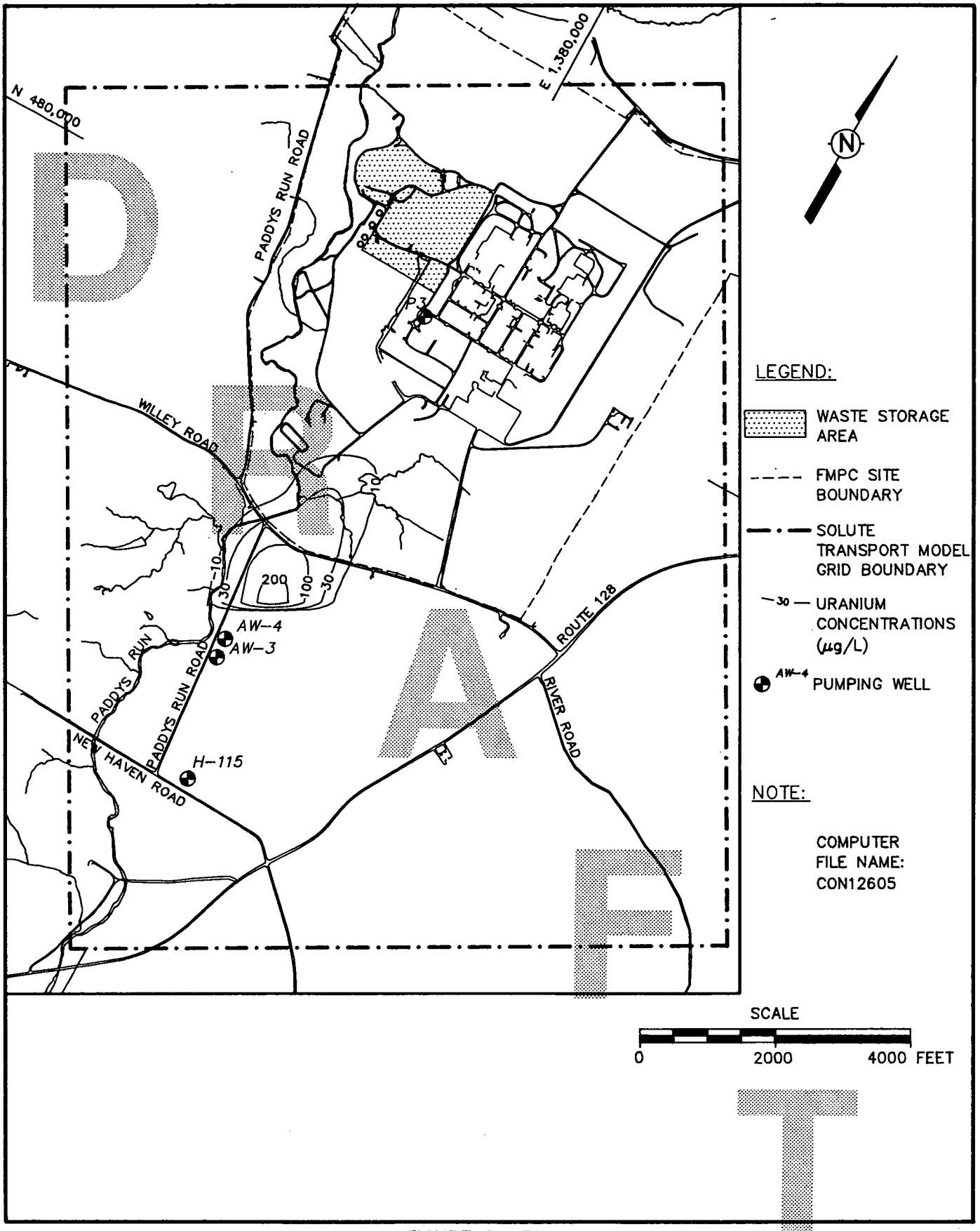


FIGURE 21-30
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
PRESENT CONDITIONS
LAYER 4
RETARDATION FACTOR 12



303317-A162 (GW) /01 (PGH)

FIGURE 21-31
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
PRESENT CONDITIONS
LAYER 5
RETARDATION FACTOR 12

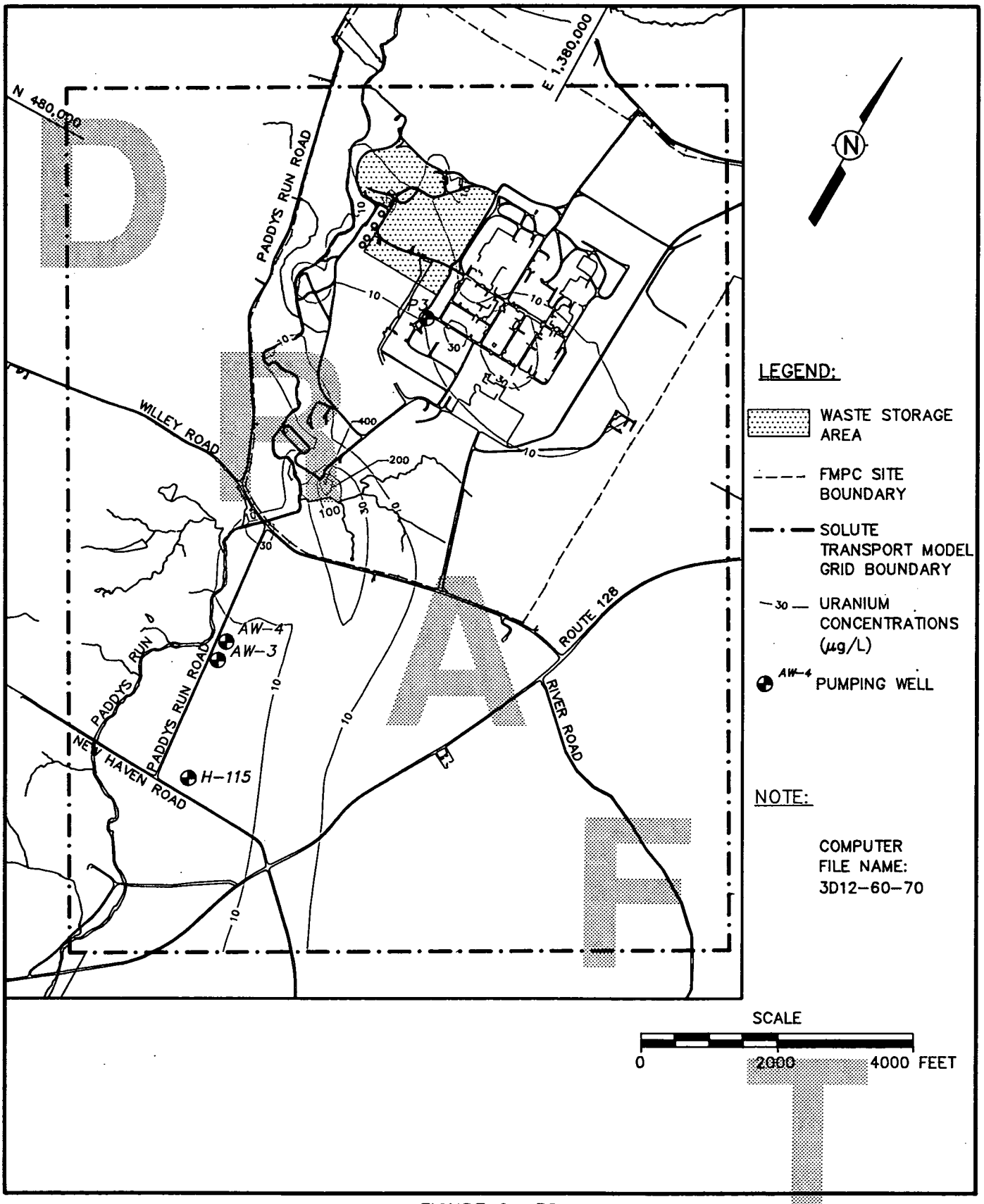
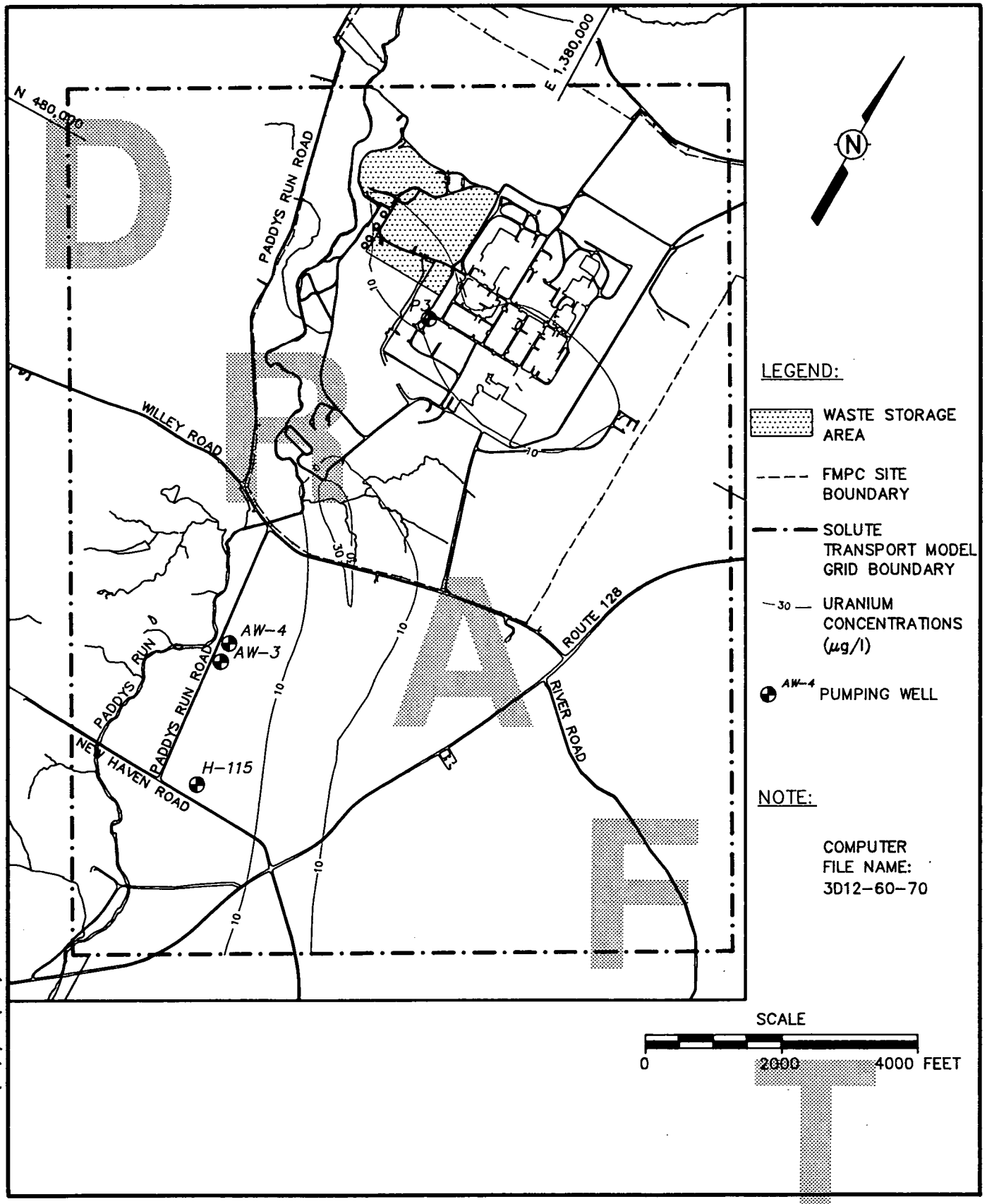
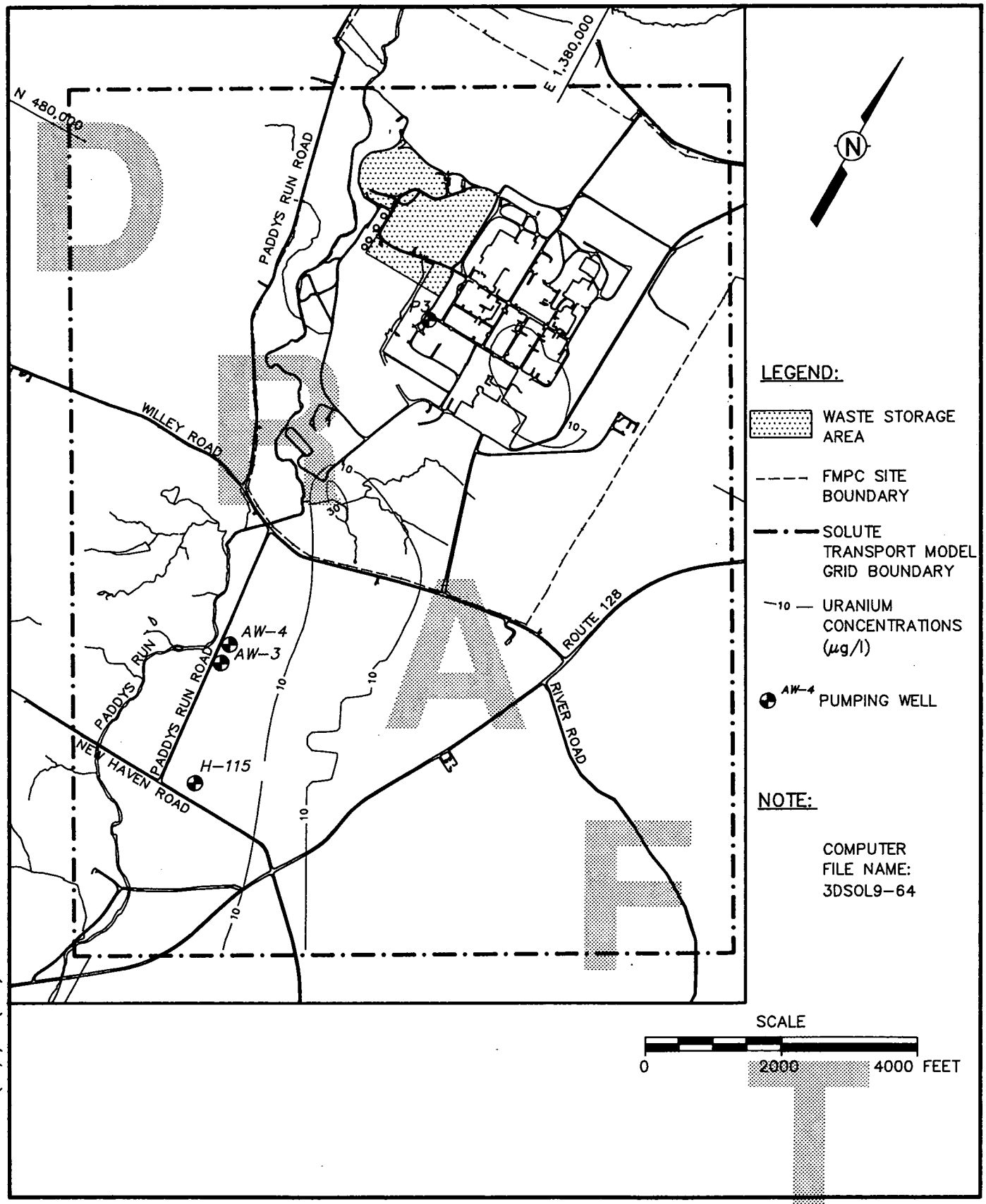


FIGURE 21-32
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
70 YEARS FROM PRESENT
LAYER 1
RETARDATION FACTOR 12



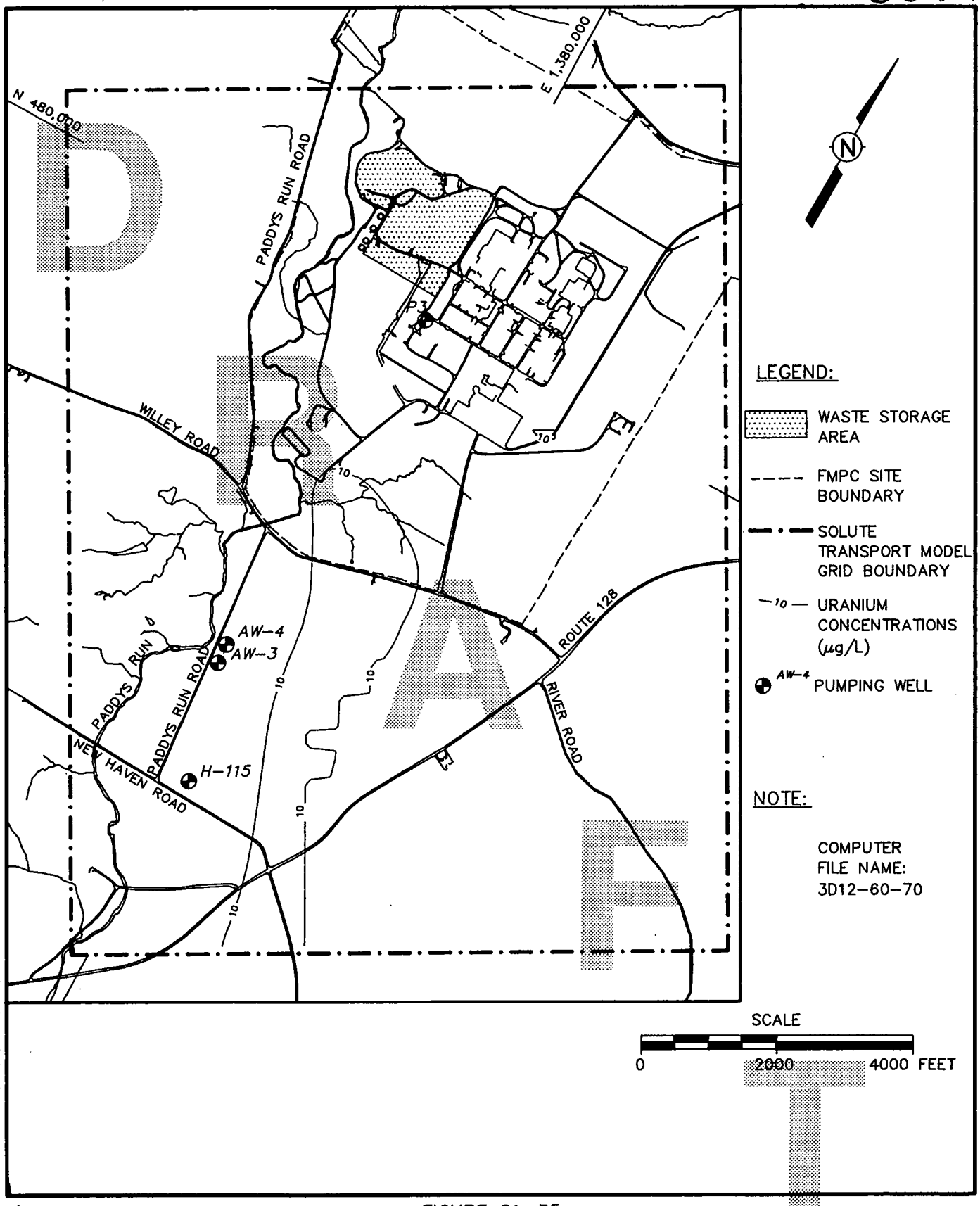
303317-A183 (GW) /01 (PGH)

FIGURE 21-33
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
70 YEARS FROM PRESENT
LAYER 2
RETARDATION FACTOR 12



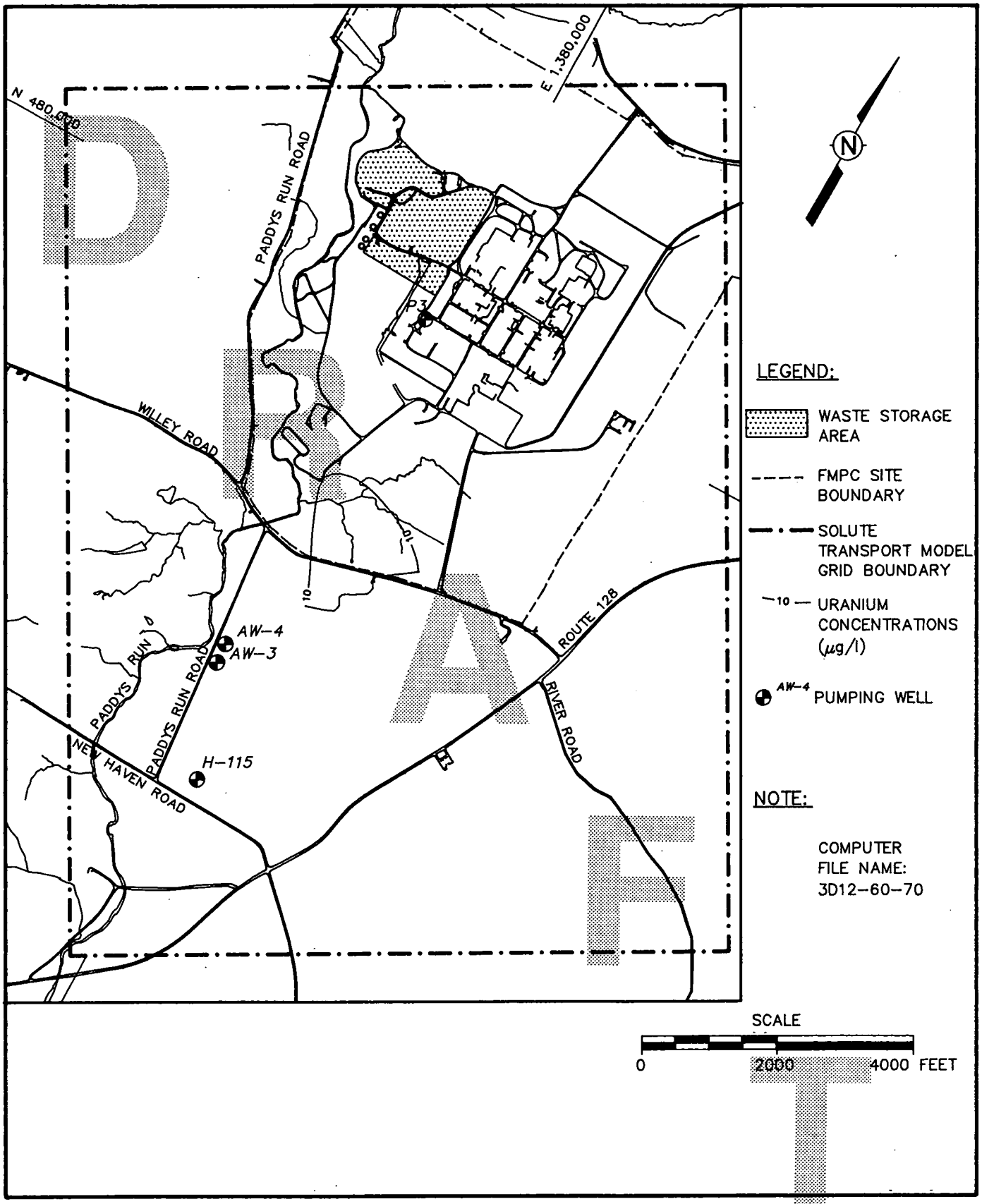
303317-A184 (GW) /01 (PGH)

FIGURE 21-34
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
70 YEARS FROM PRESENT
LAYER 3
RETARDATION FACTOR 12



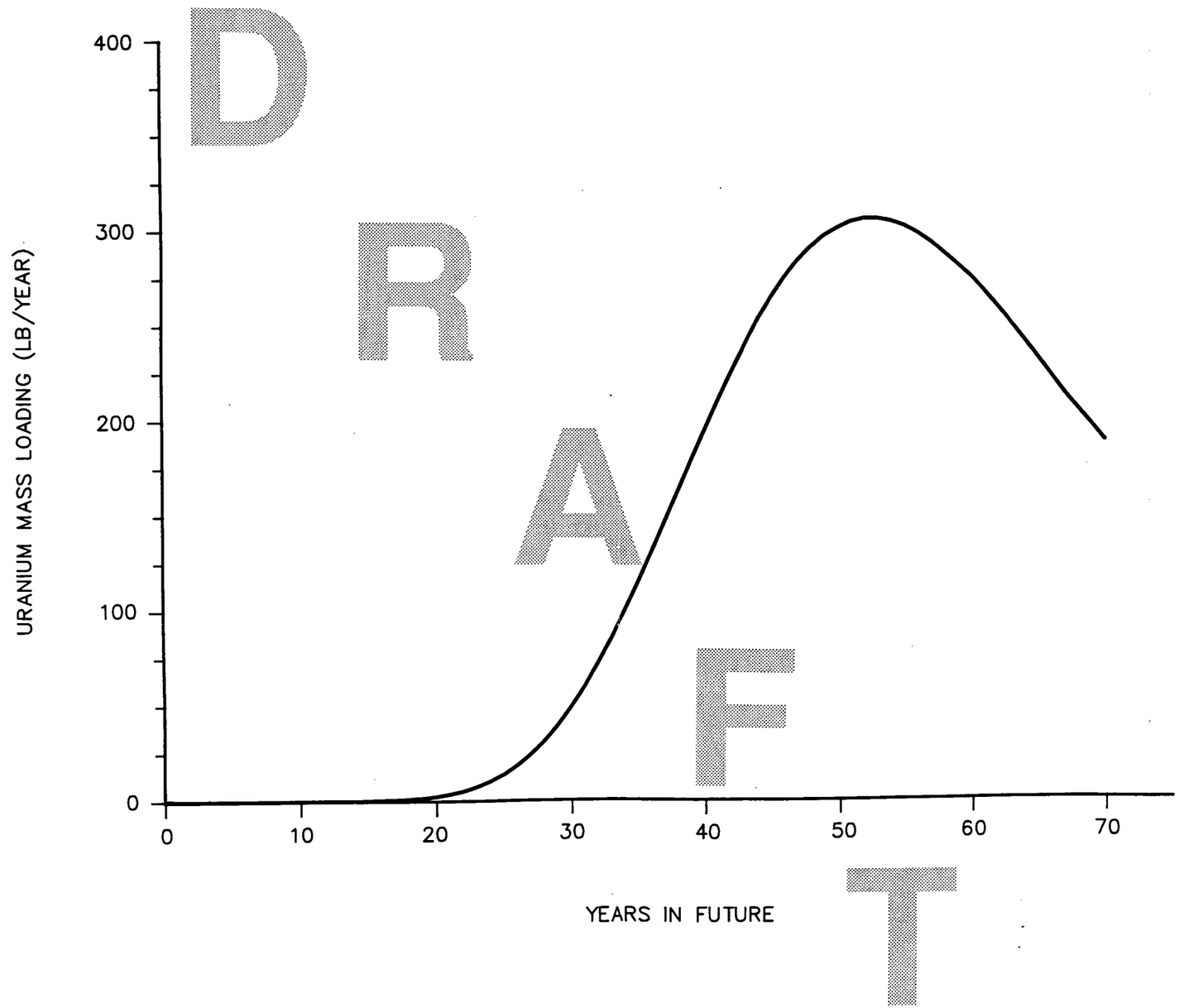
303317-A185 (GW) /01 (PGH)

FIGURE 21-35
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
70 YEARS FROM PRESENT
LAYER 4
RETARDATION FACTOR 12



303317-A186 (GW) /01 (PGH)

FIGURE 21-36
3-DIMENSIONAL SOLUTE TRANSPORT MODEL
SIMULATED URANIUM CONCENTRATIONS
70 YEARS FROM PRESENT
LAYER 5
RETARDATION FACTOR 12



DRAFT

NOTE:

YEAR ZERO (0) REPRESENTS YEAR 1990

303317-B753 (GW) (PGH)

FIGURE 21-37
ESTIMATED URANIUM DISCHARGE TO
THE GREAT MIAMI RIVER

22.0 MODELING OF LONG-TERM MIGRATION POTENTIAL

The two objectives for the analysis of the long-term migration of chemical constituents through the subsurface were to:

- Provide input for the baseline risk assessment and for the operable unit remedial investigations
- Aid in the preliminary evaluation of alternatives for the operable unit feasibility studies

To complete the baseline risk assessment for each operable unit, it was necessary to evaluate the fate and transport of constituents of concern through the groundwater pathway from the waste units to the regional aquifer and ultimately to the receptor. The focus of the RI and risk assessment portions of the analysis was to determine the extent and magnitude of groundwater contamination under two risk-based scenarios:

- Scenario 1 assumes that institutional controls are active on the site for the next 100 years and that the hypothetical receptor is located at the FMPC boundary
- Scenario 2 assumes that institutional controls will be lost after 100 years and the hypothetical receptor of groundwater is located directly at the waste site for up to 500 years into the future

For the feasibility study, it was necessary to establish the potential for future contamination from the waste source to evaluate the need for remediation and the types of remediation that will be acceptable. This type of analysis established the no-action alternative as a starting point for evaluating other alternatives.

In this chapter, the procedure and methodology for long-term migration potential is presented for Waste Pit 1 of Operable Unit 1 as an example. The results of the analysis for the rest of Operable Unit 1 and other operable units are presented in each operable unit RI report.

22.1 SUMMARY OF APPROACH

The fate and transport model progressed from the development of a conceptual model, based upon the depositional history and hydrogeology of a site, to the formulation of a mathematical representation of the conceptual model. Groundwater and vadose zone systems are complex and some simplification was required to represent these systems. Simplifications were bound to basic hydrologic principles. In application of the fate and transport results, the assumptions, limitations, and uncertainties of the model were considered.

Based on available data and work completed to date, an analytical model, ODAST, was selected for projecting the fate and transport of constituents in the vadose zone and a numeric model, SWIFT III, was selected for the flow analysis of the Great Miami Aquifer. Since the primary receptor in both cases is a hypothetical groundwater user of the Great Miami Aquifer, conservative travel times and constituent concentrations were calculated. The numeric solution combined with geochemical modeling provided a conservative order of magnitude estimate of the constituent concentrations.

22.2 MODEL DESIGN

The conceptual flow models for each operable unit varied based upon the site-specific characteristics of the underlying geologic material. These characteristics were developed based upon a review of the available RI/FS boring logs (Appendix A), appropriate cross sections (Figures 11-7 through 11-18 and 17-2 through 17-12), and fence diagrams (Appendix Q) for the various operable units.

The conceptual source-pathway-receptor relationship for Operable Unit 1 is shown in Figure 22-1. As this figure depicts, after the contaminant leaches from the waste unit, it moves downward vertically through the glacial till. The leachate, as it migrates through the till, will undergo geochemical reactions, changing the composition of the leachate. Subsequently, the leachate will migrate through the unsaturated sand and gravel portion of the buried channel sediments, resulting in further geochemical reactions and leachate compositional changes. Most of the chemical constituents present in the leachate will be retarded, causing delayed arrival. Travel times will also be controlled by hydrogeologic characteristics of the transport pathway. After reaching the Great Miami Aquifer, contaminants will migrate in the direction of the flow while undergoing dilution, dispersion, decay, and geochemical reactions. Contaminant concentration profiles for different locations and different times were computed based on these processes.

For the simulation of contaminant concentration at the receptors, the following tasks were performed:

- The constituents of concern were identified
- A conceptual hydrogeologic model was developed for each waste unit
- Leachate concentrations and attenuation reactions were estimated through the use of geochemical modeling for inorganics and partitioning coefficients for organics
- Retardation of constituents was estimated using partitioning coefficients and the hydrogeologic characteristics

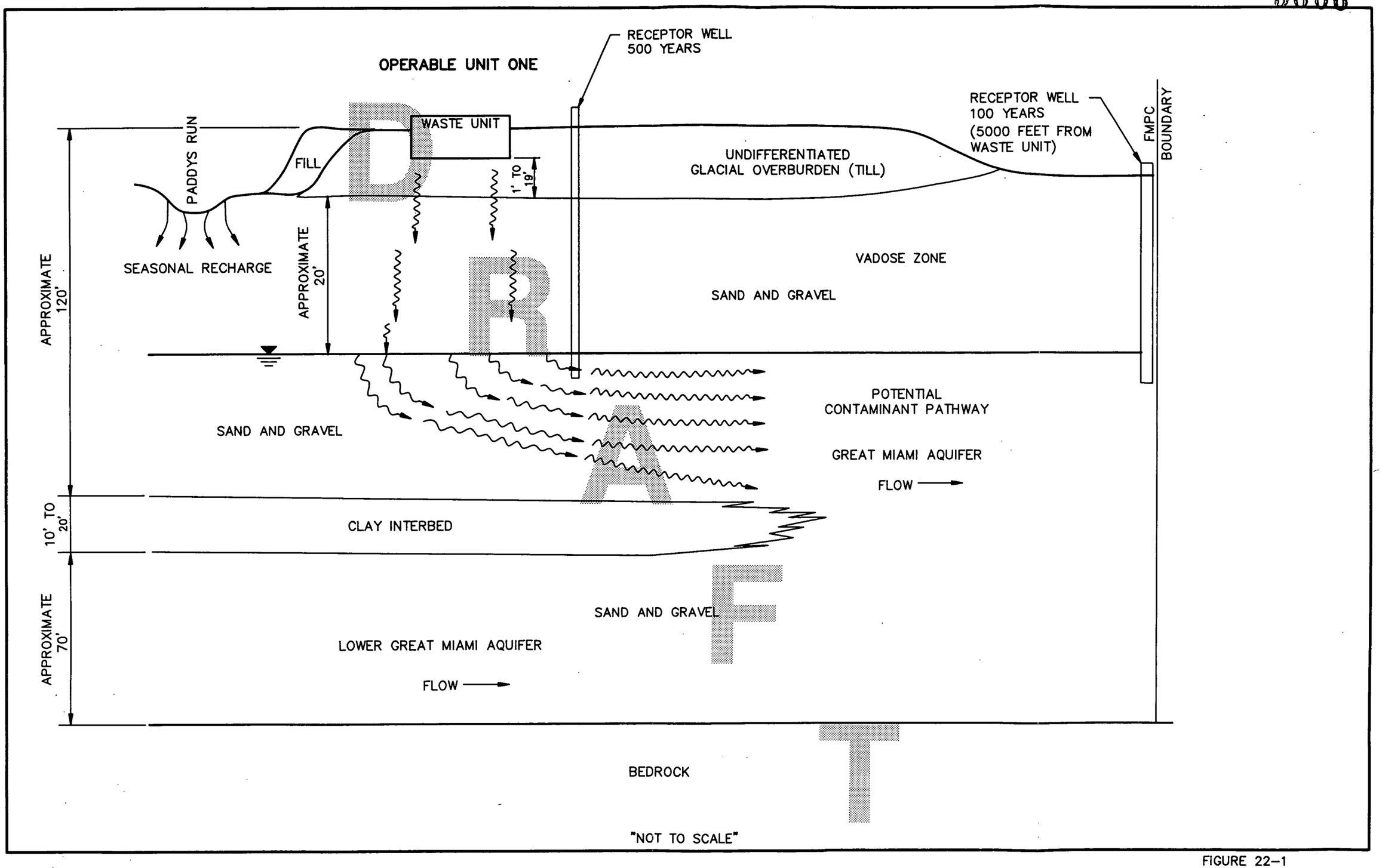


FIGURE 22–1
OPERABLE UNIT 1
CONCEPTUAL GROUNDWATER FLOW
PATHWAY

- Analytical modeling was used to determine the movement of constituents through the vadose zone
- Numerical modeling was used to estimate the movement of constituents through the Great Miami Aquifer

22.3 IDENTIFICATION OF THE CONSTITUENTS OF CONCERN

The physical characteristics and inventory of the wastes were estimated based on production history, production description, and waste composition. The waste inventories are reported in the operable unit RI report. Examples of waste inventory profiles for Waste Pit 1 of Operable Unit 1 are presented in Table 22-1. This table contains constituents of concern, their concentration, and total amount of available constituents.

22.4 COMPUTER CODES USED

The models used for flow evaluation were ST1D (IT 1990), a one-dimensional analytical solution; ODAST (Javendel et al. 1984), a one-dimensional analytical solution; and SWIFT III (Geotrans 1988), a finite-difference three-dimensional model. ST1D was used for the initial screening of constituents for mobility. ODAST was used for determining fate and transport of the remaining constituents in the unsaturated zone, while SWIFT III was used for flow in the regional aquifer. The initial source concentration was developed using leachate data, where available, and geochemical modeling for other constituents of concern. Each layer in the conceptual flow system was analyzed separately with the concentrations from the upper layers acting as the input concentrations to the lower layers. The depletion of the waste source over time and radioactive decay were taken into account in the vadose zone modeling.

The model codes ST1D and ODAST used for contaminant movement in the vadose zone are based on the solution originally developed by Ogata and Banks (1961). The ODAST code can account for retardation of contaminants, source changes, and decay. Both computer codes calculate the normalized concentrations of a given constituent in a uniform flow field from a source having a constant or varying concentration in the initial layer. ST1D and ODAST have been extensively verified against STRIP1B (Batu 1989 and Wilson and Miller 1978).

The model code used for the analysis of contaminant transport in the regional aquifer was SWIFT III. The SWIFT III code is a fully transient three-dimensional, finite-difference model which solves the coupled equations for flow and transport in geologic media. The model has been applied at the site since 1988 and has been extensively calibrated against the existing uranium groundwater contamination. The SWIFT III programs consists of a main routine and about

TABLE 22-1

CONSTITUENT AND LEACHATE CONCENTRATION
WASTE PIT 1, OPERABLE UNIT 1

Constituent	Concentration (mg/kg)	mg Available	Leachate (mg/L) (SOL)	Leach Rate (mg/Day)	Depletion In Years
U-234	9.75E-02	5.23E+06	1.26E-08	8.07E-05	1.78E+08
U-235	2.80E+01	1.50E+09	3.62E-06	2.32E-02	1.77E+08
U-238	6.32E+03	3.39E+11	8.18E-04	5.24E+00	1.77E+08
Th-228	1.33E-08	7.13E-01	7.28E-19	4.66E-15	4.19E+11
Th-230	4.26E-02	2.28E+06	7.31E-13	4.68E-09	1.33E+12
Th-232	1.07E+02	5.74E+09	1.84E-09	1.18E-05	1.33E+12
Tc-99	9.65E-04	5.17E+04	8.74E-05	5.60E-01	2.53E+02
Cs-137	1.27E-08	6.81E-01	9.96E-11	6.38E-07	2.93E+03
Ra-226	3.35E-05	1.79E+03	1.03E-06	6.60E-03	7.44E+02
Magnesium	2.72E+04	1.46E+12	1.41E+01	9.03E+04	4.43E+04
Silver	8.90E+00	4.77E+08	2.38E-04	1.52E+00	8.57E+05
Anthracene	2.28E-02	1.22E+06	7.21E-05	4.62E-01	7.24E+03
Benzo(a)anthracene	5.88E-02	3.15E+06	1.30E-05	8.34E-02	1.04E+05
Benzo(b)fluoranthene	1.46E-01	7.82E+06	3.23E-06	2.07E-02	1.04E+06
Benzo(k)fluoranthene	3.60E-02	1.93E+06	4.56E-07	2.92E-03	1.81E+06
Benzo(a)pyrene	3.60E-02	1.93E+06	3.19E-06	2.04E-02	2.59E+05
Chrysene	2.01E-01	1.08E+07	4.45E-05	2.85E-01	1.04E+05
Ethylbenzene	2.56E-02	1.37E+06	1.62E-03	1.04E+01	3.62E+02
Fluoranthene	2.49E-01	1.34E+07	6.49E-05	4.16E-01	8.83E+04
2-Methylnaphthalene	5.20E-02	2.79E+06	6.36E-04	4.07E+00	1.88E+03
Naphthalene	1.88E-02	1.01E+06	7.24E-04	4.64E+00	5.97E+02
Phenanthrene	7.23E-01	3.88E+07	2.21E-03	1.41E+01	7.51E+03
Pyrene	1.71E-01	9.18E+06	7.58E-05	4.86E-01	5.18E+04
Toluene	3.86E-02	2.07E+06	6.98E-03	4.47E+01	1.27E+02
Acetone	2.50E-01	1.34E+07	3.88E+01	2.48E+05	1.48E-01
2-Butanone	2.62E+00	1.40E+08	1.28E+02	8.21E+05	4.67E-01
Aroclor-1248	7.04E-02	3.77E+06	6.24E-06	3.99E-02	2.59E+05
Aroclor-1254	4.70E+00	2.52E+08	4.16E-04	2.67E+00	2.59E+05
Aroclor-1260	1.40E+00	7.51E+07	6.60E-03	4.23E+01	4.87E+03
DDT	3.20E-01	1.72E+07	2.84E-03	1.82E+01	2.60E+03
Bis(2-Ethylhexyl)phthalate	4.12E-01	2.21E+07	1.82E-04	1.17E+00	5.18E+04
Di-n-butyl Phthalate	7.42E-02	3.98E+06	4.16E-05	2.66E-01	4.09E+04
Di-n-octyl Phthalate	3.07E-01	1.64E+07	1.72E-04	1.10E+00	4.08E+04
Chloroform	9.18E-02	4.92E+06	8.72E-02	5.58E+02	2.42E+01
Methylene Chloride	1.32E-01	7.09E+06	6.58E-01	4.21E+03	4.61E+00
Tetrachloroethene	1.90E-02	1.02E+06	2.21E-03	1.42E+01	1.97E+02
1,1,1-Trichloroethane	3.59E-02	1.92E+06	1.08E-02	6.90E+01	7.62E+01

70 supporting subroutines. The SWIFT III code and its application is fully outlined in Chapters 20.0 and 21.0 of this report.

22.5 INPUT PARAMETER DETERMINATION

The parameters needed to perform the long-term migration analysis could be divided into flow parameters and contaminant transport parameters. Flow parameters affect the overall hydraulic flow field and the velocity of groundwater movement. The contaminant transport parameters affect the rate of migration and the fate of the contaminant.

Whenever possible, site-specific values were used for the analysis and calculation of the contaminant concentration at the hypothetical receptor location. However, certain parameters were not available for all operable units and estimates had to be made from the literature and the results of the FMPC studies. The computer concentrations provided a basis for comparing the relative risk of various chemical constituents. However, as the feasibility study process extends into the selection of a viable alternative, it will be necessary to collect site-specific data to verify the validity of the estimated values.

22.5.1 Flow Parameters

The primary parameters that affect flow in both the vadose zone and the saturated zone are moisture content, hydraulic conductivity, specific yield, and porosity. The value of these parameters for Waste Unit 1, OU1 is given in Table 22-2.

Moisture Content of the Vadose Zone

The moisture content is the amount of moisture held within the vadose zone at any given time. This moisture content, or degree of saturation, will vary continuously over time and along flow paths. It is an important characteristic that affects both the ability of a material to pass fluids (hydraulic conductivity) and the force that keeps water within the material (matric suction). This moisture content can vary from saturation to a condition known as air dryness (Hillel 1982).

At the site, no measurements were made of soil or vadose zone moisture content; therefore, two estimation techniques were used. Both of these techniques require several simplifying assumptions, as follows:

- Moisture content was evaluated as an average, steady-state condition
- The unsaturated moisture content cannot exceed the saturated moisture content

TABLE 22-2

LONG-TERM FATE AND TRANSPORT MODEL
INPUT DATA
WASTE PIT 1, OPERABLE UNIT 1

DRAFT

Parameter	Till	Unsaturated Sand and Gravel
Hydraulic Conductivity		
Horizontal (ft/day)	0.147	450
Vertical (ft/day)	0.00735	45
Thickness (feet)	5	20
Porosity (%)	34	39
Specific Yield (%)	0.06	0.25
Bulk Density (g/cc)	1.78	1.60
Field Capacity (%)	28	14
Dispersivity (ft/sq./day)	0.0012	0.00187
Organic Content (%)	4	4
Fines passing less than 200 mesh (%)	70	16

- The amount of infiltration cannot exceed the saturated hydraulic conductivity
- The saturated moisture content approximately equals the porosity
- The minimum moisture content equals the residual water content
- The residual water content closely corresponds to specific retention or field capacity

The first technique is based upon Clapp and Hornberger (1978), as presented in U.S. EPA (1988). This equation states that:

$$\Theta = (\Theta_s) * (q/K_s)^{1/(2b + 3)}$$

where:

Θ = moisture content in the vadose zone
Θ_s = saturated moisture content in the vadose zone
q = infiltration or recharge rate
K_s = saturated hydraulic conductivity
b = soil-specific exponential parameter, estimated from U.S. EPA (1988)

The second technique is based upon the relationship that:

$$Sy + \Theta_r = n$$

where:

Sy = specific yield
Θ_r = specific retention or residual water content
n = porosity

For each layer within the conceptual flow model, the moisture content was estimated using both techniques.

Hydraulic Conductivity

Perhaps the most important difference between unsaturated and saturated flow is hydraulic conductivity. When the matrix is saturated, all of the pores are water-filled and conducting, so that conductivity is at its maximum. When the matrix dries, some of the pores fill with air and the conductive portion of the unconsolidated material decreases. The larger, more-conductive pores are

the first to drain, leaving only the smaller, less-conductive pores available for water movement. Furthermore, as the water drains, the matric suction that attracts water to the particles within the matrix increases.

For these reasons, the transition from saturation to unsaturation generally entails a steep drop in hydraulic conductivity, which may decrease by several orders of magnitude as the matrix dries and suction increases. As the degree of saturation decreases, the conductivity may drop so low that very long times are required for any appreciable seepage to occur as the saturation begins to increase.

At saturation, the most conductive unconsolidated materials are those with large interconnected pores, such as sands; while the least are those with small micropores, such as silts or clays. However, under unsaturated conditions, the opposite may be true. Material with large interconnected pore spaces drains quickly and suction rapidly increases, thereby rapidly decreasing hydraulic conductivity. In a material with small pores, many of the pores retain and conduct water even under significant suction, so that the hydraulic conductivity does not decrease as quickly. Because of this, the hydraulic conductivity of a silt or clay may approach or exceed that of a sandy layer under unsaturated conditions.

Because the hydraulic conductivity is dependent on water content and matric suction in unsaturated material, measuring unsaturated hydraulic conductivity in the laboratory or field is extremely difficult. As the degree of saturation or water content changes constantly over the length of a flow path, the unsaturated hydraulic conductivity will vary continuously. Therefore, a unique value for hydraulic conductivity cannot be determined (Everett et al. 1984).

The unsaturated hydraulic conductivity is commonly estimated based upon a relationship between the soil moisture curve and saturated hydraulic conductivity using techniques such as found in van Genuchten (1978). However, at the FMPC, no measurements of water content, matric suction, or unsaturated hydraulic conductivity have been completed. Therefore, it was necessary to rely on direct measurements of saturated hydraulic conductivity for the hydraulic conductivity of the vadose zone. When these estimates were applied in the calculation of velocity, they were adjusted to reflect partial saturation. The use of the saturated hydraulic conductivities will tend to overestimate the movement of fluids through the vadose zone.

A series of slug tests were performed in water-bearing zones of the till at the FMPC. This provides a conservative estimate of hydraulic conductivity since the anticipated hydraulic

conductivity in the till would be an order of magnitude less (Norris 1962). Hydraulic conductivity measured in wells closest to Pit 1 was 0.147 foot per day (5 x 10^{-5} cm/s). Vertical hydraulic conductivities were calculated by dividing the horizontal hydraulic conductivity by 20. The thickness of till beneath Pit 1 was assumed to be 5 feet.

The average hydraulic conductivity of the Great Miami Aquifer is 450 feet per day. As described above, the aquifer is divided by a clay interbed approximately 120 feet below the surface. The receptor pathway considered for this analysis is the upper part of the Great Miami Aquifer, above the clay interbed. The upper unsaturated section of the unsaturated sand and gravel was assumed to have the same 450-foot per day (0.16 cm/s) hydraulic conductivity. The vertical hydraulic conductivity was calculated to be 45 feet per day (0.016 cm/s) by dividing the horizontal conductivity by 10. The thickness of unsaturated sand and gravel is 20 feet.

Specific Yield

The storage coefficient is a measure of the amount of water that is released from storage as the water level in an aquifer declines. For the purposes of this analysis, the specific yield was used for the estimation of the moisture content of the vadose zone material. Estimates for the specific yield were derived from published tables found in Morris and Johnson (1967) and van der Leeden et al. (1990).

Porosity

The porosity of a material is a measure of the void volume or pore space within a material as compared to the total volume. Porosity is important in determining the velocity of fluids in saturated zones and in estimating values for the moisture content. Estimates for porosity were obtained from published tables found in Morris and Johnson (1967), Driscoll (1986), and van der Leeden et al. (1990).

22.5.2 Contaminant Transport Parameters

Dispersivity

Dispersion is the spreading of contaminants due to flow-related mechanical dispersion and molecular diffusion. With the exception of fine-grained material, the mechanical dispersion is the dominant process for the types of materials at the FMPC. In subsurface systems, longitudinal dispersion, or spreading in the direction of flow, is much more important than transverse dispersion. For the analysis of flow through the vadose zone, only longitudinal dispersion was calculated using the techniques outlined in Mills et al. (1985).

Bulk density

Bulk density, which is used in the calculation of retardation factors, is the dry particle mass per unit volume of soil. Estimates of bulk density were taken from ASI/IT (1989) and Morris and Johnson (1967).

Partition Coefficient

Partition coefficients (K_d) are used to account for the reactive portion of the transport equation. The use of K_d values is subject to several important assumptions and considerations, as follows:

- There is a linear relationship between the solid and solution-phase concentrations.
- The use of K_d values assumes that the isotherm that describes reactions between the solution and the solids in the matrix is linear and passes through the point (0,0). Ideally, sufficient K_d values should be measured to evaluate the shape of the isotherm. This is particularly important in the design of groundwater treatment projects where the use of improper K_d and retardation factors will have a major effect on the projection of pumping and capture times for a contaminant plume. For initial modeling efforts, it is often necessary to use published or calculated K_d values; however, the inherent problems associated with these values must be recognized.
- K_d values are highly sensitive to changes in the major chemistry of the solution phase. It is, therefore, important that the values be measured or estimated under conditions that will represent, as closely as possible, those of the contaminant plume. It is also important to note that the major geochemistry of the plume could change over time if affected by multiple solutes that are present in any waste disposal facility.
- Under conditions of diffusive transport, the advection-dispersion equation with a linear reaction term appears to give an accurate representation of the transport process. This provides further evidence of the greater predictability of solute transport in fine-grained geologic materials.

Although the K_d values and more general formulations of the reaction term have numerous assumptions and uncertainties associated with them, they nevertheless provide a practical means of incorporating the reaction process into the transport models. The alternative of using thermodynamic modeling of potential plumes originating from waste sites is generally impractical and is likely to represent further uncertainties.

K_d values were estimated using two techniques. The first technique, which was applied to inorganics and radionuclides, involved the analysis and selection of published values. K_d values found in the literature search were carefully screened to select those values that were derived under conditions that approximated those at the FMPC (Chapter 16.0). Planned geotechnical studies at the

FMPC could refine these values prior to the application of the model to the detailed evaluation and design of groundwater remedial alternatives. The distribution coefficients for radionuclides and inorganics are given in Table 22-3. The second technique which was used for organics involved using empirical formulas (Mills et al. 1985) that relate the octanol-water partitioning coefficient to the octanol-carbon partitioning coefficient (K_{oc}). The K_{oc} value was combined with the amount of carbon and the size distribution of the vadose zone matrix to estimate a K_d value. The governing equation is given in Chapter 16.0 and composited values are presented in Table 22-4.

Retardation Factor

The retardation factors for chemical constituents were calculated using the following equation:

$$R_d = 1 + \Theta K_d/f_c$$

where:

R_d = retardation factor (dimensionless)
Θ = bulk density of solid matrix (g/cc)
K_d = distribution coefficient (cc/g)
f_c = field capacity (dimensionless)

The values of the above parameters are given in Tables 22-3 and 22-4.

22.6 MODEL APPLICATION

Initially, the ST1D program was used to eliminate the chemical constituents which will not reach the Great Miami Aquifer within 500 years. This was accomplished by calculating concentrations at the base of the vadose zones using input parameters given in Table 22-2 and different retardation factors and thicknesses. Subsequent to the identification of the constituents of concern, the concentration was calculated by the ODAST program. Initially, the concentration was calculated at the base of Layer 1, and then the results of Layer 1 were input to Layer 2 for concentration calculation at the base of Layer 2. The results were concentration versus time for each contaminant. An example of such a computation is shown in Figure 22-2. This figure shows the concentration of the Uranium 234 for Waste Unit 1 of Operable Unit 1. Similar curves have been prepared for other waste units and operable units.

TABLE 22-3

PARTITIONING COEFFICIENT AND RETARDATION FACTOR
FOR RADIONUCLIDES AND INORGANICS
WASTE PIT 1, OPERABLE UNIT 1

Constituent	Kd	Rd Vadose 1	Rd Vadose 2	Rd Aquifer
U-234	1.80E+00	1.24E+01	2.16E+01	8.38E+00
U-235	1.80E+00	1.24E+01	2.16E+01	8.38E+00
U-238	1.80E+00	1.24E+01	2.16E+01	8.38E+00
Th-228	5.80E+03	3.69E+04	6.63E+04	2.38E+04
Th-230	5.80E+03	3.69E+04	6.63E+04	2.38E+04
Th-232	5.80E+03	3.69E+04	6.63E+04	2.38E+04
Tc-99	1.18E-01	1.75E+00	2.35E+00	1.48E+00
Cs-137	1.90E+03	1.21E+04	2.17E+04	7.80E+03
Ra-226	6.96E+02	4.43E+03	7.96E+03	2.86E+03
Magnesium	1.20E+01	7.73E+01	1.38E+02	5.02E+01
Silver	1.80E+02	1.15E+03	2.06E+03	7.39E+02

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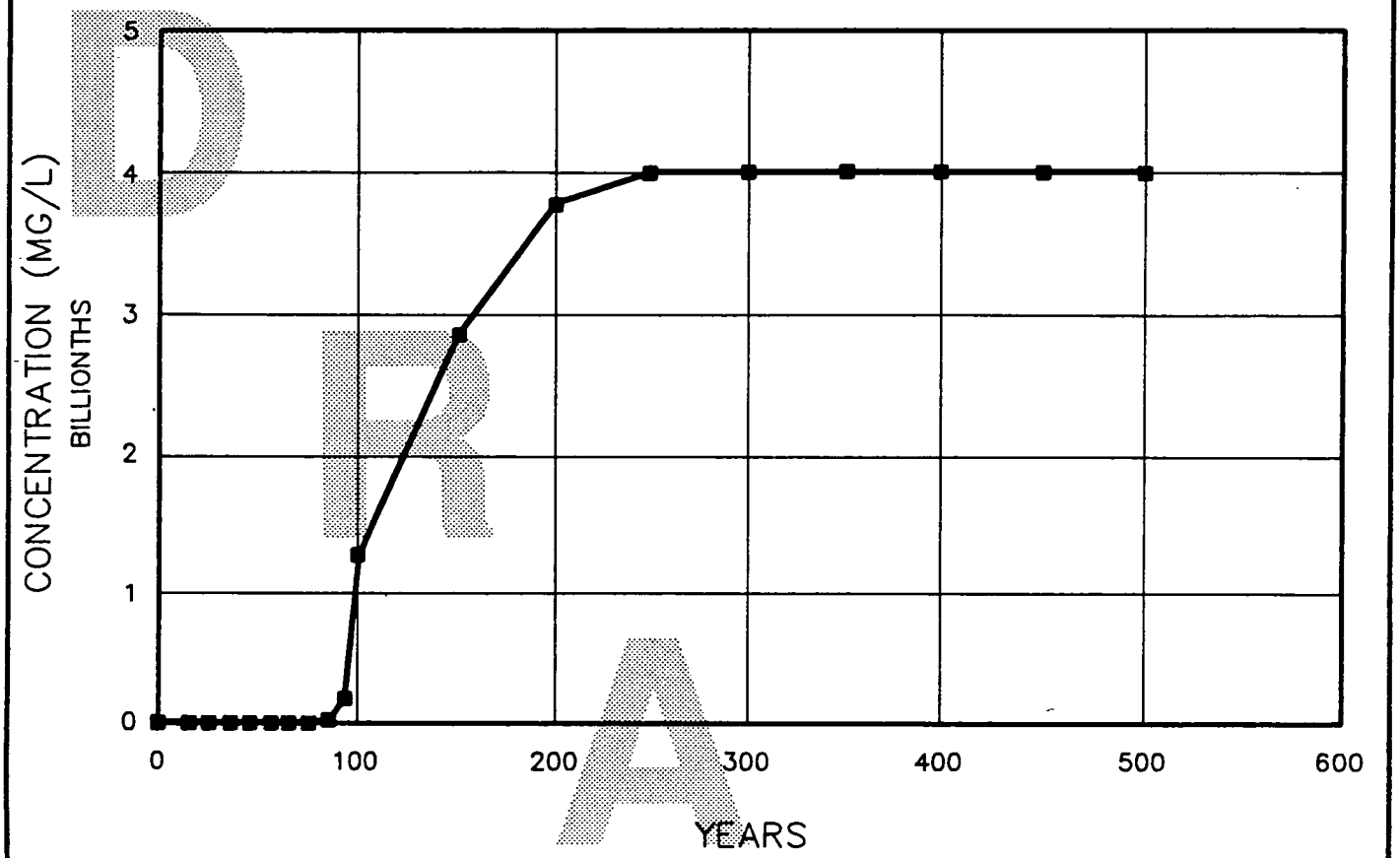
TABLE 22-4

PARTITIONING COEFFICIENTS AND RETARDATION FACTOR
FOR ORGANIC COMPOUNDS
WASTE PIT 1, OPERABLE UNIT 1

Constituent	Kow	Koc	Kd Vadose 1	Kd Vadose 2	Rd Vadose 1	Rd Vadose 2
Anthracene	2.80E+04	1.76E+04	5.36E+02	2.31E+02	3.41E+03	2.65E+03
Benzo(a)anthracene	4.00E+05	2.52E+05	7.66E+03	3.31E+03	4.87E+04	3.78E+04
Benzo(b)fluoranthene	4.00E+06	2.52E+06	7.66E+04	3.31E+04	4.87E+05	3.78E+05
Benzo(k)fluoranthene	7.00E+06	4.41E+06	1.34E+05	5.79E+04	8.52E+05	6.61E+05
Benzo(a)pyrene	1.00E+06	6.30E+05	1.92E+04	8.27E+03	1.22E+05	9.45E+04
Chrysene	4.00E+05	2.52E+05	7.66E+03	3.31E+03	4.87E+04	3.78E+04
Ethylbenzene	1.40E+03	8.82E+02	2.68E+01	1.16E+01	1.71E+02	1.33E+02
Fluoranthene	3.40E+05	2.14E+05	6.51E+03	2.81E+03	4.14E+04	3.21E+04
2-Methylnaphthalene	7.24E+03	4.56E+03	1.39E+02	5.98E+01	8.82E+02	6.85E+02
Naphthalene	2.30E+03	1.45E+03	4.40E+01	1.90E+01	2.81E+02	2.18E+02
Phenanthrene	2.90E+04	1.83E+04	5.55E+02	2.40E+02	3.53E+03	2.74E+03
Pyrene	2.00E+05	1.26E+05	3.83E+03	1.65E+03	2.44E+04	1.89E+04
Toluene	4.90E+02	3.09E+02	9.38E+00	4.05E+00	6.07E+01	4.73E+01
Acetone	5.70E-01	3.59E-01	1.09E-02	4.71E-03	1.07E+00	1.05E+00
2-Butanone	1.81E+00	1.14E+00	3.47E-02	1.50E-02	1.22E+00	1.17E+00
Aroclor-1248	1.00E+06	6.30E+05	1.92E+04	8.27E+03	1.22E+05	9.45E+04
Aroclor-1254	1.00E+06	6.30E+05	1.92E+04	8.27E+03	1.22E+05	9.45E+04
Aroclor-1260	1.88E+04	1.18E+04	3.60E+02	1.55E+02	2.29E+03	1.78E+03
DDT	1.00E+04	6.30E+03	1.92E+02	8.27E+01	1.22E+03	9.46E+02
Bis(2-Ethylhexyl)phthalate	2.00E+05	1.26E+05	3.83E+03	1.65E+03	2.44E+04	1.89E+04
Di-n-butyl Phthalate	1.58E+05	9.95E+04	3.03E+03	1.31E+03	1.92E+04	1.49E+04
Di-n-octyl Phthalate	1.58E+05	9.95E+04	3.03E+03	1.31E+03	1.92E+04	1.49E+04
Chloroform	9.33E+01	5.88E+01	1.79E+00	7.71E-01	1.24E+01	9.81E+00
Methylene Chloride	1.78E+01	1.12E+01	3.41E-01	1.47E-01	3.17E+00	2.68E+00
Tetrachloroethene	7.60E+02	4.79E+02	1.46E+01	6.28E+00	9.35E+01	7.28E+01
1,1,1-Trichloroethane	2.95E+02	1.86E+02	5.65E+00	2.44E+00	3.69E+01	2.89E+01

PIT/GW/TS.1-2/12-17/90



NOTE:
CONCENTRATION IS CALCULATED BENEATH THE WASTE UNIT AT GREAT MIAMI AQUIFER.

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00317-A246 (GW) (PGH)

FIGURE 22-2
COMPUTED CONCENTRATION OF URANIUM 234 FOR WASTE PIT 1
OPERABLE UNIT 1

An additional step in the model application was the use of the calibrated SWIFT III program. In this application, a normalized source loading was applied at each waste area, and normalized concentrations were calculated at the Great Miami Aquifer for different times, especially for 100 and 500 years.

Finally, using the ODAST and SWIFT III program results, the maximum concentrations at 100 and 500 years at the receptor locations were calculated.

22.7 RESULTS

As an example, the results from Waste Pit 1 located in Operable Unit 1 are presented in Table 22-5. The results from the remaining sites can be found in the respective RI reports. Generally, the analysis indicates that uranium and technetium are the most mobile of the radionuclides and metals present and will reach the Great Miami Aquifer. The highly soluble organics are also projected to reach the aquifer. However, the biodegradation of these organics, which can be significant in the vadose zone, was not considered.

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TABLE 22-5

COMPLETED CONCENTRATIONS
AT RECEPTOR LOCATIONS
WASTE PIT 1, OPERABLE UNIT 1

Constituent	Years To Reach Aquifer	Receptor Concentration Maximum 0 - 100 Years	Receptor Concentration Maximum 100 - 500 Years
U-234	Approx. 75	1.54E-11	3.95E-09
U-235	Approx. 75	4.81E-09	1.28E-06
U-238	Approx. 75	1.09E-06	2.95E-04
Th-228	Over 500	NA	NA
Th-230	Over 500	NA	NA
Th-232	Over 500	NA	NA
Tc-99	Approx. 20	1.10E-07	1.45E-05
Cs-137	Over 500	NA	NA
Ra-226	Over 500	NA	NA
Magnesium	Over 500	NA	NA
Silver	Over 500	NA	NA
Anthracene	Over 500	NA	NA
Benzo(a)anthracene	Over 500	NA	NA
Benzo(b)fluoranthene	Over 500	NA	NA
Benzo(k)fluoranthene	Over 500	NA	NA
Benzo(a)pyrene	Over 500	NA	NA
Chrysene	Over 500	NA	NA
Ethylbenzene	Over 500	NA	NA
Fluoranthene	Over 500	NA	NA
2-Methylnaphthalene	Over 500	NA	NA
Naphthalene	Over 500	NA	NA
Phenanthrene	Over 500	NA	NA
Pyrene	Over 500	NA	NA
Toluene	Approx. 150	NA	4.80E-04
Acetone	Approx. 10	8.29E-05	2.12E-03
2-Butanone	Approx. 15	8.76E-04	2.24E-02
Aroclor-1248	Over 500	NA	NA
Aroclor-1254	Over 500	NA	NA
Aroclor-1260	Over 500	NA	NA
DDT	Over 500	NA	NA
Bis(2-Ethylhexyl)phthalate	Over 500	NA	NA
Di-n-butyl Phthalate	Over 500	NA	NA
Di-n-octyl Phthalate	Over 500	NA	NA
Chloroform	Approx. 75	1.89E-05	2.03E-03
Methylene Chloride	Approx. 25	4.42E-05	1.13E-03
Tetrachloroethene	Over 500	NA	NA
1,1,1-Trichloroethane	Over 500	1.40E-06	8.63E-11

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PART V. CONCLUSIONS

Part V of the FMPC groundwater report presents the results and conclusion of the groundwater program, emphasizing groundwater flow systems, source-pathway-receptor relationships, and the nature and extent of contamination. The uncertainties associated with groundwater programs are also discussed in this part.

Figure V-1 presents the rationale and logic of the overall report. The report conclusion relys heavily on Parts III and IV, while Parts I and II have provided the fundamental data for the development of Parts III and IV.

Part V includes Chapters 23.0 and 24.0. The summary of the FMPC groundwater flow systems is given in Chapter 23.0. The nature and extent of contamination along with the source-pathway-receptor relationship are presented in Chapter 24.0.

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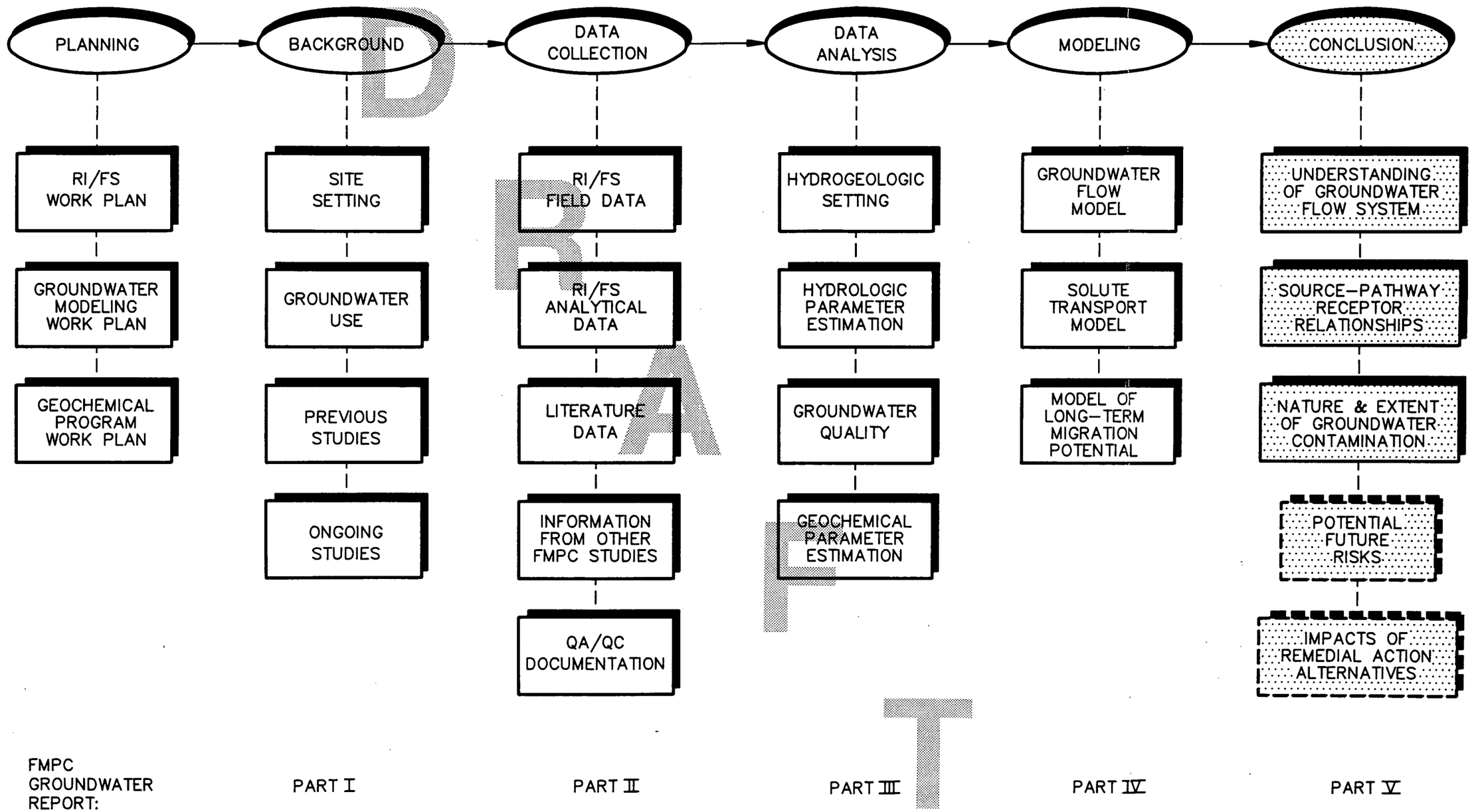

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FIGURE V-1
OVERVIEW OF CONCLUSION COMPONENT

23.0 GROUNDWATER FLOW SYSTEMS

Conclusions pertaining to groundwater flow systems are derived primarily from data and discussions presented in Chapters 12 and 20, which contain significant detail concerning groundwater flow at the FMPC. Specific conclusions concerning the groundwater flow system are discussed in the following sections:

- Rate and Direction of Groundwater Flow, Section 23.1
- Surface Water Interaction with the Great Miami Aquifer, 23.2
- Influence of the Production Wells, Section 23.3
- Area of Influence of the FMPC, Section 23.4

The hydrogeologic setting of the FMPC study area is composed of a buried channel aquifer (Great Miami Aquifer) which is deeply incised into bedrock. The buried channel varies in width from about one-half mile to over two miles and has a U-shaped cross section with a broad, relatively flat bottom, and steep valley walls. Bedrock is predominantly flat-lying shales with thin interbedded layers of limestone. Thick deposits (exceeding 200 feet) of glacial outwash material composed of coarse sands and gravels, with interbeds of silts and clays, fill the buried channel. Overlying this regional aquifer in places are variable thicknesses of loess, glacial till, glaciolacustrine, and glaciofluvial deposits (glacial overburden).

Hydrologic, geologic, and cultural features which affect groundwater flow in the regional aquifer are:

- The complex, irregularly shaped buried channel aquifer
- The thick, highly permeable aquifer with variations in both horizontal and vertical hydraulic conductivity
- The location of major pumping centers which influence groundwater elevations and flow over large areas
- Pumping at depth (partially penetrating wells) within the regional aquifer
- Large changes in horizontal hydraulic gradients across the study area
- The presence of vertical hydraulic gradients within the study area, possibly induced by local pumping
- Complex river, stream, and aquifer interactions
- Variable areal recharge through overburden, flood plain, and alluvial deposits

Groundwater flow within the glacial overburden is influenced by local variations in hydraulic conductivity, local recharge, varying depositional thicknesses, and cultural features (e.g., drains, ponds, and ditches).

23.1 RATE AND DIRECTION OF GROUNDWATER FLOW

Groundwater in the vicinity of the FMPC occurs in the glacial overburden, and sand and gravel units of the Great Miami Aquifer. Groundwater also occurs to a limited extent in bedrock. The bedrock beneath the FMPC lies at great depth and below the other sand and gravel units. The bedrock is essentially impermeable in comparison to the aquifer and therefore will not be discussed further.

The glacial overburden varies considerably in thickness and lithology. It is composed primarily of glacial till, but also contains loess, glaciolacustrine deposits, and glaciofluvial deposits. Consequently, the glacial overburden has a wide range of hydraulic properties, ranging from more highly permeable glaciofluvial layers (silt, sand, and gravel) to relatively impermeable zones of glacial till (sand through boulders in a matrix of clay and silt) and glaciolacustrine deposits (silt and clay). Perched zones of groundwater within the glacial overburden are often associated with discontinuous layers of glaciofluvial deposits.

The perched groundwater system within the glacial overburden, while not representing a practical water supply, fits the regulatory definition of an aquifer in some locations and is important as a potential contaminant pathway to surface water courses and the underlying regional aquifer.

The Great Miami Aquifer is the principal aquifer within the vicinity of the FMPC and consists of thick sequences of sand and gravel deposits located in a deep subterranean valley known as the New Haven Trough. Contained within the sand and gravel is a distinct, relatively continuous clay unit, the clay interbed. Where present, the clay interbed divides the Great Miami Aquifer into upper and lower sand and gravel units. The Great Miami Aquifer is the principal aquifer of concern in the FMPC study area and it could yield a significant quality of water.

23.1.1 Glacial Overburden

Perched groundwater occurs in localized discontinuous zones within the glacial overburden. Small-scale fluvial and beach deposits interbedded within the till form layers of relatively higher hydraulic conductivity. The hydrogeologic characteristics of the overburden vary widely throughout the study area. At the FMPC, depth to perched groundwater in the glacial overburden may range from

1 to 15 feet. The seasonal fluctuation of a perched water table is up to 10 feet, with the highest water levels occurring during the early spring and lowest during the late fall. Based on falling head tests in 1000 Series wells (Table 12-1), hydraulic conductivities in perched zones of the glacial overburden at the FMPC range from 0.009 to 0.85 ft/day (2.8×10^{-6} to 3.0×10^{-4} cm/s). Dense, fine-grained glacial till and glaciolacustrine deposits of silt and clay act as aquitards in this area.

Perched groundwater in the glacial overburden occurs primarily under unconfined conditions; however, due to the variability of materials, local confined conditions may occur. Groundwater movement in the vicinity of the FMPC in the perched system is to the south and west, toward Paddys Run and the Storm Sewer Outfall Ditch, and vertically downward. However, because the shallow perched zones are not interconnected across the facility, the movement of fluids is likely toward local surface drainage. The flow patterns within perched zones vary seasonally due to variations in recharge, with localized patterns often dependent on location-specific features such as leaky storm sewers. Shallow groundwater flow may also be influenced by the presence of agricultural drain tiles installed prior to the construction of the FMPC. Consequently, groundwater flow within the glacial overburden is discontinuous and nonuniform across the FMPC and site-wide patterns cannot be established.

23.1.2 Great Miami Aquifer

Groundwater elevation contour maps, developed from the 2000 and 3000 Series wells in the FMPC Study Area, are presented in Appendix K. The maps represent groundwater flow patterns for the period of January 1988 through April 1990. In addition, regional groundwater maps for August 1982, April 1986, and May 1988 are presented in Figures 12-1, 12-2, and 12-3. The generalized regional groundwater flow pattern is shown in Figure 2-6.

Groundwater enters the FMPC study area from three directions:

- In the northeast, groundwater moves southwest from the Ross area into the portion of the New Haven Trough now occupied by the Great Miami River.
- The second source of groundwater is the Shandon Tributary to the north of the FMPC, a tributary of the New Haven Trough that lies under Shandon. The majority of the groundwater from the Shandon Tributary flows under the Waste Storage Area and the Production Area and leaves

along the eastern boundary of the FMPC to discharge into the Great Miami River.

- The large pumping wells, located in the "Big Bend" meander on the Great Miami River east of the FMPC, produce a pronounced and persistent cone of depression. This cone of depression creates an induced eastward flow of groundwater in the northern and central portions of the FMPC (Figures 12-2 and 2-6), resulting in a groundwater divide trending northwest to southeast across the southern portion of the FMPC.

- The third source of groundwater is from the west. The recharge from the Dry Fork of the Whitewater River, located about two miles west of the FMPC, causes groundwater to move to the east toward the FMPC. This flow turns southward under the southern part of the FMPC and flows to the Great Miami River in the glaciofluvial deposits under the southern part of Paddys Run. A portion of the groundwater from the Shandon Tributary also reaches the Great Miami River via this Paddys Run Outlet (Figure 2-6).

Paddys Run affects local groundwater flow along the western boundary and in the area south of the FMPC. During dry seasons, when there is little recharge to the regional aquifer, groundwater flows southeastward. During periods of high flow in Paddys Run, stream water infiltrates the Great Miami Aquifer and creates a groundwater mound and strong southward gradients. In the northern portion of the study area, the mound also creates a local reversal causing northward flow. Both of these flows are transient conditions which only last for the duration of high flow in Paddys Run, and do not appear to significantly affect long-term contaminant migration patterns.

Bedrock highs south of the FMPC also control groundwater flow direction. The eastern and western bedrock highs force groundwater to flow due south toward the Great Miami River. In the channel between the two bedrock highs, the groundwater gradient steepens.

In addition to dominant regional groundwater direction, as discussed above, the preceding general flow patterns dominate the regional flow system; however, short-term variations in groundwater flow direction do occur. Examination of the historic groundwater data from January 1988 through April 1990 shows the transient effects that occurred due to both drought and wet conditions. These features are prevalent throughout the 2000 Series groundwater elevation maps (Appendix K), although they can be seen in the 3000 and 4000 Series maps as well. Based on the groundwater elevation maps, the following trends and conditions can be seen:

- Flow conditions across the FMPC generally trend east to southeast. These flow patterns are relatively stable, although they tend to shift northward during periods of extended recharge (e.g., April-May 1989).

- Peak groundwater levels generally occur during the spring and early summer months, which are also the major groundwater recharge months. For the period of record, the highest groundwater contours occurred during the month of June 1989, when unusually large amounts of rainfall caused groundwater elevations to rise approximately 4 to 5 feet above their mean levels. Wells proximal to Paddys Run exhibited elevations increase as much as 10 feet over their normal values, due to the large amount of groundwater recharge entering from Paddys Run (Appendix D cluster hydrographs).

- Minimum groundwater levels generally occur during the late fall and early winter months of the year. This corresponds with the dry season for southern Ohio, which generally starts in late summer or early fall and runs to late fall. For the January 1988 through April 1990 time period, minimum groundwater levels occurred during the months of November 1988 through January 1989. Levels during this time were unusually low due to drought conditions that occurred during the summer and fall of 1988. This caused groundwater levels to fall approximately 2 to 3 feet below their normal seasonal lows.

- Recharge events occurring in Paddys Run were observed in May-June 1989 and March-April 1990. Leakage from Paddys Run can be seen to cause a "mound" in the groundwater as it infiltrates through the stream bed and enters the groundwater system. This effect is transient, although it appears to occur on an annual basis as part of the recharge during the spring and early summer months (Section 12.5.2).

23.2 SURFACE WATER INTERACTION WITH THE GREAT MIAMI RIVER AQUIFER

23.2.1 Great Miami River/Groundwater Interaction

The Great Miami River and Paddys Run are two major surface water bodies that have a greater interaction with the Great Miami Aquifer. The Great Miami River is a perennial river, while Paddys Run is an intermittent stream.

The Great Miami River is a gaining stream which receives groundwater along most of its course through the study area. However, near the collector wells, the river loses water at a rate of 13 Mgd and provides approximately 64 percent of the water pumped from the wells. Particle tracking shows that the reach of the Great Miami River that would receive water coming from

beneath the FMPC gains water at a rate of about 637,000 ft^3/day (4.8 Mgd). This amount is about 32 percent of the total discharge to the total reach of the Great Miami River included in the modeled area.

If the pumping rates of the collector wells were increased by 25 percent, induced infiltration from the Great Miami River would increase by 27 percent to 16.5 Mgd. The reach influenced by the wells would extend an additional 500 feet upstream. If the pumping rates of the collector wells were decreased by 28 percent, induced infiltration from the Great Miami River would decrease by 31 percent, to 9 Mgd.

23.2.2 Paddys Run/Groundwater Interaction

Paddys Run interacts with the Great Miami Aquifer in several ways that affect groundwater flow and discharge. Paddys Run has eroded through the glacial overburden and into the top of the Great Miami Aquifer from its confluence with the Great Miami River to about the position of the metal oxide silo (Figure 11-5). It is, therefore, directly connected with the Great Miami Aquifer and generally loses water to the regional aquifer where it is perched above the regional water table. In the vicinity of the FMPC, the stream is generally dry, except during runoff periods following rainfall and snow-melt events. Sustained flow has been observed in Paddys Run during the winter and spring and by stream gaging stations monitored during the RI. During flooding periods, recharge is relatively high where the bed of the stream is on the sand and gravel of the Great Miami Aquifer. Farther north, Paddys Run is underlain by clayey glacial overburden. Therefore, relatively little recharge of the Great Miami Aquifer occurs in this reach.

Groundwater elevation contour maps for the 2000 and 3000 Series wells (Appendix K, Figures K-1 through K-56) show that the regional water table near Paddys Run fluctuated as much as 10 feet during the period January 1988 to June 1989. This fluctuation may be considered extreme, since the FMPC experienced drought conditions from spring 1988 to early winter 1989, followed by an extended period of higher than average precipitation from late winter to early summer 1989. A similar pattern has been observed in the winter of 1989/1990. As shown in Figures 12-8 and 12-9, there is a high degree of correlation between the Paddys Run hydrograph and the regional aquifer hydrographs. As the Paddys Run water elevation peaks, the groundwater shows a peak a short time later, which indicates the direct hydrologic connection between the stream and the Great Miami Aquifer.

As discussed previously, the Storm Sewer Outfall Ditch is a tributary of Paddys Run that enters the stream just above the southwest boundary of the FMPC. The bed of this stream is on sand and gravel of the Great Miami Aquifer, to about as far upstream as the storm water retention basins. As such, water in the Storm Sewer Outfall Ditch can directly infiltrate into the underlying aquifer. The Storm Sewer Outfall Ditch once carried excess storm water and associated contaminants from the Production Area prior to construction of the two retention basins, and is considered to be a significant source of historic uranium releases to the Great Miami Aquifer during the years of FMPC operations. The Storm Sewer Outfall Ditch still drains nearby uplands during sustained rainfall events and during winter and spring snowmelts, and serves as the conveyance channel in the event of overtopping of the retention basins. Otherwise, the ditch is typically dry.

23.3 INFLUENCE OF PRODUCTION WELLS

A water balance was established to determine the effect of collector wells on groundwater flow direction and rate. The results of this analysis indicate that the collector wells significantly influence the groundwater water flow direction. The same results show that about 3.3 percent of the water pumped from the collector wells comes from beneath the FMPC site. About half of this amount comes from the part of the aquifer above the clay interbed. As indicated earlier, the main source of water for the collector wells is the Great Miami River. The results for the collector wells are presented in Table 20-6.

23.4 AREA OF INFLUENCE OF THE FMPC

Modeling results showed that water entering the FMPC production well comes from the Great Miami Aquifer northwest of the site. Any influence on the aquifer above the clay interbed is too small to be detected. Particles seeded immediately above the interbed radially around the well during particle tracking analysis were not drawn downward into the interbed. They moved southeastward and showed no deflection toward the well.

The pumping well has such a slight effect on the groundwater system (pumping at a rate of 292 gpm) that reasonable increases or decreases in the pumping rate will have no significant effect on the configuration of the potentiometric surface or on seepage through the clay interbed. Simulated seepage through the clay interbed is negligible.

24.0 SOURCE-PATHWAY-RECEPTOR RELATIONSHIPS

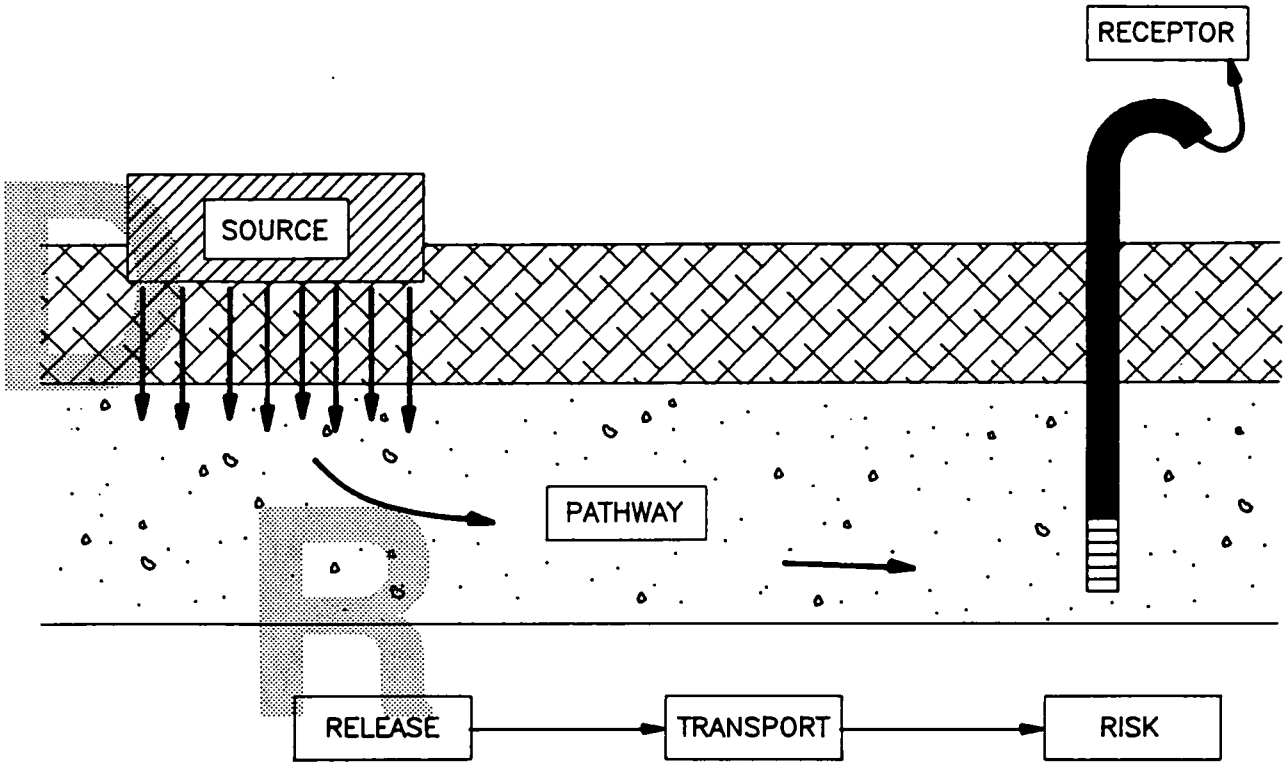
24.1 RELATIONSHIP TO GROUNDWATER PROGRAM

The principal objective of the groundwater program reported herein was to further the understanding of the sources of contaminant release to the Great Miami Aquifer, the transport of contaminants to receptors via the groundwater pathway, and the risk posed to those receptors as a result of exposure to contaminated groundwater. As indicated by the schematic in Figure 24-1, the FMPC groundwater program was designed around the concept that the Great Miami Aquifer is the primary pathway by which contaminants released from a source unit are transported to a receptor. The analysis of the aquifer as a pathway not only had to consider the advective and dispersive flux of the groundwater and associated contaminants, but also the geochemical reactions between the contaminants and the porous media that effectively attenuate contaminant migration.

For purposes of the comprehensive groundwater study, however, the Great Miami Aquifer also had to be viewed as a source of contaminants and an environmental receptor. In establishing a baseline condition for the evaluation of potential future risks, the existing contaminant plumes were treated as potential sources of contamination. Their continued migration to as yet unaffected areas was evaluated under a no-action scenario. The existing levels of contaminants in groundwater were also evaluated under both present and future scenarios in terms of the degradation of the aquifer as an environmental receptor, independent of whether or not human receptors would be impacted by use of the groundwater.

The contribution of the various tasks of the groundwater program to the understanding of source-pathway-receptor relationships at the FMPC is also summarized in Figure 24-1. In particular, the figure indicates that information provided by each of seven principal elements of the groundwater program found widespread use in the integrated analysis of contaminant release (source), transport (pathway), and risk (receptor) associated with the groundwater medium.

In the case of contaminant release, the site history and previous studies that involved sampling of the waste materials provided the most direct evidence of sources and the characteristics of waste materials and leachate. RI/FS field observations served primarily as documentation as to whether a release to the Great Miami Aquifer, or at least to perched groundwater, which could serve as a continuing source, had or had not occurred. The groundwater quality data also allowed special analyses (e.g., isotopic ratio analysis) that contributed information as to the potential source or sources of any observed contamination. The groundwater flow and solute transport models provided



SITE HISTORY	● RECORDS OF MATERIALS DISPOSED ● DISPOSAL SITE CONSTRUCTION RECORDS ● DOCUMENTATION OF RELEASE EVENTS	● LONG-TERM MONITORING RECORDS ● "FINGERPRINTING" BASED ON PROCESS KNOWLEDGE	
PREVIOUS STUDIES	● CHARACTERIZATION OF WASTE MATERIALS (DIRECT SAMPLING) ● IDENTIFICATION OF SUSPECT AREAS	● GENERAL UNDERSTANDING OF GROUNDWATER FLOW SYSTEM ● IDENTIFICATION OF FLOW DIVIDE	● KNOWN/SUSPECTED PROBLEM AREAS ● MONITORING RECORDS AT RECEPTOR LOCATIONS
HYDROLOGIC DATA	● POTENTIAL FOR GROUNDWATER CONTACT WITH WASTES ● STREAM/AQUIFER INTERACTION	● GROUNDWATER FLOW DIRECTION ● GROUNDWATER VELOCITIES ● VERTICAL GRADIENTS AND FLOW	● POTENTIAL DOWNGRADIENT RECEPTORS ● CAPTURE ZONES OF PRODUCTION WELLS
GROUNDWATER QUALITY DATA	● DIRECT EVIDENCE OF RELEASE ● "FINGERPRINTING" OF SOURCES ● PERCHED GROUNDWATER AS POTENTIAL SOURCES	● NATURE AND EXTENT OF CONTAMINATION ● BACKGROUND CONDITIONS ● EVIDENCE OF CONTAMINANT ATTENUATION	● QUANTIFICATION OF CURRENT RISK ● IDENTIFICATION OF AREAS OF POTENTIAL RISK
GEOCHEMICAL PROGRAM	● CHARACTERIZATION OF POTENTIAL LEACHATE FORMATION ● POTENTIAL ATTENUATION IN TILL	● POTENTIAL ATTENUATION IN AQUIFER ● DOMINANT SPECIES AND PROPERTIES ● EXPLANATION OF BACKGROUND CONDITIONS	● DOMINANT CHEMICAL FORMS AT RECEPTOR LOCATIONS
FLOW/SOLUTE TRANSPORT MODELING	● RECREATION OF PAST RELEASES TO DISTINGUISH SOURCES	● INTERPOLATE BETWEEN AND EXTRAPOLATE BEYOND FIELD OBSERVATIONS ● PREDICT FUTURE BEHAVIOR	● IDENTIFY AREAS OF MAXIMUM CONCENTRATION ● PINPOINT CRITICAL RECEPTOR LOCATIONS ● PREDICT CONCENTRATIONS AT RECEPTORS
LONG-TERM MIGRATION MODELING	● POTENTIAL FOR LONG-TERM LEACHATE RELEASE ● MIGRATION THROUGH TILL ZONES AND GREAT MIAMI AQUIFER	● PREDICT TRANSPORT OVER LONG TIME PERIOD (UP TO 500 YEARS)	● FUTURE RISK AT DESIGNATED RECEPTORS DUE TO FUTURE RELEASES ● RISK AFTER INSTITUTIONAL CONTROL

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FIGURE 24-1

SOURCE-PATHWAY-RECEPTOR FRAMEWORK:
RELATIONSHIP TO GROUNDWATER PROGRAM

additional input as to the most likely sources and time frames of historic and continuing releases. This was accomplished by utilizing the model to "recreate the past"; that is, to model the time variant behavior of historic release scenarios until the most plausible scenario that satisfactorily reproduced the currently observed plumes was found. On the other hand, the potential for continuing or future releases from the same sources was quantitatively addressed through the geochemical program and the long-term fate and transport model.

The analysis of the Great Miami Aquifer as an environmental pathway focused on the modeling tasks, as supported by field observations that provided both the input data base and the "actual conditions" against which model performance was evaluated. Whereas groundwater elevation data and water quality data provided direct evidence of actual field conditions at specific points in time and space, the groundwater flow and transport models allowed for the systematic interpolation between and extrapolation beyond the point observations. In short, the models provided spatially continuous views and future predictions of groundwater conditions at the FMPC. The long-term fate and transport model provided a similar mechanism for predicting future conditions over a longer time frame, but with less certainty. Each model extracted data from the geochemical program to account for the expected attenuation of a plume as it is transported from a source to a receptor. In the case of the pathway analysis, the site history and previous studies provided a historic record against which the RI/FS field and modeling efforts would be more effectively designed.

The groundwater program provided the basic data for many elements of the risk assessment. This was accomplished through a combination of field observations and modeling. Field observations provided the most direct evidence of elevated concentrations of the chemical constituents at potential receptor locations, whereas the predictive capacity of the models was necessary to estimate both existing concentrations at unmonitored locations and future conditions at all critical locations. Through the various data interpretation and modeling studies, the following types of information were generated:

- Receptor locations potentially at risk based on source locations and groundwater flow patterns
- Chemical species that would likely be dominant in the groundwater environment
- Approximate maximum concentration to which a receptor may be exposed in the future
- Time at which the maximum exposure would occur

24.2 POTENTIAL SOURCES OF RELEASE TO THE GREAT MIAMI AQUIFER

Within the context of the source-pathway-receptor relationship conceptualized in Figure 24-1, the demarcation between the source and the pathway is the point at which contaminants enter the regionally important Great Miami Aquifer. This conceptualization adds complexity to the source term in that several different source scenarios can be developed from a single point of contaminant release. Five such scenarios are shown in Figure 24-2, with each scenario further divided into a number of subpathways between the initial point of release and the eventual entry of a contaminant into the Great Miami Aquifer.

Four different release mechanisms from the source units into the environment are identified in Figure 24-2. These include the resuspension of surface contaminants into storm water runoff, episodic releases (e.g., spills) directly onto the ground surface, the release of permitted discharges of contaminated process waters or groundwater from the main effluent line, and subsurface leakage from a waste storage or operating unit. Releases via storm water runoff are further differentiated by the initial and subsequent receptors of the water, with special consideration given to the possible release through the storm water retention basins into the Storm Sewer Outfall Ditch. The interim pathways taken by these releases can vary with location. Two noteworthy variations are the local presence of perched groundwater zones that serve as intermediate collection points pending future release to the regional aquifer, and the role of Paddys Run as a collection point that is susceptible to leakage directly into the underlying sand and gravel aquifer.

The initial sources of contaminant release shown in the leftmost column of Figure 24-2 remain generally unchanged from those identified at the time of RI/FS Work Plan preparation. Through the groundwater study, however, the likelihood of past, present, and future releases of contaminants from each of these units was more firmly established so that appropriate remedial actions could be developed and prioritized. In particular, the following conclusions have resulted from the RI/FS in relation to the source scenarios shown in Figure 24-2 and discussed in the following section:

- Scenario 1a: Both historic and continuing releases of contaminants into storm water runoff have been documented in the Waste Storage Area and Production Area. The infiltration of these releases into the underlying perched groundwater and the Great Miami Aquifer is likely, but direct evidence is masked by other types of releases in these same areas. On the other hand, the historic release of contaminated storm water runoff to the Great Miami Aquifer via Paddys Run and the Storm Sewer Outfall Ditch is considered to be the principal cause of much of the groundwater contamination in the southwest portion of the FMPC and in off-FMPC areas to the south.

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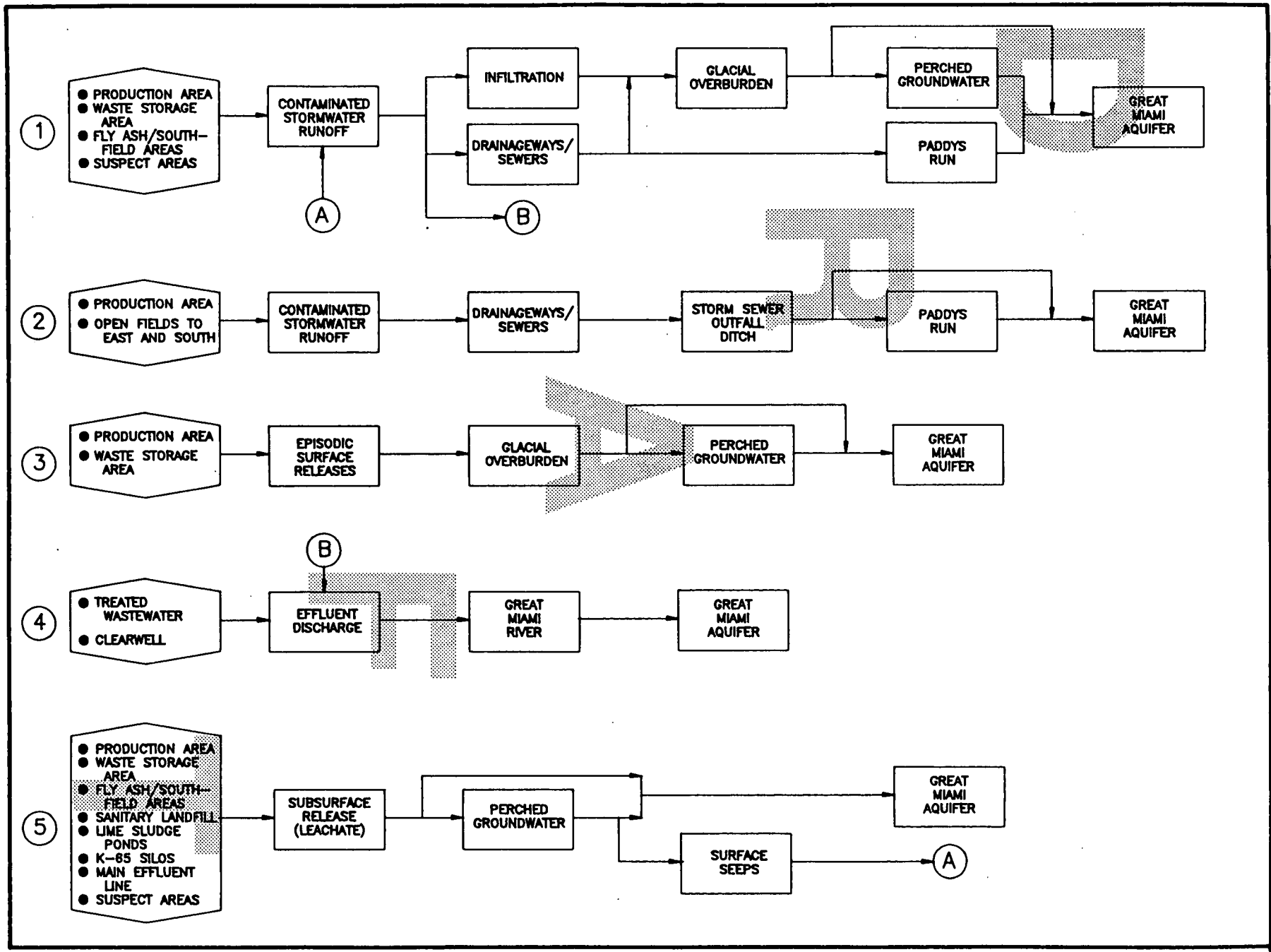

FIGURE 24-2
FMPC SOURCE RELEASE SCENARIOS

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- Scenario 1b: Contaminated storm water runoff from the fly ash area/Southfield has also been observed, but is considered to be of secondary importance to groundwater contamination when compared to releases from the Waste Storage Area and the Production Area. Contributions from suspect areas and other areas of the FMPC as a result of storm water runoff do not appear to be significant.

- Scenario 2: The contribution of leakage from the Storm Sewer Outfall Ditch to groundwater contamination is in evidence both to the east of the outfall ditch (i.e., in the immediate downgradient direction) and downgradient from the confluence of the outfall ditch with Paddys Run. The former condition was evidenced by groundwater quality observations, while the latter condition was confirmed both by direct observations and by the application of the solute transport model to recreate the most likely locations and strengths of historic sources.

- Scenarios 1 and 2: The contribution of storm water runoff to groundwater contamination has been greatly reduced in recent years by capital improvement projects, including the storm water retention basins. Planned removal actions are expected to eliminate the release of contaminated storm water runoff from the Waste Storage Area to Paddys Run, and thus to the Great Miami Aquifer.

- Scenario 3: Localized effects on groundwater quality resulting from episodic releases are difficult to distinguish from other release mechanisms. Exceptions to this are several areas within the Production Area, where observed contamination of perched groundwater has been linked to singular spills, leaks, tank overflows, or other operational problems. The elevated concentrations of uranium and other organic constituents observed in 2000 Series wells beneath the southern portion of the Production Area are also considered to have their origin in such singular events.

- Scenario 4: The scenario of groundwater contamination resulting from discharges into the Great Miami River is only applicable to those reaches of the river that recharge the aquifer, with emphasis on the reaches within the zone of influence of the production wells. A previous study of the effects of the FMPC outfall on groundwater quality in this area concluded that any effects would be indistinguishable from natural and analytical variances in the observations. Consequently, this source scenario is not considered significant.

- Scenario 5a: There is substantial evidence from groundwater quality observations and the modeling study that the release of contaminated leachate has occurred and continues to occur from the individual units within the Waste Storage Area. Although exact sources are difficult to substantiate, analyses of isotopic ratios and upgradient/downgradient relationships provide evidence that the principal sources are the waste pits and the Clearwell. Evidence also exists that the effects of these releases extend to the regional aquifer underlying the Waste Storage Area, although considerable attenuation does occur as contaminants migrate vertically through the till.

- Scenario 5b: Subsurface releases have also been substantiated in several areas within the Production Area. Total uranium concentrations in excess of 10,000 ug/L were observed in perched groundwater in the vicinity of Plants 2/3, 6, 8, and the Raffinate Area. Whereas these levels of groundwater contamination can be tied back to specific buildings or operations, the specific sources of more widespread contamination of perched groundwater beneath the Production Area to levels exceeding 100 ug/L cannot be definitized. Total uranium concentrations in the uppermost portion of the Great Miami Aquifer were significantly lower than those in the glacial overburden within the Production Area.

- Scenario 5c: Although there is limited evidence that subsurface releases have occurred from the Sanitary Landfill and the Lime Sludge ponds, it has been concluded that any effects remain limited to the overburden and perched groundwater. There is no substantial evidence that the underlying Great Miami Aquifer has been affected.

- Scenario 5d: Some evidence exists that the fly ash area and Southfield have been a source of subsurface releases to the Great Miami Aquifer. Much of this evidence is based on elevated concentrations of inorganic species characteristic of fly ash leachate and other materials potentially disposed in Southfield (e.g., high alkalinity values indicative of concrete debris). Other potential impacts, in particular the potential release of radionuclides, are masked by the location of these areas downgradient from those reaches of Paddys Run known to contribute contaminants directly into the underlying aquifer.

- Scenario 5e: There is no direct evidence that significant leakage has occurred from the K-65 silos over the period of waste storage. Elevated radium concentrations that would fingerprint the silos as a source have not been observed in groundwater, whereas elevated concentrations of uranium observed in perched groundwater near the silos are thought to be the result of a suspect area in the same vicinity.

- Scenario 5f: The results of a recent study indicate that the main effluent line to the Great Miami River leaks along portions of its length. However, measurable impacts on groundwater quality have not been observed in nearby wells. Additional monitoring of groundwater in this area is planned.

24.3 THE GREAT MIAMI AQUIFER AS A CONTAMINANT PATHWAY

The basic requirement of a groundwater source-pathway-receptor relationship is that the receptor of interest be hydraulically connected to a given source via the groundwater pathway; that is, that the receptor be hydraulically downgradient from the source. If this is not the case, then the critical link between the source and receptor is nonexistent and no pathway analysis would be warranted regardless of the magnitude and duration of the release. For a given source and receptor of interest at the FMPC, this issue would be resolved solely by consideration of the groundwater flow patterns in the Great Miami Aquifer, as presented in Chapters 12.0 and 23.0.

With reference to the source scenarios illustrated in Figure 24-2, the pathways of key interest to the Great Miami Aquifer are those that originate in the following areas: (1) Waste Storage Area; (2) Production Area; (3) fly ash area/Southfield; (4) the reach of Paddys Run from approximately the silo area to the Great Miami River; and (5) the Storm Sewer Outfall Ditch. Based on the RI/FS groundwater study, the following conclusions have been made:

- Releases from the Waste Storage Area will migrate to the east, with a slight northern component of flow during seasonal periods of high recharge. Discharge of this groundwater would be either to the Great Miami River east of the FMPC or to the SOWC collector wells.

- A total travel time of approximately 35 years has been estimated for groundwater from the Waste Storage Area to reach the collector wells. Contaminant retardation effects would extend this travel time considerably, as exemplified by an estimated travel time of 420 years for uranium at a retardation factor of 12. The leading edge of a uranium plume would, in fact, reach the collector wells in a somewhat shorter time period due to dispersive effects.

- In accordance with the eastward component of flow, the Production Area is located immediately downgradient from the Waste Storage Area. The migration of contaminants away from the Production Area would, therefore, mirror the patterns from the Waste Storage Area with a reduction in the travel time to reach the collector wells or the river. Due to the upgradient/ downgradient relationship, groundwater contamination from the Waste Storage Area would mix with any continuing releases from the Production Area prior to reaching the Great Miami River and the collector wells.

- The fly ash area and Southfield lie within a zone of transition from eastward groundwater flow to southerly groundwater flow, with some seasonal influences caused by groundwater mounding under Paddys Run. It is expected that the southern component of flow will dominate the groundwater flow pattern in this area such that any eastward flowing water will eventually be diverted to the south through the westernmost bedrock valley.

- The groundwater gradient in the immediate vicinity of the fly ash area/Southfield is relatively flat, resulting in locally small velocities on the order of one foot per day. With a retardation value of 12, the migration rate of uranium would be approximately 30 feet per year. This would indicate that contaminant releases from Paddys Run within the early years of plant operation could be currently impacting groundwater quality within the vicinity of the fly ash area/Southfield.

- The long-term migration pathways of contaminants released from Paddys Run to the Great Miami Aquifer vary depending on the specific location of the release. The northernmost impacted reaches would be associated with an eastward component of groundwater flow, in which case a plume would flow to the east across the FMPC and discharge to the Great Miami River to the east-southeast of the site. The central reaches, considered to be the most important historical sources of contaminant release, would have a slight eastward component of flow, but would be dominated by a southerly flow

component. The resultant plume would have a net southern movement toward the westernmost bedrock valley (as evidenced by the existing plume south of the FMPC), with eventual discharge to the Great Miami River west of New Baltimore. The velocity of this plume would be expected to increase by an approximate factor of three to five as the hydraulic gradient steepens through the narrow bedrock valley. A southerly flow from Paddys Run directly to the Great Miami River would occur in the lower, east-west oriented reach of the stream.

- Groundwater quality data indicate that some contaminant migration occurs to the west from Paddys Run. This is considered to be the result of seasonal mounding of the water table immediately beneath the stream. Based on particle tracking in the solute transport model, the extent of this transient phenomenon is limited in time and space, and eastward or southern plume migration occurs over the long term.

- Groundwater contaminants infiltrating into the aquifer from the Storm Sewer Outfall Ditch would migrate to the east-southeast toward the Great Miami River upstream from New Baltimore. An exception would be infiltration from the lower reaches of the outfall ditch, which would likely be dominated by the southerly component of flow toward the bedrock valley underlying Fernald. An eastward moving plume from the outfall ditch would have a velocity less than one foot per day due to the relatively flat piezometric gradient in this area, indicating that any such plume would not yet have reached the FMPC boundary.

24.4 GROUNDWATER RECEPTORS

Once a hydraulic connection between a source and a pathway is established, the key issues revert to the time required for contaminants to reach a receptor, the potential duration of the exposure, and the concentration distribution over time at the receptor location. To fully address these issues at the FMPC, consideration had to be given to both the future migration of existing groundwater contaminant plumes and the potential for future releases that would either sustain existing plumes or create new plumes of groundwater contamination. The baseline condition (i.e., the existing plumes) was established by a combination of groundwater quality observations and model results. The quantification of future releases and the resultant future behavior of the plumes had to be based solely on the predictive capability of the various models. These models were developed and calibrated to site conditions to secure confidence that the results are representative of actual conditions at the FMPC.

The potential receptors at risk must also be considered from two vantage points--hypothetical receptors assumed in the performance of the risk assessment in accordance with regulatory guidance, and actual receptors based on current land use in the vicinity of the FMPC. The risk assessment was performed based on the assumption that DOE's institutional control of the FMPC

would remain in effect for at least 100 years. Under this assumption, the most critical groundwater receptor would be hypothetically located at the FMPC property boundary at a point immediately downgradient from a given source. One exception is the most critical receptor for the South Plume, in which case the maximum plume concentration is already located beyond the FMPC boundary and is expected to remain so under anticipated contaminant release patterns. Within this context, the most critical potential receptor would be located at the predicted point of maximum concentration in the plume.

Based on the results of the RI/FS groundwater study and the associated baseline risk assessment, no current users of groundwater at off-FMPC locations are at risk as a result of FMPC discharges to the regional aquifer. The receptors that would be initially exposed to the northern (eastward migrating) plume are groundwater users to the east of the FMPC along Route 128. This plume would eventually reach the production wells located within and adjacent to the "Big Bend" in the river, causing a significant increase in the number of potential industrial receptors through the associated water distribution system. As indicated in the previous section, however, this would not be expected to occur for well over 100 years.

Model predictions of plume migration under a 70-year exposure period provide important results in terms of both the hypothetical and actual receptors. In the case of the northern plume, the simulated spreading of the uranium plume for the next 70 years under a no-action scenario indicates that the 30 ug/L contour would migrate about 400 feet to the east-southeast (Figure 21-32). This degree of movement implies that the 30 ug/L contour would not reach the FMPC boundary in a 70-year time frame. Predicted uranium concentrations at the eastern boundary after 70 years are less than 10 ug/L. These results mitigate any concern for even the hypothetical receptor at the FMPC boundary within the assumed period of institutional control.

The South Plume, as it is currently configured based on water quality observations and model predictions, would continue to migrate to the south under the no-action alternative. The velocity of plume migration would increase in relation to historical trends, however, due to the steepening of the piezometric gradient through the narrow bedrock valley. This continued migration would place the residences and commercial establishments along New Haven Road at maximum risk within the 70-year period of exposure. Groundwater users south of New Haven Road toward the Great Miami River would also be at potential risk at a later point in time, but continued plume dispersion would lower the maximum uranium concentration to which these individuals would be exposed.

The predicted movement of the South Plume would be approximately 5,000 feet over the next 70 years, down to the point where it would enter the Great Miami River (Figure 21-32). The 30 ug/L contour of the plume would be expected to reach New Haven Road in about 5 years; whereas, it would take approximately 20 years for the projected center of the plume to reach this location. The predicted uranium concentration in the center of the plume would be 100 ug/L at the time that it reaches New Haven Road. Future plume development and migration would remain at levels below the existing South Plume due to the drastic decreases in the supply of uranium from Paddys Run and the Storm Sewer Outfall Ditch relative to historic loading rates (Figures 21-27 and 21-32). It is also noteworthy that model predictions indicate that the highest levels of uranium will remain in the uppermost portions of the Great Miami Aquifer (2000 Series wells) over the 70-year time frame.

Once the assumed period of institutional control ends, it is considered that an individual would be free to reside at any point within the FMPC and to use groundwater at that location. Under this scenario, the location of the hypothetical critical receptor would be at the point where a release from a given source enters the Great Miami Aquifer, prior to plume dispersion and further mixing with less-contaminated groundwater.

The long-term effects of the waste units as new or continuing sources of contaminants to the Great Miami Aquifer are evaluated using the long-term migration model described in Chapter 22.0. Because future releases from the individual waste units are most critical to the maximum exposure that would hypothetically occur under a noninstitutional control scenario, the long-term migration model was used to predict receptor exposures for the post-70-year period. Further, because it is more likely that such releases will be controlled at the source rather than once they enter the aquifer, the RI/FS strategy has been to address the long-term migration issue within the framework of the source operable units rather than under Operable Unit 5.

As indicated in Chapter 22.0, the analysis of long-term release and migration is continuing on an operable unit-by-operable unit basis. This analysis of fate and transport is currently limited by data limitations around which future field investigations are planned. Only preliminary conclusions can be drawn at this time. In general, the results of the long-term migration model indicate that the highest future concentrations of the radionuclides and chemicals of concern will occur directly beneath waste units. Leachate formation will be controlled by the form and chemical speciation of the waste materials, whereas the rate of vertical migration will be controlled by the retardation factor and thickness of the geologic units, specifically the till underlying most waste units. The

analysis indicates that uranium, technetium, and soluble organics are most likely to reach the Great Miami Aquifer first. The existing data base on waste materials and the preliminary model results further indicate that the source of some contaminants will be depleted earlier into the future under the no-action alternative, while other radionuclides and chemicals will continue to leach throughout the period of interest.

24.5 UNCERTAINTIES

The principal objective of the groundwater program, as indicated earlier, was to further understand the sources of contaminant release to the Great Miami Aquifer, the transport of contaminants to receptors via the groundwater pathway, and the risk posed to those receptors as a result of exposure to contaminated groundwater. The results of the groundwater program, as presented in this report and summarized in this chapter, have met this objective to the degree necessary for the development of technically sound, effective remedial actions in the FS.

Nevertheless, as is the case with any study of historical and future effects on an environment that can only be remotely monitored (i.e., the subsurface groundwater environment), uncertainties remain regardless of the scope of the program and the magnitude of the available data. The most important uncertainties that remain with respect to the sources, pathways, and receptors associated with the Great Miami Aquifer are discussed in the following sections.

24.5.1 Source Uncertainties

The identity of the principal source areas at the FMPC is well understood and has been incorporated into the RI/FS operable unit strategy. These principal sources are identified in Figure 24-2. Source uncertainties remain, however, in two geographical areas and in the modeling of long-term leachate formation and release.

One geographic area of continuing concern is north of the Southfield in the vicinity of Wells 2045 and 2046. Uranium levels in excess of 100 ug/L have been observed in groundwater in this area with no source being firmly identified. One possible explanation is that large releases of uranium from Paddys Run historically occurred over a short span of time such that a highly concentrated plume was formed and migrated eastward. A plume formed in the late 1950s or early 1960s would be expected to be located in the vicinity of Wells 2045 and 2046 today. The argument against this supposition is the apparent discontinuities in the high uranium levels south of these wells and the high variability of observed uranium concentration in these wells from sampling period to sampling period. Both of these arguments would be countered if the plume was very narrow in width; thus,

a qualifier was added that a high rate of release would have had to occur over a short period of time. The resolution of this issue is important to the selection of a remedy for the groundwater operable unit only in terms of its effect on the cleanup times and the volume of water requiring treatment.

A second uncertainty is the strength and distribution of source terms within the Production Area. The RI within the Production Area was planned and conducted toward an objective of identifying the most critical continuing sources for remediation on a priority basis. Whereas the most important historic and continuing sources have been identified and related to the areas of highest groundwater contamination, extensive areas with uranium concentrations exceeding 100 ug/L have likely resulted from a combination of point and dispersed sources. Future studies will continue to define sources within the Production Area, and the eventual decontamination and decommissioning of the facilities will remove any continuing sources. In the meantime, progress will be made toward the remediation of both known sources and areas of highly contaminated perched groundwater that would otherwise serve as a continuing release to the Great Miami Aquifer.

The uncertainties associated with the modeling of leachate formation and release are associated with both uncertainties in the input parameters to the model and the lack of field data for model validation. In the former case, the incorporation of the modeling needs into the field investigation focused on uranium as the principal contaminant of concern. As EPA's guidance changed, however, numerous other constituents had to be considered in the long-term migration model to fully satisfy the risk assessment needs. Certain geochemical data for these other constituents, as for example site-specific retardation factors, were not available and had to be established based on the results of geochemical modeling and published literature values. The resultant uncertainties in model predictions of long-term concentrations at receptor locations are difficult to quantify.

An option to the lack of definitive information on site-specific model parameters is to establish an acceptable range of input values through the calibration of the model against known field conditions. In the case of the long-term transport model, however, many of the contaminants of concern have not actually been detected in groundwater or have been detected at very low concentrations. Consequently, the model predictions of future hypothetical conditions cannot be compared to current field observations to ascertain the reasonableness and representativeness of the results.

24.5.2 Pathway Uncertainties

Based on the existing literature and direct field observations, as supported by the groundwater flow and solute transport models, the hydrogeology of the Great Miami Aquifer is considered to be well understood. Uncertainties remain, however, in the understanding of the perched groundwater system. This unit is discontinuous and its hydrologic properties vary considerably throughout the site. Therefore, interpolating between and extrapolating beyond direct observations at a limited number of points is difficult. This difficulty results in uncertainties in the selection of the hydrologic properties such as hydraulic conductivity in the till, as well as in the prediction of the rate and potential extent of contaminant migration through perched groundwater units.

Another uncertainty in the analysis of pathway migration is the retardation factor for many of the radionuclides and chemicals of concern. Whereas the site-specific retardation factor for uranium is considered to be well supported, other values had to be based on literature values as supported by the results of geochemical models of chemical speciation. In addition, the geochemical properties of the aquifer materials vary across the site, further increasing the uncertainty in model results based on a single retardation factor.

24.5.3 Receptor Uncertainties

Both the groundwater flow patterns and the nature and extent of contamination in the Great Miami Aquifer are well understood. Therefore, there is a high degree of certainty in determining both current and future critical receptor locations. The principal uncertainty is associated with the long-term release and transport of contaminants under the no-action alternative. The criticality of this issue is limited, however, in that it becomes important only for time frames well into the future. It is expected that appropriate source control actions will occur within this time period.

A secondary uncertainty is the extent of uranium contamination in groundwater to the west of Paddys Run. There is a high degree of confidence that the cause of these observations is known, and that the effects are transient and limited in spatial extent. These conclusions are supported by the model results.

DRAFT

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