

above those of this soil, so no solubility controlling chromium-containing solid appears to be present in this sample.

Copper. The logarithm of the Cu^{2+} activity is plotted versus pH in Figure 4-17 for all four samples, along with the profile expected for the mineral tenorite (CuO). Tenorite is a common mineral controlling copper concentrations in ore systems (Mason and Barry, 1959) and fly ash systems (Fruchter et al., 1988). It also appears to be a solubility control in this system at pH levels above 7. Below pH 7, the supply of leachable copper appears to be exhausted before the solubility limit set by tenorite is reached. Thus it is likely that tenorite exists as a solubility control only in the ash samples since the pHs of the soil samples are in the 4 to 5 range.

Manganese. The logarithm of the Mn^{2+} activity is plotted versus pH in Figure 4-18. Also plotted are two possible controls on Mn^{2+} , rhodochrosite (MnCO_3) and pyrochroite [$\text{Mn}(\text{OH})_2$]. Obviously, neither of these two Mn(II) compounds is a solubility control in this system. Manganese chemistry is complicated by the existence of two other oxidation states, Mn(III) and Mn(IV). Although neither of these other species is particularly stable in aqueous solution, they both form very stable solid compounds that may control the activity of Mn^{2+} through redox reactions. The most common Mn solid in oxidized soils is pyrolusite (βMnO_2), a Mn(IV) compound. The geochemical calculations do not indicate that manganese in the L-Site system is controlled by this compound.

In Figure 4-19, the logarithm of Mn^{2+} activity plus PO_4^{2-} activity is plotted versus pH and compared to the profile expected for MnHPO_4 . The match is fairly good, indicating that manganese phosphate is a potential solubility control for manganese in the ash samples (cores LC-19 and LC-20-2A) for the soil below the ash (core LC-20-3). Because this phase has not been identified in fly ash systems previously, this hypothesis must remain tentative given the circumstantial nature of the evidence. However, Lindsay (1979) has reported that MnHPO_4 is a common solubility control for both Mn and phosphate in soils in the pH + pe range from 8 to 16. The pe + pH for the ashes is about 10. Phosphate was below detection in the downgradient soil (core LC-12).

Nickel. Relatively few nickel analyses are available for the four core samples, mostly at low pH levels. The results are plotted versus pH in Figure 4-20. Also plotted is a profile for bunsenite [$\text{Ni}(\text{OH})_2$], which clearly is not a solubility control for nickel in these samples.

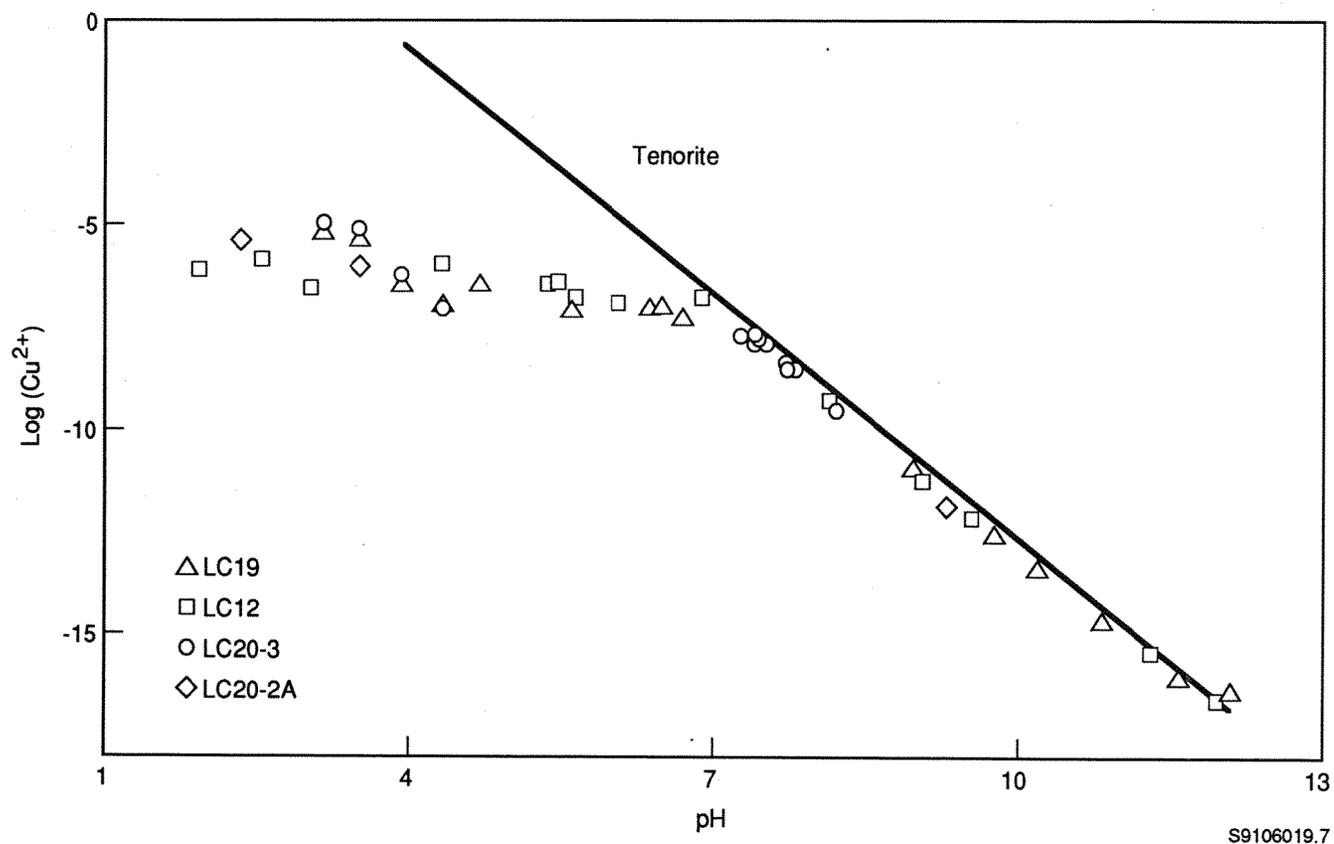


Figure 4-17. Log of Copper Activity vs pH for All L-Site Samples

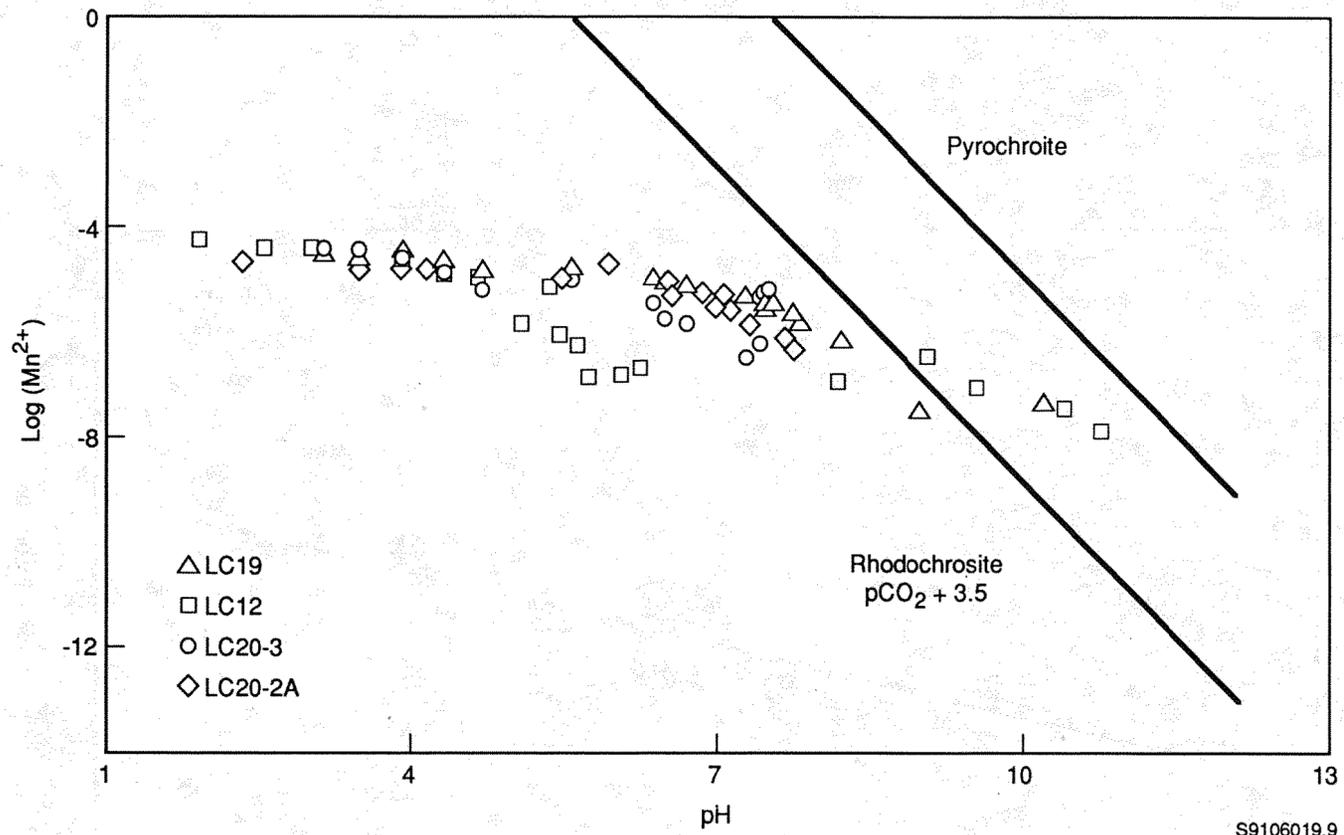


Figure 4-18. Log of Manganese Activity vs pH for All L-Site Core Samples

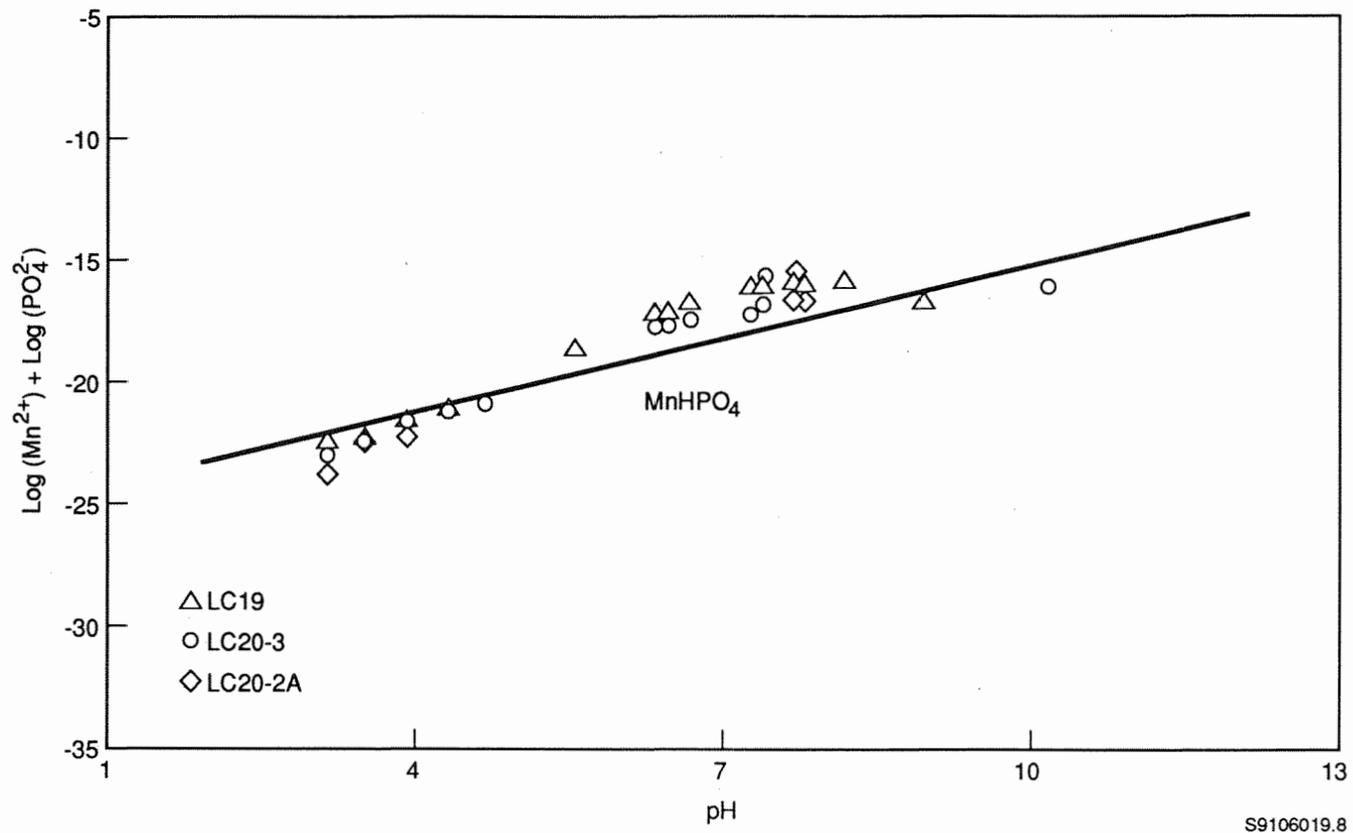


Figure 4-19. Log of Manganese Plus Phosphate Activity vs pH for All L-Site Core Samples

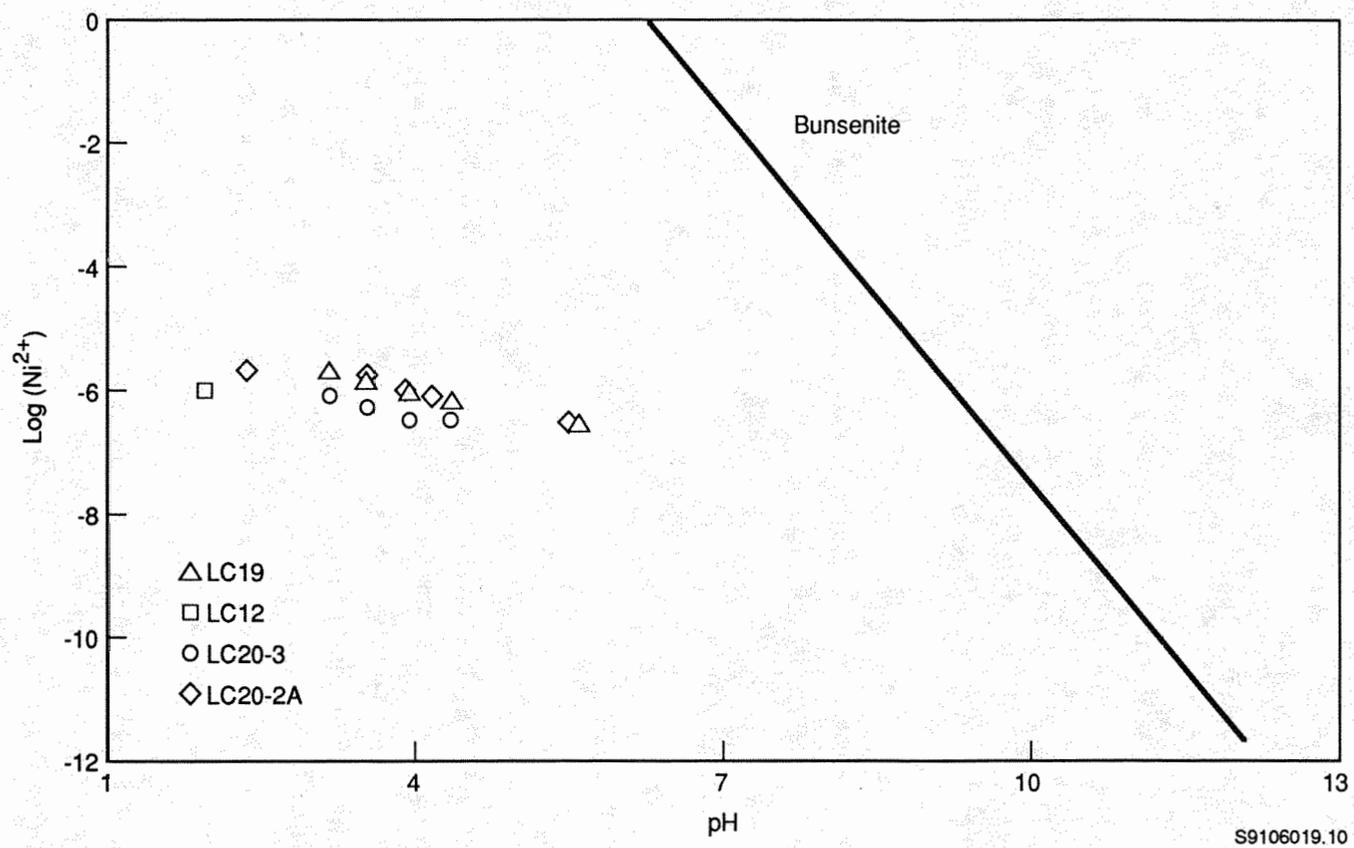


Figure 4-20. Log of Nickel Activity vs pH for All L-Site Core Samples

Strontium. The logarithm of the Sr^{2+} activity is plotted versus pH for all four core samples in Figure 4-21. Also plotted are profiles for two potential solubility controls, celestite (SrSO_4) and strontianite (SrCO_3). Neither appears to be an effective control for strontium in these samples. The down-gradient soil (core LC-12) is even more under-saturated than it appears from the diagram, because the actual pSO_4 for these samples is 3.7.

Zinc. The logarithm of the Zn^{2+} activity is plotted versus pH in Figure 4-22, along with the expected profile for zincite (ZnO). Previous work has shown zincite to be a solubility control in other fly ash samples when pH was higher than 9 as appears to be the case here. However, since the pH of all L-site samples is well below 9, no solubility controlling zinc solid phase appears to be present.

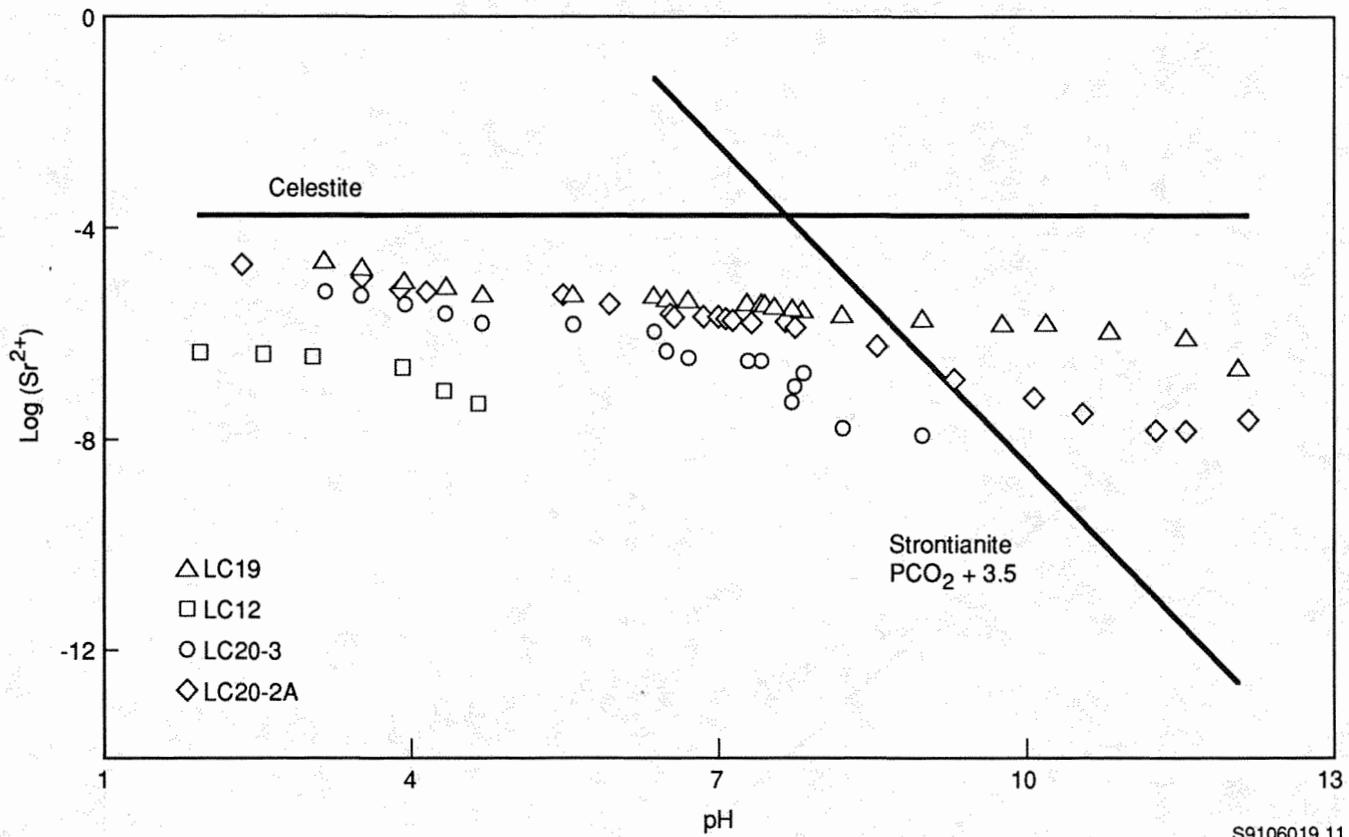
Other Trace Elements. In addition to the trace elements discussed above, arsenic, boron, and fluoride were also measured and are tabulated in Appendix F. No possible solubility controls were identified for those elements.

SUMMARY

Ash and Soil Characteristics

Soils at the L-site are mildly acidic, highly weathered products of micaceous gneiss and schists. Soil mineralogy consists of muscovite, kaolinite, quartz, and feldspar, with minor vermiculite, hornblende, and gypsum. Ash mineralogy consists of quartz, mullite and glass, with minor kaolinite, magnetite, barite, and pyrite. Physical properties of the soils were measured to determine aquifer properties. Measured vertical hydraulic conductivities ranged from about 2×10^{-7} to 2×10^{-3} cm/sec. The wide range of permeabilities is ascribed to the effects of preferential orientation of mica platelets. Porosity measurements on core samples found a mean value of about 50% porosity.

The L-site ash composition is slightly lower in calcic components and higher in ferric components than the U.S. average coal ash, reflecting the original coal mineralogy and leaching during ash sluicing and in the ash pond. Total and hydroxylamine-hydrochloride-extractable concentrations of elements in ash and soil samples were measured to determine the mobility of ash constituents in the ash pond and its underlying soils. Less than 7% of the total concentration of any element in the ash was determined to be leachable, except for the calcic elements barium, calcium, and



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Figure 4-21. Log of Strontium Activity vs pH for All L-Site Core Samples

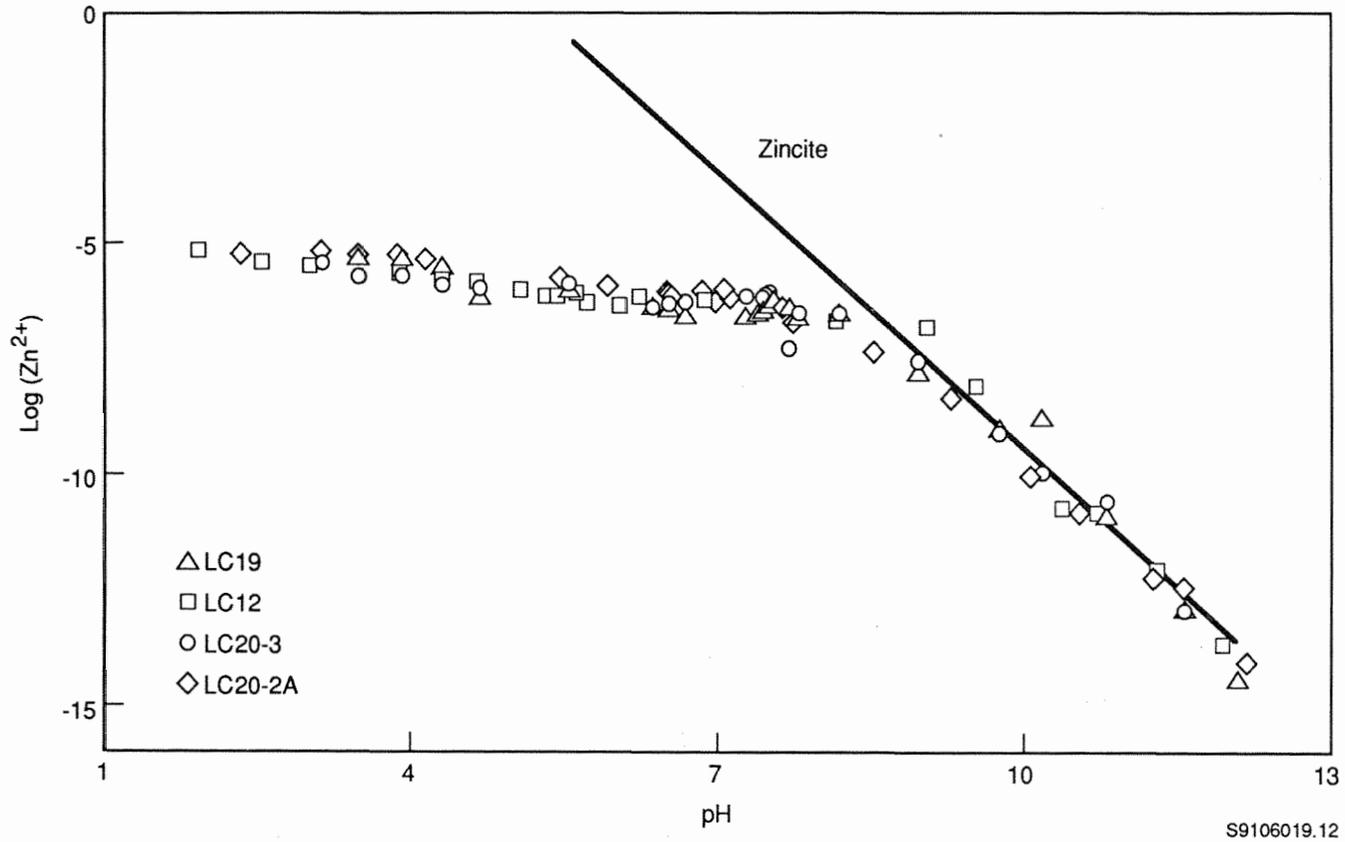


Figure 4-22. Log of Zinc Activity vs pH for All L-Site Core Samples

sodium. Soils beneath the ash had a larger leachable fraction of the total elemental concentrations, but control soils were indistinguishable from soils beneath the ash, suggesting that accumulation of ash-derived constituents in underlying soils was not important. Low soil cation exchange capacities and extractable iron concentrations suggest that the L-site soils have limited adsorptive capacity to attenuate the transport of metals in solution.

Solubility Controls

Possible solubility controls for As, B, Al, Ba, Ca, Cd, Cr, Cu, F, Fe, Mg, Mn, Ni, Sr, and Zn were tested for the L-site samples using the pH manipulation method. The results are summarized in Table 4-8. No potential solubility controls were identified for As, B, Ca, Cd, Cr, F, Mg, Ni, Sr, or Zn. Potential solubility controls were identified for Al, Ba, Cu, Fe, and Mn. The proposed solubility control for Ba is a sulfate and that for Mn is a phosphate. Oxides or hydroxides appear to be the solubility controls for Al, Cu, and Fe. No carbonates were identified as solubility controls for any of the samples. In the case of the ash samples (core LC-19 and LC-20-2A), the lack of carbonates probably indicates that these samples are not yet in equilibrium with atmospheric CO₂. In the case of the soil samples (cores LC-12 and LC-20-3), the lack of carbonates is typical of highly weathered, acidic soils like those at the L-Site. In the cases of Al, Ba, Fe³⁺, and Mn²⁺, solubility controls in the coal ashes are similar to those for the soils, so attenuation by precipitation cannot be expected for these elements in the soil, especially since the pH of the soils is lower than that of the ashes. The only real prospects for attenuation by precipitation are if Fe²⁺ and Mn²⁺ oxidize to higher oxidation states, which then form highly insoluble oxides or hydroxides that precipitate.

Impact of Codisposal on Leachable Inventories of Trace Elements

In geologic materials such as coal ash, the total concentration of an element may be a poor indication of the amount that is available to be leached out under disposal conditions. This is because at least part of the total amount of the element may be bound in highly insoluble minerals. For example, chromium contained in the mineral chromite, which is common in coal ashes, can be dissolved only under extreme chemical conditions. This chromium is not accessible to leaching by aqueous solutions at ambient temperatures, even if the solutions are fairly acidic or caustic.

In some cases, for elements whose solubility is controlled by oxides or hydroxides, an indication of the total inventory of leachables can be obtained from the pH manipulation data. For example, as shown in Figure 4-17 for copper, the fact that

Table 4-8

PROPOSED SOLUBILITY CONTROLS FOR L-SITE FLY ASHES AND SOILS

	Potential	Fly Ash	Soil
Al	Al(OH) ₃ (amorph) Gibbsite	Al(OH) ₃ (amorph)	Al(OH) ₃ (amorph)
As	none	none	none
B	none	none	none
Ba	barite (BaSO ₄)	barite (BaSO ₄)	barite (BaSO ₄)
Ca	gypsum	none	none
Cd	none	none	none
Cr	(Fe,Cr)(OH) ₃ , Cr = .01, Cr(OH) ₃	none	none
Cu	Tenorite	tenorite	none
F	none	none	none
Fe	Fe(OH) ₃ (amorph)	Fe(OH) ₃ (amorph)	Fe(OH) ₃ (amorph)
Mn	MnHPO ₄	MnHPO ₄	MnHPO ₄
Ni	none	none	none
Sr	none	none	none
Zn	zincite	none	none

the copper concentrations level out and fall off the tenorite line below pH 7 indicates that the leachable inventory of copper in these ashes has been exhausted below pH 7. Because the maximum concentration of copper is 1.21 ppm (see Table F.1) the analyses were performed in a 10:1 water to ash or soil weight ratio suspension, this concentration indicates that the leachable inventory of copper in these samples is about 12.1 ppm. This amount of leachable copper can be compared to data from other ashes (Ainsworth and Rai, 1987 and Rai et al., unpublished data).

The leachable concentrations of the various trace elements (As, B, Cd, Cu, F, Ni, and Zn) in the two ash samples were calculated using the pH manipulation data, and are summarized in Table 4-9.

The values in Table 4-9 were obtained by Battelle PNL laboratories (Ainsworth and Rai, 1987) for fly ashes from sites where low-volume wastes were not co-disposed. Therefore, co-disposal at this site appears to be having no effect on trace element inventories.

Table 4-9
 LEACHABLE INVENTORIES OF TRACE ELEMENTS IN L-SITE SAMPLES
 (mg/g ash)

	<u>L-site</u>	<u>Ranges of Values from Other Coal Ashes</u>
As	4.9	<2-10
B	7.5	10-1550
Cd	0.1	0.3-2.6
Cr	2.6	4.0-34
Cu	12.1	17-26
F	13.4	<1-58
Ni	3.2	3.0-12
Zn	10.5	35-80

Section 5

GROUNDWATER ANALYSIS

This section presents the results of the groundwater analyses made at the L-site ash pond. The results from the hydrogeologic analyses are presented first. Water quality analytical results are presented next with discussions of spatial trends of constituents in groundwater and surface water.

HYDROGEOLOGY

The hydrogeology of the L-site was characterized from boring logs from previous geotechnical investigations and from borings and wells installed as part of this investigation (Figure 5-1). None of the past geotechnical borings were completed as monitoring wells. The only wells present at the site before this investigation were nine piezometers completed in the ash-pond dams for monitoring pore pressures in the dam.

Geologic core collection and well installations were performed in three phases, so that hydrologic and geochemical data obtained from earlier phases could be used to better plan the locations of samples and monitoring wells for a later phase. The first phase of this investigation took place in December 1988 when four monitoring wells were installed downgradient of the primary ash pond. The second phase took place in March and April 1989 when 14 monitoring wells and four piezometers were installed in and around the primary ash pond. The third phase took place in February 1990 when two monitoring wells and one boring were installed downgradient of the dam in the bedrock valley.

A total of 24 monitoring wells and piezometers were installed at the L-site as part of this investigation (Figure 5-1). Monitoring wells and piezometers were installed on all sides of the primary ash pond except along the bedrock high separating the primary and secondary ponds. Most wells were installed downgradient of the pond to characterize the flow paths between the pond and river. A vertical well cluster was installed immediately downgradient of the primary pond to determine vertical head gradients and groundwater chemistry in the saprolite and bedrock deposits. Wells upgradient and east of the pond were used to define hydraulic gradients and natural background water chemistry.

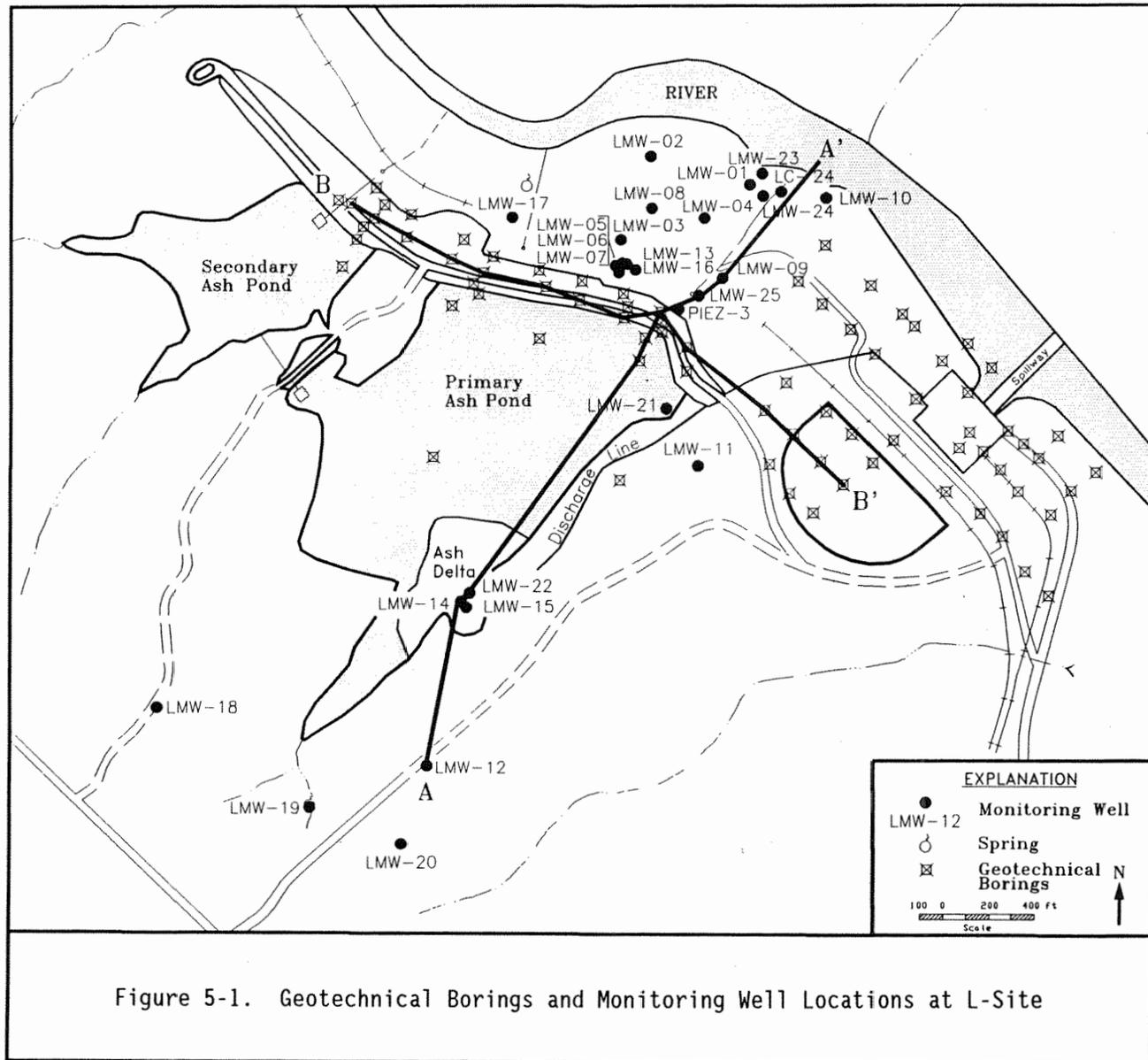


Figure 5-1. Geotechnical Borings and Monitoring Well Locations at L-Site

Monitoring wells also were installed within and beneath the ash delta to obtain groundwater, ash, and saprolite samples as close as possible to the ash discharge pipe. Constituents associated with co-management would have the greatest potential for release to the environment here.

Physical properties were determined through in situ aquifer tests consisting of slug tests and a multiple-well pumping test, and through laboratory measurements on cores. Laboratory hydrogeologic measurements included permeability, porosity, density, and grain-size analyses.

Water levels were measured continuously in five wells (LMW-5, 6, 7, 16, and 3) down-gradient of the dam with electronic pressure transducers and a data logger. Water levels were measured in all other wells every two weeks. A temporary piezometer was installed at various spatial locations in the pond sediments for one-time measurements of hydraulic-head gradients to quantify groundwater fluxes into and out of the pond.

A total of 70 cores were collected for lithologic descriptions, physical properties, and geochemical analyses. An attempt was made to collect a minimum of one core opposite the screen interval of each monitoring well for physical properties analyses. In addition, cores were collected from upgradient, downgradient, and from within the ash delta for geochemical analyses. Core collection was generally limited to the saprolite and partially weathered bedrock deposits.

Hydrologic tests were performed in a number of monitoring wells to obtain in situ hydrologic aquifer properties. Slug tests were performed in 13 of the 24 wells installed. A multiple-well pumping test was performed to obtain larger scale aquifer properties downgradient of the ash pond.

Geology

The soils and rocks at the L-site can be classified into five stratigraphic units: 1) saprolite; 2) alluvium; 3) fill material; 4) partially weathered bedrock; and 4) unweathered bedrock. In general, the materials upgradient of the ash ponds consist of unweathered bedrock, overlain by partially weathered bedrock, which is overlain by saprolite. Alluvial deposits have been encountered in borings adjacent to the river and in borings completed in tributary valleys to the river. Saprolite and alluvial deposits have been used as fill material to build up the area between the ash ponds and river.

The term "unweathered bedrock" is used in this report to mean that the bulk of the rock appears to be unweathered. Small discrete weathered zones may be present within the unweathered bedrock; however, the majority of the bedrock is unweathered. The term "partially weathered bedrock" implies that the majority of the rock has been altered, such that it is no longer as hard as the parent bedrock. In the geotechnical boring logs these rocks are referred to as soft to moderately hard.

The contact between the saprolite, partially weathered bedrock, and unweathered bedrock is qualitatively based on the depth at which moderately hard rocks and very hard rocks were first encountered in boreholes. In this investigation, the contact between the saprolite and partially weathered bedrock corresponds to the depth at which auger refusal or resistance was encountered. For the case of the previous geotechnical borings, the contact between the saprolite and partially weathered bedrock was the depth at which rotary drilling first encountered moderately hard rocks. In general, the auger method was unable to penetrate moderately hard rocks, whereas the rotary method could.

The contact between the partially weathered bedrock and unweathered bedrock was also based on drilling resistance. For both this investigation and the previous geotechnical borings, this contact was chosen as the depth at which rotary drilling refusal was encountered.

Unweathered Bedrock. The unweathered bedrock at the site was penetrated by three boreholes in this investigation and by 34 boreholes in past geotechnical investigations. The unweathered bedrock consists of fractured granitic gneisses and schists. In some cases, thin saprolite deposits (less than 0.1 feet thick) were encountered below the contact of the partially weathered and unweathered bedrock. The occurrence of saprolite below this contact is due to the chemical weathering of the bedrock along fractured zones. These saprolite deposits are typically mica-rich, indicating the parent material was a mica-rich schist.

The present surface topography approximately mirrors the unweathered bedrock topography. The bedrock valleys and ridges are oriented to the northeast, aligned with the present drainages of this area (Figure 5-2), and consistent with the predominantly northeastern regional orientation of major lineaments of the Inner Piedmont belt. Present drainages probably developed in areas that were more fractured and less resistant to weathering than the adjacent bedrock.

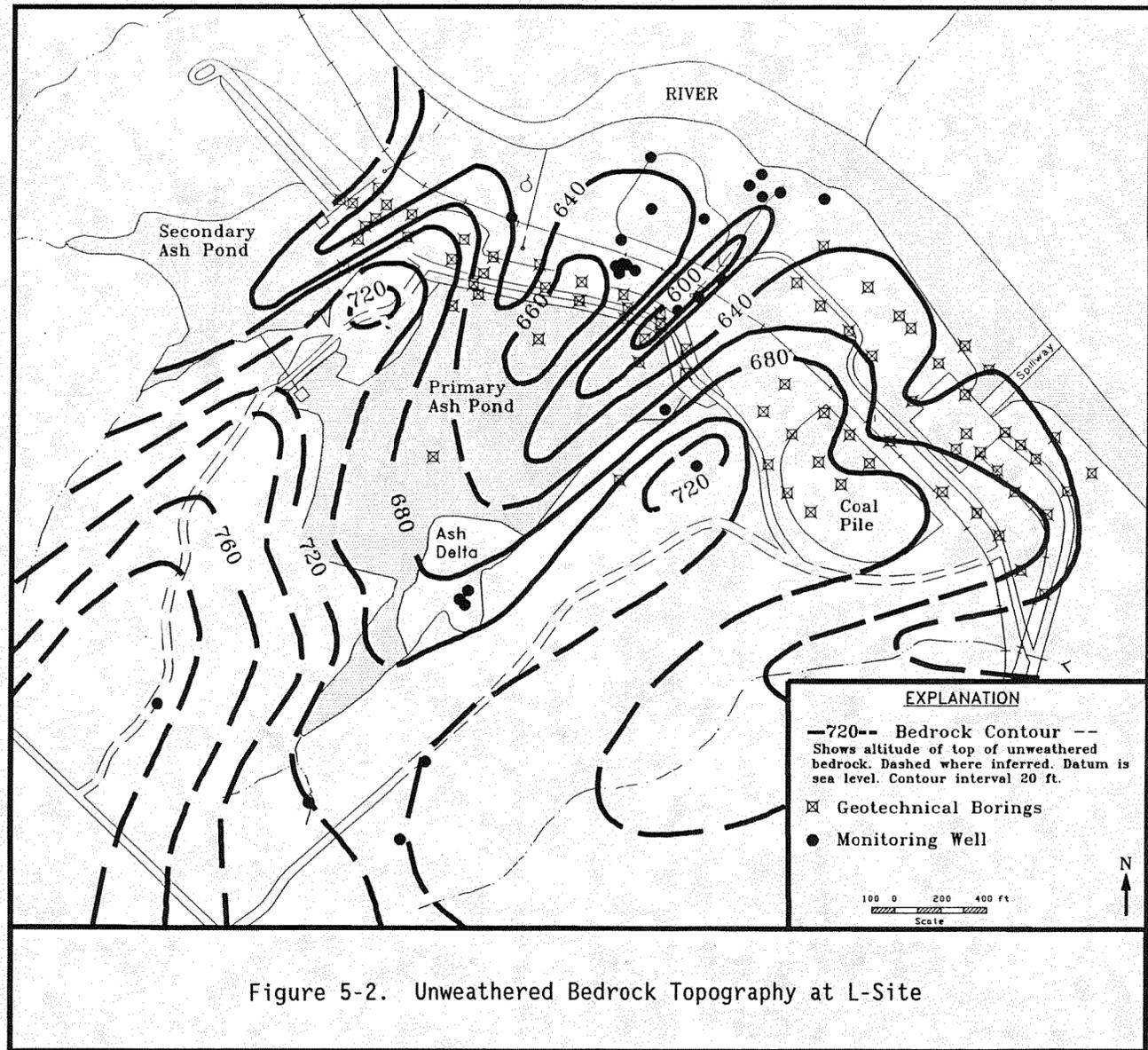


Figure 5-2. Unweathered Bedrock Topography at L-Site

Three bedrock valleys have been identified beneath the primary and secondary ash ponds (Figure 5-3). The largest of the three bedrock valleys is projected to extend from the upper reaches of the primary ash pond drainage, under the southern end of the ash pond dam to its confluence with the river.

Four geotechnical borings were drilled in the approximate center of this bedrock valley before the dam was built. None of the borings went deep enough to encounter unweathered bedrock. The deepest of the borings (D-22) went 60 feet (about 130 feet below the present dam elevation) and was terminated in unconsolidated material, at an elevation of about 601 feet. Two monitoring wells (LMW-24 and 25) and one boring (LC-24) were installed downgradient of the dam to investigate the lateral and vertical extent of the alluvium and bedrock valley (Figure 5-1). All boreholes were advanced to the top of the partially weathered bedrock; monitoring well LMW-25 was installed to a depth of 76 feet (elevation 606 feet); monitoring well LMW-24 was installed to a depth of 25 feet (elevation 631 feet); and boring LC-24 was installed to a depth of 22 feet (elevation 625 feet). The deepest part of the bedrock valley and the thickest saprolite is directly beneath the ash dam and becomes more shallow as it approaches the river (Figure 5-4). The greater depth of the bedrock valley beneath the dam is probably due to increased fracturing and weathering of a lineament in this location. In addition, the development of a tributary valley on the northeastern side of the river, opposite this bedrock valley, indicates that the lineament is locally extensive and cuts across the present river channel (Figure 5-2).

Two smaller bedrock valleys were also identified: one under the northwestern side of the primary ash pond and a second under the secondary ash pond. Limited geotechnical boring data in the vicinity of these valleys indicate that they are not as deeply incised under the ash dam or as laterally extensive as the larger valley under the southeastern side of the primary pond. The deepest unweathered bedrock was encountered at an elevation of about 634 feet in the bedrock valley under the secondary pond. The elevation of the smaller bedrock valley under the primary pond is unknown; however, geotechnical boring data indicate that its elevation is less than 634 feet. The bedrock valley under the river also probably developed along a more highly fractured, less resistant bedrock zone.

Partially Weathered Bedrock. The partially weathered bedrock is stratigraphically between the saprolite and unweathered bedrock. It represents a transition from the extensive chemical weathering of the bedrock material (saprolite) and the unweathered bedrock (gneiss and schist). This transition zone results from the more rapid

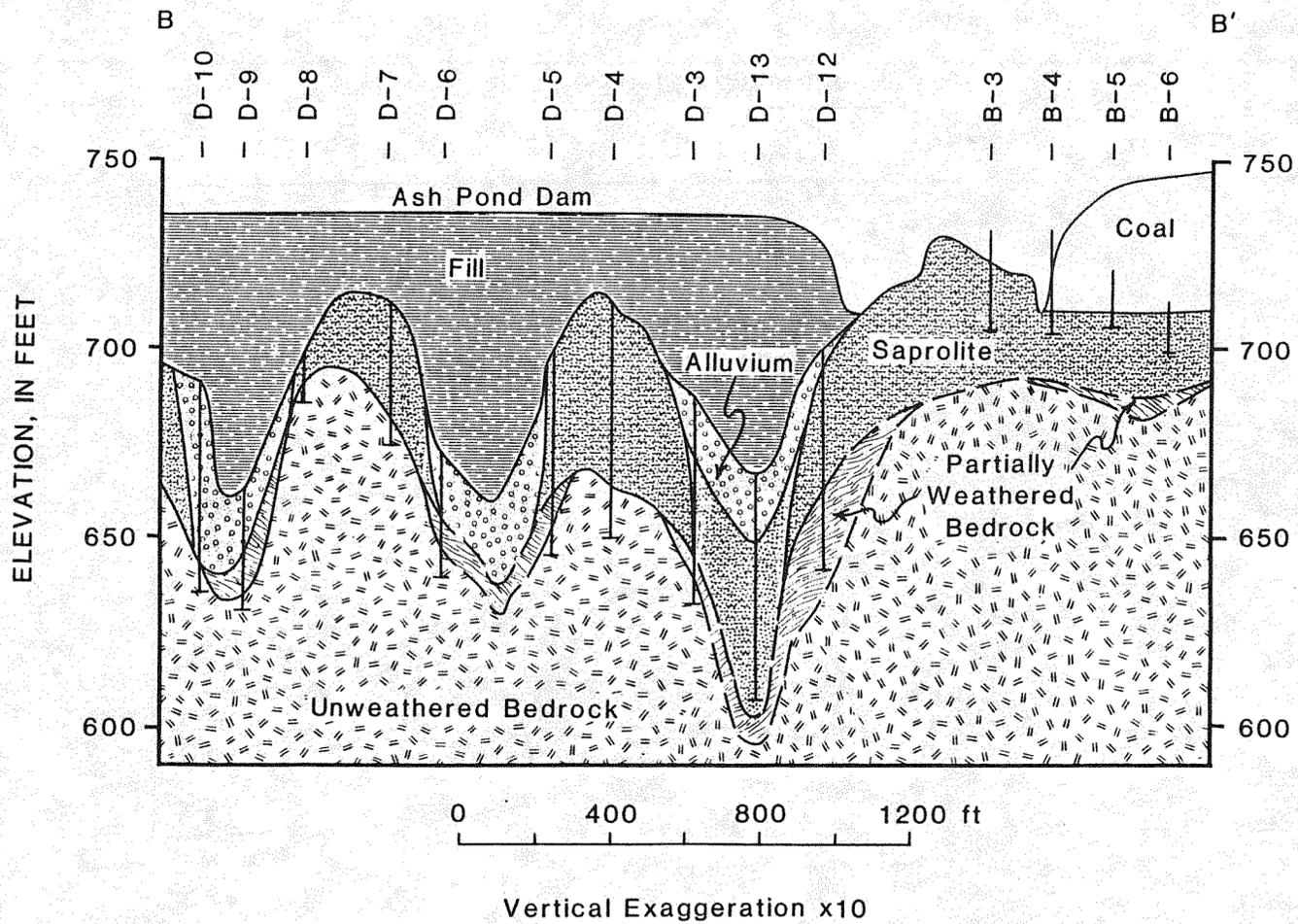


Figure 5-3. Geologic Section B-B' Parallel to Dam (Location of geologic section B-B' shown in Figure 5-1)

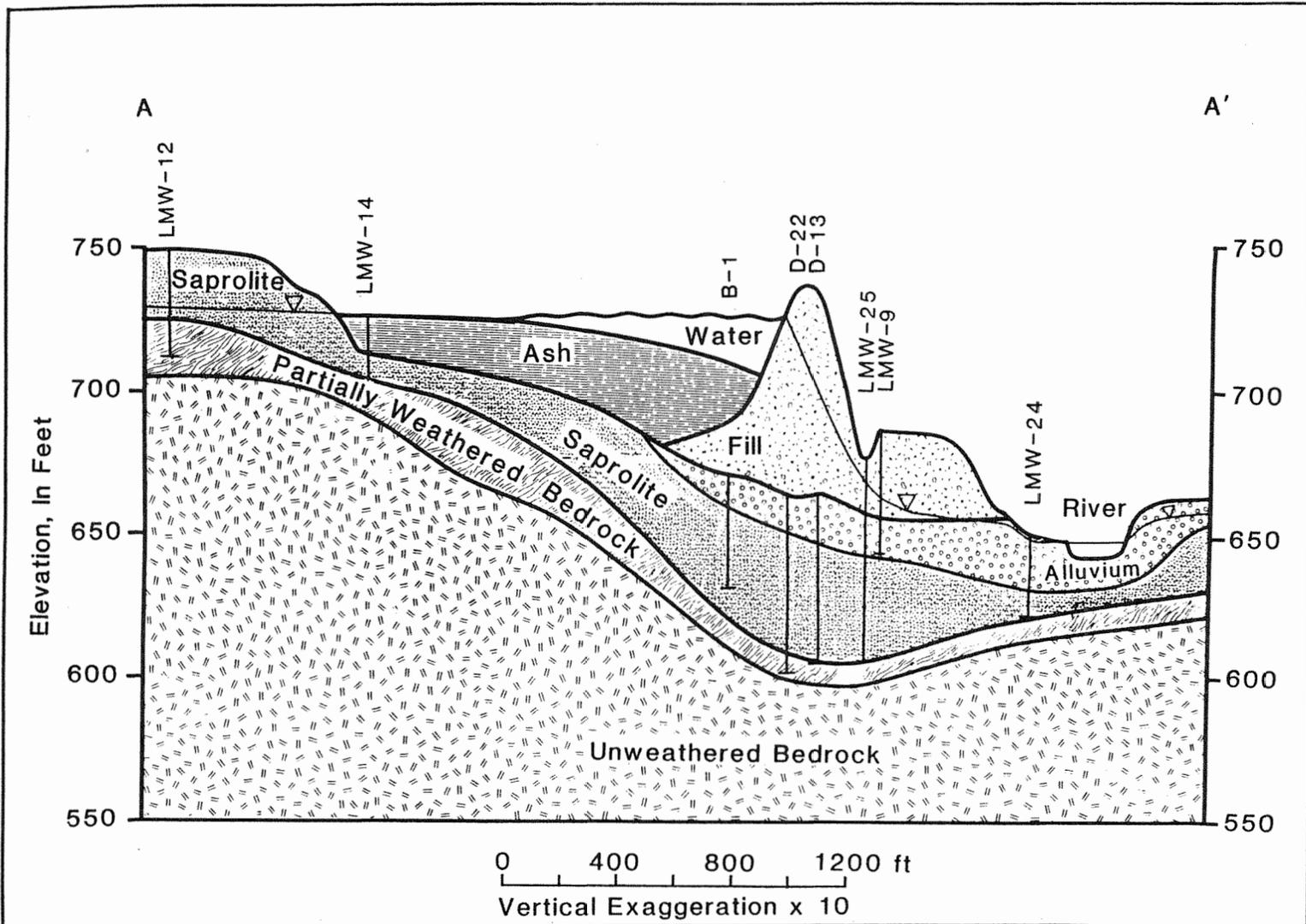


Figure 5-4. Geologic Section A-A' Perpendicular to Dam (Location of geologic section A-A' shown in Figure 5-1)

weathering of the less resistant minerals such as feldspars versus the slower weathering of the more resistant minerals such as quartz.

A transition zone clearly exists between the saprolite and bedrock; however, criteria for establishing the upper and lower boundaries of this zone can be based on a number of hydrogeologic and geochemical features. For the purposes of this investigation, the upper and lower boundaries of the transition zone are qualitatively based on the drilling resistance. The upper boundary is defined as the depth at which significant resistance or auger refusal was first encountered. The lower boundary is defined as the depth at which rotary drilling refusal was first encountered.

A number of factors influence the thickness of this transition zone including: 1) changes in the bedrock mineralogy both laterally and vertically; 2) fracture density and interconnectivity; 3) chemistry of water in contact with the rock; and 4) the flux rate of water through the rock. Given the potential variability and unpredictability of some of the factors listed above, accurate predictions of the thickness of this zone in areas where borehole data are not available are difficult to make with any degree of certainty.

The partially weathered bedrock encountered in boreholes at the site ranges in thickness from less than 1 foot to about 38 feet in the boreholes which penetrated this zone (Figures 5-3 and 5-4). The large variability in the thickness of this zone is evident in the vicinity of the primary ash pond dam. The partially weathered bedrock thickness goes from 0 feet (D-17) to 38 feet (D-16) in boreholes less than 200 feet apart. In general, the thickness of the partially weathered bedrock encountered at the site is less than 20 feet.

The rocks in the partially weathered bedrock zone range from saprolite to granitic gneisses. Materials encountered in the drilling of LMW-18 consisted of alternating layers of gneiss and sandy-silt saprolite. This alternating lithology may be attributable to alternating layers of a schist and gneiss bedrock in which the schists have been weathered to a saprolite and the gneiss is less weathered.

The partially weathered bedrock is absent in some of the boreholes drilled adjacent to the river and in borings adjacent to the bedrock valleys under the primary and secondary ponds. It has apparently been removed by erosion during the development of the bedrock valleys.

Unconsolidated Deposits. The following paragraphs describe the geology of alluvial, saprolite, and fill deposits encountered at L-site.

Alluvial Deposits. Alluvial deposits were encountered in 6 of the 21 boreholes drilled during this investigation and in 10 boreholes from previous geotechnical investigations. The identification of alluvial deposits from historical geotechnical borings data was difficult because most of the alluvial material is derived from saprolite with similar lithology. The abundance of sand-size mica grains in the saprolite makes it difficult to distinguish between alluvial sands and saprolite sands from the borehole lithologic descriptions. In addition, trace amounts of gravel can be present in both saprolite and alluvial deposits. Hence, the mention of gravel or pebbles in the sample description is not a clear indication of alluvial deposits.

Two separate criteria were used to identify alluvial deposits in historical geotechnical boring descriptions: 1) the presence of organic material in core samples obtained from greater than 10 feet below land surface, or 2) the presence of significant amounts of gravelly or pebbly sand deposits. In a number of cases, material in boreholes drilled adjacent to other boreholes in which alluvial deposits were identified did not meet either of these criteria. In general, these deposits were not classified as alluvial, even though the close proximity of the boreholes indicated that the deposits probably are alluvial.

Geologic descriptions of sediments encountered during the drilling of the geotechnical borings indicated that the alluvial deposits contained in the bedrock valley may exceed 60 feet in thickness (D-22). Monitoring wells LMW-24 and LMW-25 and boring LC-24, installed to investigate the lateral and vertical extent of alluvium, only encountered 10 to 20 feet of alluvium overlying saprolite. Hence, alluvium thicknesses projected from geotechnical borings in this bedrock valley have been adjusted to reflect the more recent data; however, the alluvium thickness may be overestimated in the adjacent bedrock valley.

Alluvial deposits were encountered in four boreholes adjacent to the river and in 12 boreholes aligned with the tributary valleys in which the ash ponds were constructed. The alluvial deposits range in thickness from about 13 feet (LMW-09) to 20 feet (D-22) (Figure 5-4). The alluvial deposits in the bedrock valley beneath the river are projected to be about 10 feet thick based on the projected elevations of the river bottom and bedrock surface. The alluvial thicknesses reported in the geotechnical borings drilled in 1973 represent conditions prior to the construction

of the ash ponds and therefore may not reflect current alluvial thicknesses in these areas. The alluvial deposits range from a light brown silty clay to a well-sorted gray pebbly sand, with trace organics present in a number of core samples.

The areal extent of the alluvial deposits in the bedrock valleys under the ash ponds is expected to be limited (Figures 5-3 and 5-4). The steep, narrow bedrock valleys would tend to restrict the alluvial deposits to the immediate valley bottoms.

Saprolite Deposits. Most of the shallow unconsolidated material in the vicinity of the L-site is saprolite, formed in place by the chemical weathering of the parent bedrock material. The saprolite at the L-site occurs as a relatively homogeneous unit at shallow depths (less than about 35 feet) and as small discrete zones (less than about 0.1 feet) in unweathered bedrock at depth. The development of saprolite at depth is probably a result of the chemical weathering of bedrock along fracture zones. In addition, the abundance of mica in saprolite deposits at depth indicates that mineral facies changes also play a major role in the development and location of saprolite deposits at depth. Boreholes for wells LMW-11 and LMW-18 encountered mica-rich saprolite deposits beneath unweathered gneiss (Appendix G).

One distinct feature of saprolite is that it commonly retains the original structure present in the unweathered parent rocks. At the L-site the parent rock appears to be a combination of granite gneiss and mica-rich schist. Foliation, banding, and weathered quartz/feldspar veins are present in core samples of the saprolite.

The saprolite encountered in boreholes at the site ranged in thickness from less than 1 foot to about 48 feet, with the thickest accumulation in the area of the primary ash dam (Figure 5-3). The maximum saprolite thickness encountered in boreholes drilled during this investigation was about 36 feet in LMW-25, downstream of the primary ash dam. During the construction of the dam, saprolitic material removed from areas of the ash ponds was used as construction material in the dam and as fill material downgradient of the dam. As a result, it is sometimes difficult to discern saprolitic fill material from undisturbed saprolite deposits in the area immediately downstream of the dam.

The saprolite at the L-site ranges from a brownish red to grayish white silty clay to sand. Laboratory grain-size analyses, performed on 21 core samples, indicate that the grain size of the saprolite is predominantly sand. The sand-sized fraction ranges from 53 to 92 percent of the samples, with a mean value of about 85 percent. The silt-sized fraction ranges from 17 to 40 percent with a mean value of about 22

percent, and the clay fraction ranges from 6 to 12 percent with a mean value of about 7 percent.

The predominant mineral group present in the saprolite and contributing to the high percent of sand-sized grains is mica. The mica content in some subsections of core is greater than 80 percent. As a result of the high mica content, field description of drill cuttings tended to classify the saprolite as more fine-grained than the classification resulting from laboratory grain-size analysis. In the field, the flat platelet shape of the mica grains caused the grains to be classified as a silt rather than a sand. Hence, the field descriptions of saprolite samples presented in Appendix G do not necessarily agree with the lithologic classification resulting from the laboratory analyses.

The platelet shape of the mica grains also affected the hydrometer analyses of the silt/clay fraction. The hydrometer analysis is based on Stoke's Law, which assumes a spherical grain shape. Because the platelet shape slows the sedimentation process of the particles, the hydrometer analyses overestimate the amount of fine-grained material present in the samples.

Weathered quartz/feldspar veins were encountered in the saprolite. As discussed in the hydrology section, the weathered quartz/feldspar veins tend to be associated with higher permeability zones within the borehole.

Fill Deposits. Fill material consisting of saprolite and alluvial deposits was also present downstream of the primary ash pond dam. The fill material ranged in thickness from 0 feet to greater than 30 feet and consisted of a pebbly silty clay to a silty sand. In a number of locations downstream of the dam, it was difficult to distinguish between saprolite fill and natural saprolite.

Hydrology

The directions and rates of groundwater flow at the L-site are controlled by a number of factors including: 1) the hydrologic properties of the saprolite, the pond bottom sediments, and the partially weathered and fractured bedrock materials; 2) the locations and magnitudes of recharge and discharge sources; and 3) the topography of the partially weathered and unweathered bedrock surfaces. A detailed quantification of the shallow groundwater hydrology at the L-site should include an evaluation of all these factors.

Potentiometric Surface Elevation and Temporal Fluctuations. The shape of the potentiometric surface can be used to estimate the average groundwater flow directions. Groundwater flows from areas of higher hydraulic head to areas of lower hydraulic head. In an isotropic and homogeneous aquifer, the average direction of groundwater flow is perpendicular to the potentiometric surface contours. However, in an anisotropic and heterogeneous aquifer, the actual flow direction may be different from that indicated by a line perpendicular to the contour lines.

The potentiometric surface in the immediate vicinity of the ash pond was constructed from average water-level elevations from May through October, 1989 (Figure 5-5; see Appendix H for complete data set). Water-level data were obtained from wells completed in saprolite, partially weathered, and unweathered bedrock materials. In general, the monitoring wells used to construct the map were completed in the upper 15 feet of the saturated material. Exceptions to this were the upgradient monitoring wells LMW-18 and LMW-12, which were completed about 40 and 25 feet into the saturated material, respectively. A private well (LMW-20) is also reported to be less than about 30 feet into the saturated material. The potential errors in the potentiometric surface elevation for LMW-18, LMW-12, and LMW-20 are projected to be less than about 2 to 3 feet.

The potentiometric surface approximately mimics the land surface topography at the L-site, ranging from a maximum elevation of about 745 feet (LMW-18) upgradient of the primary ash pond to a minimum elevation of 653 feet (LMW-10) in the immediate vicinity of the river (Figure 5-5). The general direction of groundwater flow in the immediate vicinity of the primary ash pond is northeasterly toward the river.

The primary ash pond appears to be a source of both recharge and discharge for the shallow groundwater system. Groundwater elevations in monitoring wells around the primary ash pond indicate that the pond has created a localized groundwater mound. Groundwater is flowing away from the pond on the northeastern, northern, and northwestern sides of the pond while it flows toward the pond on the southwestern (upgradient) side of the pond. Hence, somewhere between the upper reaches of the pond and the ash pond dam the groundwater flow direction switches from flowing into the pond to flowing out of the pond.

Monitoring well and piezometer water-level data from the ash delta and dam areas (Appendix H) also confirm that groundwater flows into the pond near the ash delta and out of the pond near the dam. Wells LMW-14, 15, and 22 are completed at multiple depths within the ash delta of the primary ash pond. Monitoring well LMW-14 is

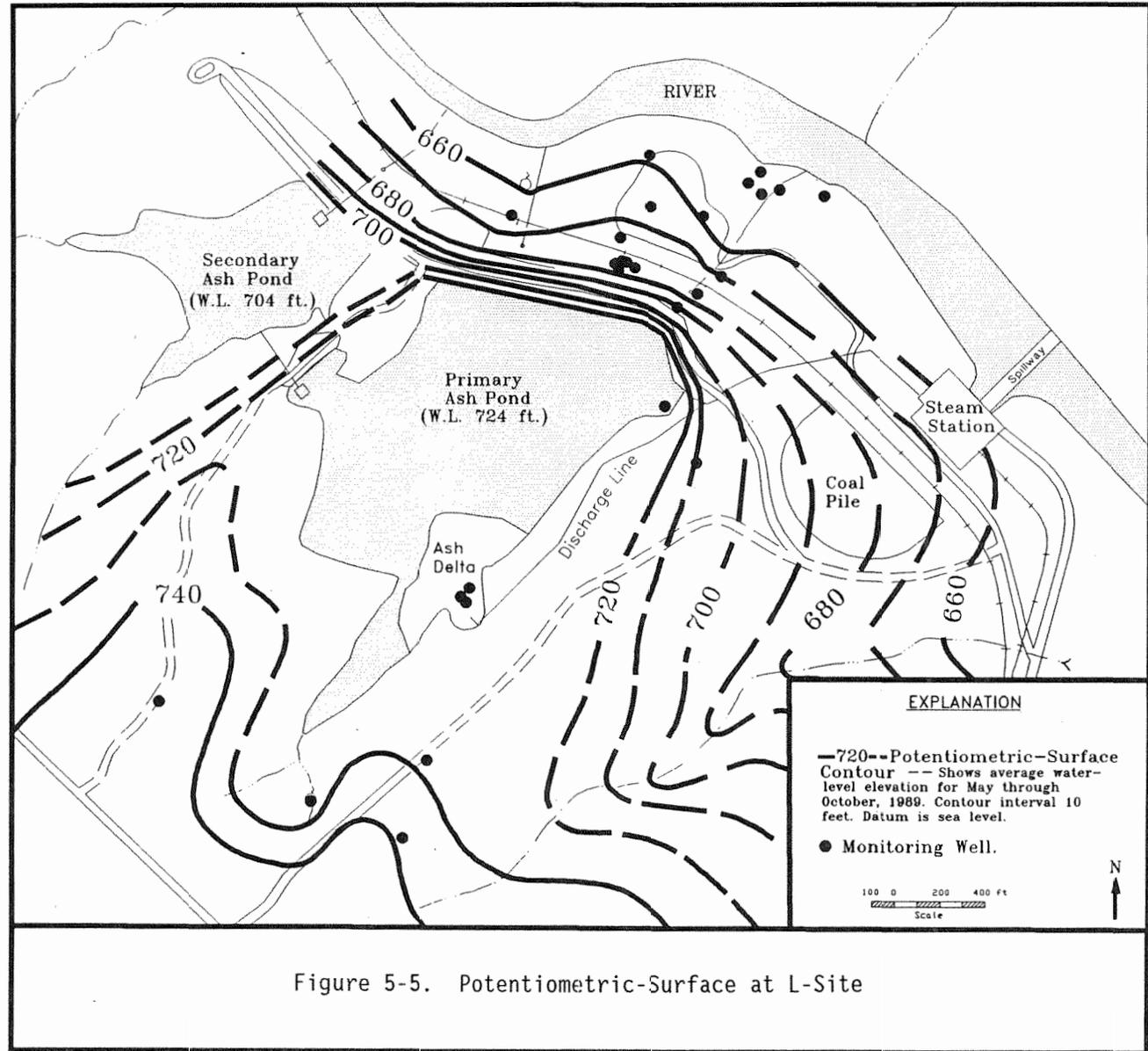


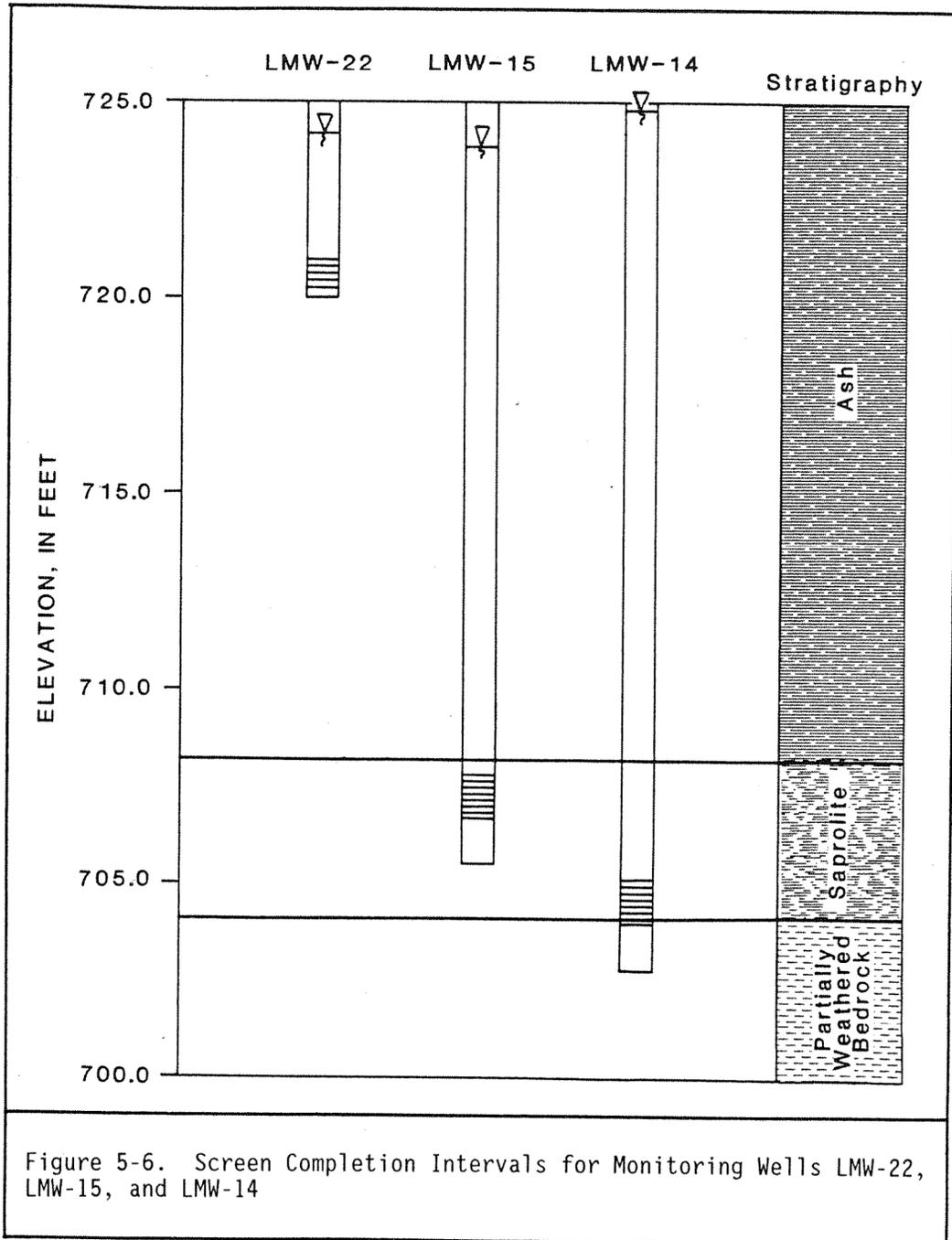
Figure 5-5. Potentiometric-Surface at L-Site

completed at the top of the partially weathered bedrock in the saprolite about 8 feet below the base of the ash; LMW-15 is completed at the contact of the ash and saprolite and LMW-22 is completed in the ash about 5 feet below the ash delta surface (Figure 5-6).

The differences in water-level elevations for this group of nested wells indicate the direction of vertical groundwater flow in this area. Water-level elevations for wells LMW-14 and LMW-22 have consistently been about 0.5 and 0.2 feet higher, respectively, than the water-level elevations in LMW-15 (Figure 5-7). The higher potentiometric-surface elevation in LMW-14 indicates that the vertical groundwater flow direction is upward toward the ash/saprolite contact and the higher potentiometric-surface elevation in LMW-22 indicates that the direction of flow is downward from the ash to the ash/saprolite contact. Hence, it appears that water is moving downward from the ash and upward from the saprolite and converging at the ash/saprolite contact.

The direction of groundwater flow at the ash/saprolite contact beneath the ash delta has periodically reversed from flow into the pond to flow out of the pond, whereas the direction of groundwater flow from the saprolite and ash deposits has consistently been into the pond. The water-level elevations in LMW-14 and LMW-22 have consistently been higher than the water-level elevation of the pond over the four-month period of record (Figure 5-7). The higher water-level elevations in LMW-14 and LMW-22 indicate that the groundwater flows from the lower part of the saprolite unit and from the upper part of the ash into the pond. Conversely, the water-level elevations in LMW-15 have fluctuated above and below the pond elevation, ranging from 0.95 feet above to 0.15 feet below. In general, the water-level elevation in LMW-15 has been above that of the pond, indicating that the vertical groundwater flow direction is toward the pond from the ash/saprolite contact. However, when the water-level elevation in LMW-15 is below that of the pond, the vertical direction of groundwater flow reverses and the flow direction is from the pond toward the ash/saprolite contact.

The consistently higher water-level elevation in LMW-14 than that of the pond indicates that the pond water is restricted to the upper saprolite zone. However, water samples from LMW-14 indicate that inorganic constituents from the ash have migrated down to the base of the saprolite. Historically, the pond stage has been reported to be as high as 727 feet (oral communication, plant personnel, 1989). A pond elevation of 727 feet would exceed the water-level elevation in LMW-14 (about 725 feet) and result in pond water moving vertically downward to the base of the saprolite.



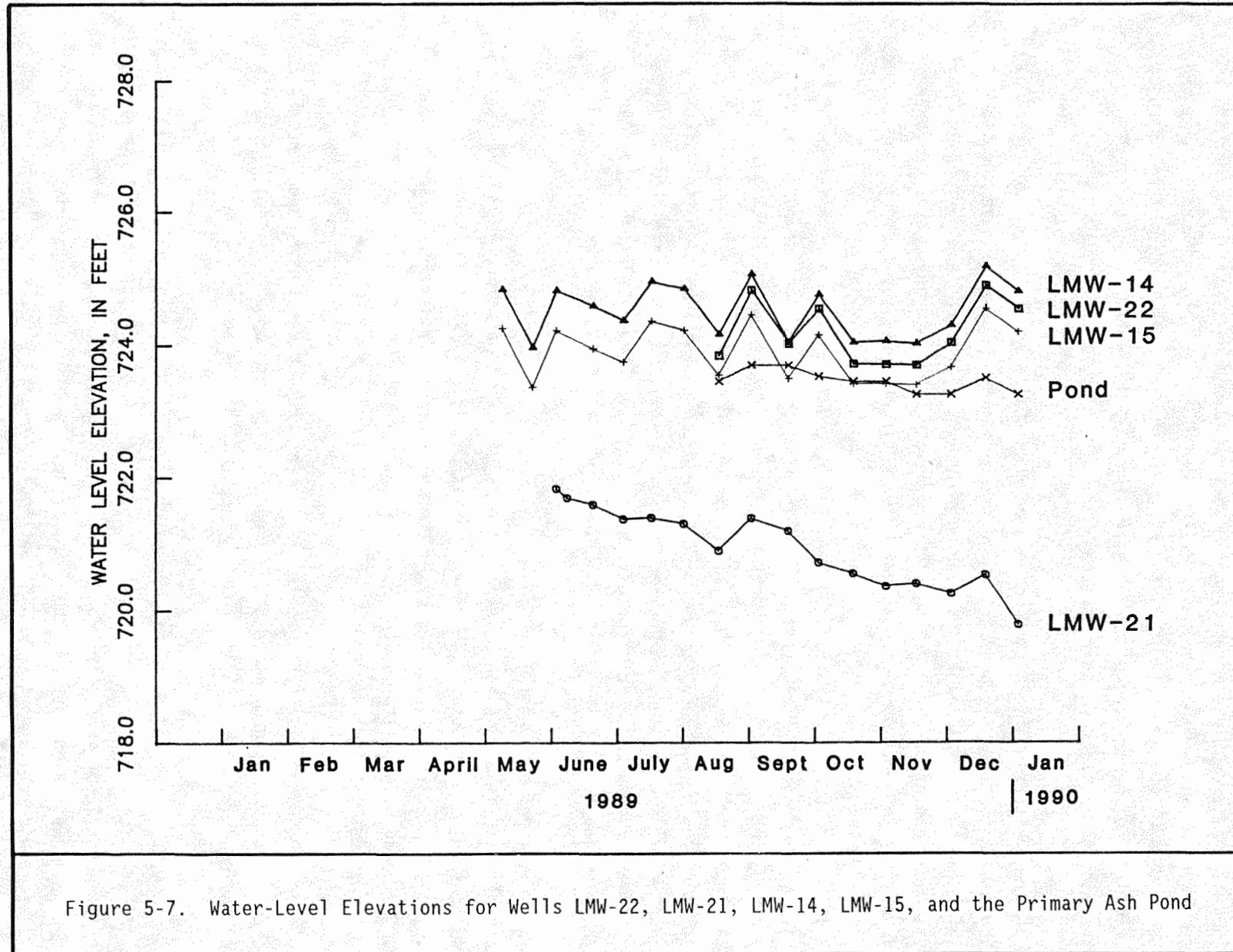


Figure 5-7. Water-Level Elevations for Wells LMW-22, LMW-21, LMW-14, LMW-15, and the Primary Ash Pond

Pond piezometer LMW-21, located on the northeastern side of the pond near the ash dam (Figure 5-1), was mechanically driven about 3.8 feet below the base of the ash pond. The water-level elevation in LMW-21 is about 3.5 feet below that of the pond (Figure 5-7); hence, the potential direction of groundwater flow is downward through the base of the pond at this location.

The large difference in hydraulic head between LMW-21 and the pond indicates that the permeability of the pond-bottom sediments is low. The continued decline in water level in LMW-21 over the seven-month period of record is an indication that this well has not yet reached equilibrium and that the hydraulic-head gradient across the pond-bottom sediments may be even larger than that currently projected.

Groundwater-level fluctuations in LMW-14, 15, and 22 appear to be affected by both natural groundwater level fluctuations and changes in the pond stage. The eight-month period of record from May through December 1989 showed water-level fluctuations of about 1.1 feet in LMW-14, 15, and 22 and water-level fluctuations of about 0.3 feet in the ash pond (Figure 5-7). The difference in the magnitude of the water-level fluctuations in the pond versus the wells is an indication that the observed groundwater level fluctuations are not strictly attributable to changes in the pond stage.

The pond stage and water-level elevations in LMW-14, 15, and 22 also do not correlate. The pond stage decreased from September to November 1989, whereas the water-level elevations in the wells increased. This is further evidence that the changes in water-level elevations in the wells are not directly related to changes in the pond stage. The flux of water into the pond appears to be dependent on natural groundwater fluctuations.

Conversely, water-level changes in LMW-21 near the ash dam correspond directly to changes in pond stage (Figure 5-7), suggesting that groundwater levels in this area of the pond are directly impacted by changes in pond stage.

Groundwater level fluctuations in monitoring wells upgradient of the primary ash pond are distinct from those downgradient of the pond. In general, the magnitude of groundwater level changes upgradient of the pond was greater than the magnitude of changes downgradient of the pond (Figure 5-8). Water levels upgradient of the pond fluctuated on average more than 1 foot; only private well LMW-20 fluctuated less than 1 foot (0.53 feet). Water levels in wells in the immediate vicinity of the dam

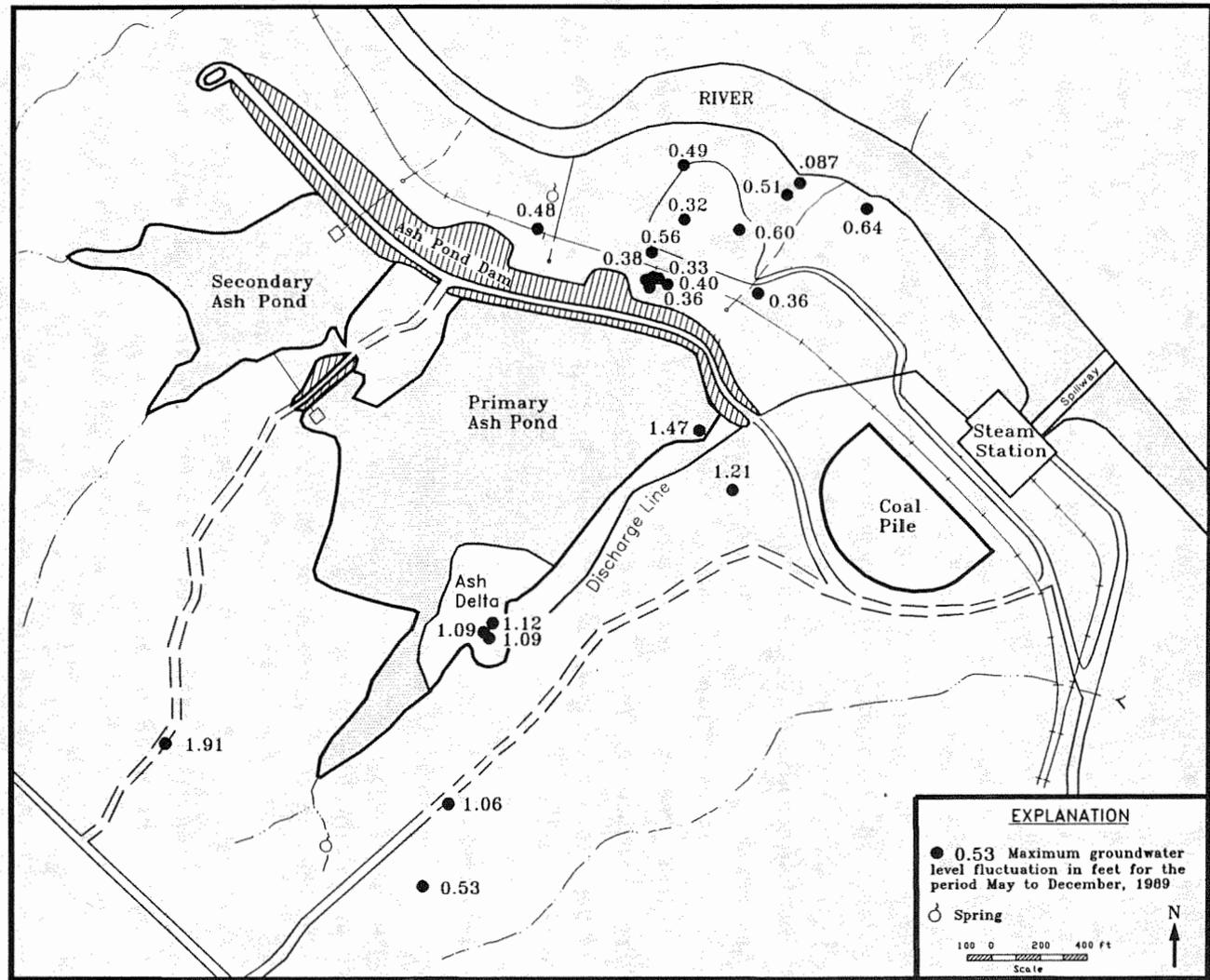


Figure 5-8. Groundwater Level Fluctuations in L-Site Monitoring Wells

fluctuated less than 0.5 feet and water levels in wells near the river fluctuated between 0.5 feet and 0.9 feet.

The primary ash pond and its associated drains in the dam comprise the primary hydrologic feature controlling groundwater levels immediately downgradient of the pond. The approximately 0.4 feet of water-level change observed in wells in the immediate vicinity of the ash dam is consistent with the 0.3 foot water-level change observed in the ash pond. Groundwater levels upgradient of the ash pond are not significantly impacted by changes in the pond level; hence, the magnitudes of the groundwater level changes upgradient of the ash pond are a reflection of the natural recharge and local groundwater level fluctuations. Similarly, water-level changes in wells in the immediate vicinity of the river are affected more by changes in river stage and bank storage than by changes in the ash-pond level. Therefore, groundwater levels upgradient of the ash pond are primarily controlled by changes in natural recharge; groundwater levels immediately downgradient of the pond are controlled by the pond stage; and groundwater levels near the river are primarily controlled by the river stage.

Water levels in a three-well cluster downgradient of the ash pond were used to evaluate the direction of vertical groundwater flow in the saprolite, partially weathered bedrock, and unweathered bedrock units. The top of the screen for LMW-07 was completed at an elevation of 650.0 feet, about 11 feet below the unweathered bedrock surface; the top of the screen for LMW-05 was completed at an elevation of 662.2 feet, at the base of the saprolite; and the top of the screen for LMW-06 was completed at an elevation of 678.0 feet, in the upper 4 feet of saturated saprolite (Figure 5-9). Differences in water-level elevations for this three-well cluster indicate that the direction of vertical groundwater flow in this area is upward from the unweathered bedrock to the partially weathered bedrock and saprolite deposits.

Water samples from LMW-07 have elevated concentrations of inorganic constituents, similar to the downgradient wells completed in the saprolite (see the latter part of this section on Water Quality). The presence of these inorganic constituents clearly indicates that pond water has moved downward into the shallow bedrock. Pond water is being forced downward into the shallow bedrock beneath the pond by the higher hydraulic head in the pond (about 724 feet). The direction of groundwater flow reverses downstream of the pond. The change in direction of flow is caused by the geometry of the pond and the presence of the toe drain system in the dam, which helps to pull groundwater up from the shallow bedrock.

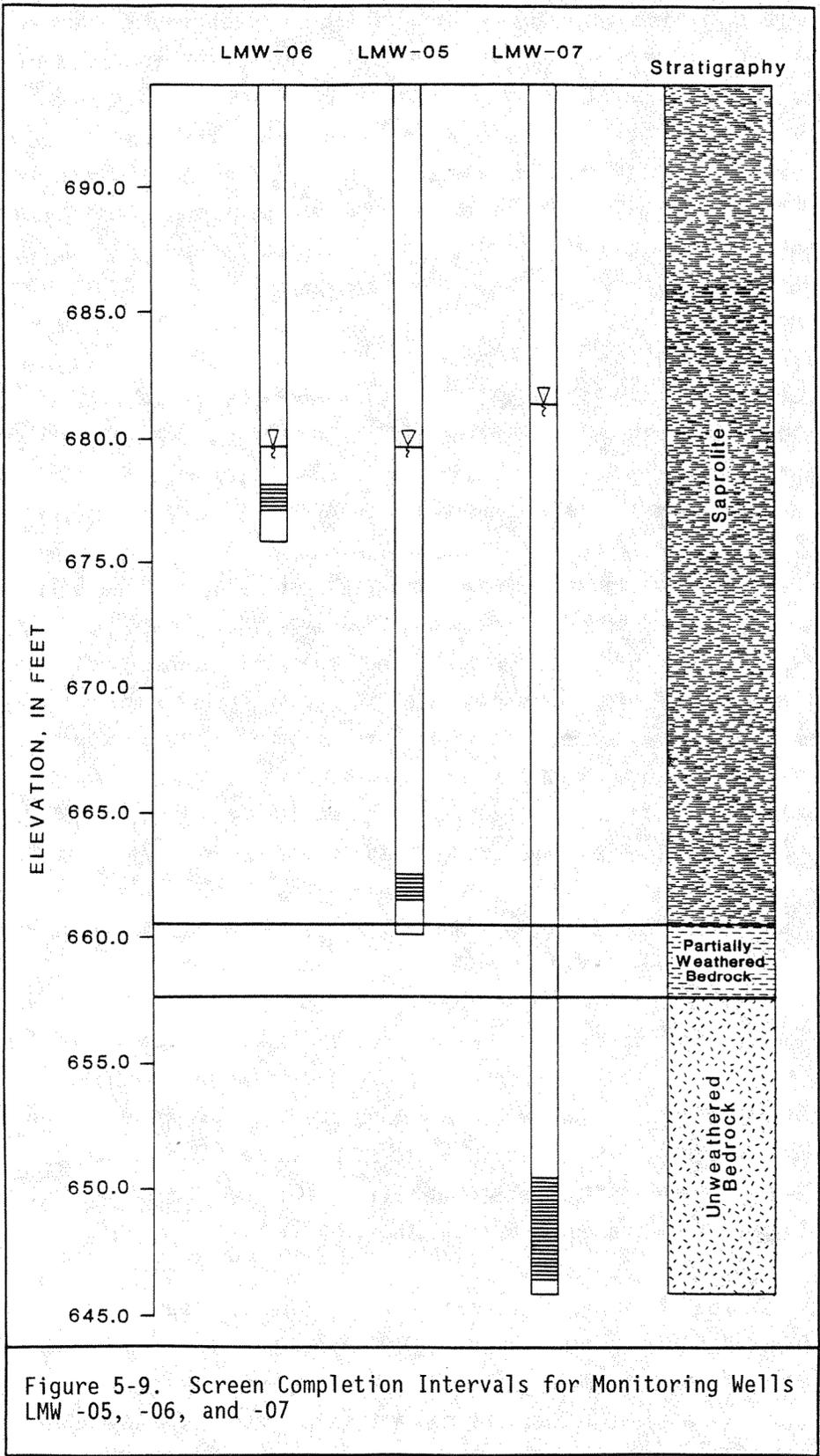


Figure 5-9. Screen Completion Intervals for Monitoring Wells LMW -05, -06, and -07

Aquifer Characteristics. Hydraulic-conductivity data, obtained from both in situ and laboratory measurements, were used to help quantify groundwater flux rates at the site. In situ permeability values were obtained by performing slug tests and a multiple-well pumping test in monitoring wells at the site. Laboratory physical properties measurements were performed on core samples of aquifer material and pond bottom sediments. Physical properties measurements performed on cores consisted of moisture content, bulk density, particle density, grain-size analysis, and permeability measurements.

Slug Tests. Slug tests provide an estimate of the horizontal hydraulic conductivity in the immediate vicinity of the wellbore. The small volume of water displaced during a slug test limits the volume of aquifer tested to that within a few feet of the wellbore. Hence, small-scale aquifer heterogeneities in the vicinity of the wellbore can significantly impact the measured hydraulic-conductivity values.

Aquifer hydraulic-conductivity values calculated from slug-test data at the L-site range over three orders of magnitude from 1.2×10^{-4} cm/sec (LMW-11) to 8.7×10^{-2} cm/sec (LMW-17) (Table 5-1). In general, the lowest hydraulic-conductivity values were measured in monitoring wells completed in the fractured unweathered bedrock and the highest values were measured in wells completed at the saprolite/partially weathered bedrock contact (Appendix I). One exception to this is monitoring well LMW-18, which appears to have a fairly high permeability (greater than 10^{-3} cm/sec) based on its rapid water-level recovery during development. The permeability of LMW-18 was not measured with a slug test because the depth to water in this well (about 78 feet) exceeded the 50-foot depth limitation of the slug-test equipment.

A type-curve match was not obtained for the slug-test data from monitoring well LMW-07, which is also completed in the unweathered bedrock. The water level in the well was bailed down to its screen interval and the recovery was monitored over time. The permeability of the fractured bedrock opposite the screen in this well was extremely low, such that it required in excess of 65 days for the water level to recover. Hence, an analysis of the slug-test data was not possible, but the hydraulic conductivity is estimated to be less than 10^{-7} cm/sec.

The highest horizontal hydraulic-conductivity values measured at the L-site were in wells LMW-08 (4.2×10^{-1} cm/sec) and LMW-17 (8.7×10^{-2} cm/sec). Both of these wells were completed at the saprolite/partially weathered bedrock contact (Table 5-1). Not all wells completed at the saprolite/partially weathered bedrock contact had high permeabilities. Monitoring well LMW-04 was completed at this contact and its

Table 5-1

HORIZONTAL HYDRAULIC-CONDUCTIVITY VALUES MEASURED FROM
SLUG TESTS AT THE L-SITE

<u>Stratigraphy</u>	<u>Well ID</u>	<u>Hydraulic Conductivity (cm/sec)</u>
Unweathered Bedrock	LMW-07	$< 10^{-7}$
Unweathered Bedrock	LMW-11	1.2×10^{-4}
Partially Weathered Bedrock	LMW-12	5.0×10^{-4}
Alluvium	LMW-09	1.2×10^{-3}
Saprolite/Partially Weathered Bedrock	LMW-04	1.2×10^{-3}
Alluvium	LMW-01	2.0×10^{-3}
Saprolite	LMW-02	3.8×10^{-3}
Saprolite	LMW-16	5.0×10^{-3}
Alluvium	LMW-10	5.0×10^{-3}
Saprolite	LMW-03	7.1×10^{-3}
Saprolite	LMW-05	1.4×10^{-2}
Saprolite	LMW-06	3.8×10^{-2}
Saprolite/Partially Weathered Bedrock	LMW-08	4.2×10^{-2}
Saprolite/Partially Weathered Bedrock	LMW-17	8.7×10^{-2}

horizontal hydraulic-conductivity value (1.2×10^{-3} cm/sec) was about two orders of magnitude less than that measured in LMW-08 and LMW-17.

Monitoring wells completed in the saprolite had horizontal hydraulic-conductivity values that ranged from 3.8×10^{-3} cm/sec to 3.8×10^{-2} cm/sec (Table 5-1). Monitoring wells completed in the alluvium had horizontal hydraulic-conductivity values that ranged from 1.2×10^{-3} cm/sec to 5.0×10^{-3} cm/sec. In general, the saprolite's permeability was higher than that of the alluvium.

Multiple-Well Pumping Test. A multiple-well pumping test was performed downgradient of the ash pond to obtain larger scale, horizontal hydraulic-conductivity values of the saprolite. Pumping well LMW-13 was screened and sand packed over the entire saturated thickness of the saprolite. Pumping well LMW-13 and monitoring wells LMW-03, LMW-05, LMW-06, and LMW-16 were instrumented with transducers to measure water-level changes during the test.

The pumping test was run for a total of 14 hours, with a maximum drawdown of about 5 feet in LMW-13. Monitoring well LMW-16 is located 23.2 feet from the pumping well and had a maximum drawdown of 0.202 feet. Monitoring wells LMW-05 and LMW-06 are located 45.7 feet and 52.5 feet, respectively, from the pumping well and had 0.054 feet of drawdown. Monitoring wells LMW-03 and LMW-07 are located 115.9 feet and 54.8 feet, respectively, from the pumping well and had no detectable drawdown during the test.

Drawdown plots and the type curve matches for the pumping well LMW-13 and monitoring wells LMW-05, LMW-06, and LMW-16 are presented in Figure 5-10. The time has been normalized by dividing it by the square of the observation-well distances from the pumping well. In an isotropic, homogeneous, infinite aquifer with fully penetrating wells, the plots of drawdown versus normalized time should fall along one curve. The net effect of dividing the time by the square of the observation well distances is to account for the additional time required for the effects of pumping to be transmitted to greater distances. A violation of one or more of the assumptions above will cause the observation well drawdowns to not all lie along the same curve.

None of the observation-well drawdown plots coincide with the curve for other observation wells or the pumping well. This clearly indicates that the assumptions of isotropy and homogeneity are probably not valid for this site. However, the assumption of an infinite aquifer was probably not violated during the test, since the effective radius of the drawdown cone was projected to be less than about 70 feet.

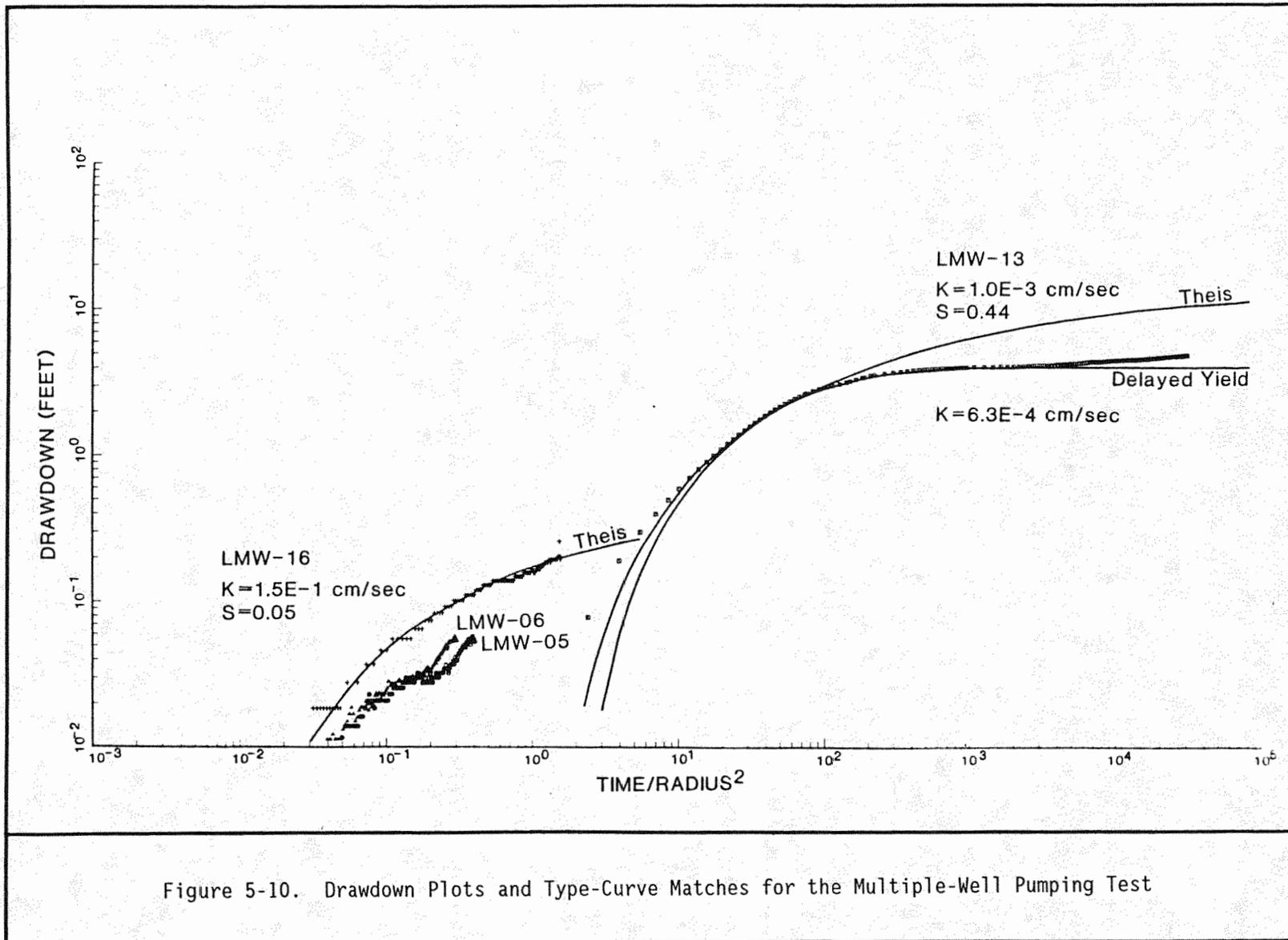


Figure 5-10. Drawdown Plots and Type-Curve Matches for the Multiple-Well Pumping Test

The assumptions of isotropy and homogeneity are rarely met in actual aquifer tests; however, analytical solutions based on these assumptions are commonly used to analyze the data. The permeability values that result from the analysis provide an estimation of the true permeabilities within approximately an order of magnitude.

A plot of drawdown versus time for the pumping well does not fit the Theis curve at early or late times (Figure 5-10). The drawdown data are above the Theis curve at early time and below the curve at late time. The early-time data approximately fit a one-to-one slope, which is indicative of wellbore storage effects, where the pumped water is predominantly coming from storage in the wellbore. This is a common phenomenon in low permeability aquifers and large diameter wells. The effects of wellbore storage are only present in early-time data and quickly disappear.

The deviation of the drawdown data below the Theis curve at late time indicates either the drawdown cone intercepting a constant-head boundary or a time delay in the vertical drainage of water as the drawdown cone propagates. Given the limited extent of the drawdown cone, it is not likely that a constant-head boundary was intercepted.

Rather, a more plausible explanation is that the lower vertical permeability of the aquifer is inhibiting the vertical drainage of water as the drawdown cone propagates laterally. At large time, after the vertical drainage of water is complete, the drawdown plot should again approximate that of a Theis curve.

The drawdown data for LMW-13 were analyzed with both a Theis method and a delayed-yield method. The Theis curve match to the drawdown data resulted in a hydraulic-conductivity value of about 1.0×10^{-3} cm/sec. The delayed-yield analysis resulted in a hydraulic-conductivity value of 6.3×10^{-4} cm/sec. The hydraulic-conductivity values resulting from the two analyses are in good agreement and differ by less than a factor of two. However, the hydraulic-conductivity value obtained from the pumping test is about an order of magnitude lower than those measured with slug tests in other wells in this area.

The Theis curve was a fairly good match to the drawdown data for LMW-16 (Figure 5-10). No effects from delayed yield were evident in the drawdown data for this well. The hydraulic-conductivity value resulting from this type-curve match was 2.0×10^{-1} cm/sec, which is about 2.3 orders of magnitude larger than that calculated for the pumping well and about 1.2 orders of magnitude larger than that calculated from the slug-test data for this well.

The large difference in hydraulic-conductivity values in LMW-16 resulting from the slug test and the pumping test data may be caused by a lower permeability skin developing on the wall of LMW-13 during the drilling of the well. One hypothesis is that the abundant mica in the saprolite would tend to be smeared and preferentially oriented perpendicular to the borehole wall as a result of the auger rotation. The orientation of the mica platelets on the borehole wall would tend to reduce the horizontal permeability by forming a low permeability skin. The effects of this low permeability skin would be more pronounced in wells being tested than in wells being used as observation wells because skin inhibits the movement of water into or out of the well. Therefore, the permeability values resulting from the slug tests will tend to be lower than the true values. Conversely, the permeability value resulting from the analysis of data from the observation wells in the pumping test will be closer to the true aquifer permeability. Therefore, the permeability value calculated from the data collected in LMW-13 (the pumping well) would tend to be less than the actual permeability of the aquifer in this area, whereas the permeability value for observation well LMW-16 will be more representative of average aquifer permeabilities in this area.

Drawdown data for observation wells LMW-05 and LMW-06 were not analyzed due to the limited amount of drawdown observed in these wells. No analytical type-curve match was possible for these wells.

Aquifer Storage Properties. Aquifer storage properties are generally made in conjunction with the estimation of permeability properties from pumping tests. Storage values obtained from slug tests are less accurate than those obtained from pumping tests because of the smaller volume of aquifer tested and borehole effects. Therefore, the slug-test data were not used to estimate storage values.

Aquifer storage values were calculated from the multiple-well pumping test conducted downstream from the ash pond. The storage value resulting from the analysis of the pumping well LMW-13 (0.44) was about one order of magnitude larger than the value resulting from the analysis of observation well LMW-16 (0.05) (Figure 5-10). This significant difference in storage values is again probably due to skin effects in the pumping well, LMW-13. Therefore, the storage value of 0.05 from observation well LMW-16 is probably closer to the actual storage value for the aquifer in this area. This relatively low storage value is not representative of an unconfined sand deposit; however, the geometry and packing of the mica platelets are effective in reducing the storage of the saprolite.

Estimates of Flux Through the Pond Bottom (Hydraulic-Gradient Method). Quantification of the flux of water through the base of the pond provides an estimate of the potential recharge to the groundwater system. One method for estimating this flux is by measuring the vertical hydraulic-conductivity and hydraulic gradient across the pond bottom sediments and calculating the flux from the Darcy equation.

Pond piezometer LMW-21, located in the primary ash pond, was driven 3.8 feet below the base of the pond (the center of the piezometer screen is at 3.3 feet). The water level in this piezometer is below that of the pond and has been declining since it was installed (Figure 5-7). The difference in hydraulic head between the pond and LMW-21 was about 3.5 feet in December 1989 and the calculated hydraulic gradient across the pond bottom sediments was 1.06. The saturated vertical hydraulic-conductivity value measured from cores of the pond bottom sediments in this area was 9.9×10^{-6} cm/sec. The resultant Darcy flux calculated from these parameter values is 1.0×10^{-5} cm/sec. Assuming an effective porosity of 30 percent, this flux value corresponds to a groundwater velocity of about 3.5×10^{-5} cm/sec (0.1 feet/day).

The rate of pond water movement through the base of the pond in other areas may deviate significantly from that calculated for the LMW-21 location. As was indicated earlier, the direction of groundwater movement is into the pond on the up-gradient side and out of the pond in the vicinity of the dam. Therefore, the hydraulic gradient is changing in magnitude and direction from the upstream end of the pond to the downstream end. In addition, the vertical hydraulic conductivity of the pond bottom sediments varies. The hydraulic conductivity of a core sample obtained from the northern side of the primary ash pond was about 1.6 orders of magnitude smaller than that measured near LMW-21.

Work by McBride and Pfannkuch (1975) demonstrated that seepage rates for lakes tend to decrease exponentially away from the shore. Assuming that this observation is also valid for man-made ponds, a decrease in flux rates would be projected for transects normal to the shoreline.

One feature that may have a significant impact on flux rates through the base of the pond is the presence of alluvial deposits in the bedrock valley beneath the pond. No data are currently available to quantify the flux of pond water in this area.

Primary Ash Pond Water Balance

An approximate water mass-balance approach was used to estimate the magnitude of various components of inflow and outflow to the primary ash pond. After initial estimates of the inflow and outflow components were calculated, a range of potential values for the components was approximated based on the projected accuracy of the calculation and pond water mass-balance restrictions.

Water inflow components to the primary ash pond include:

- Steam-plant discharges;
- Watershed runoff;
- Direct precipitation; and
- Groundwater inflow.

The steam-plant inflows to the pond were estimated from pipe discharge tables which are based on the pipe diameter, distance the stream of water travels parallel to the pipe in falling 12 inches vertically, percent cross-sectional area of pipe filled with water, and the estimated time the plant is in operation during the year. Watershed runoff was calculated by multiplying the watershed area by the estimated percent of yearly precipitation that ran off. Direct precipitation on the pond was calculated by multiplying the area of the pond by the average yearly precipitation. Groundwater inflows were calculated from the vertical hydraulic conductivity of the saprolite and measured hydraulic gradients.

Primary ash pond water outflow components include:

- Discharge to the secondary pond;
- Toe-drain discharges;
- Evaporation; and
- Groundwater recharge.

Discharges from the primary ash pond to the secondary pond were estimated from measured discharges from the secondary pond to the river (plant personnel measure this discharge on a daily basis as part of their discharge permit). Toe-drain discharges were determined directly by measuring the amount of time it took to fill a container of known volume. It was assumed that the toe-drain discharges were constant for the year. Pond-water evaporation was estimated from pan-evaporation data collected at a site about 20 miles from the plant. Two different methods were used to estimate the

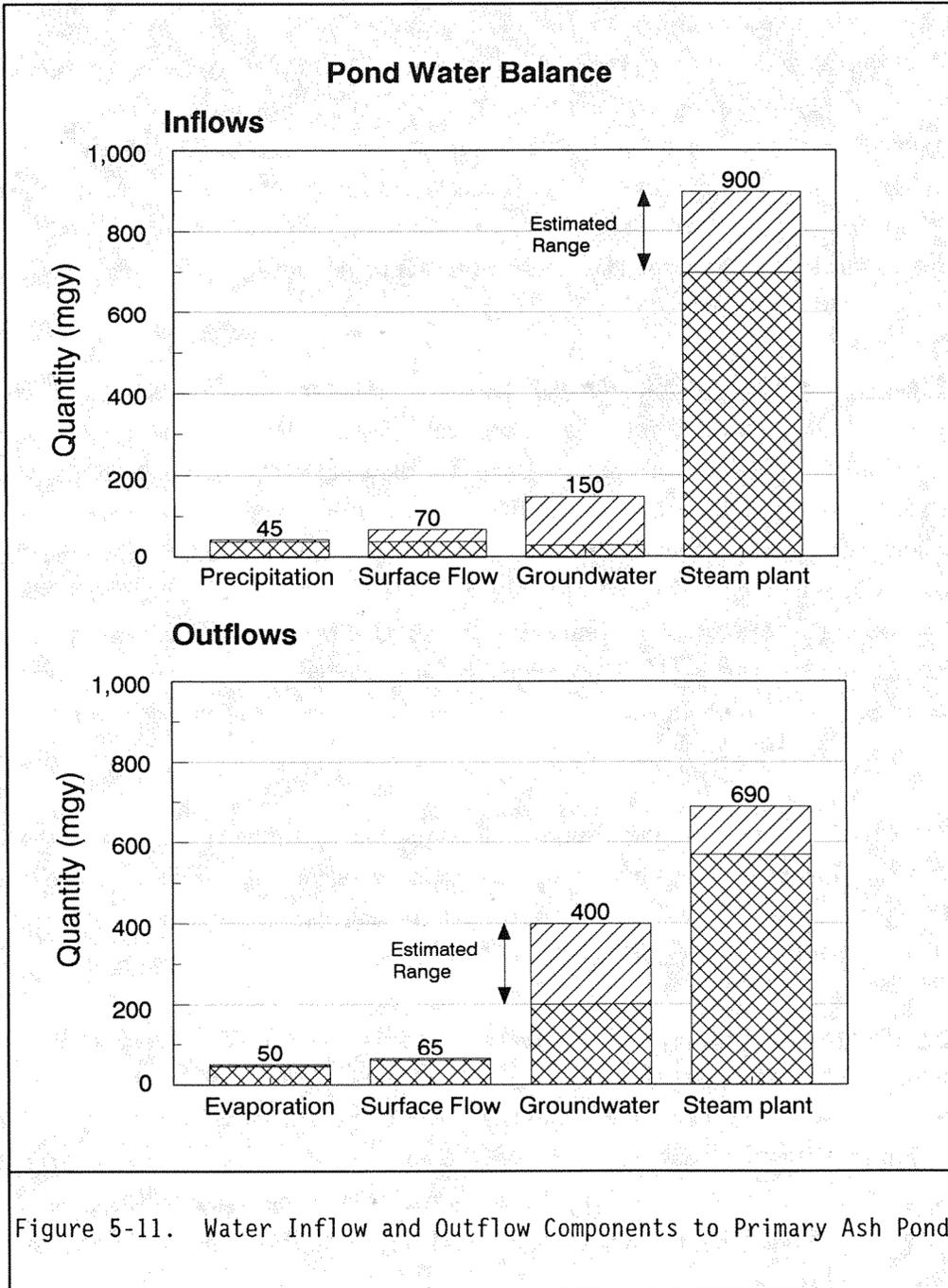
flux of water through the base of the primary ash pond to the shallow groundwater system. The first method for estimating the flux of water through the base of the pond was discussed above. The flux was estimated from a measured vertical hydraulic conductivity and hydraulic gradient of pond bottom sediments near the dam. The flux rate determined from these measurements was assumed to be constant over half the area of the pond. The second method used to estimate groundwater recharge from the pond was to integrate the horizontal flux rate calculated downgradient of the pond over the saturated thickness of deposits above the bedrock surface. This second method assumes that all shallow groundwater flow downgradient of the pond is derived from pond water.

The maximum and minimum magnitudes of the components of water inflows and outflows to the primary ash pond are shown in Figure 5-11. The single largest component of water inflow to the primary ash pond is from the steam plant. An estimated 700 to 900 million gallons per year (mgy) of water is discharged to the pond. Groundwater inflows to the pond are estimated at about 35 mgy; however, the groundwater component could be as high as 150 mgy. Contributions from surface runoff are estimated to be 40 mgy; however, this number could be as high as 70 mgy, depending on the percent of precipitation that actually runs off. Finally, direct precipitation on the ash pond is estimated to range from about 40 to 45 mgy.

The two largest components of pond-water outflows are discharges to the secondary pond and recharge to the shallow groundwater system. Discharge to the secondary pond is estimated to be between 580 and 690 mgy. The upper limit on discharge from the primary ash pond to the secondary pond is the average yearly discharge to the river from the secondary pond. Discharges to the shallow groundwater system are estimated to be between 200 and 400 mgy. Water outflows via the toe drains and through evaporation are estimated to be about 65 and 50 mgy, respectively.

The largest component of inflow to the primary ash pond is from the steam plant (about 75 percent) and the largest component of outflow is discharge to the secondary pond (about 50 percent). Discharges from the primary ash pond to the shallow groundwater system only represent about 15 to 30 percent of the total outflow.

Potential Groundwater Flow Paths and Velocities. Shallow groundwater flow paths and velocities for the L-site can be approximated from the potentiometric surface contours, total porosity, and horizontal hydraulic-conductivity values measured in monitoring wells at the site. Small changes in the values of three parameters can



result in significant changes in the calculated velocity. For example, a change in the effective porosity value from 30 percent to 10 percent results in about a factor of three increase in the groundwater velocity. Similarly, groundwater flow paths are assumed to be perpendicular to the potentiometric contour lines, which strictly speaking is only valid for an isotropic and homogeneous aquifer. Hence, this analysis will provide approximate estimates of groundwater flow paths and velocities at the L-site, given the assumptions and uncertainty in actual parameter values required for this analysis.

Groundwater Flow Paths. Approximate groundwater flow paths in the vicinity of the primary ash pond are shown in Figure 5-12. The general direction of groundwater flow is toward the pond on the upgradient side and away from the pond in the vicinity of the ash dam. Potentiometric-surface contours show a component of flow around the southeastern corner of the ash dam near LMW-11; however, water samples from monitoring well LMW-11 have approximately background concentrations of inorganic constituents. This apparent discrepancy may be due to the lack of interconnectivity between low permeability fractures in which this well is completed and fractures interconnected to the primary ash pond in this area.

The direction of groundwater flow downgradient of the ash dam is toward the river. The river appears to be the major groundwater discharge area for shallow groundwater flow. In addition, it is anticipated that the river is a discharge point for the shallow bedrock aquifer.

Groundwater Velocities. Groundwater velocity calculations are highly dependent on the effective-porosity value of the deposits; however, the effective-porosity value is one of the most difficult parameters to measure. Typically, an in situ tracer test is performed to obtain an estimate of the effective porosity. Alternatively, the total porosity can be used as an upper limit for effective porosity from which a potential range of effective-porosity values can be estimated.

At the L-site, effective porosity values of 10, 30, and 50 percent were used to calculate a range of groundwater velocities at a number of monitoring well locations (Table 5-2). For simplicity, only the velocities calculated with a 30-percent effective-porosity value will be discussed, since a 10-percent effective porosity will increase the velocity by a factor of three and a 50-percent effective porosity will reduce the velocity by about a factor of 1.6.

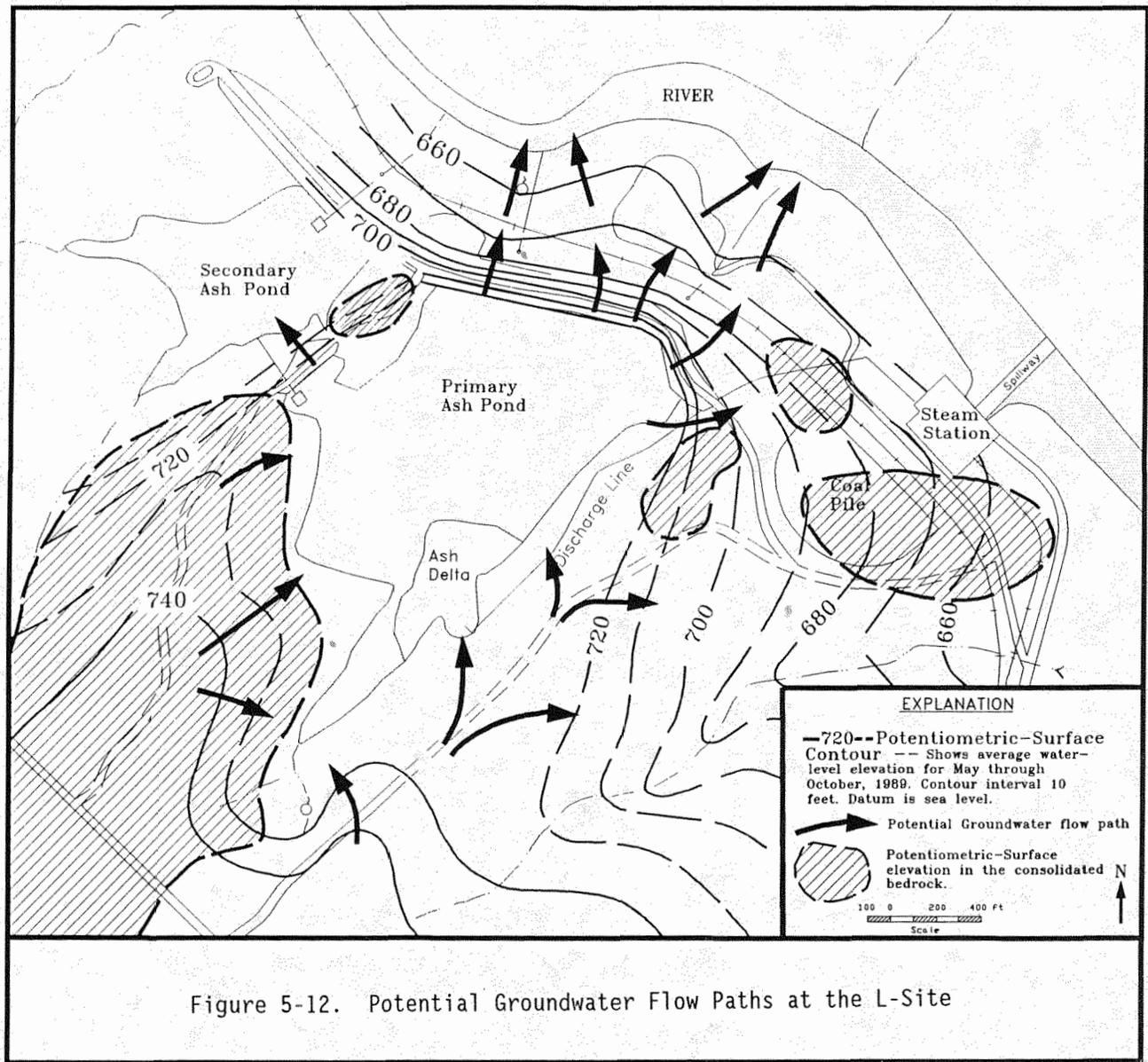


Table 5-2
CALCULATED GROUNDWATER VELOCITIES FOR THE L-SITE (ft/day)

<u>Monitoring Well</u>	<u>Hydraulic Gradient</u>	<u>Hydraulic Conductivity (ft/day)</u>	<u>Effective Porosity</u>		
			<u>0.1</u>	<u>0.3</u>	<u>0.5</u>
LMW-11	0.1	0.35	0.4	0.1	0.07
LMW-12	0.03	1.4	0.4	0.1	0.08
LMW-16	0.02	14.	3.	0.9	0.6
LMW-01	0.04	5.8	2.	0.8	0.5
LMW-09	0.07	3.5	2.	0.8	0.5
LMW-10	0.02	14.	3.	0.9	0.6
LMW-04	0.08	3.5	3.	0.9	0.6
LMW-02	0.03	11.	3.	1.	0.7
LMW-03	0.04	20.	8.	3.	2.
LMW-05	0.03	40.	10.	4.	2.
LMW-06	0.03	108.	30.	10.	6.
LMW-08	0.04	116.	50.	20.	9.
LMW-17	0.05	250.	100.	40.	30.

Estimated groundwater velocities range from about 0.1 feet/day (LMW-11 and LMW-12) to 40 feet/day (LMW-17). The lowest velocities calculated were for wells completed in the partially weathered and unweathered bed-rock deposits and the highest velocity was for a well completed at the saprolite/partially weathered bedrock contact. The mean velocity value calculated was about 0.9 feet/day below the ash dam (Figure 5-13).

In general, the large hydraulic gradient between the ash pond and river and the fairly high permeabilities of the saprolite deposits result in high velocities.

Additional wells with high velocities include LMW-06 and LMW-08, with 10 and 20 feet/day, respectively (Figure 5-13). Monitoring well LMW-06 is completed in the upper 4 feet of saturated saprolite and LMW-08 is completed at the saprolite/partially weathered bedrock contact.

In summary, the velocity calculations indicate that the area between the dam and river has much higher velocities than the area southeast of the ash pond (Figure 5-13). The higher velocities between the dam and river result from both the larger hydraulic gradients and higher permeability of deposits in this area. The area directly north of the primary ash pond appears to have higher velocities than the area to the northeast of the pond. The higher velocities north of the pond are probably due to localized changes in the geology, such as the presence of higher permeability, partially weathered quartz/feldspar veins. Groundwater flow in the saprolite and partially weathered bedrock appears to be highly dependent on the remnant structure and mineralogy of the original parent material. This is especially true for laterally extensive and partially weathered quartz/feldspar veins, which appear to correlate with high permeability zones.

Saturated Thickness

The saturated thickness of deposits above unweathered bedrock surface ranges from 0 feet near bedrock highs to greater than 100 feet in the bedrock valley under the primary ash pond (Figure 5-14). The significant thickness of saturated deposits in the bedrock valley beneath the primary ash pond is due, in part, to fill material used to construct the dam and the ash accumulation in the pond. Areas where saturated deposits above the unweathered bedrock surface are absent include the topographic high area southwest of and between the ash disposal ponds, and areas southwest of the primary ash pond.

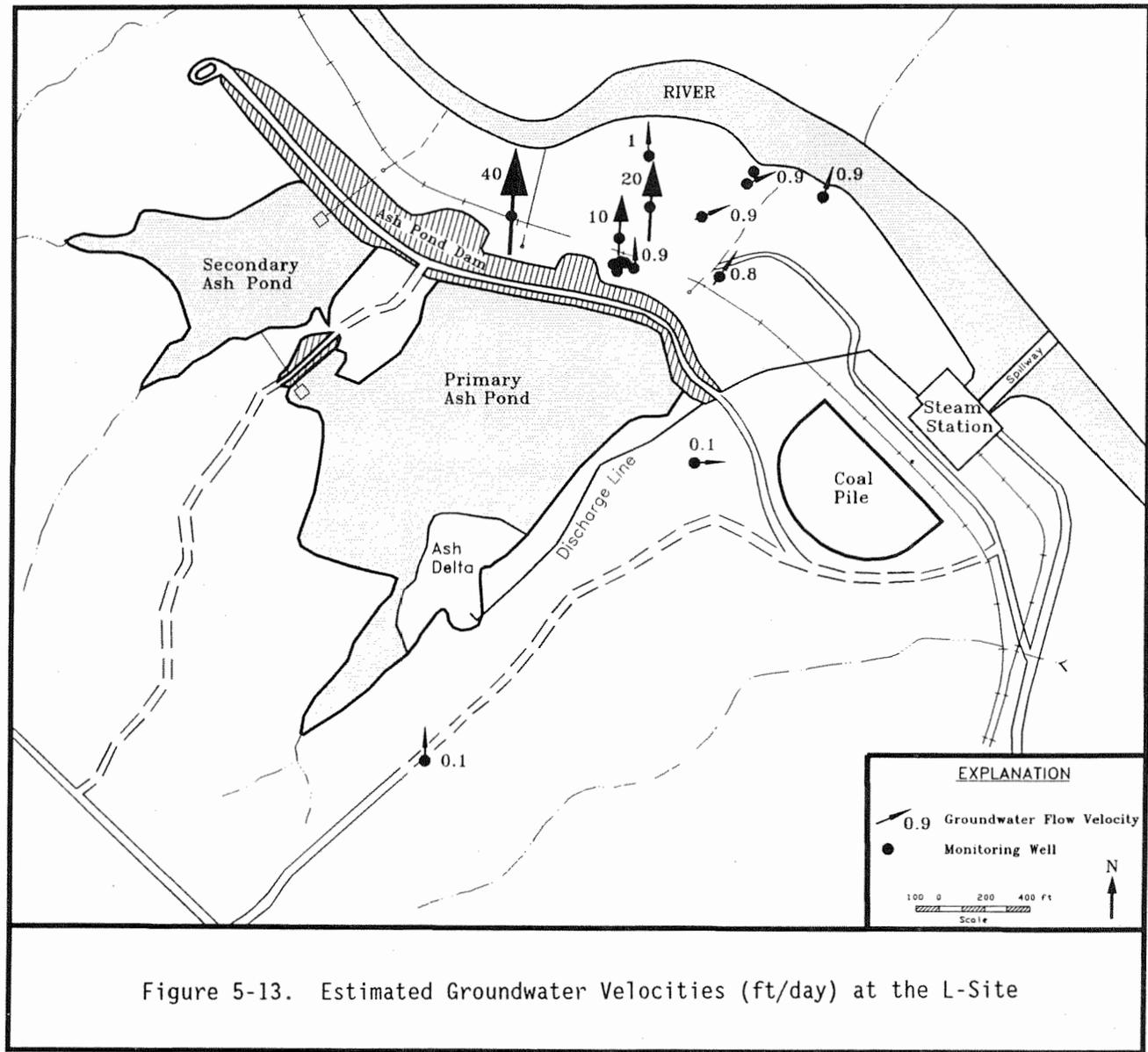
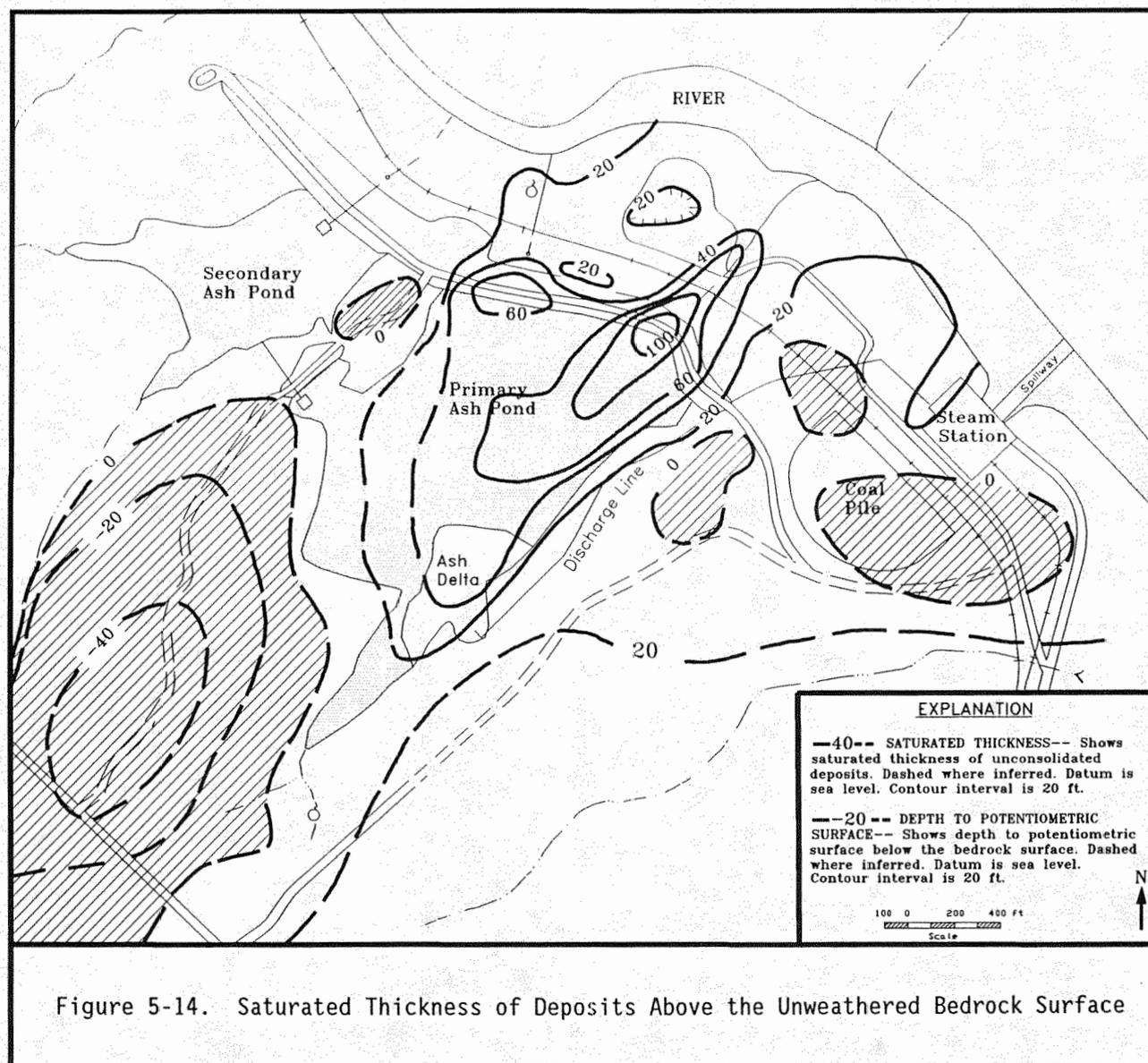


Figure 5-13. Estimated Groundwater Velocities (ft/day) at the L-Site



In addition to showing the saturated thickness of deposits above the unweathered bedrock surface, Figure 5-14 clearly shows an area southwest of the ash ponds where groundwater flow discharges from the bedrock into the unconsolidated deposits. Conversely, shallow groundwater flow may be preferentially diverted around lower permeability bedrock high areas southeast of the primary ash pond. The chemistry of groundwater samples from LMW-11 tends to indicate that shallow groundwater impacted from the ash pond is moving around the bedrock high areas immediately southeast of the primary ash ponds.

WATER QUALITY

Groundwater Quality

Tables 5-3 and 5-4 present the simple statistics for water quality results from sampling Events I and II, respectively, for 18 monitoring wells around the ash pond. Event I samples were collected in April 1989, and Event II samples were collected in July 1989. The tables present the number of samples considered (n), minimum and maximum values measured, the mean value, and the median value. The samples are presented according to where each well is located in hydrologic relation to the ash pond; i.e., upgradient or background wells, ash wells (wells which are screened in saturated substrate immediately beneath the ash pond), or downgradient wells. Figure 5-15 illustrates this designation of the monitoring wells on the L-site map.

The background groundwater quality at L-site reflects the influence of the gneissic rock prevalent in the area. The relatively low pH (median of 6.1 in background wells) and the major ion concentrations in the groundwater are typical of groundwaters associated with acidic rock (feldspar, quartz, and biotite) (Stumm and Morgan, 1970). The median value for total dissolved solids in the background wells is 100 mg/L.

The two wells placed in the ash pond are screened in the saprolite beneath the ash. Water quality in these wells shows higher levels of dissolved solids than the background groundwater (median TDS value for the ash wells is 2270 mg/L). The pH of the ash wells is acidic (median value of 4.1) with elevated iron (median of 190 mg/L) and manganese (median of 48 mg/L), which is consistent with the pyrite oxidation model proposed in Section 4.

Several other analytes are elevated in the ash wells relative to the background and downgradient wells, including Al, Be, Ca, Co, F, Mg, Mo, Ni, Sr, SO₄, and Zn. These

Table 5-3

SUMMARY STATISTICS FOR GROUNDWATER QUALITY RESULTS FROM EVENT I SAMPLES

Analyte	Detection		Background Wells				Ash Wells				Downgradient Wells						
	Limit	units	n	Minimum	Maximum	Mean	Median	n	Minimum	Maximum	Mean	Median	n	Minimum	Maximum	Mean	Median
EC (Lab)	--	umho/cm	4	58.00	118.000	93.750	99.500	2	1865	2700.000	2282.500	2282.500	12	140.0	430.000	282.083	286.500
Eh	--	mV	4	265.0	288.000	276.250	276.000	2	270.0	290.000	280.000	280.000	12	198.0	270.000	237.083	234.000
pH	--	pH units	4	5.250	6.290	5.957	6.145	2	4.010	4.170	4.090	4.090	12	4.750	7.200	5.873	5.685
Alkalinity	2.000	mg HCO ₃ /L	4	28.00	49.000	40.750	43.000	2	<2.000	< 2.000	< 2.000	< 2.000	13	5.000	90.000	37.654	33.000
TDS	5.000	mg/L	4	59.00	120.000	95.000	100.500	2	1835	2706.000	2270.500	2270.500	12	94.00	384.000	215.083	213.500
Temperature	--	C	4	16.90	24.300	19.875	19.150	2	18.50	19.300	18.900	18.900	11	17.40	21.600	19.382	19.500
Aluminum	0.200	mg/L	4	<0.200	< 0.200	< 0.200	< 0.200	2	7.600	9.000	8.300	8.300	13	<0.200	< 0.200	< 0.200	< 0.200
Ammonia	0.010	mg/L	4	0.060	0.210	0.122	0.110	2	0.100	0.100	0.100	0.100	13	0.020	0.390	0.092	0.060
Antimony	0.200	mg/L	4	<0.200	< 0.200	< 0.200	< 0.200	2	<0.200	< 0.200	< 0.200	< 0.200	13	<0.200	< 0.200	< 0.200	< 0.200
Arsenic	0.005	mg/L	4	<0.005	< 0.005	< 0.005	< 0.005	2	<0.005	< 0.005	< 0.005	< 0.005	13	<0.005	< 0.005	< 0.005	< 0.005
Barium	0.010	mg/L	4	0.013	0.067	0.029	0.018	2	0.033	0.086	0.059	0.059	13	0.017	0.310	0.105	0.069
Beryllium	0.002	mg/L	4	<0.002	< 0.002	< 0.002	< 0.002	2	0.022	0.061	0.041	0.041	13	<0.002	< 0.002	< 0.002	< 0.002
Boron	0.600	mg/L	4	<0.600	< 0.600	< 0.600	< 0.600	2	<0.600	< 0.600	< 0.600	< 0.600	13	<0.600	0.970	< 0.600	< 0.600
Bromide	0.040	mg/L	4	<0.040	0.140	0.065	0.050	2	0.050	0.090	0.070	0.070	13	0.060	0.260	0.143	0.150
Cadmium	0.005	mg/L	4	<0.005	< 0.005	< 0.005	< 0.005	2	<0.005	< 0.005	< 0.005	< 0.005	13	<0.005	< 0.005	< 0.005	< 0.005
Calcium	0.100	mg/L	4	3.607	8.818	6.613	7.014	2	208.4	400.800	304.608	304.608	12	0.802	72.144	26.419	24.649
Carbonate	7.000	mg/L	4	61.00	281.000	123.750	76.500	2	230.0	257.000	243.500	243.500	13	63.00	295.000	134.538	123.000
Chloride	0.700	mg/L	4	1.400	5.400	2.975	2.550	2	2.700	3.500	3.100	3.100	13	<0.700	8.000	4.765	5.200
Chromium	0.001	mg/L	4	<0.001	< 0.001	< 0.001	< 0.001	2	<0.001	< 0.001	< 0.001	< 0.001	13	<0.001	0.002	< 0.001	< 0.001
Cobalt	0.010	mg/L	4	<0.010	< 0.010	< 0.010	< 0.010	2	0.120	2.000	1.060	1.060	13	<0.010	0.078	0.013	< 0.010
Copper	2.000	mg/L	4	<2.000	< 2.000	< 2.000	< 2.000	2	<2.000	< 2.000	< 2.000	< 2.000	13	<2.000	< 2.000	< 2.000	< 2.000
Fluoride	0.001	mg/L	4	0.016	0.089	0.059	0.065	2	0.489	1.377	0.933	0.933	13	0.022	0.162	0.057	0.038
Iron	0.040	mg/L	4	<0.040	0.140	0.050	< 0.040	2	120.0	250.000	185.000	185.000	13	<0.040	11.000	0.989	< 0.040
Lead	0.050	mg/L	4	<0.050	< 0.050	< 0.050	< 0.050	2	<0.050	< 0.050	< 0.050	< 0.050	13	<0.050	< 0.050	< 0.050	< 0.050
Magnesium	0.100	mg/L	4	1.702	3.161	2.431	2.431	2	18.23	60.294	39.264	39.264	12	5.835	18.234	12.804	13.493
Manganese	0.010	mg/L	4	0.071	0.230	0.138	0.125	2	7.000	90.000	48.500	48.500	13	0.024	6.200	0.866	0.310
Molybdenum	0.050	mg/L	4	<0.050	< 0.050	< 0.050	< 0.050	2	0.095	0.097	0.096	0.096	13	<0.050	< 0.050	< 0.050	< 0.050
Nickel	0.020	mg/L	4	<0.020	< 0.020	< 0.020	< 0.020	2	0.027	0.660	0.344	0.344	13	<0.020	< 0.020	< 0.020	< 0.020
Nitrate/Nitrite	0.050	mg N/L	4	<0.050	2.900	1.587	1.700	2	0.100	0.210	0.155	0.155	13	0.060	1.762	0.368	0.176
Organic Carbon	0.700	mg/L	4	5.000	28.700	13.250	9.650	2	28.30	34.700	31.500	31.500	13	6.000	30.200	14.642	14.400
Phosphate, total	0.050	mg/L	4	<0.050	1.000	0.467	0.422	2	0.350	0.830	0.590	0.590	13	0.250	1.370	0.832	0.910
Potassium	0.100	mg/L	4	1.173	4.301	2.737	2.737	2	10.95	12.122	11.535	11.535	12	0.782	12.904	3.682	2.933
Selenium	0.005	mg/L	4	<0.005	< 0.005	< 0.005	< 0.005	2	<0.005	< 0.005	< 0.005	< 0.005	13	<0.005	< 0.005	< 0.005	< 0.005
Silicon	1.000	mg/L	4	9.300	18.000	14.325	15.000	2	12.00	40.000	26.000	26.000	13	4.800	18.000	12.362	12.000
Silver	0.010	mg/L	4	<0.010	< 0.010	< 0.010	< 0.010	2	<0.010	< 0.010	< 0.010	< 0.010	13	<0.010	< 0.010	< 0.010	< 0.010
Sodium	0.001	mg/L	4	3.448	8.506	6.437	6.897	2	8.276	8.736	8.506	8.506	12	4.368	14.713	9.935	9.838

Table 5-3
(Continued)

Analyte	Detection		Background Wells				Ash Wells				Downgradient Wells						
	Limit	units	n	Minimum	Maximum	Mean	Median	n	Minimum	Maximum	Mean	Median	n	Minimum	Maximum	Mean	Median
Strontium	0.003	mg/L	4	0.023	0.120	0.049	0.028	2	0.560	0.580	0.570	0.570	13	0.012	0.310	0.159	0.145
Sulfate	0.600	mg/L	4	1.400	11.000	4.025	1.850	2	1110	1630.000	1370.000	1370.000	13	45.00	171.000	103.885	114.000
Sulfide	0.150	mg/L	4	<0.150	< 0.150	< 0.150	< 0.150	2	<0.150	< 0.150	< 0.150	< 0.150	13	<0.150	0.230	< 0.150	< 0.150
Thallium	0.100	mg/L	4	<0.100	< 0.100	< 0.100	< 0.100	2	<0.100	< 0.100	< 0.100	< 0.100	13	<0.100	< 0.100	< 0.100	< 0.100
Thiourea	0.500	mg/L	4	<0.500	< 0.500	< 0.500	< 0.500	2	<0.500	< 0.500	< 0.500	< 0.500	13	<0.500	< 0.500	< 0.500	< 0.500
Vanadium	0.020	mg/L	4	<0.020	< 0.020	< 0.020	< 0.020	2	<0.020	< 0.020	< 0.020	< 0.020	13	<0.020	< 0.020	< 0.020	< 0.020
Zinc	0.020	mg/L	4	<0.020	0.039	0.025	0.026	2	0.120	0.660	0.390	0.390	13	<0.020	0.200	0.032	< 0.020

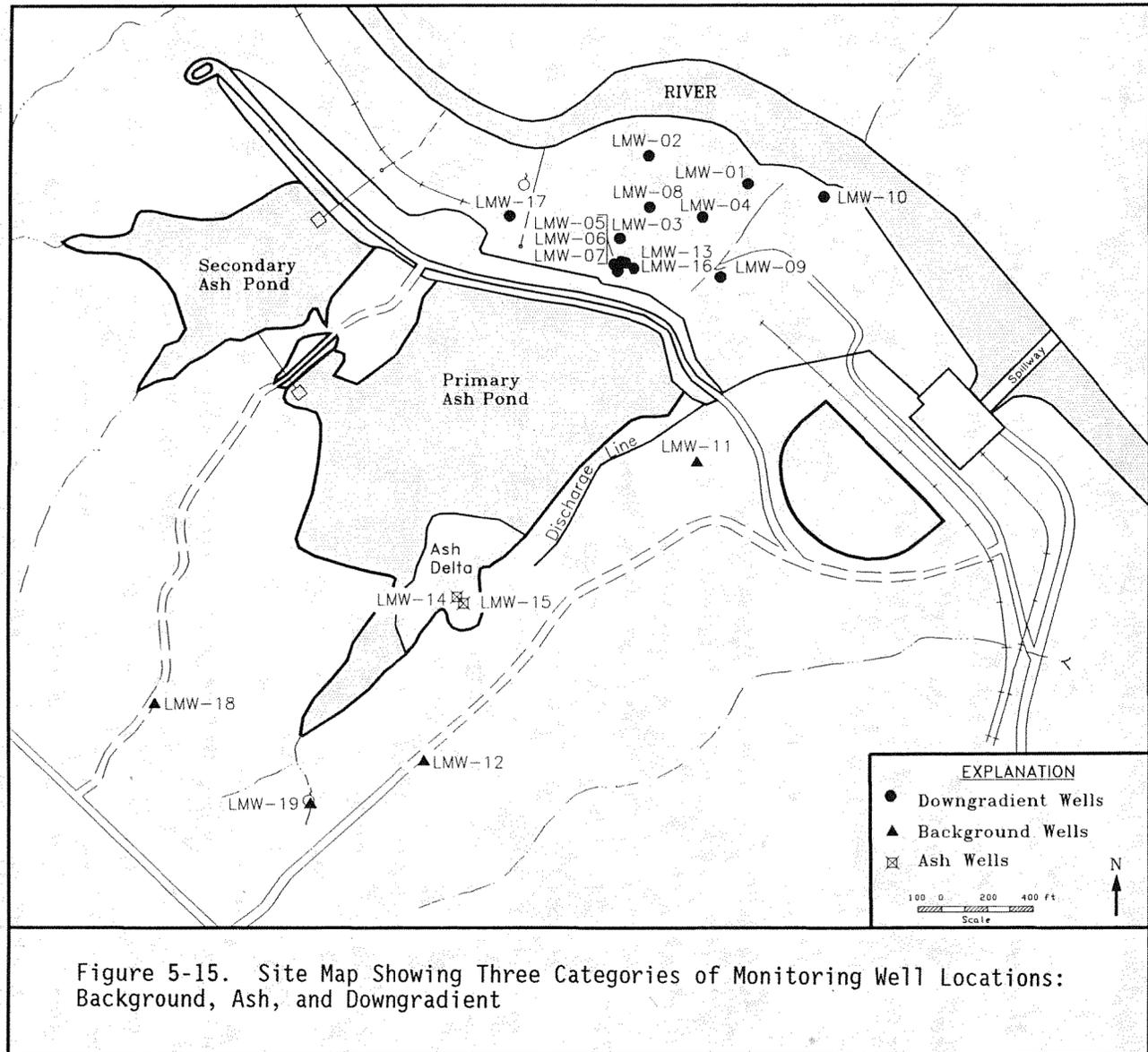
Table 5-4

SUMMARY STATISTICS FOR GROUNDWATER QUALITY RESULTS FROM EVENT II SAMPLES

Analyte	Detection		Background Wells				Ash Wells				Downgradient Wells						
	Limit	Units	n	Minimum	Maximum	Mean	Median	n	Minimum	Maximum	Mean	Median	n	Minimum	Maximum	Mean	Median
EC (Lab)	--	umho/cm	4	47.00	104.000	77.250	79.000	2	1920	3210.000	2565.000	2565.000	12	177.0	474.000	299.000	297.000
Eh	--	mV	4	144.5	519.000	341.875	352.000	2	410.0	421.000	415.500	415.500	11	91.00	572.000	367.091	364.000
pH	--	pH units	3	6.000	8.300	6.840	6.220	2	4.180	4.410	4.295	4.295	13	4.070	8.400	5.919	5.930
Alkalinity	1.000	mg HCO ₃ /L	4	25.00	53.000	37.750	36.500	2	<1.000	< 1.000	< 1.000	< 1.000	13	3.000	96.000	41.077	29.000
TDS	8.000	mg/L	4	<5.000	120.000	73.750	85.500	2	1660	2910.000	2285.000	2285.000	13	<8.000	371.000	206.462	219.000
Temperature	--	C	3	14.70	18.500	16.767	17.100	2	19.70	19.800	19.750	19.750	13	16.40	20.600	18.200	18.100
Aluminum	0.200	mg/L	4	<0.200	< 0.200	< 0.200	< 0.200	2	9.100	12.000	10.550	10.550	12	<0.200	< 0.200	< 0.200	< 0.200
Ammonia	0.010	mg/L	4	<0.010	0.100	0.046	0.040	2	0.040	0.350	0.195	0.195	12	<0.010	0.310	0.107	0.060
Antimony	0.200	mg/L	4	<0.200	< 0.200	< 0.200	< 0.200	2	<0.200	< 0.200	< 0.200	< 0.200	12	<0.200	< 0.200	< 0.200	< 0.200
Arsenic	0.002	mg/L	4	<0.002	< 0.002	< 0.002	< 0.002	2	<0.002	< 0.002	< 0.002	< 0.002	12	<0.002	0.005	< 0.002	< 0.002
Barium	0.010	mg/L	4	<0.010	0.120	0.049	0.034	2	<0.010	0.020	0.015	0.015	13	0.021	0.310	0.095	0.068
Beryllium	0.002	mg/L	4	<0.002	< 0.002	< 0.002	< 0.002	2	0.029	0.067	0.048	0.048	12	<0.002	< 0.002	< 0.002	< 0.002
Boron	0.600	mg/L	4	<0.600	< 0.600	< 0.600	< 0.600	2	<0.600	< 0.600	< 0.600	< 0.600	12	<0.600	0.900	< 0.600	< 0.600
Bromide	0.080	mg/L	4	<0.040	< 0.040	< 0.040	< 0.040	2	<0.040	< 0.040	< 0.040	< 0.040	12	<0.080	0.203	0.128	0.131
Cadmium	0.005	mg/L	4	<0.005	< 0.005	< 0.005	< 0.005	2	<0.005	< 0.005	< 0.005	< 0.005	12	<0.005	< 0.005	< 0.005	< 0.005
Calcium	0.100	mg/L	4	2.700	9.800	5.538	4.825	2	200.0	470.000	335.000	335.000	12	0.780	60.000	26.232	24.500
Carbonate	6.000	mg/L	4	69.00	133.500	88.125	75.000	2	256.0	310.000	283.000	283.000	12	53.00	321.000	121.000	103.000
Chloride	0.700	mg/L	4	2.000	4.400	2.931	2.662	2	2.300	2.400	2.350	2.350	12	2.200	9.400	5.067	4.550
Chromium	0.001	mg/L	4	<0.001	< 0.001	< 0.001	< 0.001	2	<0.001	< 0.001	< 0.001	< 0.001	12	<0.001	0.002	< 0.001	< 0.001
Cobalt	0.010	mg/L	4	<0.010	< 0.010	< 0.010	< 0.010	2	0.210	2.500	1.355	1.355	12	<0.010	0.076	0.011	< 0.010
Copper	0.001	mg/L	4	<0.001	< 0.001	< 0.001	< 0.001	2	0.004	0.005	0.005	0.005	12	<0.001	0.007	0.002	< 0.001
Fluoride	0.010	mg/L	4	0.015	0.125	0.071	0.072	2	0.520	1.700	1.110	1.110	12	0.020	0.300	0.080	0.042
Iron	0.040	mg/L	4	<0.040	< 0.040	< 0.040	< 0.040	2	110.0	280.000	195.000	195.000	12	<0.040	15.000	1.385	< 0.040
Lead	0.002	mg/L	4	<0.002	0.010	0.003	< 0.002	2	<0.002	< 0.002	< 0.002	< 0.002	12	<0.002	< 0.002	< 0.002	< 0.002
Magnesium	0.100	mg/L	4	1.200	3.200	2.063	1.925	2	19.00	70.000	44.500	44.500	12	2.000	22.000	10.858	9.400
Manganese	0.010	mg/L	4	<0.010	0.160	0.053	0.024	2	10.00	120.000	65.000	65.000	12	<0.010	4.000	0.839	0.320
Molybdenum	0.050	mg/L	4	<0.050	< 0.050	< 0.050	< 0.050	2	0.100	0.100	0.100	0.100	12	<0.050	< 0.050	< 0.050	< 0.050
Nickel	0.020	mg/L	4	<0.020	< 0.020	< 0.020	< 0.020	2	0.050	0.770	0.410	0.410	12	<0.020	< 0.020	< 0.020	< 0.020
Nitrate/Nitrite	0.050	mgN/L	4	0.590	1.473	0.828	0.625	2	<0.050	0.230	0.128	0.128	12	<0.050	0.910	0.182	0.120
Organic Carbon	0.800	mg/L	4	<0.800	4.300	2.100	1.650	1	<0.800	< 0.800	< 0.800	< 0.800	12	1.600	12.600	5.883	6.300
Phosphate, total	0.050	mg/L	4	<0.050	0.900	0.312	0.161	2	<0.050	0.050	< 0.050	< 0.050	12	<0.050	2.400	0.333	< 0.050
Potassium	0.100	mg/L	4	1.500	4.000	2.400	2.050	2	12.00	16.000	14.000	14.000	12	0.660	12.000	3.897	3.650
Selenium	0.002	mg/L	4	<0.002	< 0.002	< 0.002	< 0.002	2	<0.002	< 0.002	< 0.002	< 0.002	12	<0.002	< 0.002	< 0.002	< 0.002
Silicon	1.000	mg/L	4	7.000	16.000	11.750	12.000	2	14.00	38.000	26.000	26.000	12	4.000	19.000	11.217	11.000
Silver	0.030	mg/L	4	<0.030	< 0.030	< 0.030	< 0.030	2	<0.030	< 0.030	< 0.030	< 0.030	12	<0.030	< 0.030	< 0.030	< 0.030
Sodium	0.001	mg/L	4	4.775	16.000	8.394	6.400	2	12.00	12.000	12.000	12.000	12	4.700	21.000	12.225	12.000

Table 5-4
(Continued)

Analyte	Detection		Background Wells					Ash Wells					Downgradient Wells				
	Limit	Units	n	Minimum	Maximum	Mean	Median	n	Minimum	Maximum	Mean	Median	n	Minimum	Maximum	Mean	Median
Strontium	0.003	mg/L	4	0.020	0.055	0.034	0.029	2	0.610	0.640	0.625	0.625	12	0.010	0.280	0.159	0.165
Sulfate	0.500	mg/L	4	0.975	6.400	3.569	3.450	2	1200	2060.000	1630.000	1630.000	12	45.00	171.000	103.000	111.000
Sulfide	0.334	mg/L	4	<0.330	< 0.330	< 0.330	< 0.330	2	<0.324	< 0.324	< 0.324	< 0.324	12	<0.334	< 0.334	< 0.334	< 0.334
Sulfite	2.000	mg/L	3	<2.000	< 2.000	< 2.000	< 2.000	0	< .	< .	< .	< .	10	<2.000	< 2.000	< 2.000	< 2.000
Thallium	0.100	mg/L	4	<0.100	< 0.100	< 0.100	< 0.100	2	<0.100	< 0.100	< 0.100	< 0.100	12	<0.100	< 0.100	< 0.100	< 0.100
Thiosulfate	0.200	mg/L	4	<0.300	0.500	< 0.300	< 0.300	2	0.500	0.600	0.550	0.550	11	<0.200	0.800	0.405	0.400
Thiourea	0.500	mg/L	4	<0.500	< 0.500	< 0.500	< 0.500	2	<0.500	< 0.500	< 0.500	< 0.500	12	<0.500	< 0.500	< 0.500	< 0.500
Vanadium	0.020	mg/L	4	<0.020	< 0.020	< 0.020	< 0.020	2	<0.020	< 0.020	< 0.020	< 0.020	12	<0.020	< 0.020	< 0.020	< 0.020
Zinc	0.020	mg/L	4	<0.020	0.130	0.046	0.021	2	0.100	0.790	0.445	0.445	12	<0.020	0.400	0.085	< 0.020



elements have been shown to be associated with coal ash leachates (Rai, et al., 1987).

Of those constituents found in elevated levels in ash wells relative to background wells, only a handful were significantly elevated in wells downgradient of the ash pond. Calcium, filterable residue (total dissolved solids), magnesium, strontium, and sulfate concentrations were significantly higher in the downgradient wells than in background groundwater samples.

Table 5-5 summarizes the groupings of groundwater concentrations according to the well locations just discussed. The summary table was constructed using the Duncan's multiple-range test to detect differences in groundwater quality between the three well classes. The assumptions underlying the Duncan multiple-range test are, essentially, those of the analysis of variance. These are that the data from each group must be normally distributed and have equal variances. Since these assumptions did not always apply to the L-site data, a non-parametric Duncan's test was performed. In non-parametric tests no assumption is made about the underlying distribution of the data. In short, these methods involve ranking or categorizing the data and performing standard statistical tests on these ranks or categories. Thus, in performing a non-parametric Duncan's multiple range test all of the observations for each analyte were put into a single array, in increasing order. Ranks were then assigned to the values in the array. The Duncan's multiple-range test was then performed on the ranks of the data.¹

The Duncan's test was performed at a significance level of $\alpha = 0.05$. This means that the probability of incorrectly concluding well group means are different, when in fact they're not, is 0.05 or 5%. In other words, the probability of making an error of this type is small (5%) and there is a 95% chance that the correct decision will be made. It should be noted that non-parametric tests have a greater risk of a Type II error. That is, there is a greater probability of concluding there is no difference between the means, when in fact there is.

Co-management Effects. Table 5-6 presents estimated solids loadings of the streams routed to the ash pond. These loadings were calculated from available plant data,

¹The Duncan's multiple-range test was not done to determine well groupings. In performing the Duncan's test we first assigned each of the wells to one of the three well groups (i.e., ash wells, downgradient wells, and background wells). The Duncan's test then determines the differences between these three groups by analyte. The cluster analysis was performed to determine how the wells grouped (i.e., to confirm our choice of well groups).

Table 5-5

COMPARISON OF EVENT II GROUNDWATER RESULTS USING
A NON-PARAMETRIC DUNCAN'S MULTIPLE RANGE TEST^a

Group	Description	Analytes
A	Below Detection Limits for All Wells	Antimony, Arsenic, Boron, Cadmium, Copper, Lead, Selenium, Silver, Thallium, Thiourea, Vanadium
B	Ash Wells=Downgradient Wells=Background Wells	Ammonia, Barium, Bromide, Chloride, Chromium, Nitrate/Nitrite, Phosphate - total, Potassium, Silicon, Sodium, Sulfide by IT, Temperature
C	Ash Wells>Downgradient Wells=Background Wells	Aluminum, Beryllium, Cobalt, Fluoride, Iron, Manganese, Molybdenum, Nickel
D	Ash Wells>Downgradient Wells>Background Wells	Calcium, EC (Lab), Filterable Residue, Magnesium, Strontium, Sulfate
E	Ash Wells>Downgradient Wells (Background Wells are intermediate but not significantly different from either group)	Zinc
F	Ash Wells>Background Wells (Downgradient Wells are intermediate but not significantly different from either group)	Carbonate, Organic Carbon
G	Ash Wells=Background Wells>Downgradient Wells	Eh
H	Background Wells=Downgradient Wells>Ash Wells	Alkalinity, pH

^a $\alpha = 0.05$

Table 5-6

ESTIMATED SOLIDS LOADINGS OF STREAMS ROUTED TO ASH POND

<u>Stream</u>	<u>Suspended Solids (mg/L)</u>	<u>Dissolved Solids (mg/L)</u>	<u>Stream Volume (m³/yr)</u>	<u>Suspended Solids Loading (kg/yr)</u>	<u>Dissolved Solids Loading (kg/yr)</u>
Fly Ash	37,000 ^a	60 ^b	600,000 ^c	22 million	36,000
Bottom Ash	37,000 ^a	60 ^b	82,000 ^c	3 million	5,000
Boiler Blowdown	-- ^d	15 ^e	80,000 ^f	-- ^d	100
Boiler Chemical Cleaning Waste	-- ^g	18,000 ^e	340 ^h	-- ^g	6,000
Coal Pile Runoff	-- ^g	660 ^e	25,000 ^f	-- ^g	15,000
Cooling Tower Basin Sludge	Solids	-- ^d	15 ^f	18,000	--
Demineralizer Regenerant	44 ^e	6,060 ^e	5,000 ^h	200	30,000
Fireside Wash	-- ^g	3,500 ^e	5,000 ^f	-- ^g	17,000
Floor and Yard Drains	-- ^g	-- ^g	16,000 ^f	-- ^g	-- ^g
Makeup Water Treatment Sludge	Solids	-- ^d	1 ^h	1,400	-- ^d
Pyrites	Solids	-- ^d	23 ^h	27,000	-- ^d
Sump Drain	-- ^g	-- ^g	-- ^g	-- ^g	-- ^g

Conversion Factors:

$$\frac{\text{cubic meter}}{\text{year}} \times 263.8 = \frac{\text{gallons}}{\text{year}}$$

$$\frac{\text{kilogram}}{\text{year}} \times 1.1 \times 10^{-3} = \frac{\text{tons}}{\text{year}}$$

Table 5-6
(Continued)

Footnotes

^aAssumes 3.7% solids in ash sluice stream (data from similar plant in EPRI EA-3610).

^bBased on filterable residue (total dissolved solids) measured in ash pond water.

^cSluice volume had to be back-calculated from the plant data on suspended solids loading, and the assumed 3.7% suspended solids in the sluice stream.

^dConsidered insignificant.

^eSection 2 of EPRI CS-5281 (Holcombe, et al., 1987).

^fEstimates for 400 MW plant from EPRI CS-5281 (Holcombe, et al., 1987).

^gUnknown.

^hPlant data.

The values in the table without footnotes are calculated.

literature data for similar streams from other plants, and measurements made on this project. Because of the variety of sources for the data, and the very limited data available, results in Table 5-6 should be considered approximations only. However, the results do indicate the comparative contributions of the coal combustion by-products and the various high- and low-volume waste streams to the ash pond.

As expected, the fly ash and bottom ash streams contribute the large majority of solid material to the pond. These two streams also contribute most of the dissolved material as ash leachate. The other significant contributors of total dissolved solids are the demineralizer regenerant and boiler cleaning waste streams (the volume estimated for the fireside washwater has a large uncertainty and is probably on the high end).

The major constituents found in the groundwaters beneath the ash pond are commonly found in ash leachates: SO_4 , Ca, CO_3 , Mg, Si, K, and Na. Exceptions are Fe and Mn, which are major species in the groundwaters beneath the ash, but are not typically seen at these levels in ash leachates. It is suspected that these two elements are a result of the oxidation and leaching of pyrite (FeS_2) present in the ash and soils in the pond (this hypothesis was further discussed in Section 4).

There is little or no evidence that low volume waste streams have had a distinguishable effect on groundwater concentrations relative to the ash sluice water. Demineralizer regenerant contains the salts removed from the demineralizer columns used to treat water for the boilers, and the salts from the regenerating acids and bases, sulfuric acid and caustic. Constituents contributed by the demineralizer regenerant stream would include sulfate, sodium, calcium, magnesium and other constituents of the local water supply, in other words, the same set of constituents contributed by the ash. It would therefore be difficult to distinguish between constituents in the groundwater contributed by the ash and by demineralizer regenerant.

Boiler cleaning wastes, on the other hand, contain quite a different set of constituents. In particular, the chemical cleaning wastes from the waterside cleaning of the boiler tubes contain elevated concentrations of several heavy metals, notably copper and iron. Table 5-7 presents the approximate composition of boiler chemical cleaning wastes routed to the L-site ash pond. Each boiler at L-steam station is cleaned approximately every three years, resulting in roughly 90,000 gallons of cleaning waste and associated rinses annually.

Table 5-7
 CALCULATED CHEMICAL COMPOSITION OF
 BOILER CHEMICAL CLEANING WASTE AT L-STEAM STATION

<u>Constituent</u>	<u>Concentration, mg/L^a</u>
Ammonium	4,400
Bromate	700
Chloride	18,000
Citric Acid	400
Copper	200
Fluoride	1,200
Iron	1,500
Nickel	50
Thiourea	2,500
Zinc	25

^aEstimated from pounds of deposit removed in 2-stage cleaning, composition of cleaning solution, and approximate total cleaning and rinse-water volume of 90,000 gallons (6/1/88 cleaning).

Referring back to Table 5-5, comparison of groundwater results using the Duncan's multiple-range test, only four of the cleaning waste constituents listed in Table 5-7 appear elevated in groundwaters beneath the ash pond: F, Fe, Ni, and Zn. However, these four elements also could be a result of pyrite oxidation or contributed by the ash leachate. The remaining boiler cleaning waste constituents do not show a statistically significant difference in concentration between ash pond wells and background or downgradient wells.

Temporal and Spatial Trends. There is little difference in mean analyte concentrations between samples collected in April and July of 1989 (Table 5-3 and 5-4, respectively). Plant personnel have noticed pH differences in the ash pond water with the seasons, largely as a result of algal blooms. However, no significant changes in groundwater quality were detected during the short (three month) time between sampling events. As can be seen in Tables 5-3 and 5-4, a few analytes have different mean values in the two events, including Eh, Ba, organic carbon, and phosphate. Eh, organic carbon, and phosphate were all higher in Event II than Event I. The reason for this difference is not clear. It could be due to increased biological activity with warmer temperatures in the spring and summer months. In any case, the influence on groundwater quality is negligible. The differences in barium concentrations between the two sampling events are not statistically significant given the variability of barium concentrations within each sampling event.

The general spatial trend of dissolved constituents in the groundwater can be described as increasing concentrations of ash-derived constituents immediately beneath the ash pond, with decreasing concentrations downgradient of the pond.

This trend is exemplified in Figures 5-16 and 5-17, where contour plots of calcium and sulfate in downgradient groundwaters are shown. The convergent algorithm used, known as multi-snap, is a very stable algorithm for modeling random or clustered data. The concentration values for the control points (well locations) are snapped to a grid node. This is done using a distance-weighting technique such that control points closer to the node have a larger effect on the outcome of the node concentration values. The CPS® program performs multiple iterations of snapping control points to grid nodes. During each iteration, the goodness-of-fit between the grid and data is monitored to determine if more iterations are necessary.

Calcium and sulfate concentrations are a result of ash leaching. Both constituents decrease rapidly away from the ash pond. Calcium decreases from a median of 304 mg/L in groundwaters beneath the ash pond to a median of 25 mg/L in downgradient

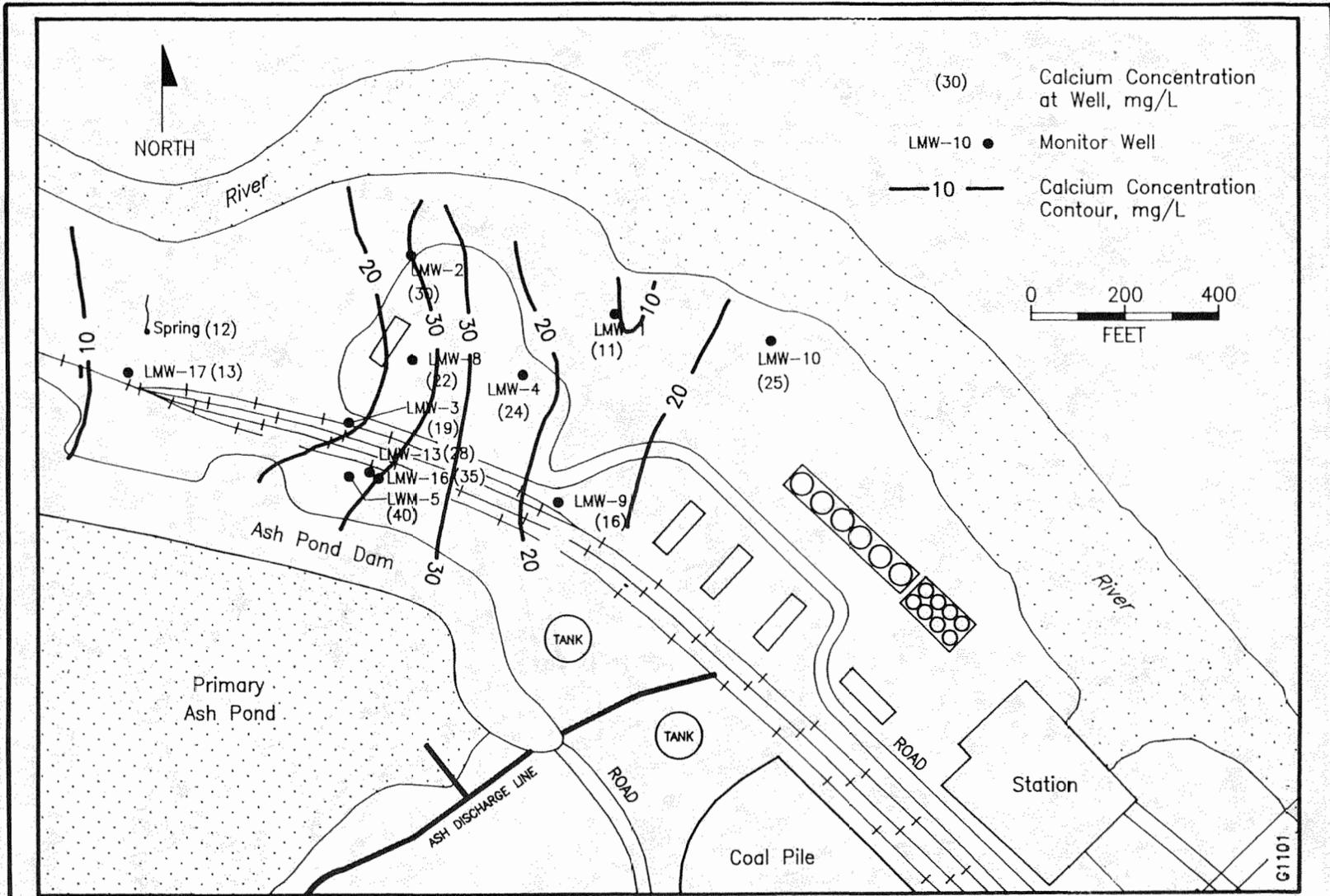


Figure 5-16. Isoconcentration contours of calcium in groundwater downgradient of ash pond (Event II). River calcium concentration is 2 mg/L. River not used as concentration boundary.

G1101

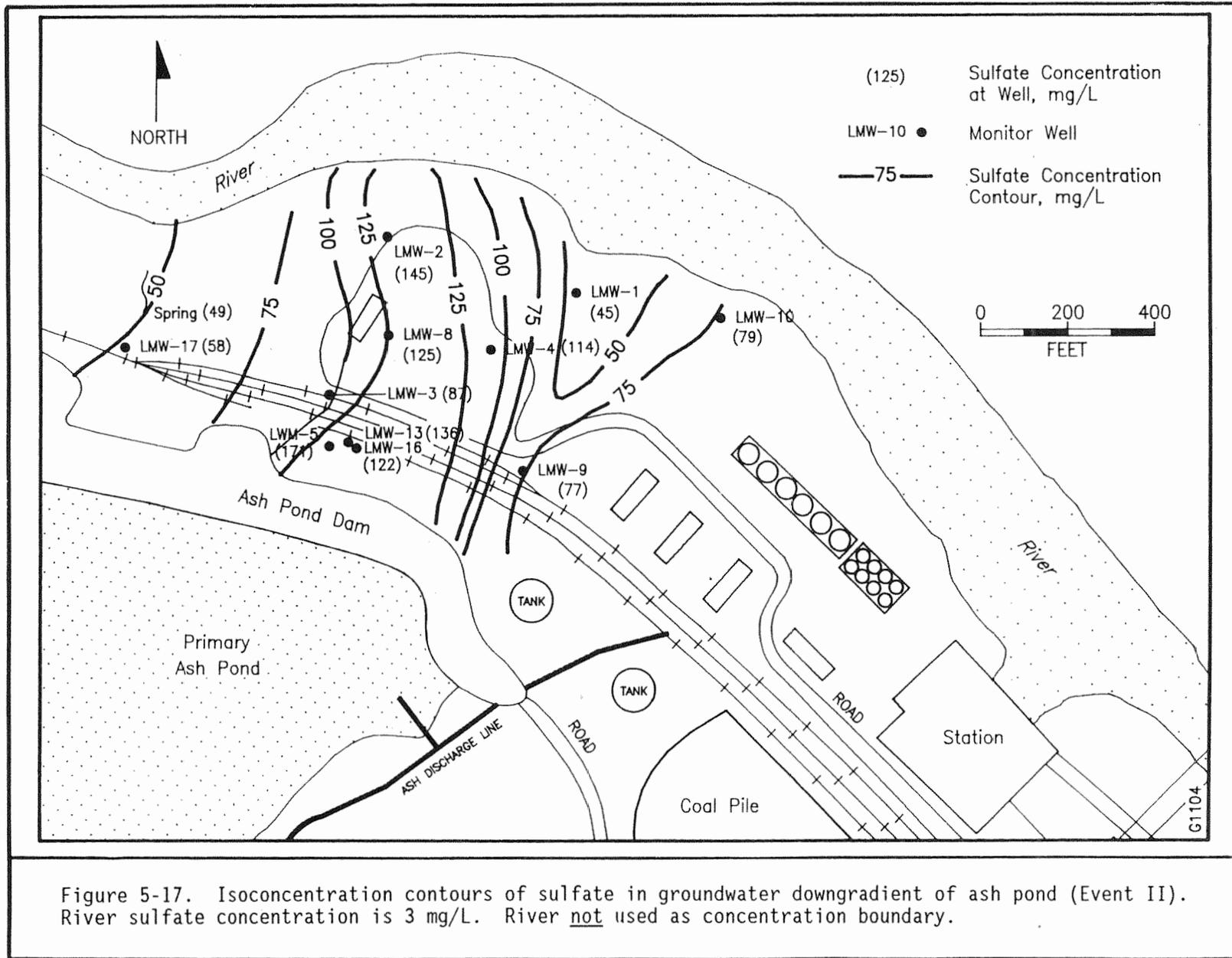


Figure 5-17. Isoconcentration contours of sulfate in groundwater downgradient of ash pond (Event II). River sulfate concentration is 3 mg/L. River not used as concentration boundary.

wells. Sulfate decreases from a median of 1370 to 120 mg/L away from the pond. In the downgradient wells the trend in concentrations is less apparent. The contours show a fairly constant concentration gradient from the pond toward the river, with decreasing concentrations laterally away from the center line. The range in calcium concentrations in the 12 downgradient wells is 0.8 to 72 mg/L and the sulfate range is 45 to 170 mg/L--a relatively narrow concentration range.

Table 5-8 presents the spatial trends in groundwater concentrations along a transect of wells running parallel to the direction of groundwater flow (see Figure 5-18 for location of transect). Again, the general trend is shown of increased concentrations immediately beneath the pond and decreasing concentrations downgradient of the pond.

In general, the assessment was designed to evaluate effects on the shallow groundwaters: essentially a two-dimensional study. However, one nest of wells, LMW-05, 06 and 07, was installed in the same location at different depths to evaluate the vertical concentration gradients in the groundwater at one location downgradient of the ash pond. Well LMW-06 was screened at about 2 feet below the potentiometric surface, LMW-05 at about 17 feet, and LMW-07 at about 33 feet.

Figures 5-19 and 5-20 present the concentrations of select analytes in the vertical nest of wells. There are obvious differences in water quality in these three wells, as evidenced by the increasing pH, alkalinity, and total dissolved solids with depth. These differences are a result of the different influence of the ash pond at each screened level. At this location between the ash pond and the river, there is an upward flow component to the groundwater. As a result, the wells screened at the deeper level (LMW-07 and LMW-05) reflect more directly the influence of the ash pond. The shallower well, LMW-06, has lower concentrations of dissolved solids, possibly as a result of dilution from recharge.

Pond and Surface Water Quality

The majority of the water from the L-site ash ponds leaves through surface discharges. The major discharge is from the secondary ash pond to the river. Table 5-9 presents the results of water quality analyses on the secondary pond discharge, the river quality immediately upstream of the discharge, and the river quality downstream of the discharge and ash ponds. The ash pond water quality is distinct from the river water in its elevated levels of ash-derived constituents such as calcium, magnesium, sulfate and other dissolved solids. No measurable effects of the pond discharge on the river water chemistry are apparent from these analyses. Quantifi-

Table 5-8
GROUNDWATER CONCENTRATION ALONG UPGRADIENT-DOWNGRADIENT TRANSECT
(Event II Values, mg/L)

	<u>LMW-19</u>	<u>LMW-15</u>	<u>LMW-16</u>	<u>LMW-04</u>	<u>LMW-01</u>
Ca	3.6	210	35	24	11
SO ₄	1.4	1110	120	110	45
Sr	0.023	0.56	0.20	0.14	0.074
TDS	99	1800	210	230	120
Screen depth, ft.		17	23	37	22

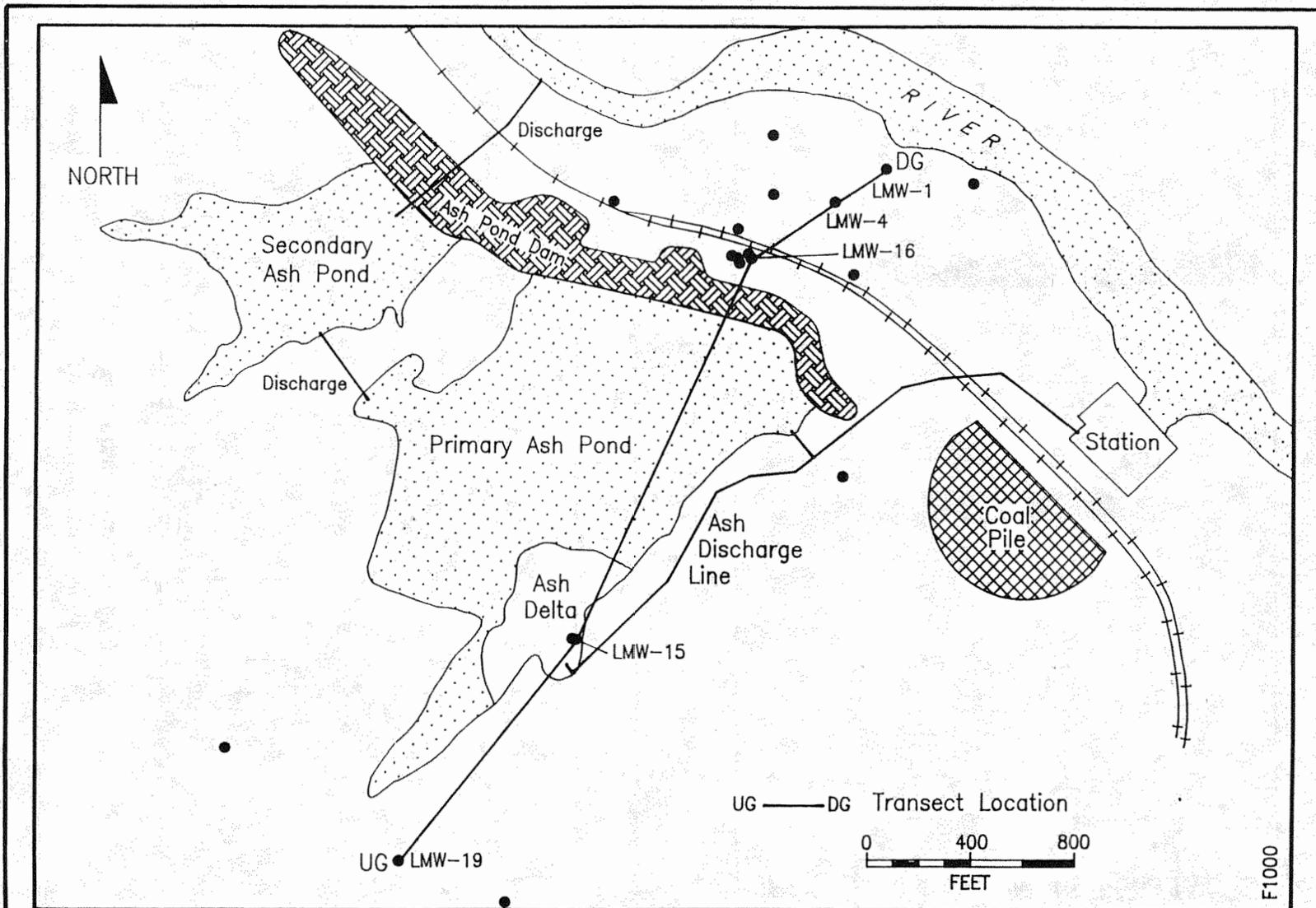


Figure 5-18. L-Site Map Showing Well Transect Used in Table 5-8

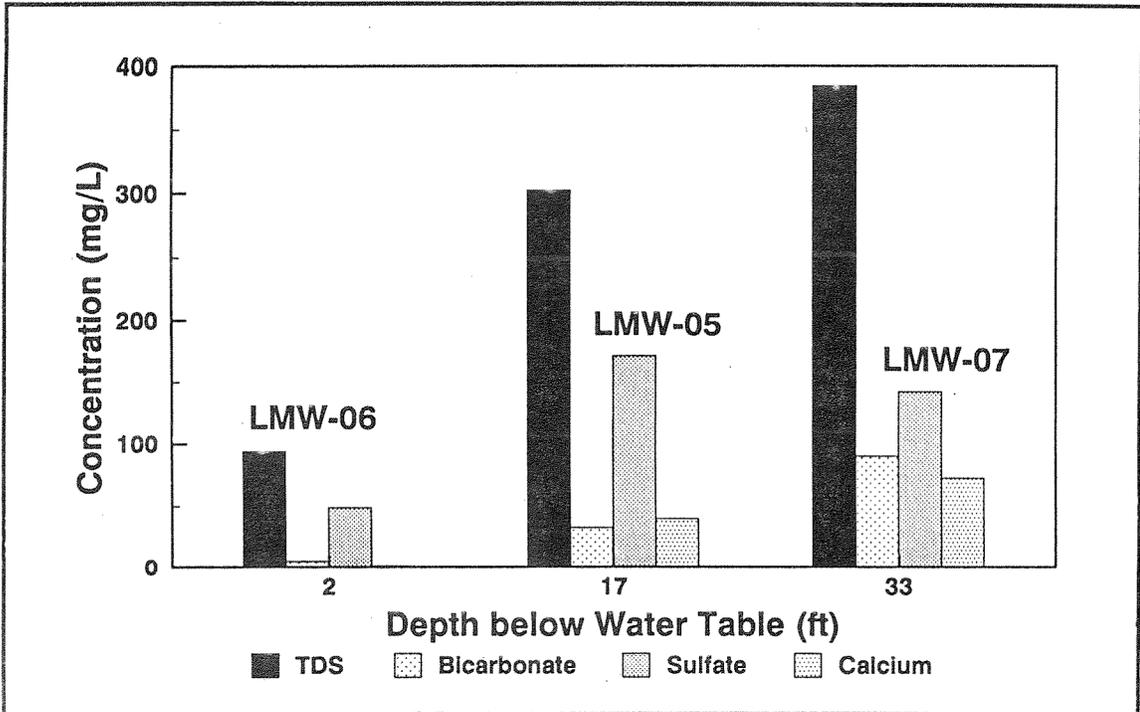


Figure 5-19. L-Site Event II, Groundwater Concentrations at Depth

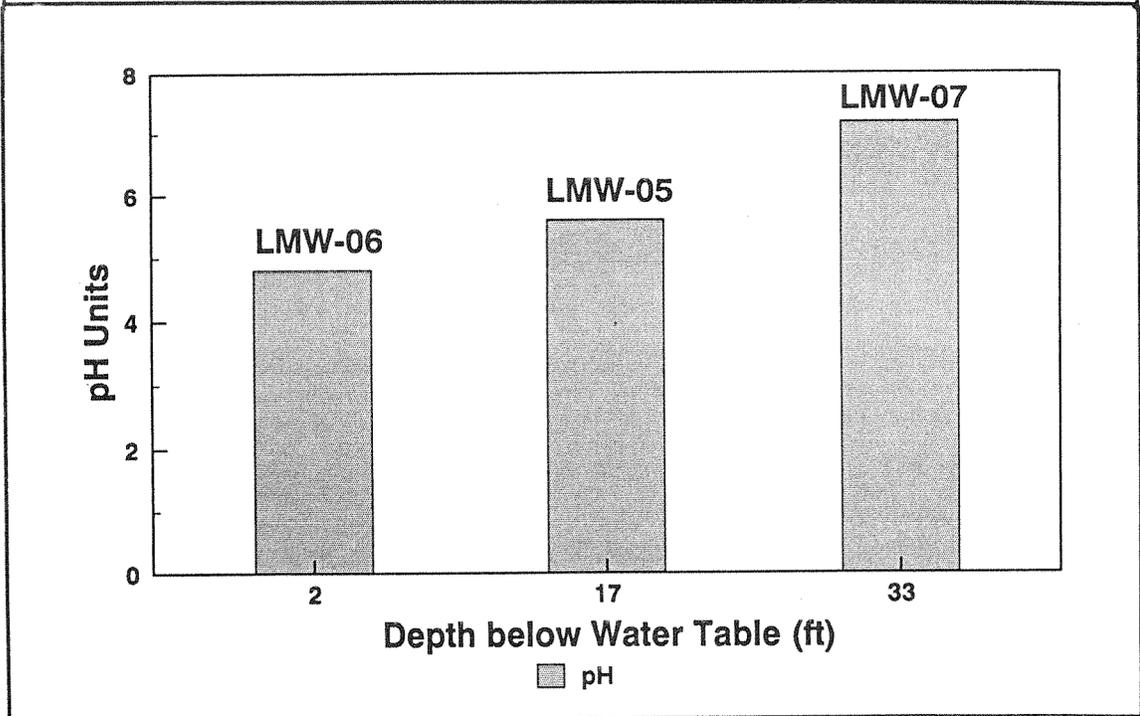


Figure 5-20. L-Site Event II, Groundwater pH at Depth

Table 5-9

CHEMICAL CHARACTERISTICS OF RIVER AND POND WATERS FROM L-SITE, EVENT II

Parameter	Units	Upstream River 26-July-89	Downstream River 26-July-89	Secondary Pond Discharge 26-July-89
Water Quality Variables				
Filterable Residue	mg/L	381.0	32.0	95.0
Oxid-Red (Eh)	mV	256	253	258
Temperature (Field)	Degrees C	26.0	25.9	29.2
pH-value	pH Units	6.6	7.0	9.9
Anions and Ammonia				
Alkalinity	mg HCO ₃ /L	15.0	13.0	27.0
Ammonia	mg N/L	0.080	0.010	0.040
Bromide	mg/L	<0.040	0.050	<0.040
Carbonate	mg HCO ₃ /L	16.0	18.0	12.0
Chloride	mg/L	1.7	1.7	1.9
Fluoride	mg/L	0.045	0.046	0.11
Nitrate/Nitrite	mg N/L	0.27	0.30	0.050
Organic Carbon	mg C/L	3.1	2.8	2.6
Phosphate, total	mg P/L	<0.050	<0.050	<0.050
Sulfate	mg/L	2.3	3.4	39.0
Sulfide	mg S/L	<0.15	<0.15	0.16
Thiourea	mg/L	<0.50	<0.50	<0.50
Elements by AA				
Arsenic	mg/L	<0.0050	<0.0050	0.10
Calcium	mg/L	2.0	2.0	11.2
Chromium	mg/L	<0.0010	<0.0010	0.0012
Copper	mg/L	<0.0020	0.0034	0.0077
Magnesium	mg/L	0.73	0.73	1.2
Potassium	mg/L	1.2	0.78	5.9
Selenium	mg/L	<0.0050	<0.0050	<0.0050
Sodium	mg/L	2.8	3.4	8.3
Elements by ICPEs				
Aluminum	mg/L	<0.20	<0.20	0.37
Antimony	mg/L	<0.20	<0.20	<0.20
Arsenic	mg/L	<0.30	<0.30	<0.30
Barium	mg/L	0.10	0.20	0.13
Beryllium	mg/L	<0.0020	<0.0020	<0.0020
Boron	mg/L	<0.60	<0.60	<0.60
Cadmium	mg/L	<0.0050	<0.0050	<0.0050
Cobalt	mg/L	<0.010	<0.010	<0.010
Iron	mg/L	0.28	0.16	<0.040
Lead	mg/L	<0.050	<0.050	<0.050
Manganese	mg/L	0.034	0.049	<0.010
Molybdenum	mg/L	<0.050	<0.050	0.053
Nickel	mg/L	<0.020	<0.020	<0.020
Potassium	mg/L	<3.0	<3.0	<3.0
Silicon	mg/L	5.4	5.3	4.9
Silver	mg/L	<0.010	<0.010	<0.010
Strontium	mg/L	0.017	0.022	0.30
Thallium	mg/L	<0.10	<0.10	<0.10
Vanadium	mg/L	<0.020	<0.020	0.026
Zinc	mg/L	0.032	0.029	<0.020

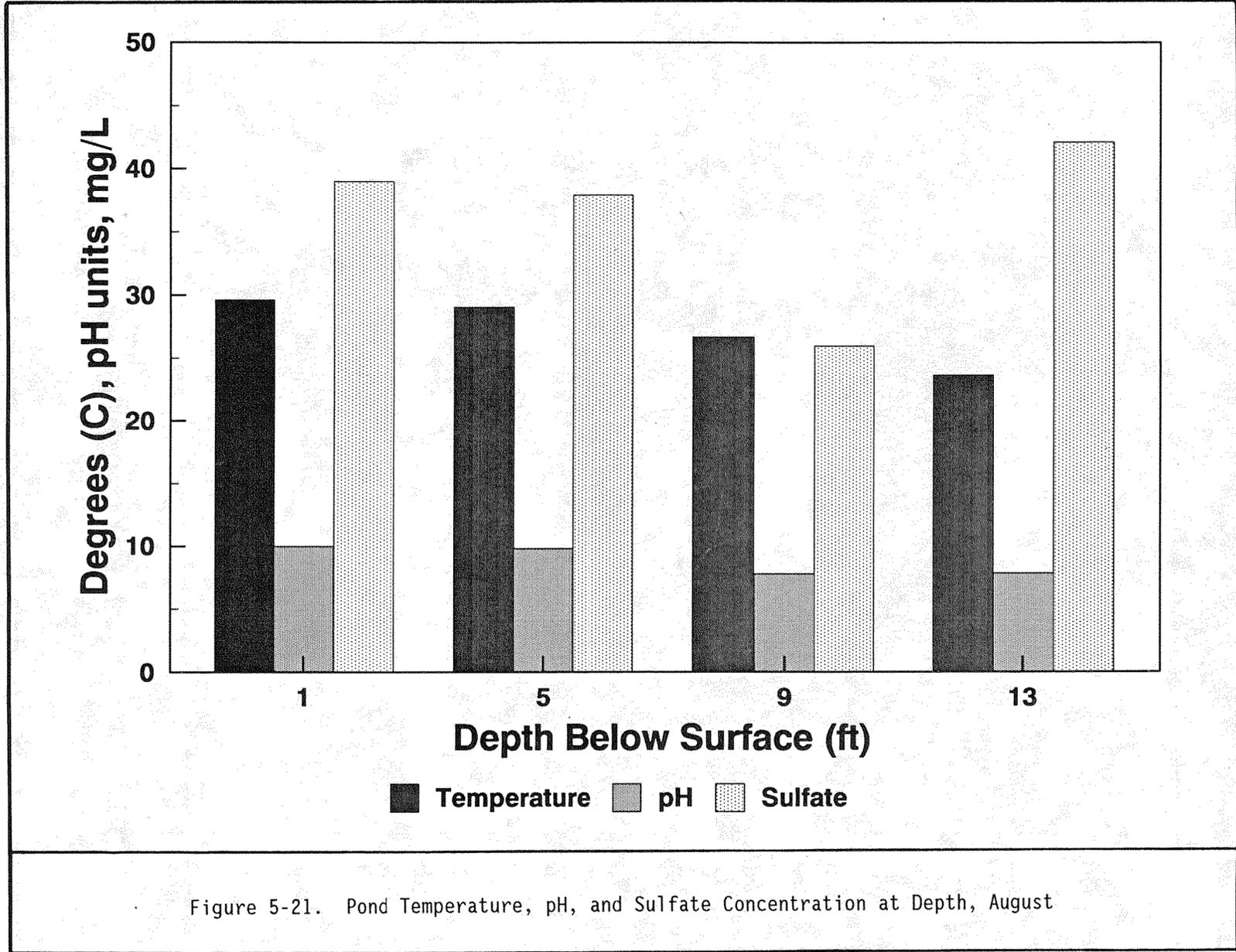
able concentrations of transition elements were found in the pond water, including part per billion levels of arsenic, chromium, copper, molybdenum, and vanadium. However, these elements were all present in concentrations of less than one part per million and, in all cases except arsenic, were measured at levels less than five times the analytical detection limit. Thus, their concentrations are not quantifiable.

Samples were collected during July and August of 1989 at several other locations in the ash ponds to determine if the pond water quality differed significantly with location. To determine the areal variability of water quality, samples were collected at the primary pond discharge, at two points within the primary pond (by boat), as well as from the secondary pond discharge. The results are presented in Appendix C and show no statistically significant differences in the water chemistry over the areal extent of the pond.

Samples also were taken from the two boat sampling locations at several different depths to determine vertical variability in pond chemistry (Figure 5-21). There is an obvious trend in decreasing pH with depth in the ash pond. Other major constituents did not show any apparent concentration dependence on depth. The pH dependence on depth is a result of algal species producing changes in pond water quality during the summer months. The algae remove carbon dioxide from the pond water during their photosynthetic activities, thus creating alkaline pHs near the pond surface. At depth in the pond, sunlight cannot promote the photosynthetic activity and the algae cannot thrive.

Figure 5-22 presents a chart summarizing the temporal changes in pond water quality. Again, the concentration of major dissolved salts does not change with the seasons, but pH depends on the seasons as a result of algal blooms. It is unknown why sulfate is higher in the winter pond sample than in the spring and summer samples.

The sluice water from the ash discharge pipes also was sampled. These results are presented in Table 5-10. Sluice water flows almost continuously from the discharge pipes, even though ash is only discharged intermittently. The sluice water consists of several of the low volume wastes discussed in Section 2. The notable results from the sluice water analyses are the elevated pH and calcium values during ash sluicing. This is expected since the sluice water leaches the readily soluble species from the ash. The surprising finding is the low sulfate concentration during sluicing. Sulfate only increased to 9.6 mg/L during ash sluicing versus 2.4 mg/L when ash was not being sluiced. This is only one sample, but it may suggest that



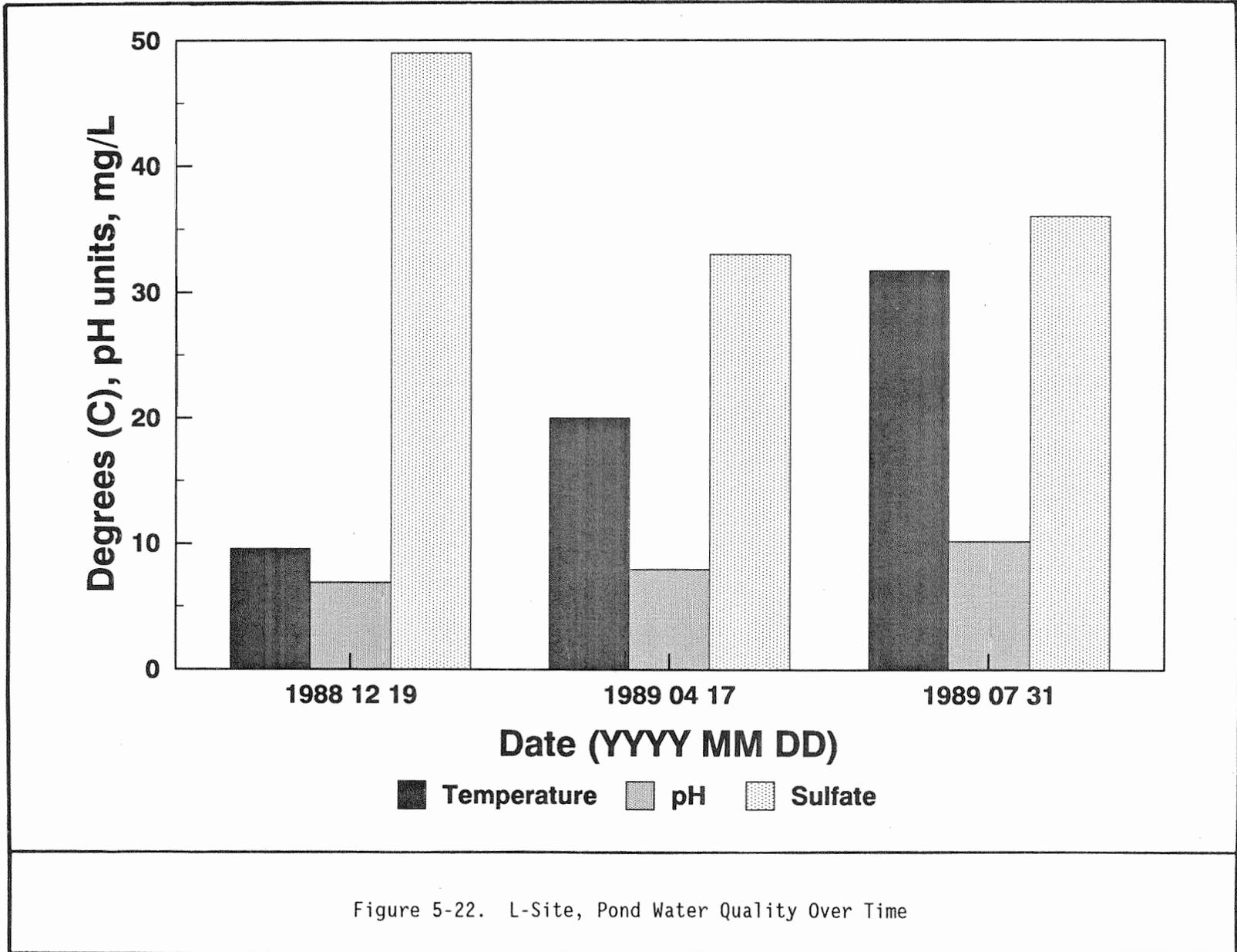


Figure 5-22. L-Site, Pond Water Quality Over Time

Table 5-10
 CHEMICAL CHARACTERISTICS OF SLUICE WATERS FROM L-SITE, EVENT II

<u>Parameter</u>	<u>Units</u>	<u>North Sluice Line</u>		<u>South Sluice Line</u>
		<u>No Ash 26-July-89</u>	<u>With Ash 7-July-89</u>	<u>No Ash 26-July-89</u>
Oxid-Red (Eh)	mV	257	249	256
Temperature (Field)	Degrees C	26.7	28.0	27.2
pH	pH Units	6.8	8.4	6.9
Sulfate	mg/L	2.9	9.6	4.1
Calcium	mg/L	2.4	208	2.8

the higher sulfate concentrations found in the pond water (about 50 mg/L) are a result of kinetically slower reactions between the ash and the pond water (e.g., pyrite oxidation).

Groundwater samples were collected from three sources on the western-most edge of the primary ash pond dam: monitoring well LMW-17, a toe-drain in the dam to relieve pore pressure, and a spring in the vicinity of LMW-17. Their major ion concentrations are presented in Table 5-11. From this limited sampling, the groundwater from LMW-17, toe-drain, and the spring water appear to be very similar in chemical composition.

SUMMARY OF HYDROLOGY AND GROUNDWATER CHEMISTRY

The L-site hydrology is a product of the geology and topography of the area and the management of the utility ash ponds. Groundwater moves outward from the primary pond except on the southwestern, upgradient end of the pond near the ash discharge area. Groundwater flow at the ash saprolite contact in this area may be either into the pond or outward from it depending on the relative levels of the pond and the groundwater.

The topography of bedrock valleys buried under alluvial deposits and the locations of weathered quartz/feldspar veins in the saprolite zone appear to influence the flow pattern of the shallow groundwater. Hydraulic conductivity is generally greatest along the zone of transition from saprolite soil to partially weathered bedrock, and in quartz/feldspar veins. Calculated groundwater velocities range up to 600 ft/day in one of the bedrock valleys.

Water balance calculations indicate that inflows to the ash pond are dominated by utility discharges, which represent between 70 percent and 90 percent of total inflows. Groundwater discharge to the ponds, surface flows, and direct precipitation make up the balance. Between 60 percent and 70 percent of the discharge from the ponds is through the secondary pond surface discharge; most of the remainder occurs as recharge to the shallow groundwater.

Groundwater within the ash deposits is more acidic and has higher concentrations of ash-derived major constituents such as SO_4 , Ca, Fe, Mg, K, and Na, and trace species including F, Ni, and Zn. The increased acidity, Fe, and SO_4 in groundwater in the ash have been associated with the oxidation of pyrite rejects co-managed with the coal ash.

Table 5-11

CHEMICAL CHARACTERISTICS OF GROUNDWATER IN THE VICINITY OF WELL LMW-17, EVENT II

<u>Parameter</u>	<u>Units</u>	<u>LMW-17 31-July-1989</u>	<u>Toe Drain B 28-July-1989</u>	<u>SEEP-A 31-July-1989</u>
Oxid-Red (Eh)	mV	239	234	225
Temperature (Field)	Degrees C	17.4	17.1	17.6
pH	pH units	5.7	6.7	6.4
Sulfate	mg/L	58	57	49
Calcium	mg/L	13	14	12

Downgradient groundwater quality measurements on samples collected during two sampling events show limited effects on groundwater quality outside of the immediate vicinity of the ash deposits.

Only a few analytes show concentrations in downgradient wells statistically elevated with respect to upgradient wells. These analytes, calcium, magnesium, strontium, and sulfate, generally decrease in concentration between the ash pond and the river, and show a relatively narrow "tongue" of elevated concentration in groundwater between the pond and the river.

The downgradient concentrations of metals associated with pyrite oxidation or boiler chemical cleaning wastes are below detection limits or are statistically indistinguishable from upgradient background concentrations. No impact from the co-management of low volume wastes was detected in groundwater outside the ash ponds.

Section 6

SUMMARY AND CONCLUSIONS

SITE DESCRIPTION

The L-site is located at a 400 MW coal-fired steam station in the southeastern U.S. Fly ash and bottom ash are sluiced to a disposal pond system consisting of two unlined settling basins, constructed in 1973. The ponds discharge to a river. Several low volume wastes are co-managed in the pond system, including pyrite rejects, demineralizer regenerant, and boiler chemical cleaning wastes.

HYDROGEOLOGY

The hydrogeology of the L-site was characterized from boring logs from previous geotechnical investigations and from boring logs and wells installed as part of this investigation. A total of 24 monitoring wells and piezometers were installed at the L-site and 70 geologic core samples were collected for physical properties and geochemical analyses.

The shallow geology at the L-site can be classified into five stratigraphic units: 1) saprolite; 2) alluvium; 3) fill material; 4) partially weathered bedrock; and 5) unweathered bedrock. The unweathered bedrock consists of granitic gneisses and mica-rich schists. The unweathered bedrock surface approximately mirrors the present surface topography. Two bedrock valleys are present under the primary ash pond and a third bedrock valley is present under the secondary ash pond.

A partially weathered bedrock deposit lies above the unweathered bedrock. This unit represents a transition from the extensive chemical weathering of the bedrock material (saprolite) and the unweathered bedrock.

Saprolite and alluvium deposits lie stratigraphically above the partially weathered bedrock. The saprolite represents a higher degree of chemical weathering than the partially weathered bedrock. Most of the saprolite samples contain abundant sand-sized mica grains which significantly impact the hydrologic properties of the aquifer. Alluvial deposits are similar in lithology to the saprolite and are restricted to the relatively narrow bedrock valleys. Fill material is stratigraphically

cally above the saprolite and alluvium and is restricted to the areas immediately downgradient of the ash ponds.

The potentiometric surface approximately mimics the land-surface topography at the L-site ranging from a maximum elevation of about 745 feet upgradient of the primary ash pond to a minimum elevation of 653 feet in the immediate vicinity of the river. The general direction of groundwater flow is northeasterly towards the river. The river appears to be a major groundwater discharge area for the shallow groundwater flow system.

The primary ash pond appears to be a source of both recharge and discharge for the shallow groundwater flow system. Shallow groundwater is discharging into the pond on the upgradient side of the pond and pond water is discharging to the groundwater system on the downgradient end near the dam. The groundwater flow direction is into the ash pond in the area of the ash delta; however, a small increase in pond level or a lowering of groundwater levels results in a reversal in groundwater flow directions in this area.

The saturated hydraulic-conductivity values were measured both in situ and in the laboratory. The in situ horizontal hydraulic-conductivity values, obtained from slug tests and a multiple-well pumping test, range over three orders of magnitude from 1.2×10^{-4} cm/sec to 8.7×10^{-2} cm/sec. In general, the larger horizontal hydraulic-conductivity values were measured in wells completed at the saprolite/partially weathered bedrock contact and the smallest values were measured in the unweathered bedrock. Weathered quartz/feldspar veins appear to correlate with some of the larger horizontal hydraulic-conductivity values measured in wells.

The vertical saturated hydraulic conductivities for unconsolidated deposits at the L-site were measured in the laboratory and range over four orders of magnitude from 2.26×10^{-7} cm/sec to 1.97×10^{-3} cm/sec. In general, the vertical hydraulic conductivity was two to three orders of magnitude smaller than the horizontal hydraulic conductivity. This significant difference between the vertical and horizontal directions is partially due to the preferential orientation of the mica platelets in the saprolite, which preserve the fabric of the parent rocks. Differences in the scale of measurement of laboratory and field hydraulic conductivity determinations may also influence these results.

The flux of water through the base of the pond was quantified by measuring the vertical hydraulic gradient and hydraulic conductivity of pond bottom sediments. The

calculated flux of water through the base of the pond near the dam was 1.0×10^{-5} cm/sec, which for an effective porosity of 30 percent, corresponds to a groundwater velocity of 3.5×10^{-5} cm/sec (0.1 feet/day).

A water-balance calculation indicated that approximately 75 percent of the water inflow to the primary ash pond is from the steam plant and approximately 50 percent of the outflow is surface-water discharge to the secondary ash pond. Only about 15 to 30 percent of the pond water discharges through the base of the pond to the shallow groundwater system.

Groundwater velocities were estimated to range from about 4×10^{-3} cm/sec (0.1 ft/day) to greater than 1.2×10^{-2} cm/sec (40 ft/day), with a mean velocity of about 0.9 ft/day. These high velocities are due to the large hydraulic gradient between the pond and river, and the high hydraulic-conductivity values of the saprolite.

ASH AND SOIL CHEMISTRY

The chemistry of the ash disposed in the L-site ponds is fairly representative of coal ash produced in the United States. Major element concentrations in the ash are typical of wet-sluciced eastern coal ash. Total and leachable concentrations of trace elements in the ash are also typical of reported values for coal ash.

Concentrations of several constituents in ash leachates, including Al, Ba, Cu, Fe, and Mn, appear to be controlled by solubility relations with the ash. In the underlying soils, similar solubility controls affect concentrations of Al, Ba, Fe, and Mn so that precipitation reactions are not expected to influence mobility of dissolved constituents in leachate moving from the ash into the underlying soils. Major and trace element concentrations in hydroxylamine hydrochloride extracts of ash and soil samples are consistent with this hypothesis, and show no enrichment in leachable material at the ash/soil interface, where precipitation reactions are most likely.

Oxidation of reduced iron, and its subsequent precipitation from solution as an oxide, appears to be the only potential precipitation reaction which could substantially affect ash leachate composition. No direct evidence for this reaction was detected in field measurements. Groundwater flow reversals in the area of the ash delta may impair mixing of oxidized groundwater and ash leachate containing reduced iron.

Other geochemical measurements, including cation exchange capacities and amorphous iron adsorption capacities, made on soil samples from the L-site indicate low atten-

uative capacity for these soils through adsorption mechanisms. This is consistent with the relatively coarse grain size and high degree of chemical weathering of the saprolite soils.

GROUNDWATER CHEMISTRY

The groundwater quality at L-site was evaluated by analyzing samples collected in April and July of 1989 from the monitoring well network around the primary ash pond. Four of the monitoring wells were screened in background groundwater, i.e., they were hydraulically isolated from the ponds. Two monitoring wells were placed in the ash delta of the pond, near the ash discharge line, and were screened in the saprolite beneath the ash. The remaining wells were located downgradient of the primary ash pond. A suite of analytes were measured in the samples. The analytes were selected based on earlier research on utility waste (Ainsworth and Rai, 1987, and Holcombe et al., 1987).

To summarize the groundwater chemistry, the following analytes were below analytical detection limits in all groundwater monitoring wells:

Analytes Below Detection Limits (mg/L)			
Analyte	Detection Limit	Analyte	Detection Limit
Antimony	0.20	Selenium	0.002
Arsenic	0.002	Silver	0.03
Boron	0.60	Thallium	0.10
Cadmium	0.005	Thiourea	0.50
Copper	0.001	Vanadium	0.02
Lead	0.002		

The following analytes were not significantly different in concentration between background wells and wells downgradient from the ash pond:

Analytes Where Background = Downgradient Concentrations		
Aluminum	Cobalt	Nitrate/Nitrite
Ammonia	Fluoride	Phosphate
Barium	Iron	Potassium
Beryllium	Manganese	Silicon
Bromide	Molybdenum	Sodium
Chloride	Nickel	Sulfide
Chromium		

The following analytes were significantly higher in concentration in wells downgradient from the ash pond than in background wells:

<u>Analyte</u>	<u>Median Downgradient Concentration</u>	<u>Median Background Concentrations</u>
Calcium	24	4.8
Magnesium	9.4	1.9
Strontium	0.16	0.03
Sulfate	110	3.4

The ash pond's effects on downgradient groundwater quality are limited to increased concentrations of Ca, Mg, Sr, and SO₄ over background groundwater. Of these constituents, only SO₄ currently has published water quality secondary standards for drinking water. The mean SO₄ concentrations measured downgradient of the ash pond were approximately half the water quality limits.

CO-MANAGEMENT OF WASTES

The co-management of low and high volume waste streams in the L-site ash ponds has not had a measurable effect on groundwater concentrations downgradient of the ponds during the past 17 years. The low volume waste streams with the greatest volume are estimated to be floor and yard drains (4.2 million gpy), boiler blowdown (21 million gpy), demineralizer regenerant (1.3 million gpy), and fireside wash water (1.6 million gpy). These streams generally contain low concentrations of dissolved solids and would not be expected to impact water quality from co-management. Of these four streams, demineralizer regenerant has the greatest potential to have measurable effects. This stream contains the salts removed from the demineralizer columns used to treat water for the boilers, and the salts from the regenerating acids and bases. Constituents contributed by the demineralizer regenerant stream would include Ca, Mg, Na, SO₄ and other constituents of the local water supply. This is the same set of constituents contributed by the ash. It would therefore be difficult to distinguish between constituents contributed by the ash and by demineralizer regenerant.

Boiler chemical cleaning wastes, on the other hand, contain quite a different set of constituents. In particular, the cleaning wastes contain elevated concentrations of copper, iron and several other heavy metals. Each boiler at L-site is cleaned every

three years, resulting in approximately 90,000 gallons of cleaning waste and associated rinses annually, or about 0.2% of the total waste volume. However, none of the heavy metals associated with the cleanings were found to be statistically different in concentration between downgradient and background groundwater. Four of the constituents which could be attributed to the cleaning waste appear elevated in groundwaters immediately beneath the ash delta; F, Fe, Ni, and Zn. However, these four constituents could also be a result of pyrite oxidation or contributed by the ash leachate. The concentrations of these four constituents in downgradient groundwater samples do not differ significantly from their concentrations in background samples.

Pyrite rejects from coal cleaning are estimated to be generated at a rate of about 23 m³/yr, compared to almost 700,000 m³/yr of coal ash. Potential impacts from pyrite codisposal in the L-site pond include high Fe and SO₄, and lower pH. These parameters were measured in the extracted pore fluids in the ash cores and the soil directly beneath the ash delta. The measurements suggest localized pyrite oxidation is occurring in the ash delta. However, examination of the downgradient core and up- and downgradient well waters indicated no measurable Fe and no statistically significant lowering of the pH. Apparently, chemical reactions in the soil beneath the pond are precipitating the Fe and neutralizing the acidity. Also, the ash delta area of the pond has exhibited reversals in groundwater flow during the period of investigation, suggesting water movement may be slow or stagnant in this location. In any case, we did not measure any impact on downgradient groundwater quality from pyrite co-management in this pond.

CONCLUSIONS

Waste management practices at the L-site are typical of those found at many U.S. utility power plants. Available data suggest that the volumes and compositions of the ash and low volume wastes managed in the unlined ash ponds fall within the normal ranges for a plant of this size. The siting of the plant and ash ponds near a major river is also typical, since power plants require large volumes of cooling water for operation. The hydrological conditions at the L-site are site-specific and, while no single property appears to be unusual, it is not possible to make general comparisons. The geology of the site is typical of the piedmont region of the southeastern United States where metamorphic bedrock has undergone extensive chemical weathering.

Effects of the L-site ash-disposal ponds on downgradient groundwater chemistry are limited to relatively small increases in the concentrations of a few common chemical

species. No statistically significant increase in concentrations of ash-derived metals was found in downgradient groundwater, and no measurable impact on river water chemistry was detected.

Groundwater within the disposed ash has statistically elevated concentrations of sulfate, metals including iron, nickel, and zinc, and increased acidity. Oxidation of co-disposed pyrite appears to be the source of these constituents. Slow or stagnant groundwater circulation within this portion of the disposed ash and/or further oxidation and coprecipitation of dissolved iron and other metals prevents the migration of these constituents into downgradient groundwater.

No other effects of the co-managed low-volume wastes were detected.

Section 7

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Appendix A
BOILER CHEMICAL CLEANINGS

Table A-1

PREVIOUS CHEMICAL CLEANINGS

<u>Boiler #</u>	<u>Date</u>	<u>Type of Cleaning</u>
L #1	February 1951	acid only (mill scale, etc.)
L #2	June 1951	acid only (mill scale, etc.)
L #1	March 1957	acid with copper complexer
L #2	July 1957	acid with copper complexer
L #3	November 1958	acid only (mill scale, etc.)
L #1	March 1960	bromate/acid with copper complexer
L #2	July 1960	bromate/acid with copper complexer
L #3	December 1961	bromate/acid only
L #1	May 1963	bromate/acid with copper complexer
L #2	July 1963	bromate/acid with copper complexer
L #3	October 1964	bromate/acid with copper complexer
L #1	April 1966	bromate/acid/bromate
L #2	May 1966	bromate/acid/bromate
L #2	April 1967	bromate/acid/bromate
L #3	July 1967	bromate/acid/bromate
L #1	January 1968	bromate/acid/bromate
L #3	May 1969	bromate/acid/bromate
L #2	September 1969	bromate/acid with copper complexer/ bromate
L #1	July 1970	bromate/acid with copper complexer
L #3	July 1971	bromate/acid with copper complexer/ bromate
L #2	October 1971	bromate/acid with copper complexer
L #1	March 1972	bromate/acid with copper complexer
L #2	October 1973	bromate/acid with copper complexer
L #1	February 1974	acid with copper complexer/bromate
L #3	May 1974	acid with copper complexer/bromate
L #2	September 1975	bromate/acid with copper complexer
L #1	May 1976	acid with copper complexer/bromate
L #3	September 1977	acid with copper complexer/bromate
L #2	January 1979	bromate/acid with copper complexer
L #1	May 1980	bromate/acid with copper complexer
L #3	September 1980	bromate/acid with copper complexer
L #2	May 1982	bromate/acid with copper complexer
L #3	January 1983	bromate/acid with copper complexer
L #1	August 1983	bromate/acid with copper complexer
L #2	September 1985	bromate/acid with copper complexer
L #3	February 1986	bromate/acid with copper complexer
L #1	January 1987	bromate/acid with copper complexer
L #2	June 1988	bromate/acid with copper complexer

Table A-2

CHEMICAL CLEANING PROCESS AND CLEANING FREQUENCY

Boilers No. 1,2, and 3 are typically cleaned on a 3-year time interval. The boiler is cleaned by sequentially using an ammonium bromate solvent for removal of copper and copper oxides, and then a solution of inhibited hydrochloric acid, ammonium bifluoride and copper complexer for removal of iron oxides, silica, and residual copper. Finally the boiler metal is neutralized and passivated with a solution of soda ash.

Table A-3

CHEMICALS USED FOR CLEANING

The following chemicals and quantities are used for the boiler chemical cleanings:

L #1 (33,000 gal. Natural Circulation Boiler):

1st Stage:	550 lbs.	Sodium Bromate
	1000 lbs.	Ammonium Carbonate
	900 gal.	Ammonium Hydroxide, 26° Be'
2nd Stage:	4500 gal.	Hydrochloric Acid, 20° Be' (31.5%), inhibited for use at 160°F in 5% solution containing 3/4% Copper complexer and 1/2% Ammonium Bifluoride
	1830 lbs.	Copper Complexer (Thiourea or equivalent)
	1400 lbs.	Ammonium Bifluoride
	300 lbs.	Citric Acid
	2800 lbs.	Soda Ash

L #2 (33,000 gal. Natural Circulation Boiler):

1st Stage:	550 lbs.	Sodium Bromate
	1000 lbs.	Ammonium Carbonate
	900 gal.	Ammonium Hydroxide, 26° Be'
2nd Stage:	4500 gal.	Hydrochloric Acid, 20° Be' (31.5%), inhibited for use at 160°F in 5% solution containing 3/4% copper complexer and 1/2% Ammonium Bifluoride
	1830 lbs.	Copper Complexer (Thiourea or equivalent)
	1400 lbs.	Ammonium Bifluoride
	300 lbs.	Citric Acid
	2800 lbs.	Soda Ash

L #3 (26,700 gal. Controlled Circulation Boiler):

1st Stage:	550 lbs.	Sodium Bromate
	1000 lbs.	Ammonium Carbonate
	850 gal.	Ammonium Hydroxide, 26° Be'
2nd Stage:	3700 gal.	Hydrochloric Acid, 20° Be' (31.5%), inhibited for use at 160°F in 5% solution containing 3/4% copper complexer and 1/2% Ammonium Bifluoride
	1680 lbs.	Copper Complexer (Thiourea or equivalent)
	1100 lbs.	Ammonium Bifluoride
	300 lbs.	Citric Acid
	2200 lbs.	Soda Ash

Table A-4

DEPOSIT COMPOSITION

The following deposit quantities were removed during the indicated chemical cleanings.

Boiler #	Date	Pounds of Deposit Removed*			
		Fe	Cu	Ni	Zn
L #1	2/6/74	/1074	/138	/27	/14
	5/12/76	/1485	/309	/-	/-
	5/7/80	1410/1046	377/330	78/63	9/28
	8/3/83	1511/1485	131/148	27/30	20/24
	1/14/87	1405/1238	74/151	13/16	8/-
L #2	9/24/75	/1459	/216	/46	/8
	1/17/79	/1046	/291	/104	/49
	5/26/82	1838/1571	195/41	63/55	16/14
	9/25/85	2181/1651	158/192	46/18	20/-
	6/1/88	1414/882	197/96	48/24	19/15
L #3	5/28/74	/2608	/423	/65	/49
	9/28/77	/2116	/514	/198	/414
	9/24/80	2900/979	638/482	112/-	83/-
	1/5/83	5052/3224	464/531	140/108	97/81
	2/12/86	3955/1670	223/366	67/-	41/-

*Numbers to the right of diagonal line - Contractor's results.
 Numbers to the left of diagonal line - results.

Table A-5

COAL PILE RUNOFF COMPOSITIONS
(mg/L)

Plant: Coal: ¹	I Bituminous	J Subbituminous	L Subbituminous
Elemental Analysis	14	0.15	0.54
Aluminum	<0.02	<0.02	<0.02
Antimony	<0.002	<0.002	0.006
Arsenic	0.04	0.078	0.043
Barium	0.007	<0.001	<0.001
Beryllium	<0.05	0.8	0.95
Boron	0.004	<0.001	0.001
Cadmium	72	45	270
Calcium	0.005	<0.005	<0.005
Chromium	0.17	<0.006	<0.006
Cobalt	0.06	0.002	0.002
Copper	2.6	0.38	13.3
Iron	<0.08	<0.002	0.015
Lead	19	18	68
Magnesium	3.2	0.023	1.2
Manganese	0.0003	0.0003	<0.0002
Mercury	0.005	<0.002	<0.002
Molybdenum	0.21	<0.003	3
Nickel	1.8	0.76	4.1
Potassium	<0.002	<0.002	<0.002
Selenium	4.4	1.8	11
Silicon	0.0018	0.0023	0.0012
Silver	12	85	48
Sodium	<0.09	<0.09	<0.09
Thallium	<0.003	<0.003	<0.003
Vanadium	0.59	<0.003	0.01
Zinc			
Water Quality			
pH (units)	3.1	9.3	8.4
Acidity (as CaCO ₃)	180	<10	<10
Alkalinity (as CaCO ₃)	<1	93	310
COD	<5	<5	<5
Chloride	20	8	34
Fluoride	0.61	0.24	0.24
Nitrate	2	6	<2
Nitrite (as N)	<0.02	0.03	<0.02
Sulfate	480	420	740
TDS	660	970	1500
TOC	2	2	4

¹All coals contain less than 1.5 percent sulfur

Table A-6

WHOLE-SAMPLE CHARACTERIZATION OF UNTREATED COOLING
TOWER BASIN SLUDGE--WATER QUALITY PARAMETERS

Plant No.:	1	6	15
Physical State	Slurry	Slurry	Slurry
Wt. Percent Solids (%)	23	5.1	23.3
pH (pH Units)	7.1	6.99	6.1
Total Organic Carbon ($\mu\text{g/mL}$)	16	100	300
Chemical Oxygen Demand ($\mu\text{g/mL}$)	72	560	370
Alkalinity ($\mu\text{g/mL CaCO}_3$)	560	1,140	1,900
Sulfate (SO_4 , $\mu\text{g/mL}$)	<1	3.6	6
Chloride ($\mu\text{g/mL}$)	330	30	66
Fluoride ($\mu\text{g/mL}$)	3.9	<1	1.4
Nitrate (NO_3 , $\mu\text{g/mL}$)	75	<1	19
Nitrite (NO_2 , $\mu\text{g/mL}$)	<0.002	0.010	0.14

Table A-6 (Continued)

TOTAL ELEMENTAL ANALYSIS OF UNTREATED COOLING TOWER
BASIN SLUDGE--ICAP SCAN

Plant No.:	1	6	15
Element	($\mu\text{g}/\text{gm}$)	($\mu\text{g}/\text{gm}$)	($\mu\text{g}/\text{gm}$)
Physical State	Slurry ¹	Slurry ¹	Slurry ¹
Silver	3.8	4.1	0.3
Aluminum	120,000	39,000	120,000
Arsenic	<60	<60	<60
Boron	1,300	520	94
Barium	6,900	1,500	990
Beryllium	1.1	<.9	2.3
Calcium	220,000	51,000	15,000
Cadmium	2.0	1.9	1.7
Cobalt	73	20	12
Chromium	<2	240	32
Copper	180	17,000	27
Iron	99,000	50,000	73,000
Mercury	<30	<30	<30
Potassium	12,000	12,000	20,000
Lithium	57	16	29
Magnesium	36,000	11,000	11,000
Manganese	710	710	1,000
Molybdenum	14	8.1	<0.3
Sodium	17,000	12,000	7,600
Nickel	110	890	54
Phosphorous	4,300	<400	3,300
Lead	<100	<100	<100
Sulfur	3,300	3,000	5,800
Antimony	<60	<60	<3
Selenium	<200	<200	<8
Tin	<100	<100	<100
Strontium	4,200	160	430
Titanium	13,000	2,100	4,000
Vanadium	170	<5	120
Tungsten	<30	<30	<30
Zinc	73	1,500	110

¹Slurries were evaporated to dryness before analysis. The results are reported as $\mu\text{g}/\text{gm}$ of dry solids.

Table A-7

WHOLE-SAMPLE CHARACTERIZATION OF UNTREATED AND TREATED DEMINERALIZED REGENERANT--WATER QUALITY PARAMETERS

Plant No.:	3	13
	Treated ¹	Untreated
Physical State	Liquid	Liquid
Wt. Percent Solids (%)	0.27	0.002
pH (pH Units)	7.0	4.42
Total Organic Carbon ($\mu\text{g/mL}$)	2	<10
Chemical Oxygen Demand ($\mu\text{g/mL}$)	31	38
Alkalinity ($\mu\text{g/mL}$) CaCO_3	93	<1
Sulfate (SO_4 , $\mu\text{g/mL}$)	3,000	2
Chloride ($\mu\text{g/mL}$)	49	0.74
Fluoride ($\mu\text{g/mL}$)	2.6	0.64
Nitrate (NO_3 , $\mu\text{g/mL}$)	<1	<1
Nitrate (NO_2 , $\mu\text{g/mL}$)	1.1	<0.002

¹Treated with caustic

Table A-7 (Continued)

TOTAL ELEMENTAL ANALYSIS OF UNTREATED AND TREATED
DEMINERALIZED REGENERANT--ICAP SCAN

Element	Plant No.:	3	13
		Treated ¹ ($\mu\text{g}/\text{mL}$)	Untreated ($\mu\text{g}/\text{mL}$)
Physical State		Liquid	Liquid
Silver		0.16	0.004
Aluminum		0.12	<.05
Arsenic		<.06	<.05
Boron		0.29	0.041
Barium		0.038	0.051
Beryllium		0.001	<.001
Calcium		43	<.04
Cadmium		0.009	0.010
Cobalt		0.016	<.006
Chromium		.001	0.014
Copper		0.029	<.001
Iron		<.008	<.008
Mercury		<.03	<.03
Potassium		3.1	0.010
Lithium		0.059	<.001
Magnesium		17	<.03
Manganese		0.42	<.002
Molybdenum		0.049	0.004
Sodium		1,200	1.2
Nickel		0.054	<.003
Phosphorous		<.1	<.1
Lead		<.08	<.08
Sulfur		900	0.060
Antimony		<.03	<.03
Selenium		<.08	<.08
Tin		<.1	<.1
Strontium		0.54	0.002
Titanium		0.012	<.005
Vanadium		0.11	<.003
Tungsten		<.03	<.03
Zinc		<.003	<.003

¹Treated with caustic

Table A-8
FIRESIDE WASTE COMPOSITIONS
(mg/L)

Plant: Sample Description:	A	D	K	T	
	Fireside Waste	Fireside Waste	Air Preheater ¹ Waste	+1 hour	+16 hours
Elemental Analysis					
Aluminum	0.66	13	2.8	21	<0.05
Antimony	<0.021	<0.2	<0.02	0.31	0.087
Arsenic	<0.002	<0.002	<0.002	<0.002	0.016
Barium	0.083	0.26	0.061	0.089	0.09
Beryllium	<0.001	<0.01	<0.001	0.012	<0.001
Boron	<0.02	<0.5	0.2	<0.5	0.29
Cadmium	0.004	0.027	0.017	0.2	<0.002
Calcium	89	270	10	381	14
Chromium	<0.005	0.32	0.21	0.53	<0.005
Cobalt	0.26	0.79	0.22	5.4	<0.006
Copper	0.12	0.63	0.33	7.1	0.012
Iron	8.6	87	36	513	0.52
Lead	0.007	<0.002	0.082	0.051	0.008
Magnesium	47	950	0.008	2710	17
Manganese	0.84	3.6	1.1	1.3	0.002
Mercury	<0.0002	0.0003	<0.0002	<0.0002	<0.0002
Molybdenum	0.003	1.1	0.036	0.074	0.43
Nickel	5.7	89	6.5	0.59	0.032
Potassium	4.4	<0.5	8.8	335	5.5
Selenium	<0.002	<0.002	<0.002	<0.002	<0.002
Silicon	2.9	13	0.49	6.1	0.98
Silver	0.0022	0.016	0.019	0.022	<0.002
Sodium	160	700	420	5790	750
Thallium	<0.09	1	<0.09	0.27	<0.09
Vanadium	0.14	180	6.8	4.7	19
Zinc	0.45	4.3	0.71	25	0.007
Water Quality					
pH (units)	6.5	3.3	3.9		10.1
Acidity (as CaCO ₃)	23	480	170		<10
Alkalinity (as CaCO ₃)	35	<1	<1		1400
Ammonia (as N)					
COD	230	5	<5		18
Chloride	130	67	350		13
Fluoride	2.1	0.28	0.21		1.2
Nitrate	<0.1	25	10		2
Nitrite (as N)	<0.1	0.22	<0.02		0.06
Sulfate	560	6400	990		340
TDS	1040	8900	2000		2200
TOC		3	4		3

¹Unfiltered sample, filtered sample analysis was essentially identical to unfiltered.

²Samples taken one and sixteen hours into wash episode.

Table A-9
PYRITE COMPOSITIONS
($\mu\text{g/g}$)

Plant:	N	R	Q	S	Pyrite Standard	
	Coal:	Subbituminous	Bituminous	Bituminous		Bituminous
Elemental Analysis						
Aluminum		16100	8130	17100	48800	
Antimony		12	<40	<40	<40	
Arsenic ¹		36	3460	1180	200	
Barium		209	<2	<2	3800	
Beryllium		0.56	<2	<2	15	
Boron		37	<100	<100	NA	
Cadmium		<4	<4	<4	<4	
Calcium		10800	1670	15300	3670	
Chromium		320	520	550	44	
Cobalt		3.3	15	3.2	20	
Copper		6.2	9.2	57	100	
Iron		26000	374000	193400	86900	435900
Lead ¹		1.6	0.27	1560	65	
Magnesium		7960	610	1800	3000	
Manganese		410	170	280	130	
Mercury ¹		0.04	0.76	0.86	0.2	
Molybdenum		1.3	11	<0.4	15	
Nickel		7.4	59	2.6	37	
Phosphorus		340	<500	<500	<300	
Potassium ¹		6380	1190	7460	11000	
Selenium ¹		0.5	<0.5	<0.5	<4	
Silver		<0.4	5.8	1.8	<2	
Sodium		5120	940	8180	730	
Sulfur (%)		0.94	46.34	18.12	14.04	49.5
Titanium		420	150	3.4	2500	
Thallium		<180	<180	<180	<180	
Vanadium		11	<30	67	160	
Zinc		19	78	34	44	
Heating Value (Btu/lb)		6100	3200	4900	4200	2174

¹Atomic absorption analysis. All other elements by inductively coupled argon plasma emission spectrometry (ICP).

NOTE: Whole-sample digestions by perchloric acid, except for iron and silicon analyses which were prepared for analysis by lithium metaborate fusion.