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October 16, 2020

VIA ELECTRONIC FILING

Ms. Kimberley A. Campbell, Chief Clerk
North Carolina Utilities Commission
Dobbs Building
430 North Salisbury Street
Raleigh, North Carolina 27603

Re: DEP Late-Filed Exhibit No. 10
Docket No. E-2, Sub 1219

Dear Ms. Campbell:

Per the request of the North Carolina Utilities Commission during the Duke Energy Progress, LLC ("DEP") evidentiary hearing, enclosed for filing on behalf of DEP in the above-referenced proceeding is Late-Filed Exhibit No. 10, which includes a copy of the 2001 EPRI Report that DEP witness Williams referenced in her testimony regarding the three case studies on cap and place.

Please do not hesitate to contact me should you have any questions. Thank you for your assistance with this matter.

Very truly yours,

/s/Mary Lynne Grigg

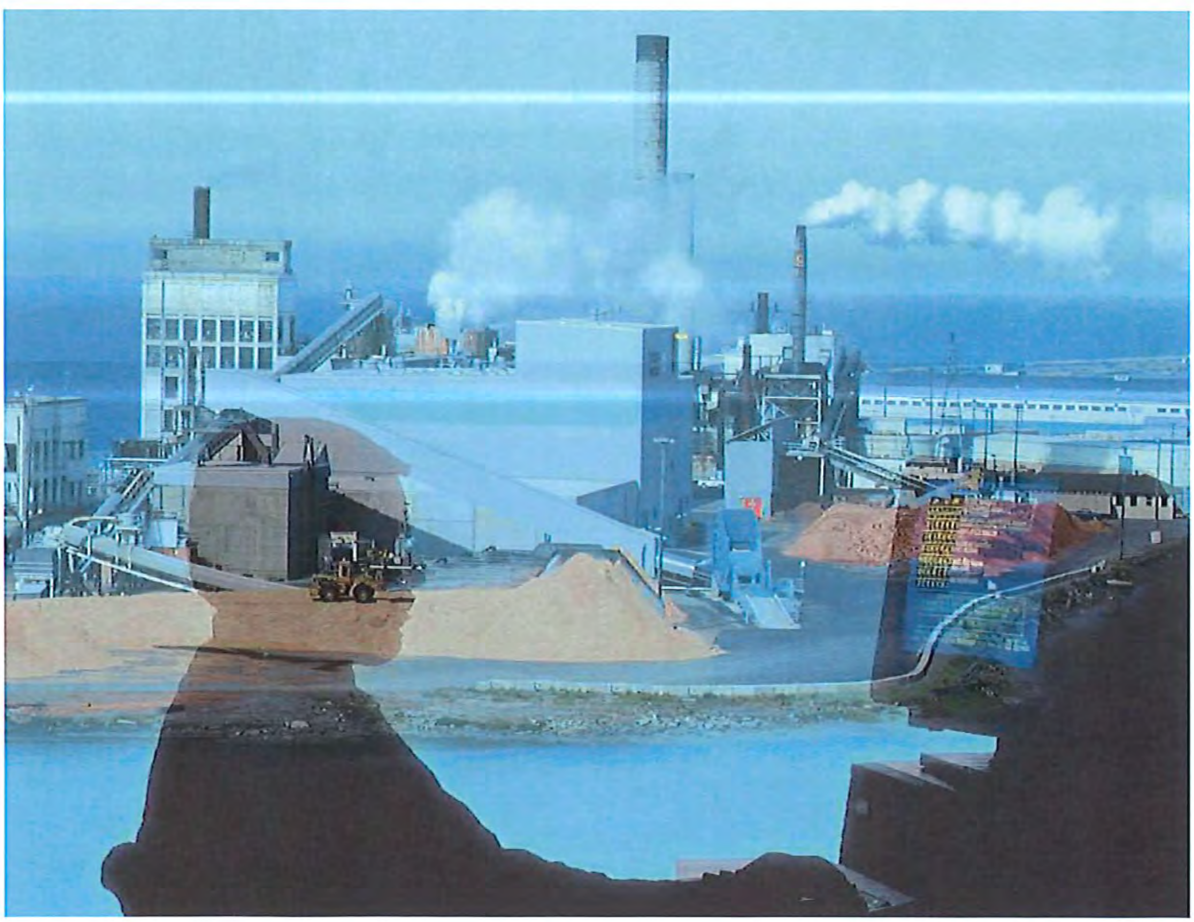
MLG:kma

Enclosures

EPRI

Evaluation and Modeling of Cap Alternatives at Three Unlined Coal Ash Impoundments

Technical Report



Evaluation and Modeling of Cap Alternatives at Three Unlined Coal Ash Impoundments

1005165

Final Report, September 2001

EPRI Project Manager
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Evaluation and Modeling of Cap Alternatives at Three Unlined Coal Ash Impoundments, EPRI, Palo Alto, CA: 2001. 1005165.

REPORT SUMMARY

Site closure plans for coal ash impoundments often call for the installation of an engineered cap, intended to reduce water infiltration into the closed facility and therefore protect groundwater quality by reducing the volume of leachate released. However, caps are costly and do not always provide significant benefit, so their inclusion should be considered carefully. This report describes research conducted at three different ash impoundments to evaluate various cap options and document the effectiveness of the selected alternatives.

Background

During the 1990s, EPRI participated in a series of tailored collaboration projects in which alternatives for compacted clay caps were explored at three coal ash impoundments. In all three cases, the impoundments were unlined and concentrations of ash indicator parameters were higher in downgradient groundwater than in upgradient groundwater. A hydrogeologic investigation was performed at each site to determine geology and groundwater flow and to delineate groundwater impacts associated with the impoundment. These results were used with groundwater flow models to predict the effectiveness of alternatives to compacted clay caps. In each case, the modeling indicated that dewatering would provide sufficient mass reduction to achieve acceptable concentrations of ash indicator parameters in downgradient groundwater, and that the additional benefits of a compacted clay cap were negligible. Based on these results, one impoundment was closed with a native soil cap and two were closed with no cap.

Objective

- to examine groundwater quality trends at three closed ash impoundments where alternatives to a compacted clay cap were used for site closure
- to determine whether the alternative closures achieved groundwater quality goals

Approach

Groundwater quality has been monitored since closure at all three impoundments. These groundwater quality results were compared to preclosure conditions to determine whether concentrations have decreased. The postclosure monitoring results were also compared to the results predicted by modeling.

Results

Two of the three impoundments achieved significant groundwater quality improvements after dewatering and closure. Of these two impoundments, one was capped and the other was not capped, suggesting that the cap had little bearing on overall closure performance. The key factor for achieving concentration reduction at these two facilities was dewatering the ash. Groundwater quality improvement closely paralleled improvement predicted using groundwater models at these two sites.

Groundwater quality did not improve at one of the three impoundments. This site differed from the other two in that a portion of the ash was below the current water table, the full extent of which was not known prior to closure of the site, and was not reflected in the closure modeling. Dewatering and closure were not effective at this site because leaching continued from the saturated ash. In this particular case, concentrations actually increased because the contact time of groundwater moving through the saturated ash increased when the hydraulic gradient of the pond was removed. A cap would have had little or no effect on this process.

EPRI Perspective

These results demonstrate that compacted clay or synthetic caps, often required under state solid waste disposal regulations, are not always necessary for groundwater protection when closing unlined coal ash impoundments. Lined impoundments will generally require a cap with permeability at least as low as the liner in order to avoid development of a saturated ash layer at the base of the fill. The costs and benefits of capping alternatives should be carefully weighed prior to closure. Keys to that analysis are the availability of capping materials, rate of mass loading, position of the ash relative to the water table, and hydrogeologic conditions. Models for estimating leachate release and transport, such as EPRI's FOWL-GH and MYGRT codes, are valuable tools in that analysis.

1005165

Keywords

Boron
Sulfate
Impoundment
Cap
Coal ash
Groundwater quality
Groundwater modeling

ABSTRACT

During the 1990s, EPRI participated in a series of tailored collaboration projects in which alternatives for compacted clay caps were explored at three unlined coal ash impoundments. A hydrogeologic investigation was performed at each site and the resulting data were input to groundwater flow models to predict the effectiveness of alternatives to compacted clay caps. In each case, the modeling indicated that dewatering would provide sufficient mass reduction to achieve acceptable concentrations of ash indicator parameters in downgradient groundwater, and that the additional benefits of a compacted clay cap were negligible. Based on these results, one impoundment was closed with a native soil cap and two were closed with no cap.

Two of the three impoundments achieved significant groundwater quality improvements after dewatering and closure, with observed groundwater quality closely paralleling model predictions. Of these two impoundments, one was capped and the other was not capped, suggesting that the cap had little bearing on overall closure performance. The key factor for achieving concentration reduction at these two facilities was dewatering the ash. Groundwater quality did not improve as predicted at the third site where conditions were later found to differ from those modeled.

These results demonstrate that compacted clay or synthetic caps are not always necessary for groundwater protection when closing unlined coal ash impoundments. The costs and benefits of capping alternatives should be carefully weighed prior to closure. Keys to that analysis are the availability of capping materials, rate of mass loading, position of the ash relative to the water table, and hydrogeologic conditions.

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1
INTRODUCTION

Background and Objective

Approximately half of the high-volume coal combustion by-products produced by electric power generation in the United States are sluiced to impoundments (EPRI, 1997). Typically, fresh ash from the collection hoppers is sluiced to the impoundment, where the ash particles settle to the bottom of the pond and the sluice water is decanted. Readily leachable compounds on the surface of the ash particles dissolve during sluicing and are discharged in dilute concentration at the sluice water decant point (EPRI, 1994). As a result, ash that has settled in impoundments generally has less readily leachable mass than fresh ash, and leachate from sluiced ash tends to have lower dissolved concentrations than leachate from fresh ash.

Table 1-1 compares field leachate concentrations for ash landfills and impoundments compiled from a variety of EPRI reports. Because the results are for different sources and were analyzed using different methods, slight variations are not significant; however the overall trend shows that landfill leachates typically have higher median concentrations than impoundment leachates, particularly for the ash indicator parameters of boron, calcium, sodium, and sulfate.

At closure, ash ponds are usually dewatered, covered, and revegetated. Removing the pond water (dewatering) reduces potential leachate loading to groundwater in two ways. First, it removes mass contained in the pond water. Second, dewatering reduces the volume of leachate released by reducing the hydraulic head that drives downward movement of water through the ash.

These observations suggest that leachate mass released from dewatered and closed impoundments will be greatly reduced compared to the leachate mass released during active operation because: (1) leachable mass in the remaining ash is relatively low; and (2) the volume of leachate water is greatly reduced. In some circumstances, this can sufficiently reduce the contaminant mass released from an impoundment such that engineered layered caps with compacted clay or synthetic materials may not be warranted.

Table 1-1
Comparison of Landfill and Impoundment Leachate Concentrations

		Landfill		Impoundment	
Parameter	Unit	Count	Median	Count	Median
Boron	mg/L	191	2.8	123	0.72
Calcium	mg/L	191	160	123	62
Chloride	mg/L	168	34	121	14
Fluoride	mg/L	79	0.40	87	0.41
Iron	mg/L	55	0.010	119	0.054
Magnesium	mg/L	191	4.1	123	4.1
Manganese	mg/L	68	0.0008	118	0.0089
pH	SU	191	9.4	98	8.5
Potassium	mg/L	169	30	75	11
Sodium	mg/L	191	489	78	37
Specific Cond.	umhos/cm	155	3,520	115	557
Sulfate	mg/L	190	1,480	122	158

Notes:

Data from EPRI reports as queried from the CBEAS database

Data include fly ash and mixed coal ash samples, exclude pyritic samples, and reflect a variety of sites and coal sources

Data include field-collected leachate samples end porewaters displaced from core samples by centrifuge or pressure

The objective of this research was to examine groundwater quality trends at three closed ash impoundments, where alternatives to a compacted clay cap were used for site closure, and to determine whether the alternative closures achieved groundwater quality goals. The three impoundments are located at two power plants in the midwestern United States. All three impoundments are unlined and had documented releases of leachate to groundwater while in service. The HA impoundment was removed from service late in 1993 and capped with native soils, and the two HN impoundments were removed from service late in 1996 and were not capped. Groundwater quality was monitored before and continuously since the impoundments were closed, providing an extensive dataset for determining closure effects.

2
HA IMPOUNDMENT

Background

Site Description

The HA Power Station has five 47-MW oil-burning units that are on permanent standby, and one 410-MW dry-bottom, pulverized coal-burning unit. The oil units began operation in 1958 and the coal unit began operation in 1978. Fly ash from the coal unit is collected by an electrostatic precipitator and sluiced to an on-site coal ash management facility. Fly ash is not collected at the oil units, which are rarely used. Bottom ash from all units is sluiced to the ash management facility. In addition to these high-volume by-products, the ash management facility receives miscellaneous low-volume plant wastes. There are three on-site ash management facilities. The North Pond received bottom ash from the oil units prior to 1978 and continues to receive low-volume wastes. The South Impoundment was the primary ash management facility from 1978 until it was closed late in 1993, and received fly ash, bottom ash, and overflows from the North Pond. The East Impoundment has been the primary ash management facility since 1993.

The closed South Impoundment, hereafter referred to as the HA impoundment, is the subject of this investigation. Several previous investigations have been performed at this facility, and provide the basis for this background discussion:

- An unpublished 1982 investigation by a state resource agency that defined hydrogeologic conditions at the impoundment.
- An EPRI study for the sponsoring utility in 1993 and 1994 that evaluated several closure options for the impoundment.
- EPRI, 2000. *Evaluation of Comanagement of Low-Volume Utility Wastes with High-Volume Coal Combustion By-Products: HA Plant.* EPRI Report Number 1000720.

The HA South Impoundment is located on the east bank of a large regional river (Figure 2-1). The impoundment is situated 1,000 feet (300 m) from the river on a terrace that is 20 feet (6 m) higher than normal river stage. This area is characterized by relatively flat topography, with occasional hills formed from post-glacial sand dunes. The surrounding area is rural and land use is mostly agricultural, primarily corn and soybeans.

The region is humid and annual average precipitation is about 35 inches (89 cm). Peak precipitation occurs during May and June. Average temperatures range from 27°F (-2.8°C) in January to 78°F (26°C) in July.

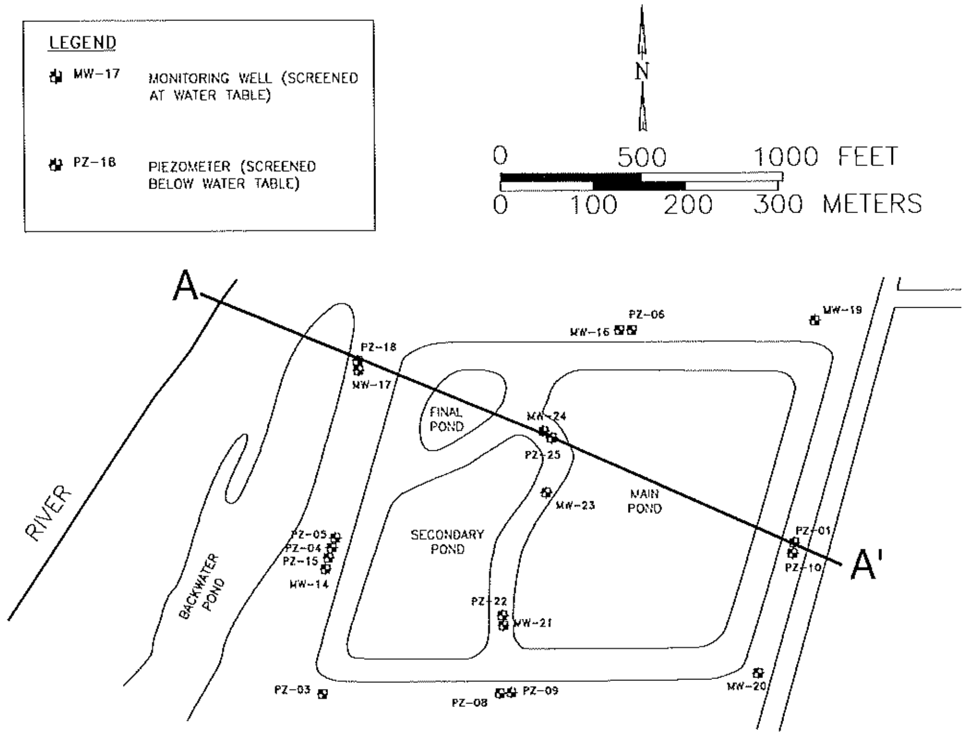


Figure 2-1
HA South Impoundment site map

The impoundment features berms constructed from locally occurring silty-sand soils, and covers an area of 30 acres (12 hectares). The berms are 30 feet (9 m) higher than the surrounding landscape and maximum ash thickness inside the berms is about 30 feet (9 m). During operation, cooling tower blowdown was used to sluice fly ash, bottom ash, and low-volume wastes to the impoundment, which consisted of three ponds. The sluice line discharged to the main pond where most of the solid by-products settled. A secondary pond received decant water from the primary pond, and a final pond received decant water from the secondary pond (Figure 2-1). Water exiting the final pond discharged to the river via an NPDES permitted discharge. The operational history of this impoundment is as follows:

- 1977: Impoundment construction.
- 1978: Coal ash sluicing initiated, primarily ash from low-sulfur eastern Kentncky and western United States coal.
- 1989: Berms surrounding the main pond raised by 10 feet to final elevation 30 feet above surrounding landscape.
- November 1993: Impoundment removed from service.
- 1994: Impoundment dewatered by gravity drainage and capped.

Prior to capping, ash in the secondary and final ponds was excavated and placed in the dewatered main pond. The cap consists of 3 to 4 feet (0.9 to 1.2 meters) of native silty sand soil, seeded with native grasses. Vegetation on the cover is thick, with no bare spots.

Leachate Characteristics

Coal ash leachate was sampled and characterized in 1996, after the impoundment had been closed for three years. Results of that characterization (Table 2-1) show that the leachate is dominated by sulfate and calcium, typical of coal ash. Trace elements detected in a majority of the nine leachate samples included arsenic, barium, copper, nickel, vanadium, and zinc. Chromium and lead were detected in two and one of the nine samples, respectively, and cadmium, selenium, and silver were not detected. Boron, an indicator constituent at coal ash sites, had a relatively low median concentration of 0.77 mg/L.

Hydrogeology

The HA impoundment overlies a highly permeable sand and gravel aquifer that is 80 to 90 feet (24 to 27 meters) thick. The upper 15 to 20 feet (4.6 to 6.1 meters) of the aquifer consist of well-sorted fine- to medium-grained wind-deposited dune sand. Below the dune sand is poorly sorted fine to coarse sand and gravel glacial outwash (Figure 2-2).

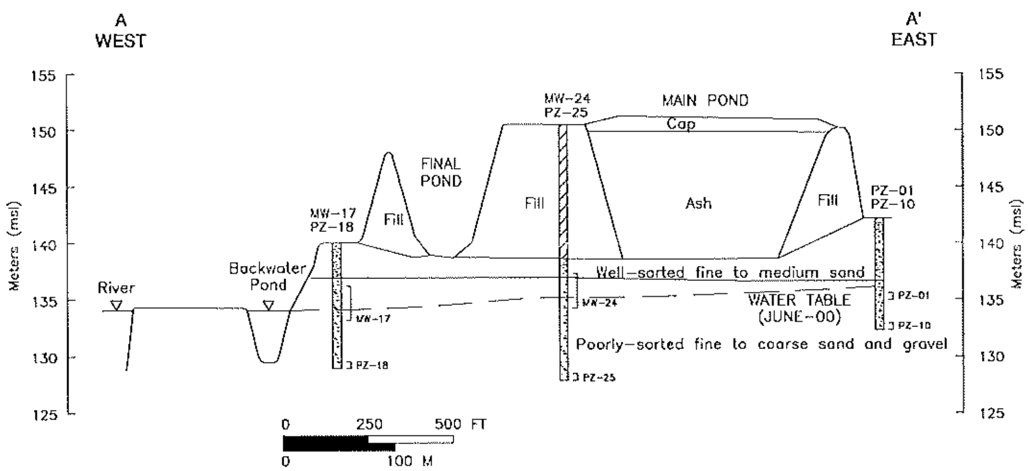


Figure 2-2
Cross-section of the HA impoundment

Table 2-1
HA South Impoundment Leachate Quality

		Porewater			
Analyte	Units	% Detects	Low	Median	High
Aluminum	mg/L	78%	<0.050	0.29	22
Arsenic	mg/L	78%	<0.005	0.019	0.35
Barium	mg/L	100%	0.022	0.18	2.7
Boron	mg/L	100%	0.12	0.77	11
Bromide	mg/L	56%	<0.10	0.20	0.69
Cadmium	mg/L	0%	<0.003	<0.003	<0.003
Calcium	mg/L	100%	25	78	226
Chloride	mg/L	100%	2.9	20	198
Chromium	mg/L	22%	<0.003	<0.003	0.006
Copper	mg/L	67%	<0.010	0.013	0.11
Fluoride	mg/L	44%	<2.0	<0.10	0.47
Iron	mg/L	33%	<0.050	<0.050	1.1
Lead	mg/L	11%	<0.005	<0.005	0.012
Magnesium	mg/L	100%	0.058	5.4	39
Manganese	mg/L	56%	<0.003	0.006	0.73
Molybdenum	mg/L	56%	<0.050	0.15	0.24
Nickel	mg/L	78%	<0.005	0.006	0.020
Nitrate	mg/L	67%	<0.15	0.15	9.9
Nitrite	mg/L	22%	<0.10	<0.10	0.37
Phosphate	mg/L	56%	<0.25	0.58	7.1
Potassium	mg/L	100%	5.1	14	173
Selenium	mg/L	0%	<0.020	<0.020	<0.020
Silicon	mg/L	100%	1.6	3.7	6.5
Silver	mg/L	0%	<0.001	<0.001	<0.001
Sodium	mg/L	100%	7.5	26	355
Strontium	mg/L	100%	0.22	1.0	9.2
Sulfate	mg/L	100%	8.4	107	478
Sulfite	mg/L	0%	<1.3	<0.25	<0.25
Thiosulfate	mg/L	0%	<0.25	<0.25	<0.25
Vanadium	mg/L	89%	<0.003	0.046	0.25
Zinc	mg/L	89%	<0.025	0.044	0.084
Carbon, inorganic	mg/L	100%	2.2	40	87
Carbon, organic	mg/L	100%	2.1	6.1	87
pH	pH	100%	6.9	8.0	12
Spec. Cond.	umhos/cm	100%	31	461	4,465

Note:

Based on nine leachate samples (EPRI, 2000)

Depth to groundwater varies from around 12 feet (3.7 meters) near the river to more than 44 feet (14 meters) upgradient of the impoundment. Groundwater elevations typically range from 440 feet to 450 feet (134 to 137 meters) above mean sea level. Groundwater flow is west toward the river (Figure 2-3). Geometric mean hydraulic conductivity is 2 x 10^{-2} cm/s in the dune sand and 9 x 10^{-2} cm/s in the outwash deposits. Using a representative hydraulic gradient of 0.005 and an estimated effective porosity of 0.33, groundwater velocities range from 300 to 1,400 ft/yr (90 to 430 m/yr).

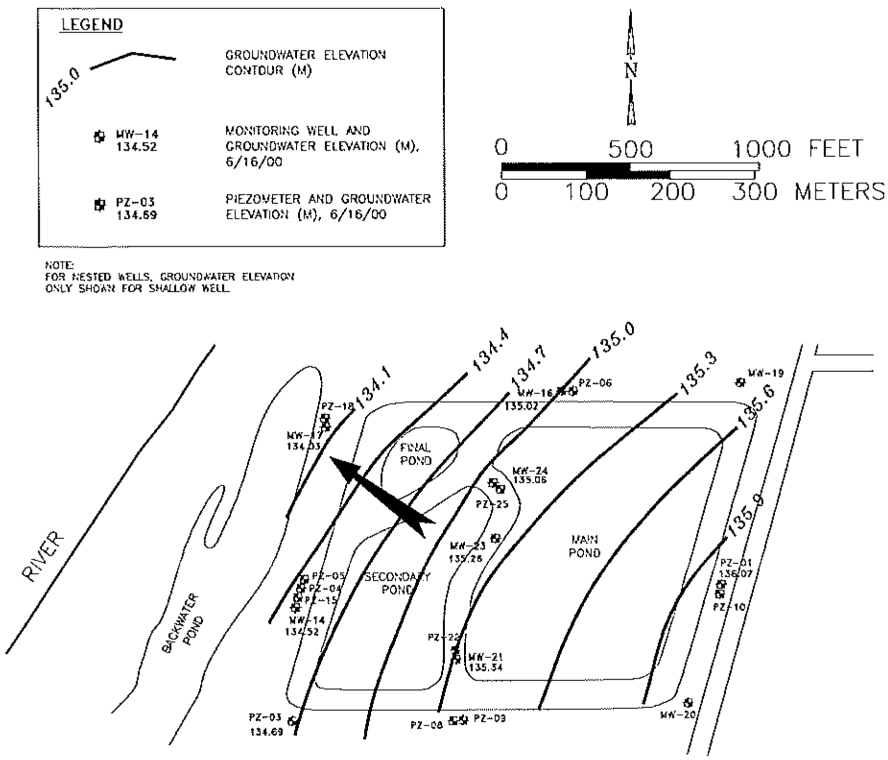


Figure 2-3
HA groundwater flow

Groundwater Quality

Groundwater quality has been monitored in 20 monitoring wells that surround the HA impoundment. The wells sampled and frequency of sampling have changed over time (Table 2-2).

Table 2-2
HA Groundwater Monitoring

Sample Dates	Wells Sampled Upgradient	Intermediate	Downgradient	Frequency	Analytes
Jun-93 through Jun-94	PZ-01 PZ-10 MW-19 MW-20	PZ-06 PZ-08 PZ-09 MW-16 MW-21 PZ-22 MW-23 MW-24 PZ-25	PZ-03 PZ-04 PZ-05 MW-14 PZ-15 MW-17 PZ-18	Monthly	Alkalinity, B, Ca, Cl, hardness, Fe, Mg, Mn, K, Na, SO4, TDS, specific conductance, pH, ORP, dissolved oxygen, temperature, groundwater elev.
Aug-94 through Dec-94	PZ-01 PZ-10 MW-20	PZ-06 PZ-08 MW-16 MW-21 PZ-22 MW-23 MW-24 PZ-25	PZ-03 PZ-04 PZ-05 MW-14 PZ-15 MW-17 PZ-18	Bimonthly	Alkalinity, B, Ca, Cl, hardness, Fe, Mg, Mn, K, Na, SO4, TDS, specific conductance, pH, ORP, dissolved oxygen, temperature, groundwater elev.
Mar-95 through Jul-98	PZ-01	PZ-06 MW-16 MW-23 MW-24 PZ-25	PZ-03 PZ-04 PZ-05 MW-14 PZ-15 MW-17 PZ-18	Quarterly	Alkalinity, B, Ca, Cl, hardness, Fe, Mg, Mn, K, Na, SO4, TDS, specific conductance, pH, ORP, dissolved oxygen, temperature, groundwater elev.
May-99 to present	PZ-01	PZ-06 MW-16 MW-23 MW-24 PZ-25	PZ-03 PZ-04 PZ-05 MW-14 PZ-15 MW-17 PZ-18	Semi-Annually	Alkalinity, B, Ca, Cl, hardness, Fe, Mg, Mn, K, Na, SO4, TDS, specific conductance, pH, ORP, dissolved oxygen, temperature, groundwater elev.

Notes:

PZ-22 and PZ-10 occasionally sampled after December 1994 for spacial studies

Groundwater Quality Prior to Closure

Most monitored analytes (boron, chloride, manganese, potassium, sodium, specific conductance, sulfate, and TDS) had higher concentrations in downgradient wells than in upgradient wells while the impoundment was in service (Table 2-3). Boron concentrations frequently exceeded the state groundwater quality standard of 2.0 mg/L in intermediate wells along the berm separating the main impoundment from the secondary and final ponds, and occasionally exceeded the standard in wells downgradient of the impoundment (Figure 2-4). Sulfate concentrations were below the state groundwater standard of 400 mg/L; however, several wells had concentrations higher than upgradient wells and the distribution of high concentrations was similar to the boron distribution. Only one other constitnent, manganese, had concentrations higher than its state groundwater standard (0.15 mg/L). The distribution of elevated manganese concentrations was different than boron and sulfate distributions. Manganese concentrations were high in all wells along the river, including PZ-03, where boron and sulfate concentrations were low. Manganese concentrations were relatively low in PZ-06 and MW-16, where boron concentrations were highest (Figure 2-4).

Table 2-3
Summary of Groundwater Quality at HA While Impoundment Was In Service

		Upgradient			Downgradient		
Analyte	Unit	min	median	max	min	median	max
Alkalinity	mg/L	110	170	250	54	160	300
Boron	mg/L	<0.20	<0.20	0.28	<0.20	0.87	3.8
Calcium	mg/L	14	60	90	29	61	110
Chloride	mg/L	4.5	10	32	2.4	34	92
Hardness	mg/L	160	230	340	150	260	480
Iron	mg/L	<0.050	<0.050	<0.050	<0.050	<0.050	1.6
Magnesium	mg/L	15	22	27	9.0	22	41
Manganese	mg/L	<0.10	<0.10	0.15	<0.10	0.19	1.5
pH	pH	6.9	7.4	8.1	6.0	7.4	8.1
Potassium	mg/L	0.062	1.2	1.9	0.60	5.1	15
Sodium	mg/L	<1.0	4.0	18	<1.0	25	89
Specific Conductance	umhos/cm	372	523	921	389	691	1,057
Sulfate	mg/L	19	24	35	23	140	290
Total Dissolved Solids	mg/L	200	290	410	200	435	670

Notes:

Wells PZ-01, PZ-10, MW-19, and MW-20 used for upgradient, all other wells included in downgradient

Means calculated from results of six sample events from June 1993 through November 1993

Excludes one low and one high pH outlier

Predictive Modeling

A negotiated settlement with the state specified an engineered clay cap for this facility unless it was demonstrated that an alternative cap would be equally effective. Therefore, groundwater flow and transport were modeled during the 1993-1994 study to predict effects of an engineered clay cap and two alternative caps constructed from locally abundant sandy soils. Two models were used to test the three cap alternatives. The Hydrologic Evaluation of Landfill Performance module (HELP: Schroeder, et al., 1984) in FOWL-GH (EPRI, 1993a) was used to predict the volume of leachate percolating from the impoundment during dewatering and after capping. The effects of this leachate on future groundwater concentrations were then simulated using a finite-element flow and transport model (PCTRANS; EPRI, 1993b). Both models were calibrated to predict boron and sulfate concentrations along a cross section parallel to A-A' in Figure 2-1, and sensitivity analysis was performed to test the effects of uncertain parameters on model results. Model input data are listed in Appendix A.

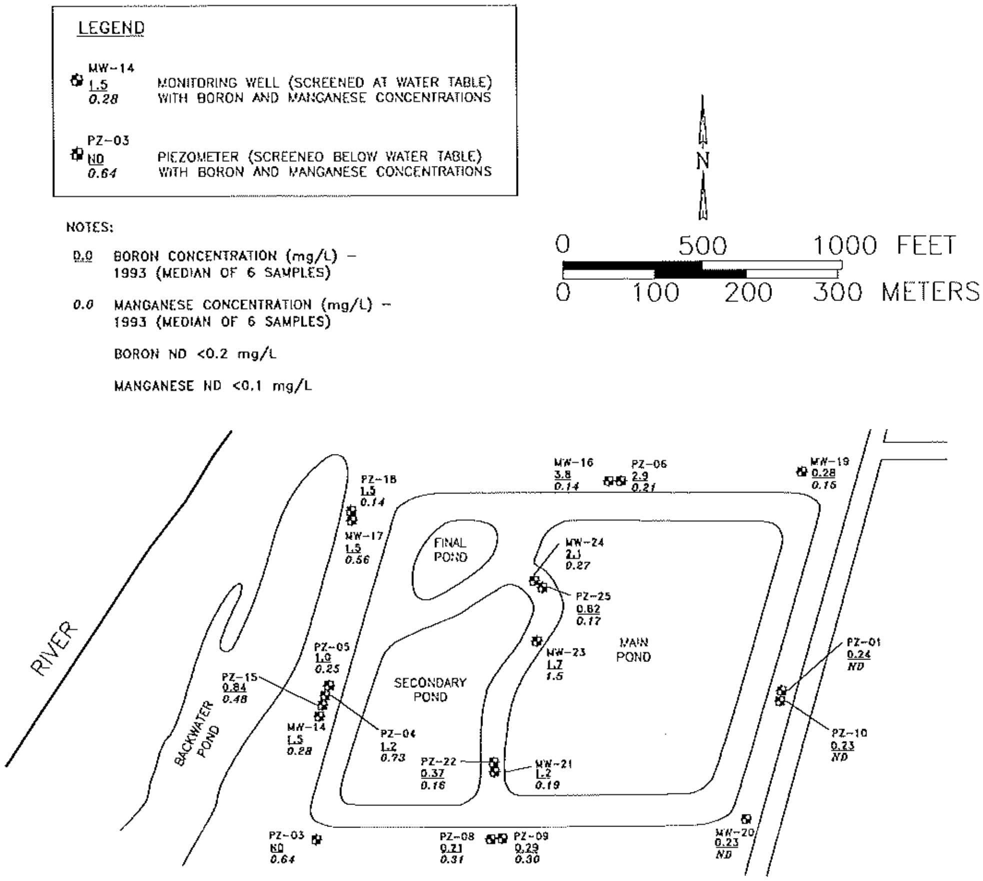
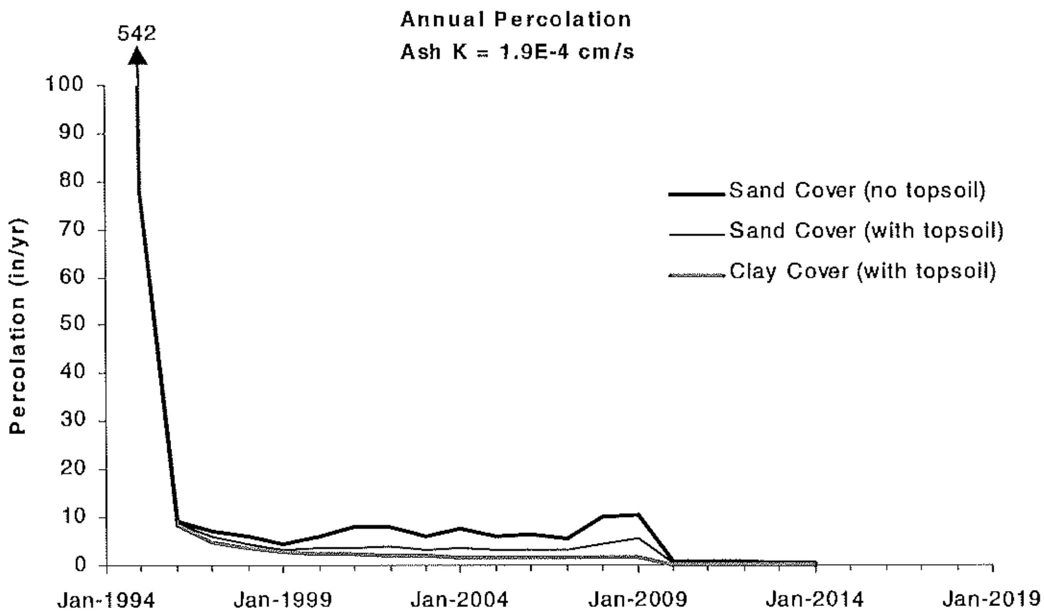


Figure 2-4
Median boron and manganese concentrations at HA while the impoundment was in service

FOWL-GH Results

Modeling was performed before the imponndment was removed from service. It was assumed that the impoundment would dewater within one year and that a cap would be constructed during the second year. Therefore, the cap was not simulated until the beginning of the third year. The impoundment actually dewatered in several months and the cap was constructed in the following summer, so that time from cessation of ash sluicing to completion of the cap was roughly one year.

Model predictions for the first two years, when no cap was simulated, suggested that ash dewatering would cause the percolation rate from the impoundment to decrease by 94 to 98 percent, depending on the hydranlic conductivity of the ash (Figure 2-5). With the caps in place, long-term percolation rates were predicted to decrease by 95 percent, relative to the active case, for a sand cover and 98 percent for a clay cover. Model results suggested that the decrease in percolation rate attributable to either cap would be small in comparison to that resulting from dewatering the impoundment.



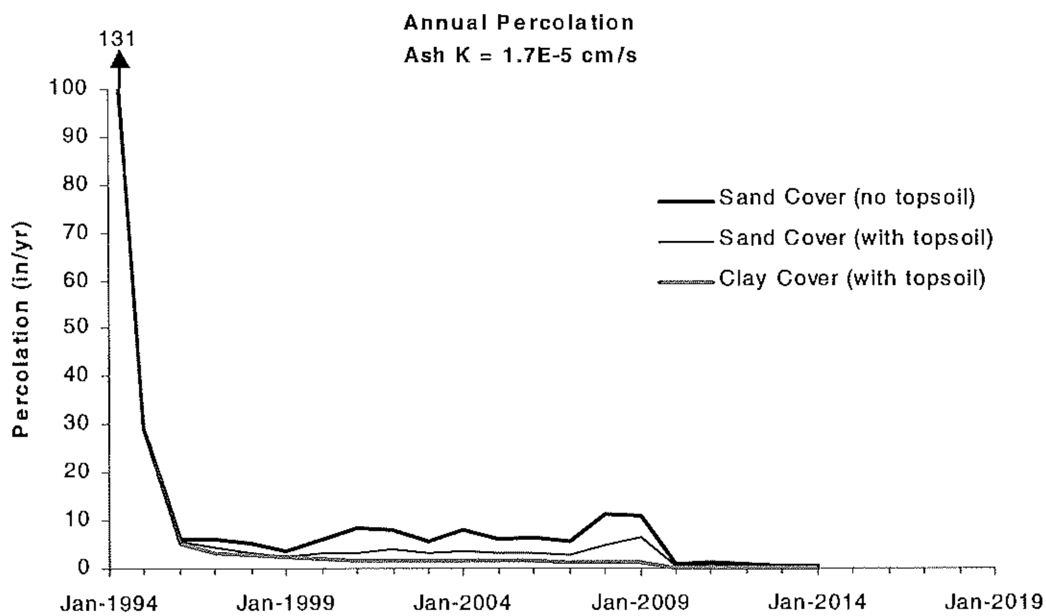


Figure 2-5
Predicted percolation rates at HA

PCTRANS Results

The percolation rates generated during the FOWL-GH simulations were input to a flow and transport model to predict the effect that reducing leachate percolation rates would have on downgradient groundwater quality. The site was modeled in profile along the transect depicted in Figure 2-2. The profile model allowed better delineation of vertical transport than possible with a plan view model. Prior to predictive simulations, the flow and transport models were calibrated to produce head and concentration distributions that matched values observed while the impoundment was in service. The maximum FOWL-GH-predicted percolation rates (calculated during the first days of the dewatering simulation) were input as leachate flux values for the active impoundment, and other model variables such as recharge and hydraulic conductivity were originally estimated from field measurements and refined during calibration. Excellent calibration results were achieved. Groundwater elevations at the six monitoring wells along the modeled profile were calibrated to within ±0.3 feet (±0.1 m) of measured values, and calibrated boron and sulfate concentrations were within the range of variability observed while the impoundment was in service.

Once the model was calibrated, it was used to predict transport of boron and sulfate after impoundment closure. For both constituents, initial leachate concentrations were assumed to remain constant while leachate percolation rate decreased. The prediction model results suggested that the boron concentrations would decrease to levels lower than groundwater quality standards within four years after removing the impoundment from service, and sulfate concentrations would decrease to background levels after five years, regardless of cap design (Figure 2-6). While there were model-predicted differences in downgradient concentrations associated with the three caps, those differences were small in comparison to the overall concentration decrease, and were within the range of variability observed at the site.

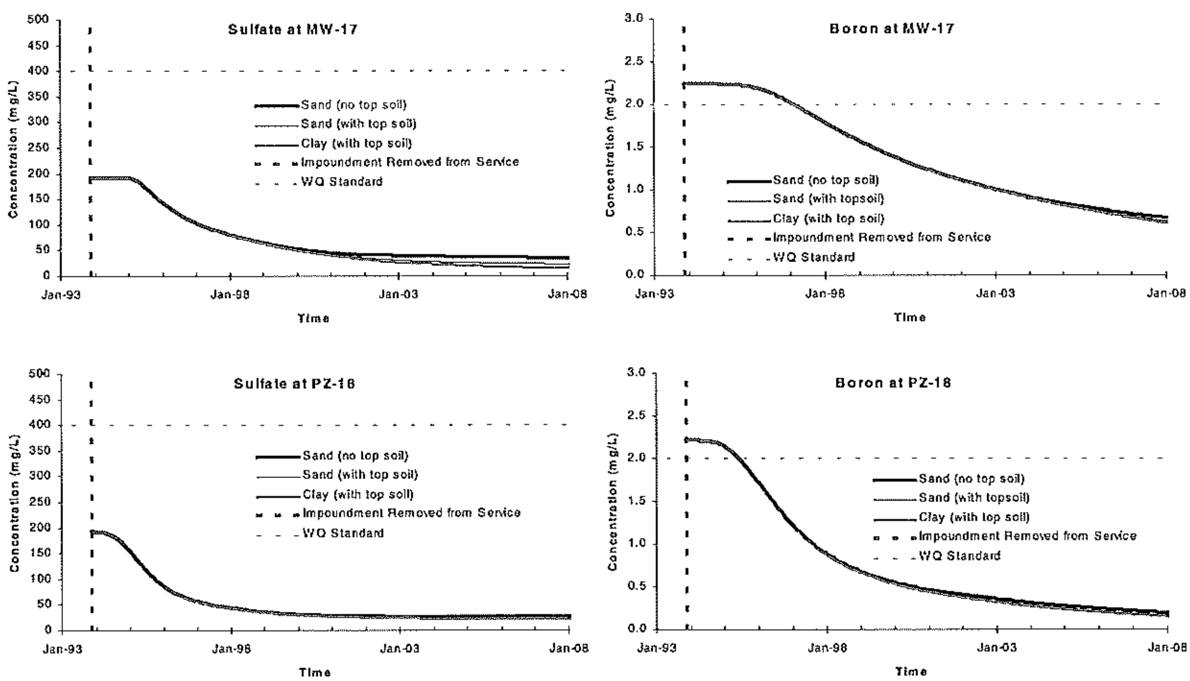


Figure 2-6
Predicted boron and sulfate concentrations at HA downgradient monitoring wells

Groundwater Quality Trends Since Closure

Based on the results of the modeling, the HA impoundment was capped with 3 to 4 feet of sandy soils that were locally available, and the cap was seeded with native grasses. Concentrations of most constituents decreased significantly during the seven years since this impoundment was closed (Table 2-4). Boron and sulfate concentration decreases have mirrored model predictions (Figure 2-7). Boron concentrations are now within state standards (Figure 2-8) and sulfate concentrations are approaching background concentrations (Figure 2-9). The only analyte that still exceeds state standards is manganese, which has decreased, although not as much as the other analytes (Figure 2-10). The differing behavior of manganese may be related to two causes: (1) manganese release and migration is controlled by redox and dissolution/precipitation reactions, and decreases in concentration resulting from ash leachate migration may be retarded relative to boron and sulfate, which are relatively mobile; and (2) some of the elevated manganese concentrations may be due to releases from native soils and sediments, as well as geochemical reactions caused by the intermixing of river water and groundwater. The latter explanation is likely for the elevated manganese concentrations observed in PZ-03, which was not affected by ash leachate from the impoundment, as indicated by low concentrations of ash indicator parameters boron and sulfate. EPRI is currently conducting research at the HA and HN impoundments to further characterize the occurrence and source of manganese in groundwater.

Table 2-4
Comparison of HA Downgradient Groundwater Quality Before and After Closure

		1993 (preclosure)			2000 (postclosure)			% change
Analyte	Unit	min	median	max	min	median	max	medians
Alkalinity	mg/L	54	160	270	140	170	280	6%
Boron	mg/L	<0.20	1.0	3.8	<0.050	0.093	1.2	-91%
Calcium	mg/L	29	62	92	47	55	95	-11%
Chloride	mg/L	2.4	34	92	<5.0	14	53	-59%
Hardness	mg/L	150	265	410	180	210	340	-21%
Iron	mg/L	<0.050	<0.050	1.6	<0.025	<0.025	0.88	---
Magnesium	mg/L	9.0	23	39	15	17	25	-26%
Manganese	mg/L	<0.10	0.18	0.73	<0.005	0.13	0.34	-26%
pH	pH	6.0	7.4	8.1	7.0	7.5	8.5	1%
Potassium	mg/L	0.89	5.8	15	0.78	2.3	3.4	-60%
Sodium	mg/L	<1.0	21	89	2.7	5.7	22	-73%
Specific Conductance	umhos/cm	389	696	1,057	365	445	751	-36%
Sulfate	mg/L	47	122	290	19	54	65	-56%
Total Dissolved Solids	mg/L	210	435	670	200	270	440	-38%

Notes:

Comparison based on wells sampled in both 1993 and 2000 (PZ-3, PZ-4, PZ-5, PZ-6, MW-14, PZ-15, MW-16, MW-17, PZ-18, PZ-22, MW-24, PZ-25)

1993 medians differ from those on Table 2-3 because wells that were not sampled in 2000 are excluded.

HA Impoundment

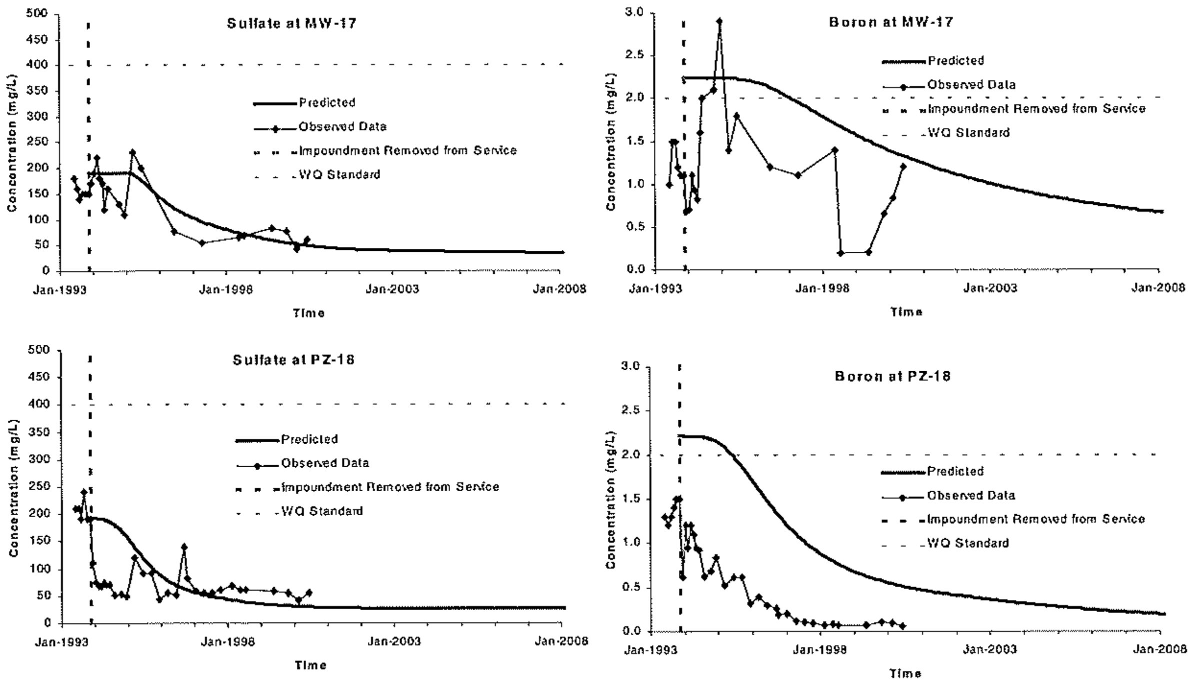


Figure 2-7
Comparison of observed and predicted boron and sulfate concentrations at HA

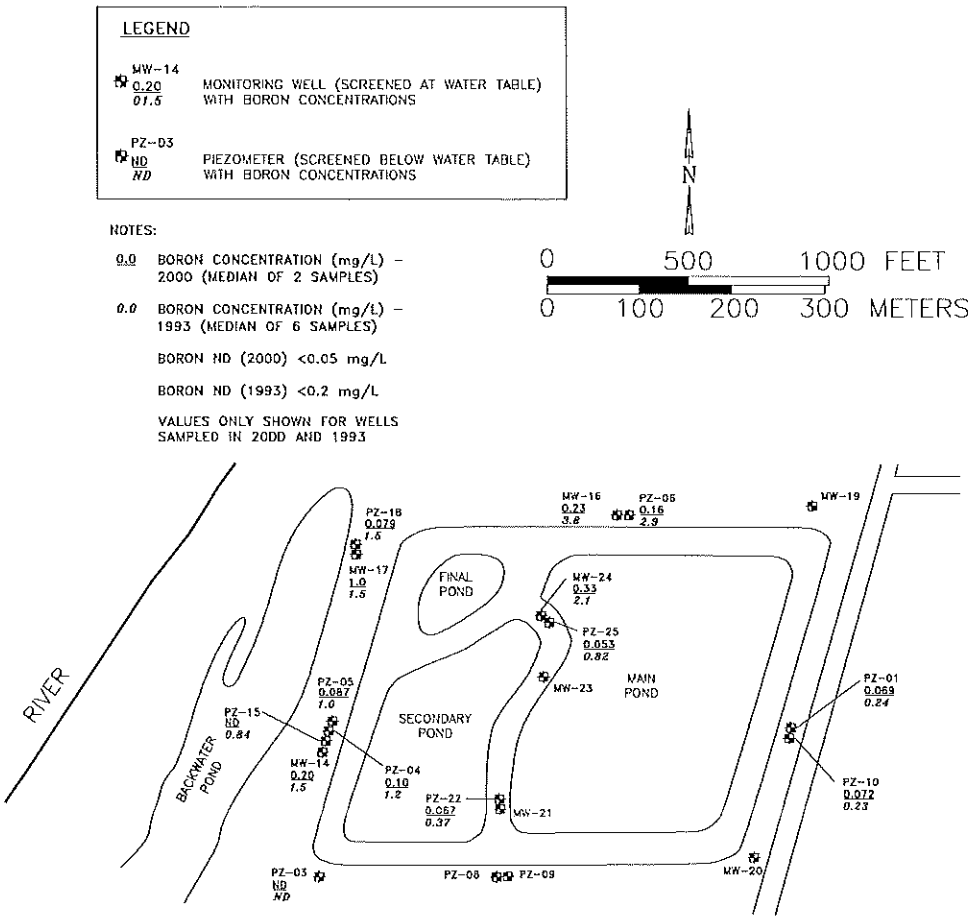


Figure 2-8
Comparison of median boron concentrations in 1993 (preclosure) and 2000 (seven years after closure)

HA Impoundment

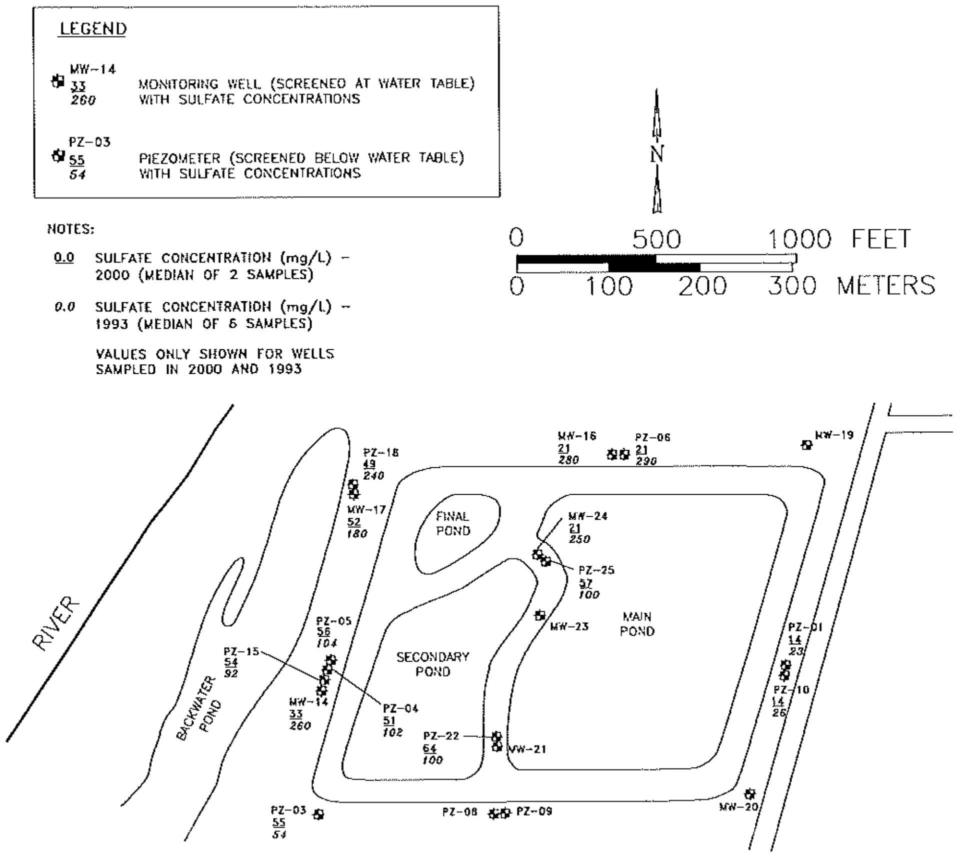


Figure 2-9
Comparison of median sulfate concentrations in 1993 (preclosure) and 2000 (seven years after closure)

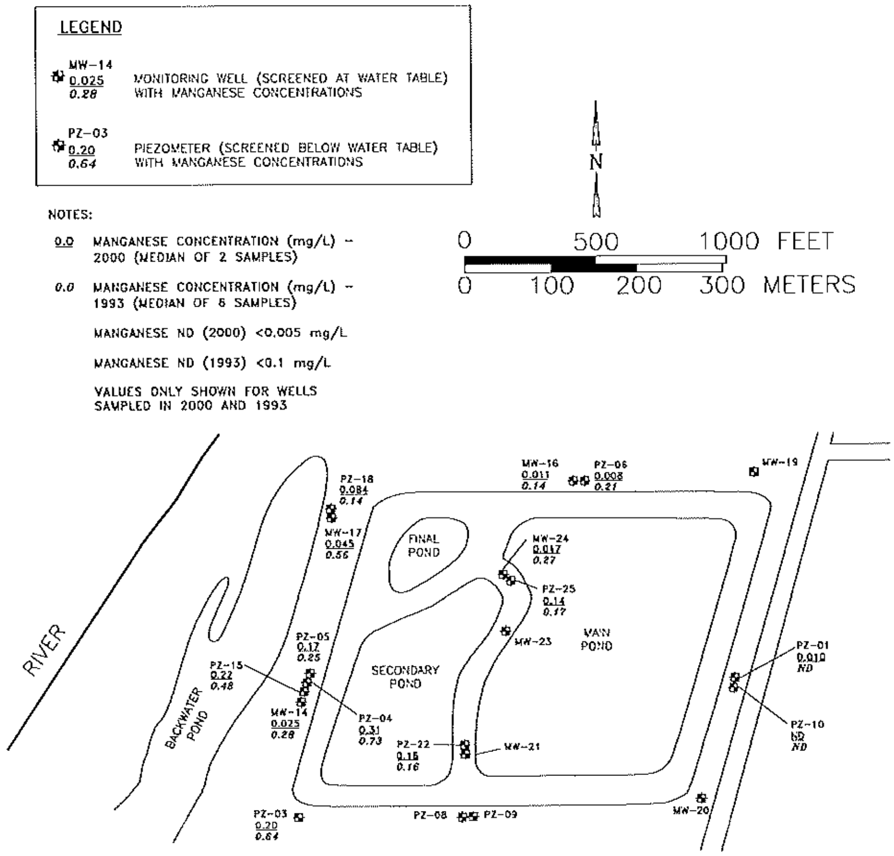


Figure 2-10
Comparison of median manganese concentrations in 1993 (preclosure) and 2000 (seven years after closure)

Groundwater was also analyzed for minor and trace elements in 1997, three years after the impoundment was removed from service. The results of that sampling (Table 2-5) showed that most analyzed trace elements (aluminum, arsenic, cadmium, chromium, copper, lead, molybdenum, nickel, silver, vanadium) were not detected in groundwater. Only two trace elements with MCLs (barium and selenium) were detected. Barium was detected in all eight downgradient samples and both upgradient samples; its maximum concentration of 0.098 mg/L was lower than the MCL of 2.0 mg/L by a factor of 20. Selenium was detected in three of the eight downgradient samples; its maximum concentration of 0.013 mg/L was lower than the MCL of 0.050 mg/L by a factor of 4. Nitrate and fluoride were the only other constituents with MCLs that were detected in groundwater. The concentration of nitrate in upgradient groundwater (35 and 53 mg/L) was higher than in downgradient groundwater (median of 19 mg/L), indicating that its source is associated with upgradient agricultural activities rather than the impoundment. The maximum fluoride concentration of 0.74 mg/L was lower than the MCL of 4.0 mg/L by a factor of 5.

Site Summary

The concentration of the primary ash indicator parameters, boron and sulfate, at the unlined HA impoundment decreased by 91 percent and 56 percent respectively in the seven years since the impoundment was removed from service, dewatered, and capped with native sandy soils. Concentrations of both indicators are now lower than state water quality standards.
The percentage decrease for sulfate is less than for boron because sulfate concentrations now occur at near-background levels. These decreases are similar to decreases predicted by groundwater modeling that was performed prior to closure, which predicted negligible difference between compacted clay and native soil caps and indicated that dewatering would have more significant effects on postclosure groundwater quality improvement than the type of cap. Postclosure sampling for trace metals in groundwater near this impoundment found that most were not present in detectable concentrations and those that were detected were at concentrations a factor of 5 or more below their respective health standards. The only MCL that was exceeded (nitrate) was also exceeded in upgradient groundwater, apparently due to agricultural activity in the area. In addition, one element (manganese) exceeds a state water quality standard, although its source may be associated with the nearby river.

Table 2-5
Results of 1997 HA Groundwater Sample Event That Included Analysis of Trace Elements

		Upgradient		Downgradient			
Analyte	Unit	min	max	min	median	max	MCL
Aluminum	mg/L	<0.050	<0.050	<0.050	<0.050	<0.050	
Arsenic	mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	0.050
Barium	mg/L	0.005	0.010	0.006	0.023	0.098	2.0
Boron	mg/L	0.079	0.079	0.036	0.13	1.2	
Bromide	mg/L	<0.10	<0.10	<0.10	<0.10	0.14	
Cadmium	mg/L	<0.003	<0.003	<0.003	<0.003	<0.003	0.0050
Calcium	mg/L	40	47	51	58	85	
Chloride	mg/L	6.0	7.7	3.5	9.4	27	
Chromium	mg/L	<0.003	<0.003	<0.003	<0.003	<0.003	0.10
Copper	mg/L	<0.010	<0.010	<0.010	<0.010	<0.010	1.3
Fluoride	mg/L	0.049	0.085	<0.10	0.24	0.74	
Iron	mg/L	<0.050	<0.050	<0.050	<0.050	0.91	
Lead	mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	zero
Magnesium	mg/L	15	17	17	19	22	
Manganese	mg/L	<0.003	<0.003	<0.003	0.18	0.41	
Molybdenum	mg/L	<0.50	<0.50	<0.50	<0.50	<0.50	
Nickel	mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	
Nitrate nitrogen	mg/L	35	53	<0.15	19	52	10
Nitrite nitrogen	mg/L	<0.10	<0.10	<0.10	<0.10	<0.10	1.0
pH (field)	pH	7.3	7.3	7.0	7.2	7.5	
Phosphate	mg/L	<0.25	<0.25	<0.25	<0.25	0.72	
Potassium	mg/L	<1.0	1.2	<1.0	2.6	4.1	
Selenium	mg/L	<0.010	<0.010	<0.010	<0.010	0.013	0.050
Silicon, diss	mg/L	6.5	7.2	2.9	4.6	5.6	
Silver	mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	
Sodium	mg/L	2.2	3.6	3.2	5.3	15	
Specific Conductance	umhos/cm	344	383	388	457	680	
Strontium	mg/L	0.041	0.077	0.053	0.29	2.0	
Sulfate	mg/L	15	20	24	51	71	
Sulfite	mg/L	<0.25	<0.25	<0.25	<0.25	<0.25	
Sulfur	mg/L	5.1	6.9	7.7	16	23	
Vanadium	mg/L	<0.003	<0.003	<0.003	<0.003	<0.003	
Zinc	mg/L	0.028	0.028	<0.025	<0.025	0.19	

Notes:

Source, EPRI (2000)

Two upgradient samples (PZ-10 and MW-20)

Eight downgradient samples (PZ-03, PZ-04, PZ-06, MW-14, PZ-15, MW-16, MW-17, MW-22)

MCLs from [http://www.epa.gov/safewater/mcl.html#inorganic], blank if none

3
HN EAST IMPOUNDMENT

Background

Site Description

The HN Power Station has two dry-bottom, pulverized coal-burning units. Unit 1 began operation in 1953 and has a capacity of 70 MW. Unit 2 began operation in 1959 and has a capacity of 210 MW. Each unit has an electrostatic precipitator for collection of fly ash, which is sluiced to an on-site coal ash management facility. Bottom ash from both units and miscellaneous low-volume plant wastes are also sluiced to the ash management facility. Prior to 1997, Units 1 and 2 utilized separate ash management facilities. Unit 1 discharged to the West Impoundment and Unit 2 discharged to the East Impoundment. The West Impoundment and unlined portions of the East Impoundment were removed from service late in 1996, and all coal combustion by-products are now managed in the lined portion of the East Impoundment. This section focuses on the unlined portion of the East Impoundment, hereafter noted as the HNE impoundment. Effects of closure on groundwater quality at the West Impoundment are described in Section 4.

In 1995 and 1996, EPRI and the sponsoring utility performed a hydrogeologic and model investigation at this impoundment to determine the effects of various closure options on downgradient groundwater quality. That investigation provided the basis for this background discussion.

The HNE impoundment is located on the south bank of a large regional river (Figure 3-1). The impoundment is situated 300 feet (100 m) from the river on a terrace that is 15 feet (5 m) higher than normal river stage. This area is characterized by relatively flat topography, except near the river and major tributary valleys. The surrounding area is rural and land use is mostly agricultural, primarily corn and soybeans; however, the parcels immediately adjacent to the impoundment are either industrial or undeveloped.

The region is humid and annual average precipitation is about 34 inches (86 cm). Peak precipitation occurs from June through September. Average temperatures range from 20°F (-6.7°C) in January to 74°F (23°C) in July.

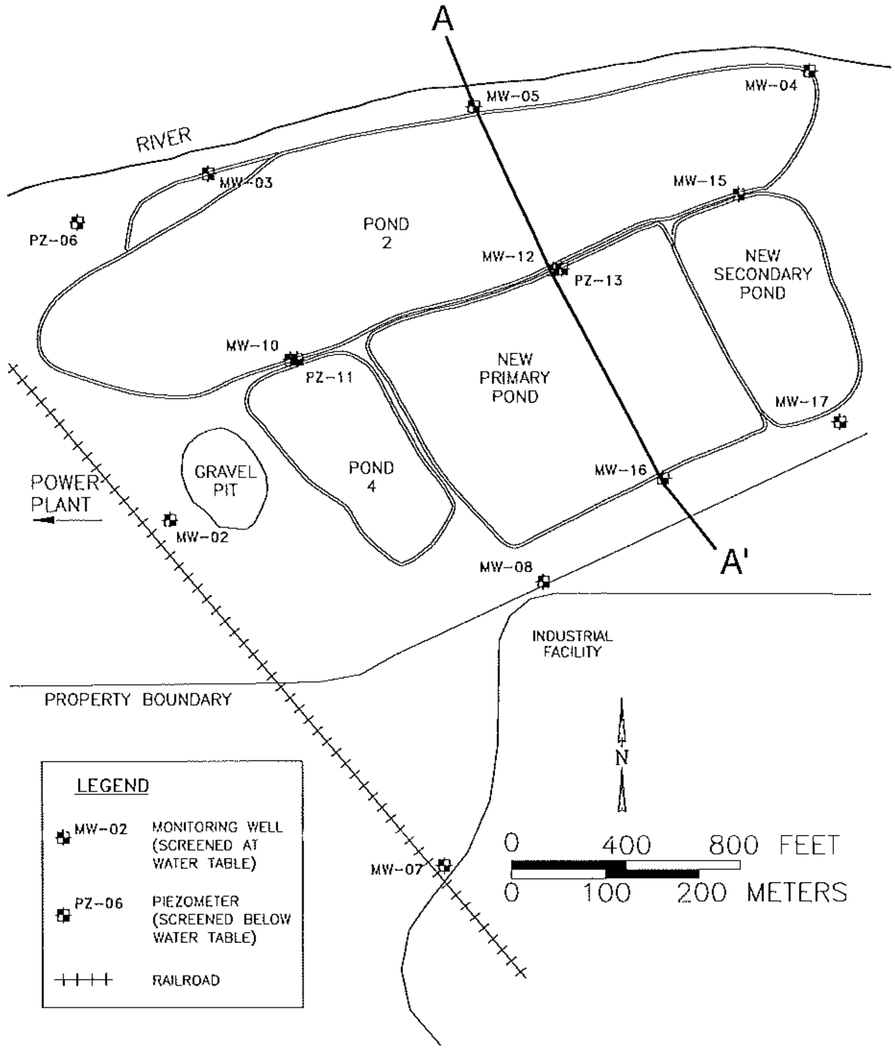


Figure 3-1
HNE site map

The impoundment consists of four ponds. Two new, clay-lined, primary and secondary ponds are currently active. There is no evidence of a leachate release from the new ponds and they are not the focus of this investigation. The other two ponds, Ponds 2 and 4, are unlined. Pond 2 was the primary management pond until it was removed from service late in 1996. The south wall of Pond 2 abuts a second river terrace, and the north, east, and west ends of the pond are contained by berms rising 40 feet (13 m) above grade. The berms are constructed from locally occurring sandy soils. Pond 4 was filled in an abandoned gravel quarry approximately 30 feet deep. Pond 2 covers an area of 30 acres (12 hectares) and Pond 4 covers an area of 8.3 acres (3.4 hectares).

During operation, cooling tower blowdown was used to sluice fly ash, bottom ash, and low volume wastes to the active pond. There was no surface water discharge during the period that Ponds 2 and 4 were active, and precipitation exceeds evaporation in this area; therefore, all sluice water (2 million gallons per day) exfiltrated via groundwater. Coal ash sluiced to Ponds 2 and 4 was a by-product of high-sulfur Illinois coal. The operational history of this impoundment is as follows:

- 1958: Pond 2 constructed.
- 1978: Pond 2 berms raised by 10 feet.
- mid 1980s: Pond 4 filled.
- 1989: Pond 2 berms raised by an additional 10 feet.
- December 1996: Pond 2 removed from service, new lined pond placed in service.

Ponds 2 and 4 were not capped after they were removed from service, to facilitate mining of ash for utilization. Both ponds dewatered by gravity drainage, and the ash is now dry at the surface and supports limited, spotty vegetation. The top of ash elevation in Pond 2 slopes downward from west to east, and is lower than the surrounding berms. There are no controls to collect storm water so all runoff collects in the lower, eastern portion of the pond where it infiltrates through the ash. The top of ash elevation in Pond 4 is flat, and lower than the berms, and storm water that collects on the surface infiltrates through the ash.

Leachate Characteristics

There are no leachate data available for this impoundment. However, groundwater beneath the impoundment was mounded while Pond 2 was in service; therefore groundwater quality observed in central wells MW-12 and MW-15, where gradients were downward from the impoundment, was assumed to be representative of leachate concentrations.

Similar to the HA impoundment, leachate from the HNE impoundment was dominated by sulfate and calcium (Table 3-1). Boron concentrations ranged from 9.4 to 22 mg/L, which were considerably higher than at HA. Factors responsible for the higher ash indicator concentrations at HNE include: (1) different coal sources—Illinois coal was used at HN, while Kentucky and western U.S. coal was used at HA; and (2) dissolved constituents washed from the ash during sluicing at HA were removed from the impoundment via pond water discharge at a NPDES permitted outfall, while there was no surface water discharge to remove dissolved constituents from HNE.

Table 3-1
Estimated HNE Leachate Quality

Analyte	Unit	Low	Median	High
Alkalinity	mg/L	38	68	140
Boron	mg/L	9.4	15	22
Calcium	mg/L	91	130	270
Chloride	mg/L	48	75	150
Hardness	mg/L	252	293	334
Iron	mg/L	<0.050	<0.050	0.050
Magnesium	mg/L	<5.0	<5.0	7.0
Manganese	mg/L	<0.005	<0.030	<0.10
pH	pH	8.2	9.6	10
Potassium	mg/L	9.6	14	18
Sodium	mg/L	35	64	110
Specific Conductance	umhos/cm	606	1,109	1,261
Sulfate	mg/L	150	340	600
Total Dissolved Solids	mg/L	490	840	960

Notes:

Based on 27 samples collected at MW-12 and MW-15 in 1995 and 1996

Hydrogeology

The HNE impoundment overlies a highly permeable aquifer that is more than 100 feet (30 meters) thick. The aquifer consists of a poorly sorted mixture of silty-sandy gravel, with cobble zones and with boulders up to several feet in diameter (Figure 3-2).

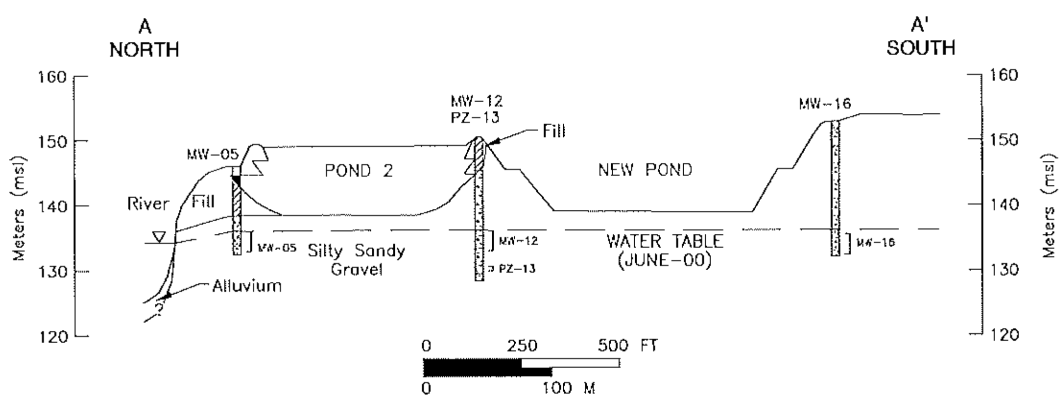


Figure 3-2
HNE cross-section

Depth to groundwater varies from 20 to 40 feet (6 to 12 meters) along the river to more than 50 feet (15 meters) south of the impoundment. Groundwater elevations typically range from 445 feet to 450 feet (136 to 137 meters) above mean sea level. Groundwater flow while the impoundment was active was radial from a mound that existed beneath Pond 2. The mound is dissipating, and was still evident at MW-12 in 2000 (Figure 3-3). Geometric mean hydraulic conductivity of the aquifer is 1.7 x 10^{-1} cm/s. Gradients near the impoundment range from 0.003 to 0.0008. Assuming an effective porosity of 0.2, groundwater velocities range from 700 to 2,600 ft/yr (210 to 790 m/yr).

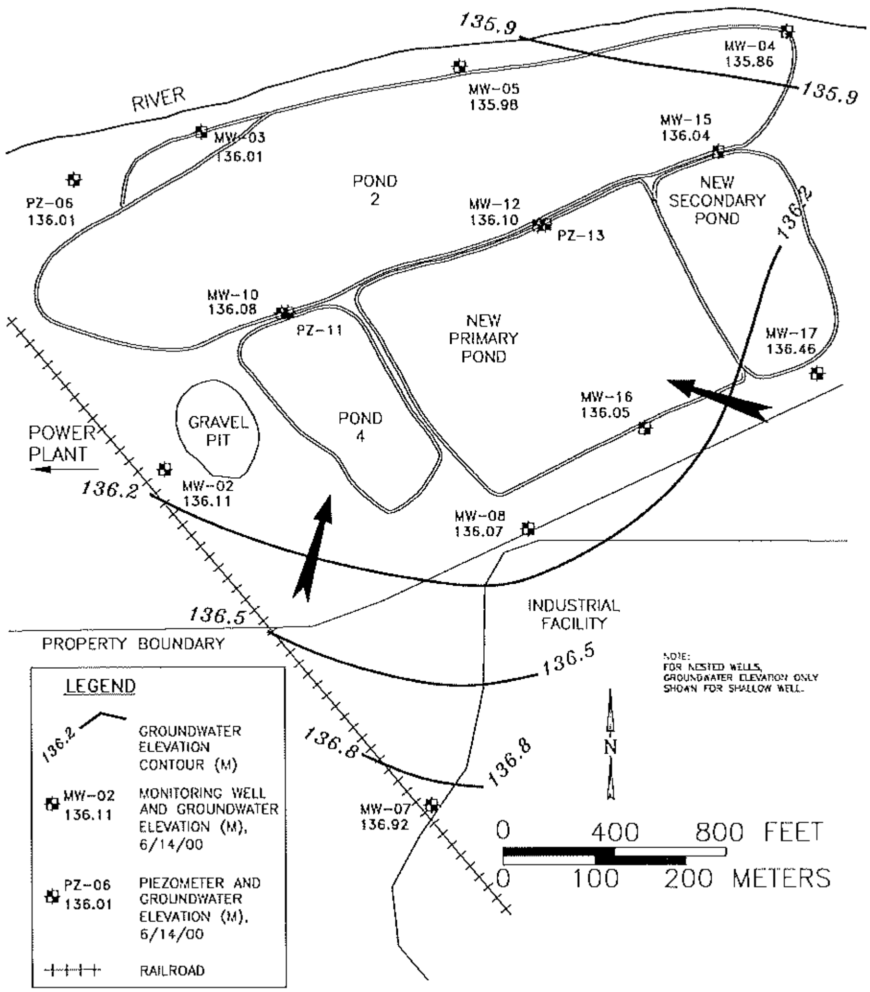


Figure 3-3
HNE groundwater flow

Groundwater Quality

Groundwater quality has been monitored in 14 monitoring wells that surround the impoundment. Only the frequency of monitoring has changed over time (Table 3-2).

Table 3-2
HNE Groundwater Monitoring

Sample Dates	Wells Sampled: Upgradient	Wells Sampled: Intermediate	Wells Sampled: Downgradient	Frequency	Analytes
Mar-95 through June-96	MW-07 MW-08 MW-17	MW-02 MW-10 PZ-11 MW-12 PZ-13 MW-15 MW-16	MW-03 MW-04 MW-05 PZ-06	Monthly	Alkalinity, B, Ca, Cl, hardness, Fe, Mg, Mn, K, Na, SO4, TDS, specific conductance, pH, ORP, dissolved oxygen, temperature, groundwater elev.
Nov-96 to present	MW-07 MW-08 MW-17	MW-02 MW-10 PZ-11 MW-12 PZ-13 MW-15 MW-16	MW-03 MW-04 MW-05 PZ-06	Quarterly	Alkalinity, B, Ca, Cl, hardness, Fe, Mg, Mn, K, Na, SO4, TDS, specific conductance, pH, ORP, dissolved oxygen, temperature, groundwater elev.

Notes:

MW-08 is currently considered upgradient. However elevated boron concentrations indicate that it was affected by mounding while Pond 2 was in service.

Groundwater Quality Prior to Closure

The following discussion describes groundwater quality prior to 1997, when Pond 2 was active and before the new lined ponds were in service.

Groundwater downgradient of Ponds 2 and 4 had higher average concentrations of boron, chloride, potassium, sodium, specific conductance, sulfate, and total dissolved solids than upgradient groundwater (Table 3-3). Upgradient groundwater had higher concentrations of alkalinity and magnesium than downgradient groundwater.

Sulfate and boron exceeded state groundwater standards. Sulfate slightly exceeded the standard at least once at eight wells, with no value greater than 600 mg/L. Boron consistently exceeded the standard at all four downgradient wells, all intermediate wells, and at MW-16. Boron concentrations were also high, relative to background, at MW-08. The high boron concentrations at MW-08 and MW-16 are evidence that the mounding beneath Pond 2 reversed groundwater flow as far south as those wells.

Boron and sulfate concentrations were highest in monitoring wells toward the eastern end of Pond 2 (Figures 3-4 and 3-5), where water accumulated and leachate flux was highest. Comparison of downgradient groundwater to Pond 2 surface water samples indicated similarities in that boron, chloride, potassium, sodium, sulfate, and total dissolved solids were elevated relative to npgradient groundwater while magnesium and alkalinity were lower than upgradient concentrations (Table 3-3). However, the waters were dissimilar in that concentrations of the two ash indicator parameters, boron and sulfate, were a factor of two to four higher in the groundwater samples than in the pond water. This increase was due to additional leaching as the pond water percolated through the ash and possibly due to differences in the composition of the ash at the time the pond water samples were obtained compared to older ash at depth in the pond. The pond water also had detectable concentrations of iron and manganese, which were typically below detection limits in upgradient and downgradient groundwater samples.

Table 3-3
HNE Groundwater Quality

		Upgradient			Downgradient			Pond
Analyte	Unit	min	median	max	min	median	max	Water
Alkalinity	mg/L	110	325	380	34	100	450	140
Boron	mg/L	<0.10	0.16	0.98	0.11	11	22	4.9
Calcium	mg/L	67	110	120	42	120	270	102
Chloride	mg/L	8.5	27	52	18	70	170	121
Hardness	mg/L	452	452	452	252	379	593	
Iron	mg/L	<0.050	<0.050	0.48	<0.050	<0.050	1.6	0.67
Magnesium	mg/L	32	49	54	<5.0	9.4	58	28
Manganese	mg/L	<0.005	<0.030	<0.10	<0.005	<0.030	0.27	0.045
pH	pH	7.0	7.9	8.6	6.6	8.6	10	
Potassium	mg/L	1.3	2.0	3.3	1.0	14	56	11
Sodium	mg/L	5.9	11	13	6.6	60	110	91
Specific Conductance	umhos/cm	599	833	966	450	1,074	1,399	
Sulfate	mg/L	55	67	90	56	320	600	230
Total Dissolved Solids	mg/L	330	520	560	340	785	1,000	748

Notes:

Wells MW-07 and MW-17 used for upgradient, all other wells included in downgradient

Means calculated from results of 18 sample events from December 1994 through November 1996

Pond water values are average of five samples taken during the week of 2/26/95

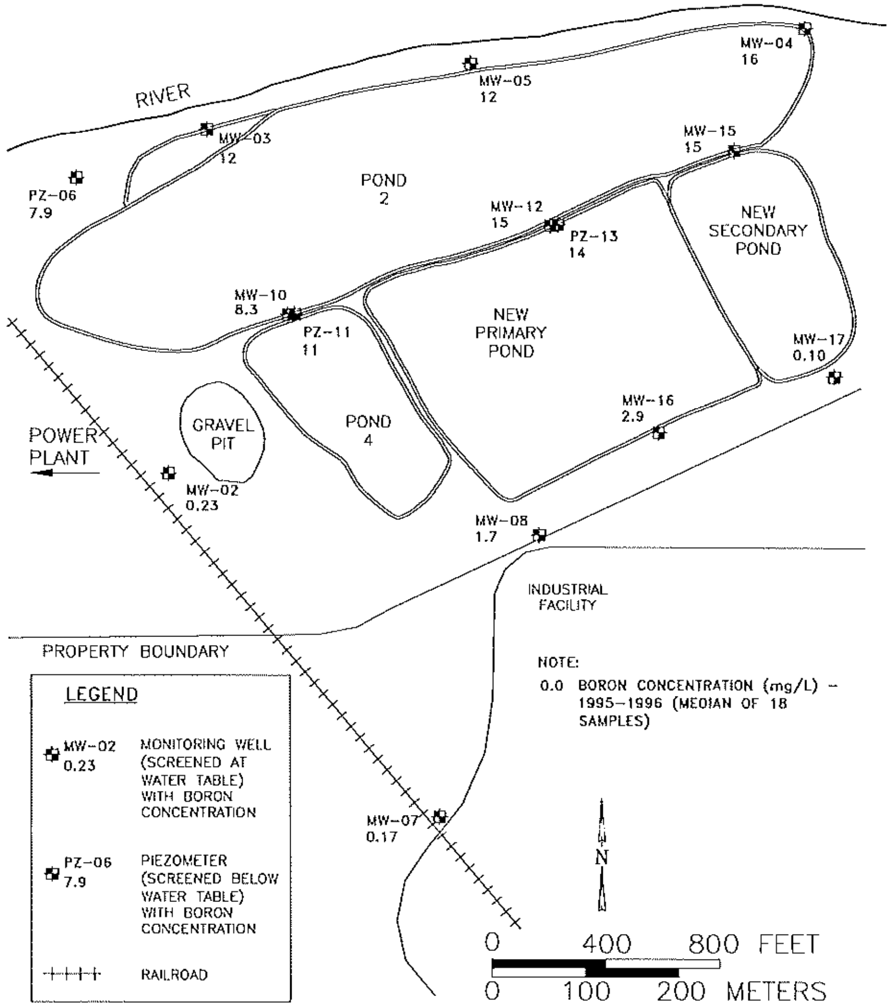


Figure 3-4
HNE boron distribution while Pond 2 was in service

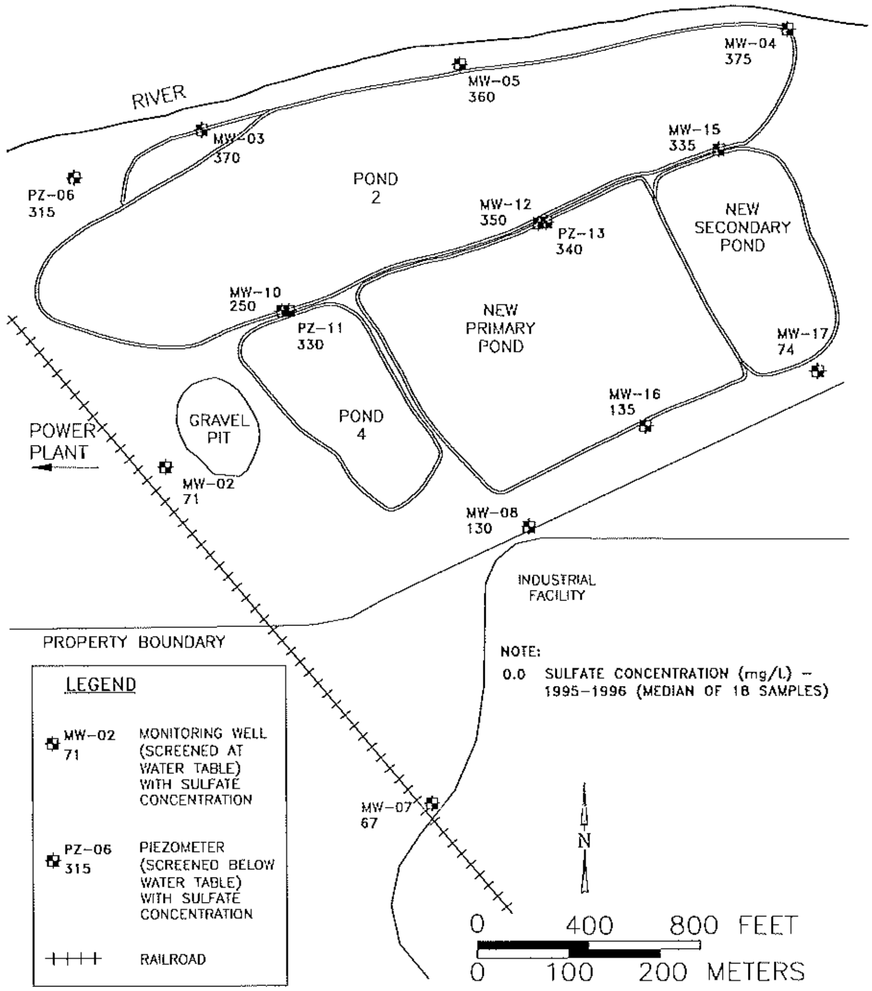


Figure 3-5
HNE sulfate distribution while Pond 2 was in service

Predictive Modeling

A negotiated settlement with the state specified an engineered clay cap for this facility unless it was demonstrated that an alternative cap would be equally effective. Therefore, groundwater flow and transport at this impoundment was modeled during the 1995-1996 study to predict effects of four alternative closure strategies: the compacted clay cap, two variations of a cap constructed from native sandy soil, and no cap. Two models were used to test the four closure alternatives. The Hydrologic Evaluation of Landfill Performance model (HELP; Schroeder, et al., 1994) was used to predict the volume of leachate percolating from the impoundment to groundwater during dewatering and after capping. The effects of this leachate on future groundwater quality were then simulated using a finite-difference flow model (MODFLOW; McDonald and Harbaugh, 1988) coupled with a transport model (MT3D; Zheng, 1992). The HELP model was calibrated to allow a percolation flux of 2 million gallons per day, similar to the volume of sluice water exiting the impoundment. MODFLOW and MT3D were calibrated to predict observed boron concentrations while the impoundment was in service, and sensitivity analysis was performed to test the effects of uncertain parameters on model results. Model input data are listed in Appendix A.

HELP Results

Modeling assumed that the ash would be allowed to dewater for one year, at which time a cap could be added. Model predictions for the first year, when no cap was simulated, suggested that percolation rate from the impoundment would decrease by 98 percent due to dewatering (Figure 3-6). Long-term percolation rates were predicted to decrease by more than 99 percent, relative to the in-service impoundment, regardless of the type of cap simulated or even if no cap was simulated, suggesting that the primary cause of decreasing percolation was dewatering the impoundment.

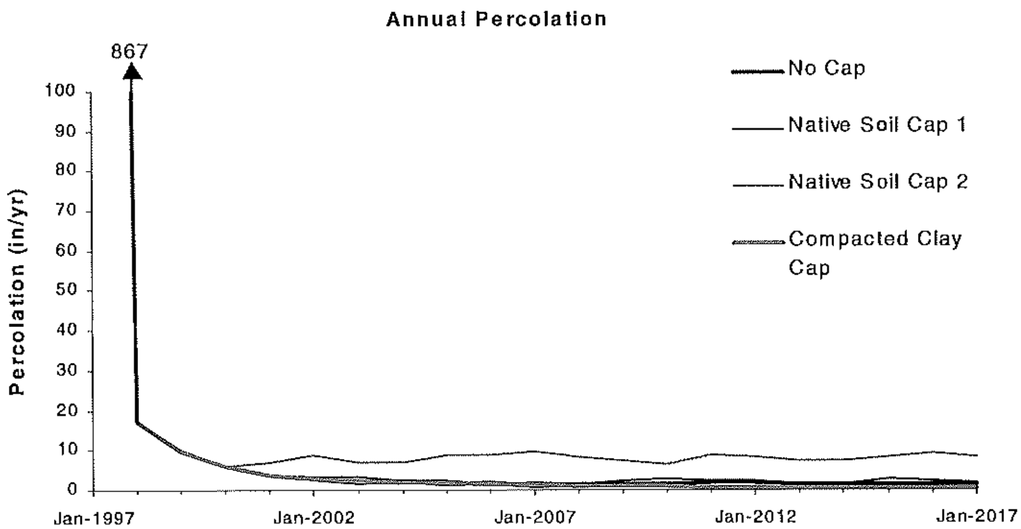


Figure 3-6
Predicted percolation rates for HNE

MODFLOW/MT3D Results

The percolation rates generated during the HELP simulations were input to a flow and transport model to predict the effect that reducing leachate percolation rates would have on groundwater quality—specifically boron concentrations. The site was modeled three-dimensionally to simulate horizontal and vertical variations in flow and transport. Prior to predictive simulations, the flow and transport models were calibrated to produce head and concentration distributions that matched values observed while the impoundment was in service. The maximum HELP-predicted percolation rates (calculated during the first days of the dewatering simulation) were input as leachate flux values for the active impoundment, and other model variables such as recharge and hydraulic conductivity were originally estimated from field measurements and refined during calibration. Calibration results were good for a model of this scope. Groundwater elevations at all monitoring wells surrounding the site were within 1.4 feet (0.4 m) of their respective target elevations (i.e., elevations representative of typical flow conditions while the impoundment was in service), and most were within 0.6 feet (0.2 m). Calibrated boron concentrations were within the range of variability observed while the impoundment was in service.

Initial boron concentrations in the leachate were assumed to remain constant while leachate percolation rate decreased. The modeling predictions suggested that three of the four closure scenarios would result in similar decreases in groundwater concentrations. One scenario (native soil cap 2) was not as effective as the other closure scenarios because a shallow gravel layer modeled as part of the cap facilitated rapid downward drainage (percolation). The no cap, native soil 1, and compacted clay cap scenarios all resulted in modeled boron concentrations decreasing to levels lower than groundwater quality standards within six years after removing the impoundment from service (Figure 3-7). It was assumed that sulfate would meet its standard more quickly than boron because it is slightly more mobile than boron and because its concentration exceeded the standard by less than a factor of 1.5, while boron concentrations exceeded the standard by as much as a factor of 10.

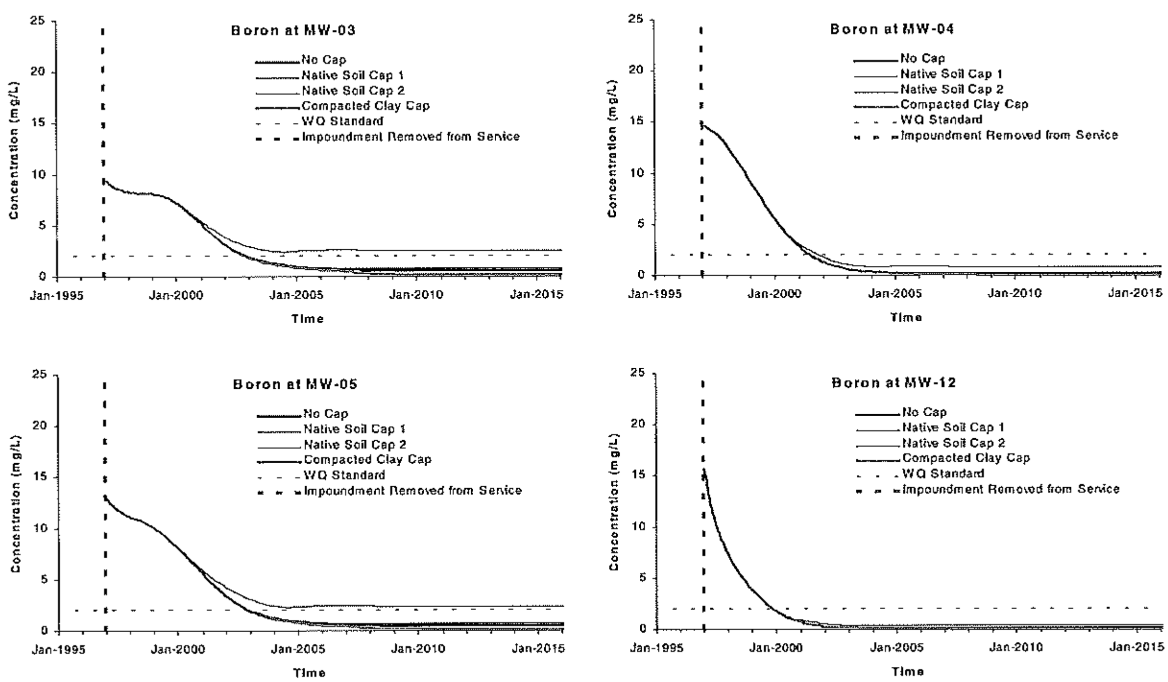


Figure 3-7
Predicted boron concentrations at HNE

Groundwater Quality Trends Since Closure

Concentrations of most constituents have decreased significantly during the four years since this impoundment was removed from service (Table 3-4). Boron concentration decreases have been consistent with model predictions at all wells except for MW-05, and recently MW-04 (Figure 3-8). Boron concentrations are now within the state standard of 2.0 mg/L at all wells except downgradient wells MW-03, MW-04, MW-05, and intermediate well PZ-13 (Figure 3-9). Sulfate concentrations are within the state standard of 400 mg/L at all wells (Figure 3-10).

Inversely to the decreasing boron and sulfate concentrations, median alkalinity and magnesium concentrations in intermediate and downgradient wells increased by more than 200 percent. Prior to closure, concentrations of these parameters in leachate were very low, and since groundwater beneath this impoundment was replaced by leachate, intermediate and downgradient groundwater concentrations were also very low. The return of these parameters to near background concentrations indicates the partial dissipation of the mound, and return of groundwater flow beneath portions of this impoundment.

Table 3-4
Comparison of Downgradient Groundwater Quality Before and After Removing HNE From Service

		1995-96 (preclosure)			2000 (postclosure)			% change
Analyte	Unit	min	median	max	min	median	max	medians
Alkalinity	mg/L	34	82	330	52	275	380	235%
Boron	mg/L	0.11	12	22	0.059	1.5	12	-88%
Calcium	mg/L	42	120	270	72	110	140	-8%
Chloride	mg/L	24	72	170	26	55	65	-24%
Hardness	mg/L	252	363	508	220	385	500	6%
Iron	mg/L	<0.05	<0.05	0.83	<0.025	<0.025	0.43	---
Magnesium	mg/L	<5	2.3	49	<0.5	35	43	1400%
Manganese	mg/L	<0.03	<0.03	0.27	<0.005	<0.005	0.035	---
pH	pH	7.1	8.7	10	6.6	7.3	9.6	-17%
Potassium	mg/L	1.5	15	56	1.9	12	33	-23%
Sodium	mg/L	6.6	62	110	12	34	56	-46%
Specific Conductance	umhos/cm	450	1,081	1,399	599	923	1,187	-15%
Sulfate	mg/L	56	340	600	72	150	260	-56%
Total Dissolved Solids	mg/L	340	800	1,000	400	605	800	-24%

Notes:

Comparison based on wells sampled in both 1995-96 and 2000 (MW-02, MW-03, MW-04, MW-05, PZ-06, MW-10, PZ-11, MW-12, PZ-13, MW-15)

1995-96 medians differ from those on Table 3-3 because MW-08 and MW-16 are excluded.

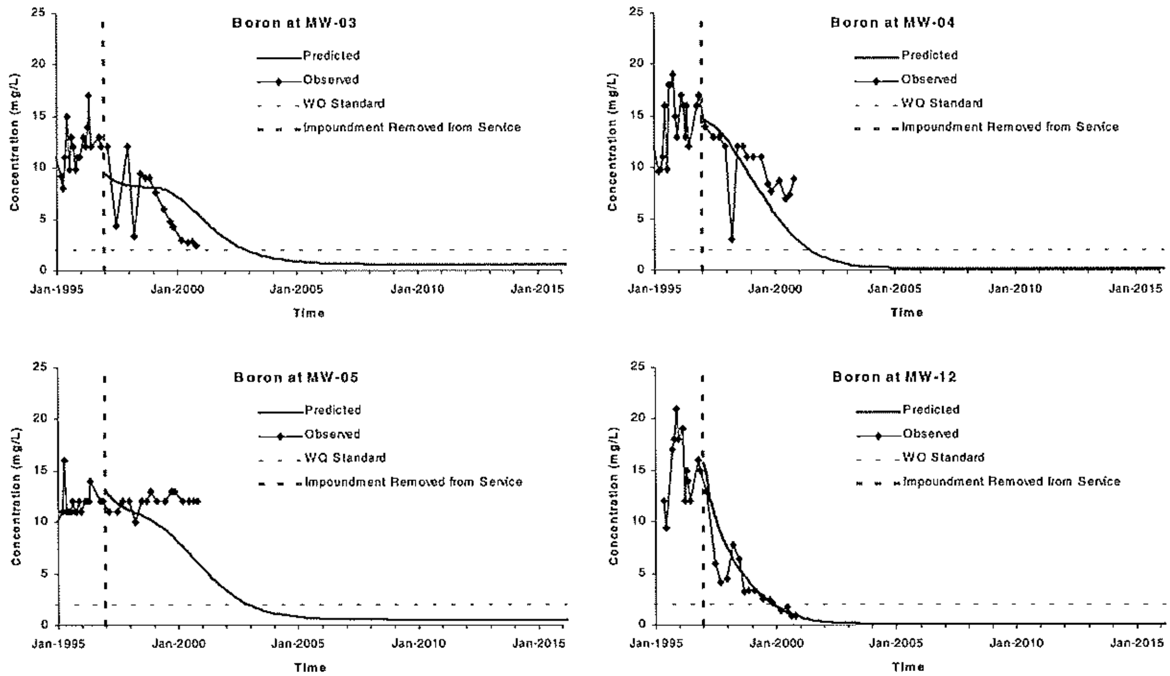


Figure 3-8
Comparison of model-predicted and observed boron concentrations at HNE

Site Summary

Downgradient concentrations of the primary ash indicator parameters, boron and sulfate, decreased by 88 and 56 percent since Ponds 2 and 4 were removed from service and dewatered. These decreases have occurred even though the ponds were not capped, and no steps were taken to collect surface runoff of precipitation that falls on the dewatered ponds.

The modeling assumed that runoff would be collected before it could pond and infiltrate. MW-04 and MW-05 are downgradient of the portion of Pond 2 where the top of ash elevation is lowest, and the relatively high boron concentrations observed at these wells are a result of leaching caused by infiltrating precipitation runoff that collects in the low area. Boron and sulfate concentrations at other wells that are not downgradient of the low area have decreased as predicted by the modeling and are on track to be within regulatory standards within the predicted six-year time period.

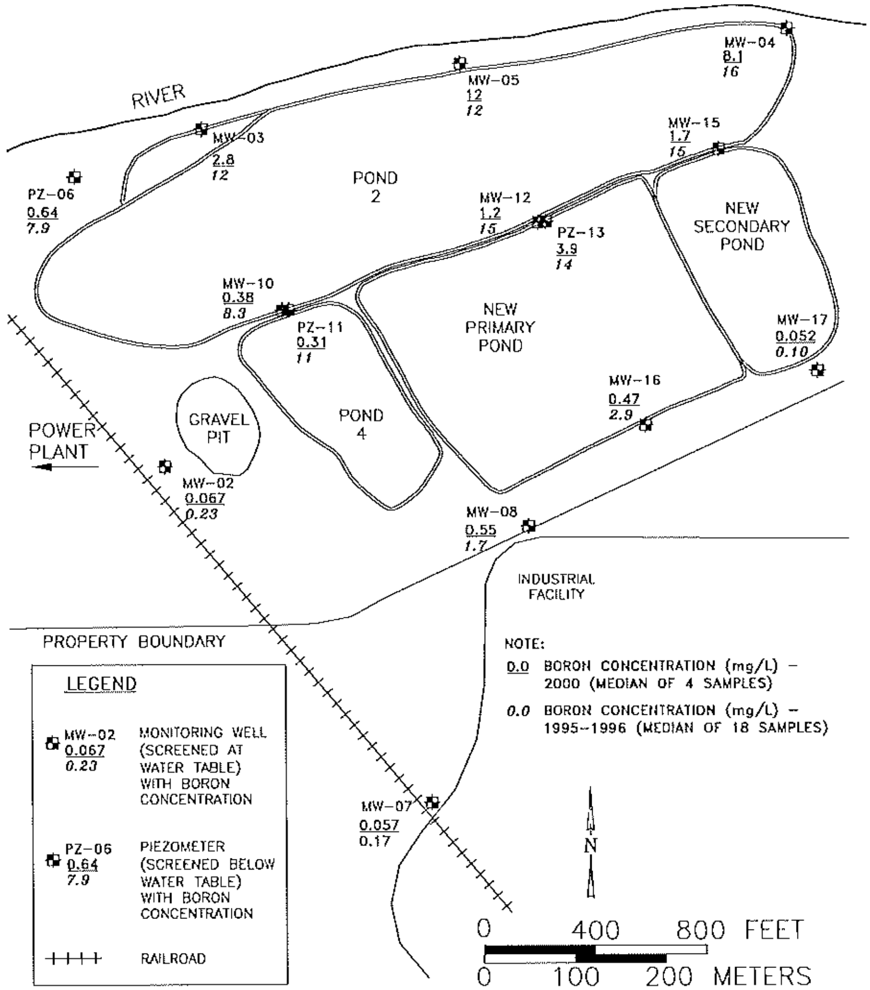


Figure 3-9
Comparison of median boron concentrations in 1995-96 (preclosure) and 2000 (four years after closure)

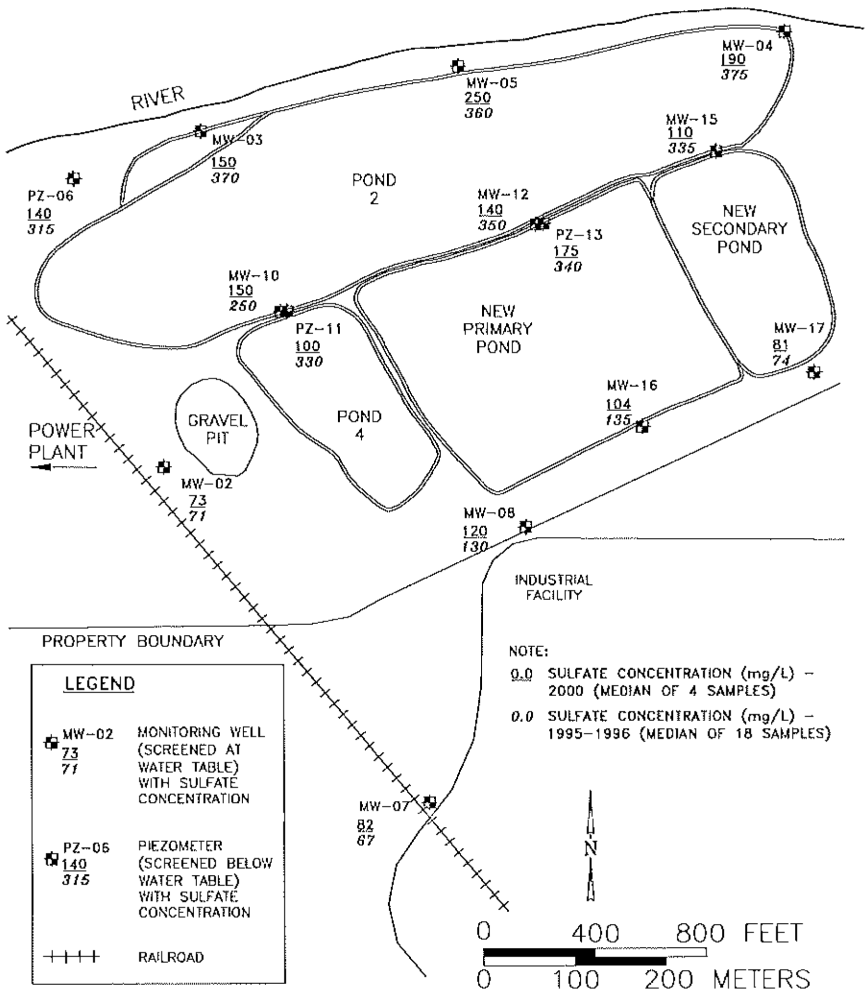


Figure 3-10
Comparison of median sulfate concentrations in 1995-96 (preclosure) and 2000 (four years after closure)

4
HN WEST IMPOUNDMENT

Background

Site Description

The HNW impoundment was the subject of an EPRI investigation in 1996 and 1997, in which hydrogeology was mapped and modeling was performed to predict the effects of various closure options on downgradient groundwater quality. That investigation provided the basis for this background discussion.

The HNW impoundment is located on the south bank of a large regional river (Figure 4-1), about 4,000 feet (1,200 m) downriver from the HNE impoundment discussed in Section 3. The HNW impoundment is situated less than 200 feet (60 m) from the river. The eastern third of the impoundment is on a terrace about 15 feet (5 m) higher than normal river stage. The western two-thirds are on lowlands that are about 5 to 10 feet (2 to 3 m) higher than normal river stage. The surrounding area is rural and land use is either agricultural, primarily corn and soybeans, or natural area.

The region is humid and annual average precipitation is about 34 inches (86 cm). Peak precipitation occurs from June through September. Average temperatures range from 20°F (-6.7°C) in January to 74°F (23°C) in July.

The impoundment consists of three ponds. Pond 1, at the eastern end of the impoundment, primarily contains bottom ash and slag. This pond overlies the terrace. Pond 3, in the central portion of the impoundment, contains mixed coal ash. A polishing pond is located at the western end of the impoundment. All of the ponds are bermed, and the berms, which are 15 feet (5 m) above grade, were constructed from locally occurring sandy soils. Pond 1 covers an area of 9.3 acres (3.8 hectares), Pond 3 covers an area of 16.4 acres (6.6 hectares), and the secondary pond covers an area of 4.7 acres (1.9 hectares).

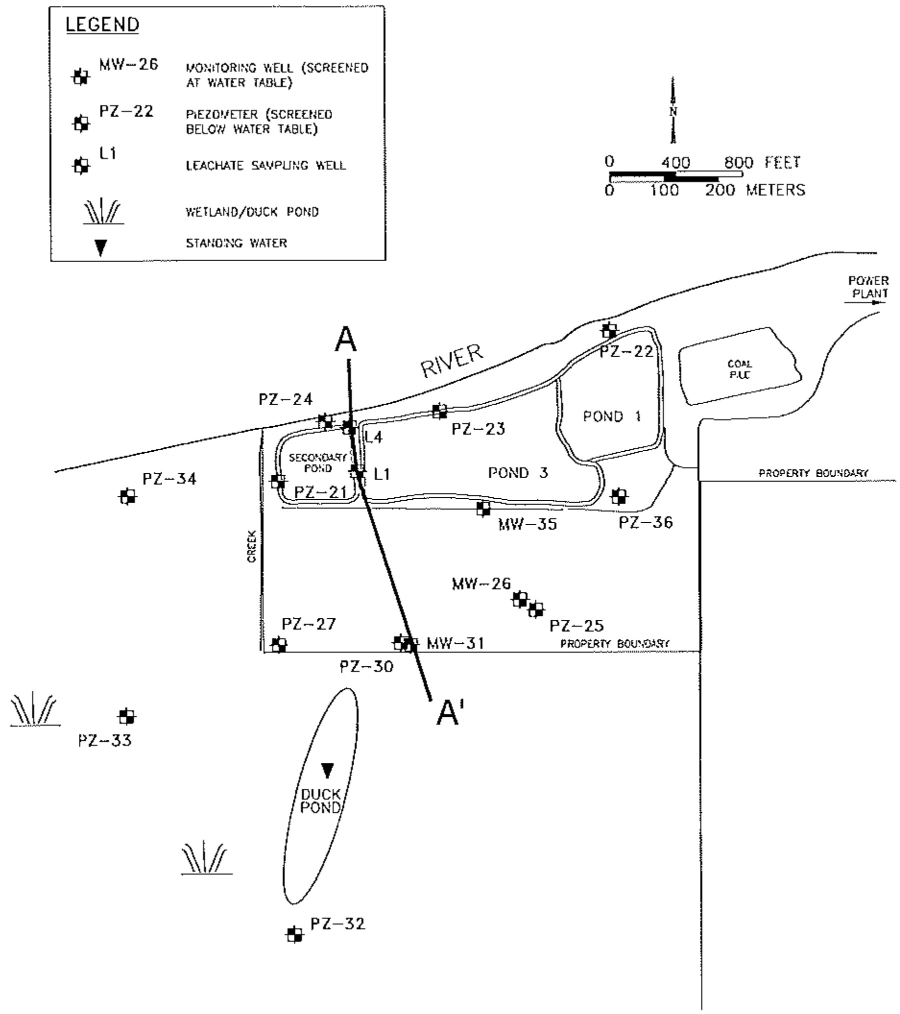


Figure 4-1
HNW site map

During operation, cooling tower blowdown was used to sluice fly ash, bottom ash, and low-volume wastes to the impoundment. There was initially a surface water discharge from this impoundment; however, that discharge stopped after the impoundment was reworked in 1989. At the time that the impoundment was removed from service, all sluice water (1.4 million gallons per day) exfiltrated from the impoundment via groundwater. Coal ash sluiced to this impoundment was a by-product of high-sulfur Illinois coal. The operational history of this impoundment is as follows:

- 1952-55: Pond 1 constructed.
- 1968: Pond 3 constructed.
- 1979: Berms raised by 3 feet.
- 1988-89: Ponds 1 and 3 consolidated and secondary pond added. It was after this consolidation that surface water discharge from the impoundment ceased.

- December 1996: Impoundment removed from service.

The HNW impoundment was not capped after it was removed from service, to facilitate future mining and utilization of ash. The impoundment dewatered by gravity drainage and the ash is dry over most of the surface, except for the secondary pond, which always contains water, and the western end of Pond 3, where runoff water collects and is ponded most of the year. The ash at the surface supports limited, spotty vegetation. The top of ash elevation in Pond 1 is uneven but has no prononnced slope. The top of ash elevation in Pond 3 slopes downward from east, where the ash inlet was located, to west. Ash elevation in all three ponds is lower than the surrounding berms, and there are no controls to collect storm water, so all runoff collects in the lower, western portion of Pond 3 where it infiltrates through the ash.

Leachate Characteristics

This facility has two leachate sampling wells (L1 and L4), both on the berm separating Pond 3 and the secondary pond. The leachate wells are finished in silty materials underlying the ash. Because the impoundment is mounded and gradients beneath the impoundment are downward, water sampled from these wells is leachate; however, the leachate is migrating through organic sediments prior to entering the well screen, and some constituents such as manganese, iron, and trace metals may undergo chemical reactions. L4 has higher boron and sulfate concentrations than L1, which may also be affected by infiltrating pond water from Pond 3 and the secondary pond. Infiltrating pond water will tend to reduce boron and sulfate concentrations by dilution because the pond water has lower concentrations than the leachate.

Leachate at HNW has similar quality to leachate at HNE, where ash is derived from the same coal, although concentrations of boron, calcium, and sulfate are higher in L4 than in the monitoring wells where leachate quality was observed at HNE. The relatively high concentrations in L4 leachate are due to long groundwater contact time with the ash. Migration rates in this area are relatively slow due to a silty clay layer that underlies this portion of the HNW impoundment.

Table 4-1
HNW Leachate Quality

Analyte	Unit	L1 Low	L1 Median	L1 High	L4 Low	L4 Median	L4 High
Alkalinity	mg/L	360	490	580	86	120	150
Boron	mg/L	8.6	15	17	15	27	31
Calcium	mg/L	160	190	210	95	200	260
Chloride	mg/L	76	88	130	52	76	97
Hardness	mg/L	635	645	656	648	659	669
Iron	mg/L	1.2	1.8	5.8	<0.050	0.13	0.72
Magnesium	mg/L	38	42	46	13	19	25
Manganese	mg/L	6.5	7.2	7.7	0.32	0.51	1.8
pH	pH	6.5	7.3	8.3	7.6	8.6	9.0
Potassium	mg/L	0.75	1.5	2.6	13	17	19
Sodium	mg/L	65	81	86	54	61	73
Specific Conductance	umhos/cm	1,036	1,395	1,438	1,159	1,340	1,540
Sulfate	mg/L	110	150	250	400	520	690
Total Dissolved Solids	mg/L	910	960	980	970	1,100	1,300

Notes:

Based on 14 samples collected at L1 and L4 from September 1995 through November 1996

Hydrogeology

The HNW impoundment overlies an aquifer consisting of a poorly sorted mixture of silty-sandy gravel that is more than 100 feet (30 meters) thick (Figure 4-2). The western half of the impoundment directly overlies this aquifer. The eastern half overlies an old river channel, subsequently filled with fine-grained and organic channel deposits, which overlies the aquifer.

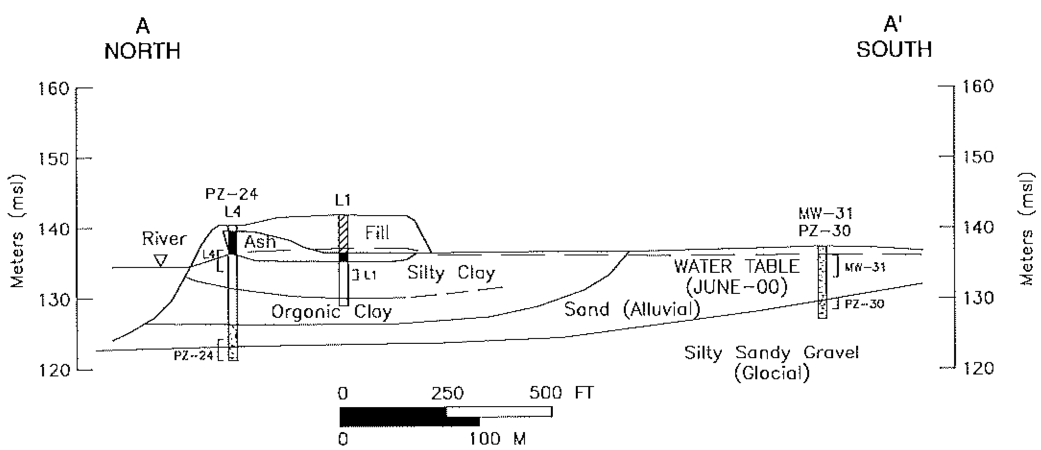


Figure 4-2
HNW cross-section

Depth to groundwater varies from less than 5 feet (3 meters) in the lowlands south and west of the impoundment to 15 to 20 feet (5 to 7 meters) in wells on the impoundment berm and in upland wells. Groundwater elevations typically range from 442 feet to 447 feet (135 to 136 meters) above mean sea level.

Soil boring logs performed down the middle of Pond 3 indicate that the base of ash elevation is as low as 439.5 feet (134.0 m) msl. Assuming a representative groundwater elevation of 446.5 feet (136.1 m), these data indicate as much as 7 feet (2.1 m) of saturated ash beneath this impoundment (Table 4-2). This is the only one of the three impoundments reported here where ash remained saturated (i.e., below the water table) after dewatering.

Table 4-2
Saturated Ash Thickness at HNW

Boring	Location	Elevation Base of Ash	Elevation Groundwater	Saturated Ash Thickness (ft)
L1	Pond 3 & Secondary berm	439.5	446.5	7.0
L2	Pond 1, center	454.4	446.5	none
L3	Pond 3, south central	444.8	446.5	1.7
L4	Pond 3 & Secondary berm	446.4	446.5	0.1
L5	Pond 3, east	444.3	446.5	2.2
23S	Pond 3, center	441.3	446.5	5.2
A1	Pond 3, north central	446.8	446.5	none
A3	Pond 3, east	443.7	446.5	2.8

Notes:

Groundwater elevation representative as of June 2000

Groundwater flow while the impoundment was active was radial from a mound that existed beneath Pond 3. As of 2000, that mound persists based on evidence at wells PZ-23 and MW-35, although in a reduced level (Figure 4-3). There appears to be a pronounced gradient toward the southwest, a conservation area with wetlands where surface water elevations are managed. Wells PZ-32 and PZ-33 are in this area and are therefore downgradient of the impoundment; however, ash indicator parameter concentrations in these wells are low, either because the ash plume has not migrated that far or because it is discharging to the wetlands between the impoundment and these wells. Therefore, PZ-32 and PZ-33 are used in this comparison as background wells.

Geometric mean hydraulic conductivity in the sand and gravel aquifer is 1.7×10^{-2} cm/s. Gradients distant from the mound range from 0.001 to 0.0006. Assuming an effective porosity of 0.2, groundwater velocity near this impoundment is 50 to 90 ft/yr (15 to 27 m/yr).

Groundwater Quality

Groundwater quality has been monitored in 14 monitoring wells that surround the impoundment. The wells sampled and frequency of monitoring has varied over time (Table 4-3).

Groundwater Quality Prior to Closure

Groundwater downgradient of the impoundment had higher median concentrations of boron, chloride, potassium, specific conductance, sulfate, and TDS than background groundwater (Table 4-4). Background groundwater had higher concentrations of alkalinity, iron, and manganese than downgradient groundwater.

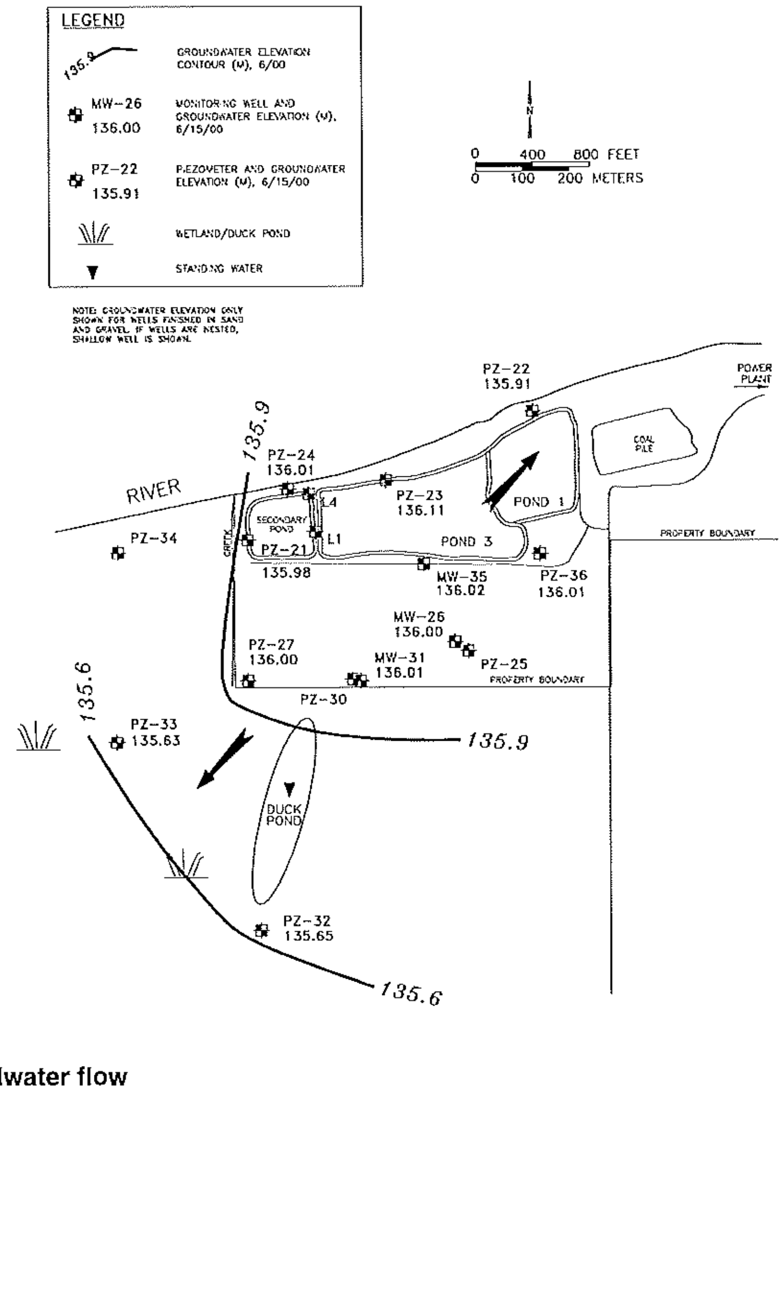


Figure 4-3
HNW groundwater flow

Table 4-3
HNW Groundwater Monitoring

Sample Dates	Wells Sampled: Background	Wells Sampled: Intermediate	Wells Sampled: Downgradient	Frequency	Analytes
Sep-95 through Dec-96	PZ-32 PZ-33 PZ-34	PZ-25 MW-26 PZ-27 PZ-30 MW-31	PZ-21 PZ-22 PZ-23 PZ-24	Monthly	Alkalinity, B, Ca, Cl, hardness, Fe, Mg, Mn, K, Na, SO4, TDS, specific conductance, pH, ORP, dissolved oxygen, temperature, groundwater elev.
Feb-97 to present	PZ-32 PZ-33 PZ-34	PZ-25 MW-26 PZ-27 PZ-30 MW-31 MW-35 PZ-36	PZ-21 PZ-22 PZ-23 PZ-24	Quarterly	Alkalinity, B, Ca, Cl, hardness, Fe, Mg, Mn, K, Na, SO4, TDS, specific conductance, pH, ORP, dissolved oxygen, temperature, groundwater elev.

Notes:

PZ-32, PZ-33, and PZ-34 first sampled in September 1996

MW-35 and PZ-36 first sampled in November 1999

Table 4-4
HNW Groundwater Quality

		Background			Downgradient			Pond
Analyte	Unit	min	median	max	min	median	max	Water
Alkalinity	mg/L	210	300	550	28	200	390	84
Boron	mg/L	<0.10	0.070	0.20	0.12	4.6	10	12
Calcium	mg/L	64	92	270	56	100	200	140
Chloride	mg/L	13	37	51	14	49	130	67
Hardness	mg/L	287	335	394	274	410	800	
Iron	mg/L	<0.025	0.092	4.7	<0.050	<0.005	6.6	0.088
Magnesium	mg/L	30	35	60	<5.0	38	96	18
Manganese	mg/L	0.10	0.60	2.2	<0.030	0.025	1.4	0.033
pH	pH	6.8	7.0	7.9	6.6	7.6	9.5	
Potassium	mg/L	0.72	2.1	2.8	<0.50	3.1	32	15
Sodium	mg/L	21	24	40	9.8	24	78	57
Specific Conductance	umhos/cm	535	739	1,507	275	887	2,070	
Sulfate	mg/L	<5.0	50	190	34	215	600	360
Total Dissolved Solids	mg/L	330	440	960	290	665	1,300	715

Notes:

Wells PZ-32, PZ-33, and PZ-34 used for upgradient, all other wells, except L1 and L4, included in downgradient

Means calculated from results of 14 sample events from September 1995 through November 1996

Boron, iron, manganese, sulfate, and TDS exceeded state groundwater standards. Sulfate exceeded the standard at least once in four wells. Boron consistently exceeded the standard at all four downgradient wells and three of the five intermediate wells. TDS exceeded the standard once at PZ-23. Iron and manganese exceeded the standard in wells finished beneath the organic river channel fill sediments; however, the exceedances did not correlate with proximity to the impoundment, indicating that these exceedances are naturally occurring.

Boron and sulfate concentrations were highest in downgradient monitoring wells between the impoundment and the river (Figures 4-4 and 4-5). In addition, a plume of elevated boron and sulfate concentrations extended southwest from the impoundment toward PZ-27.

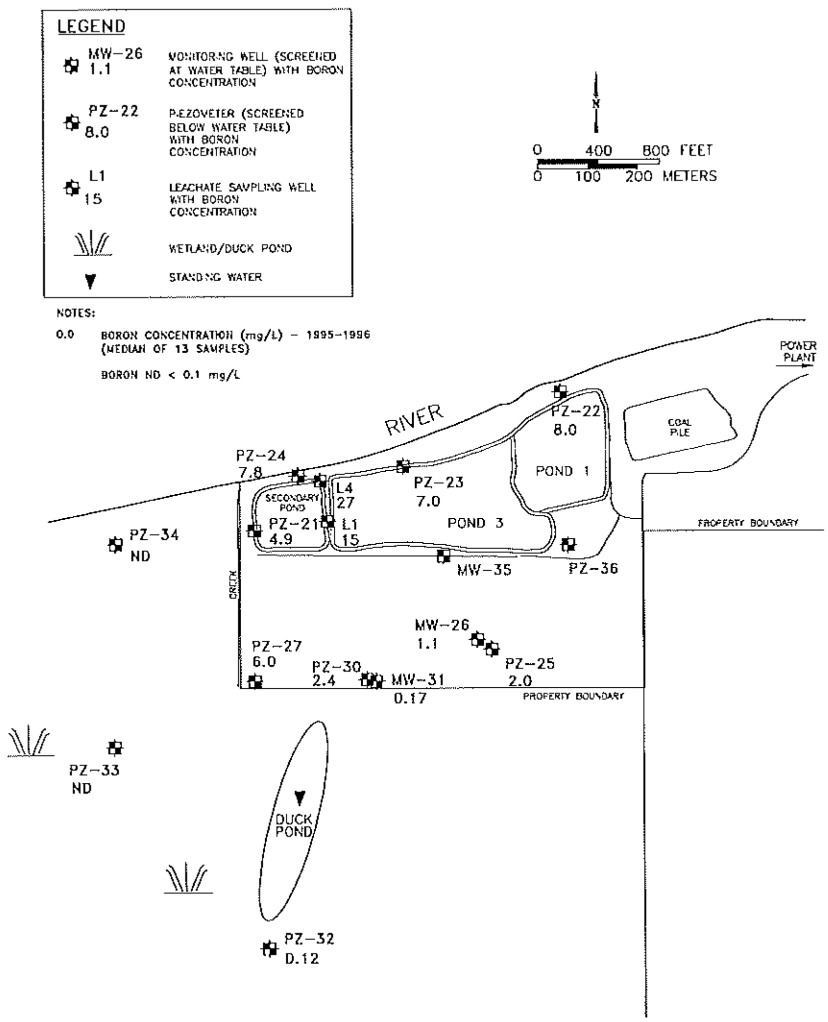


Figure 4-4
HNW boron distribution while Pond 3 was in service

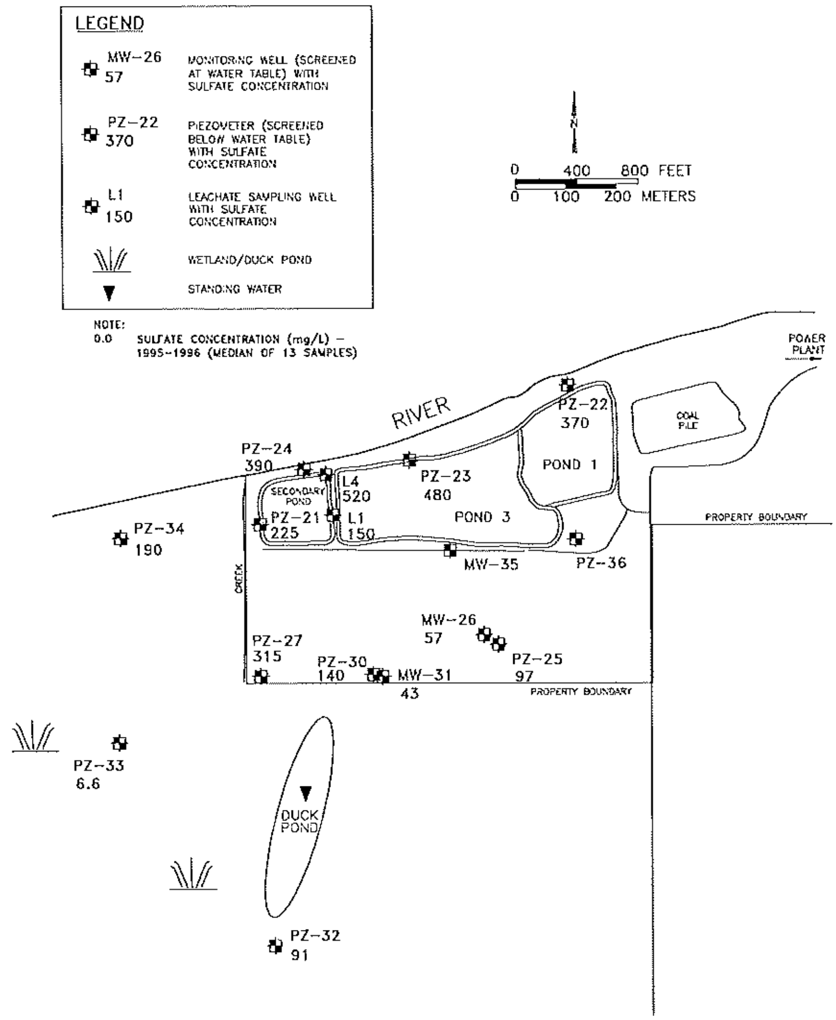


Figure 4-5
HNW sulfate distribution while Pond 3 was in service

Predictive Modeling

As at the other two impoundments, a clay cap was required for this impoundment unless a demonstration could be made that an alternative would be equally effective. Therefore, groundwater flow and transport were modeled during the 1996-1997 study to predict effects of three alternative closure strategies: the compacted clay cap, a native soil cap, and no cap. Two models were used to test the three closure alternatives. The Hydrologic Evaluation of Landfill Performance model (HELP; Schroeder, et al., 1994) was used to predict the volume of leachate percolating from the impoundment to groundwater during dewatering and after closure. The effects of this leachate on future groundwater concentration were then simulated using a finite-difference flow model (MODFLOW; McDonald and Harbaugh, 1988) coupled with a transport model (MT3D; Zheng, 1992). The HELP model was calibrated to allow a percolation flux of 1.4 million gallons per year, similar to the volume of sluice water exiting the impoundment. MODFLOW and MT3D were calibrated to predict observed boron concentrations while the impoundment was in service, and sensitivity analysis was performed to test the effects of uncertain parameters on model results. Model input data are listed in Appendix A.

HELP Results

Modeling assumed that the ash would be allowed to dewater for one year, at which time a cap could be added. Model predictions for the first year, when no cap was simulated, suggested that percolation rate from the impoundment would decrease by 98 percent due to dewatering (Figure 4-6). Long-term percolation rates were predicted to decrease by more than 99 percent, relative to the active case, regardless of the type of cap simulated or even if no cap was simulated, suggesting that the primary cause of decreasing percolation was dewatering the impoundment.

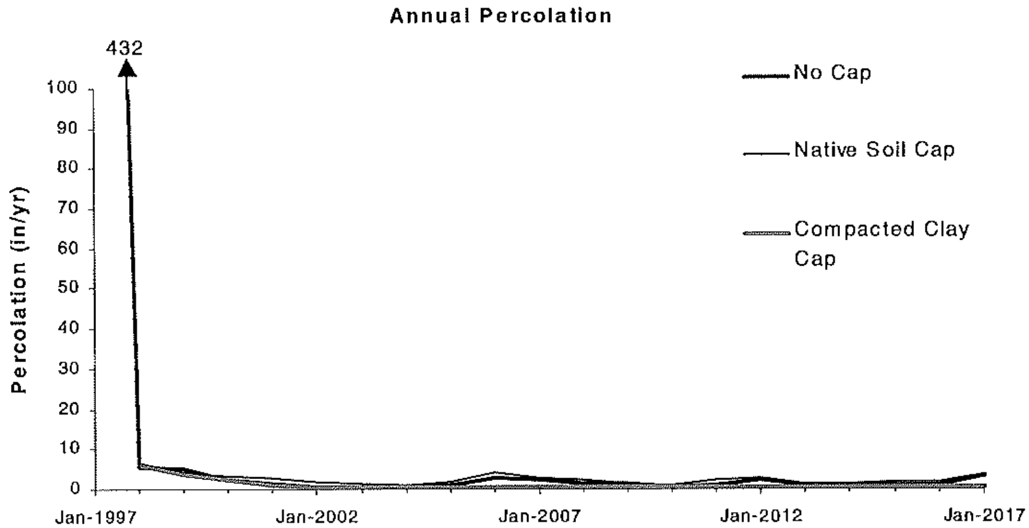


Figure 4-6
Predicted percolation rates at HNW

MODFLOW/MT3D Results

The percolation rates generated during the HELP simulations were input to a flow and transport model to predict the effect that reducing leachate percolation rates would have on groundwater quality—specifically boron concentrations. The site was modeled in three-dimensions to simulate horizontal and vertical variations in flow and transport. Prior to predictive simulations, the flow and transport models were calibrated to produce head and concentration distributions that matched values observed while the impoundment was in service. The maximum HELP-predicted percolation rates (calculated during the first days of the dewatering simulation) were input as leachate flux values for the active impoundment, and other model variables such as recharge and hydraulic conductivity were originally estimated from field measurements and refined during calibration. Calibration of flow and transport at HNW was more difficult than at the other two sites, although the results were adequate for the data available at the time the modeling was performed. Predicted groundwater elevations for most monitoring wells surrounding the site were within 0.6 feet (0.2 m) of their respective target elevations, although predicted elevation at one well (MW-22) was 4.0 feet (1.2 m) lower. Calibrated boron concentrations were highest in wells with elevated concentrations and lowest in wells with background concentrations, but were not all within the range of variability observed while the impoundment was in service.

Initial boron concentrations in the leachate were assumed to remain constant while leachate percolation rate decreased. The model results suggested that boron concentrations would decrease to levels lower than groundwater quality standards within five years after removing the impoundment from service (Figure 4-7), with little difference between the three closure scenarios. It was assumed that sulfate would meet its standard more quickly than boron because it is slightly more mobile than boron and because its concentration exceeded the standard by less than a factor of 1.5, while boron concentrations exceeded the standard by as much as a factor of 5. The groundwater model did not account for potential effects of saturated ash, because its extent at the time of modeling was thought to be confined to a small area beneath the western half of Pond 3 where it was underlain by the silty-clay confining unit.

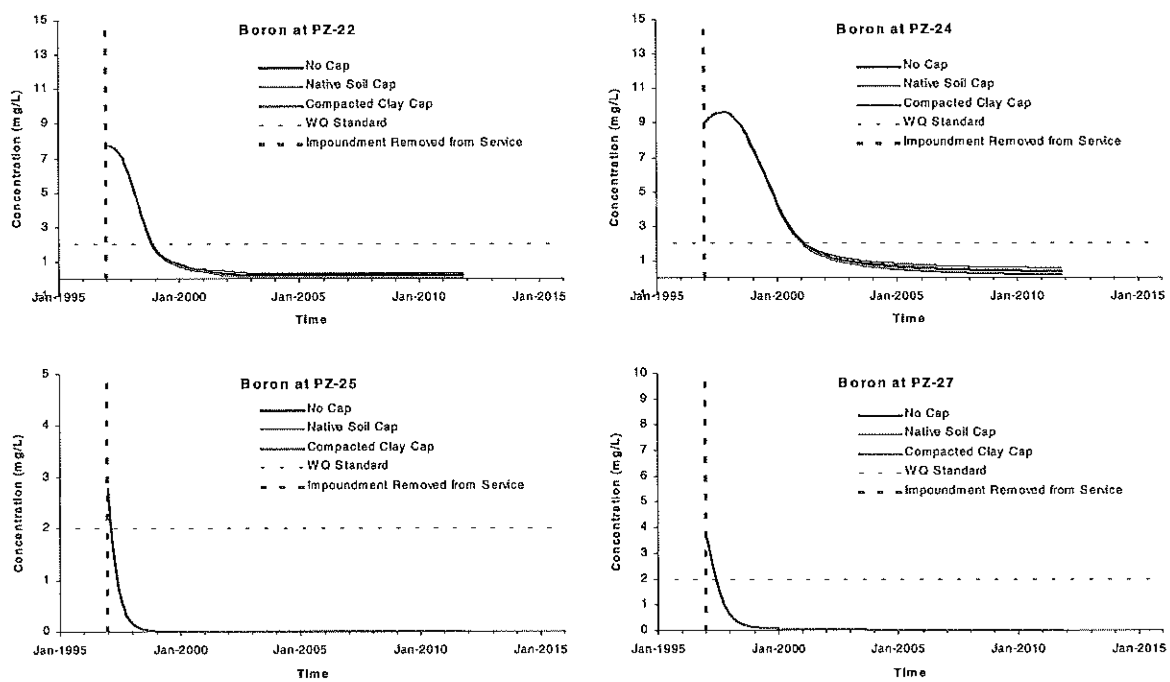


Figure 4-7
Predicted boron and sulfate concentrations at selected HNW monitoring wells

Groundwater Quality Trends Since Closure

Median concentrations of most constituents have increased during the four years since this impoundment was removed from service (Table 4-5), and boron concentrations have not followed the downward trends predicted by the modeling (Figure 4-8). Ash indicator parameter (boron and sulfate) concentrations have decreased at wells near the fringe of the plume (PZ-25, MW-26, PZ-30, and MW-31) because the groundwater mound has partially dissipated and there is less head to drive the plume in the direction of these wells (Figures 4-9 and 4-10). However, the concentrations of ash indicator parameters at wells near the impoundment (PZ-21, PZ-22, PZ-23, PZ-24) have increased because boron and sulfate continue to leach from saturated ash beneath the impoundment and groundwater contact time is increasing due to dissipation of the mound and resulting hydraulic gradient reduction.

Table 4-5
Comparison of Downgradient Groundwater Quality Before and Four Years After Removing HNW From Service

		1995-96 (preclosure)			2000 (post closure)			% change
Analyte	Unit	min	median	max	min	median	max	medians
Alkalinity	mg/L	28	200	390	38	205	470	3%
Boron	mg/L	0.12	4.5	10	0.065	6.1	10	36%
Calcium	mg/L	56	100	200	51	110	170	10%
Chloride	mg/L	14	49	130	12	66	83	34%
Hardness	mg/L	274	410	800	210	420	770	2%
Iron	mg/L	<0.050	<0.005	6.6	0.025	0.35	5.4	---
Magnesium	mg/L	<5.0	38	96	4.2	35	99	-9%
Manganese	mg/L	<0.005	<0.03	1.4	0.007	0.12	0.96	---
pH	pH	6.6	7.6	9.5	7.1	7.4	8.9	-3%
Potassium	mg/L	<0.50	3.1	32	0.61	2.7	23	-13%
Sodium	mg/L	9.8	24	78	17	39	63	63%
Specific Conductance	umhos/cm	275	869	2,070	486	984	1,493	13%
Sulfate	mg/L	34	180	600	33	285	590	58%
Total Dissolved Solids	mg/L	290	650	1,300	280	715	1,200	10%

Notes:

Comparison based on wells sampled in both 1995-96 and 2000: (PZ-21, PZ-22, PZ-23, PZ-24, PZ-25, MW-26, PZ-27, PZ-30, MW-31)

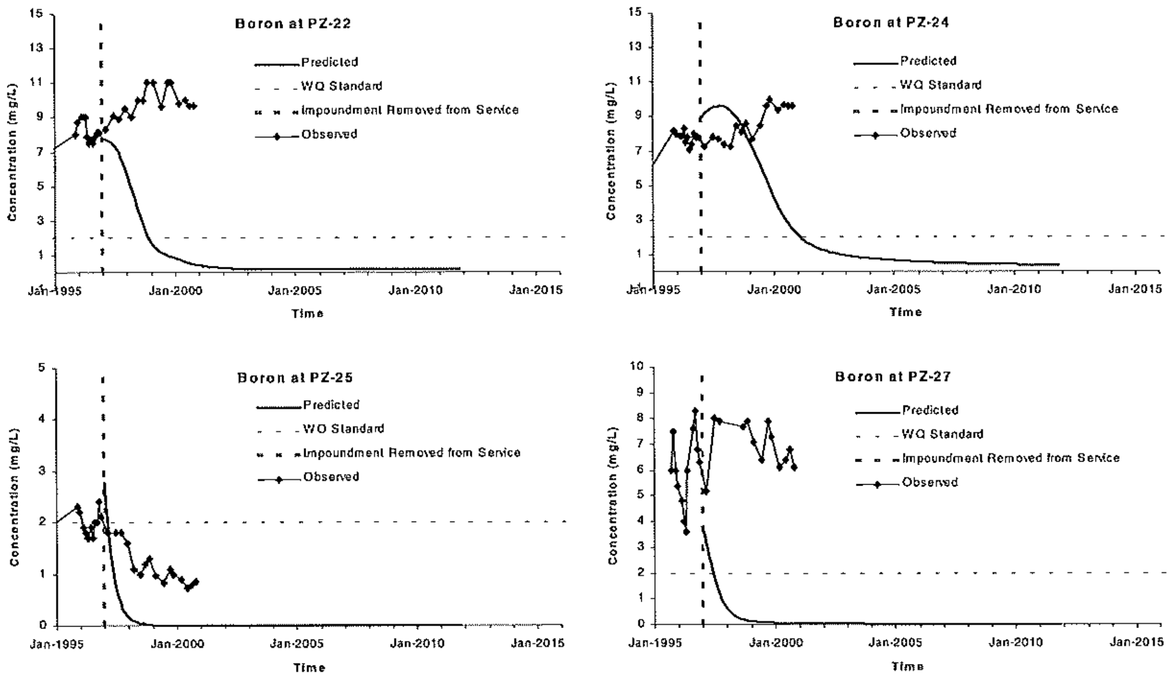


Figure 4-8
Comparison of model-predicted and observed boron concentrations at HNW

Site Summary

The HNW impoundment was closed with no cap, under the assumption that ash would be mined in the future. The model predicted that concentrations of ash indicator parameters would decrease to levels below state standards within five years. After four years, concentrations in wells near the impoundment have not decreased, due to continued leaching from ash that was filled below the water table. The model did not account for leaching from saturated ash, the full extent of which was discovered subsequent to the modeling and closure.

The observation that there is continued mounding beneath this facility suggests that there is more groundwater recharge occurring beneath the impoundment than outside the impoundment, as might be expected since no actions were taken to prevent storm water runoff from collecting in low areas on the impoundment where it can then infiltrate through the ash. A cap of any type would facilitate storm water runoff from the impoundment; however, it is unlikely that such an action would result in a reduction in groundwater elevation by more than a foot, meaning that ash beneath much of Pond 3 would continue to remain saturated, regardless of cap type.

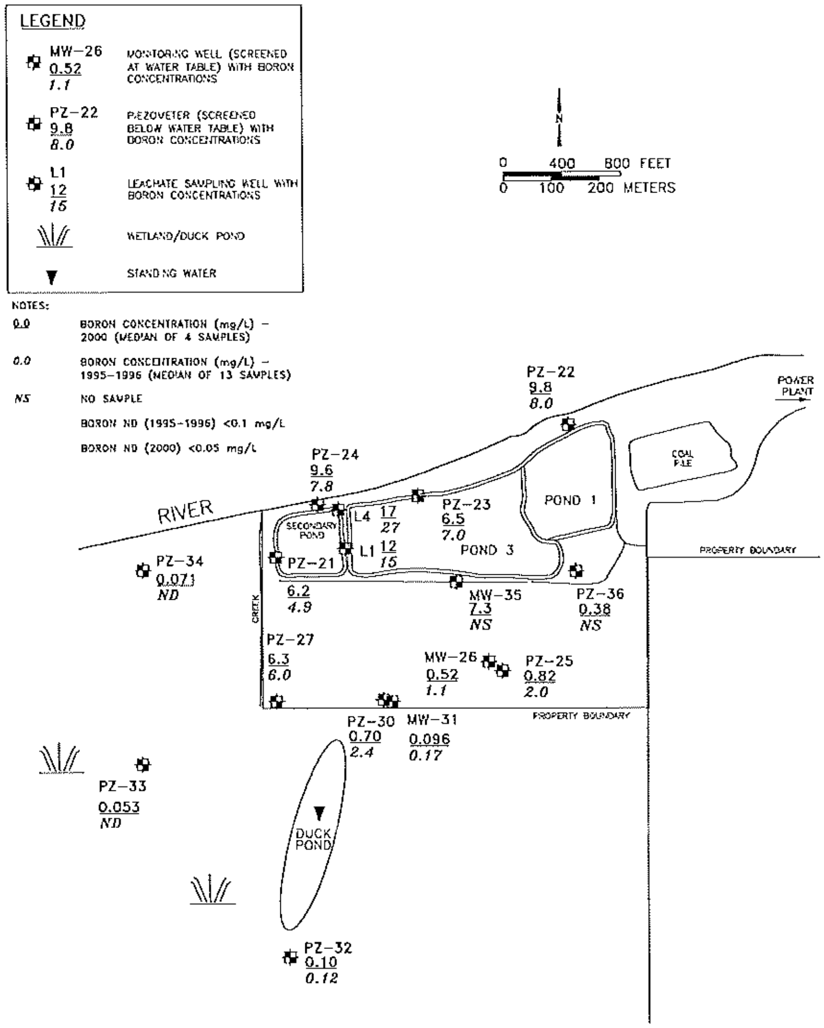


Figure 4-9
Comparison of median boron concentrations in 1995-96 (preclosure) and 2000 (four years after closure)

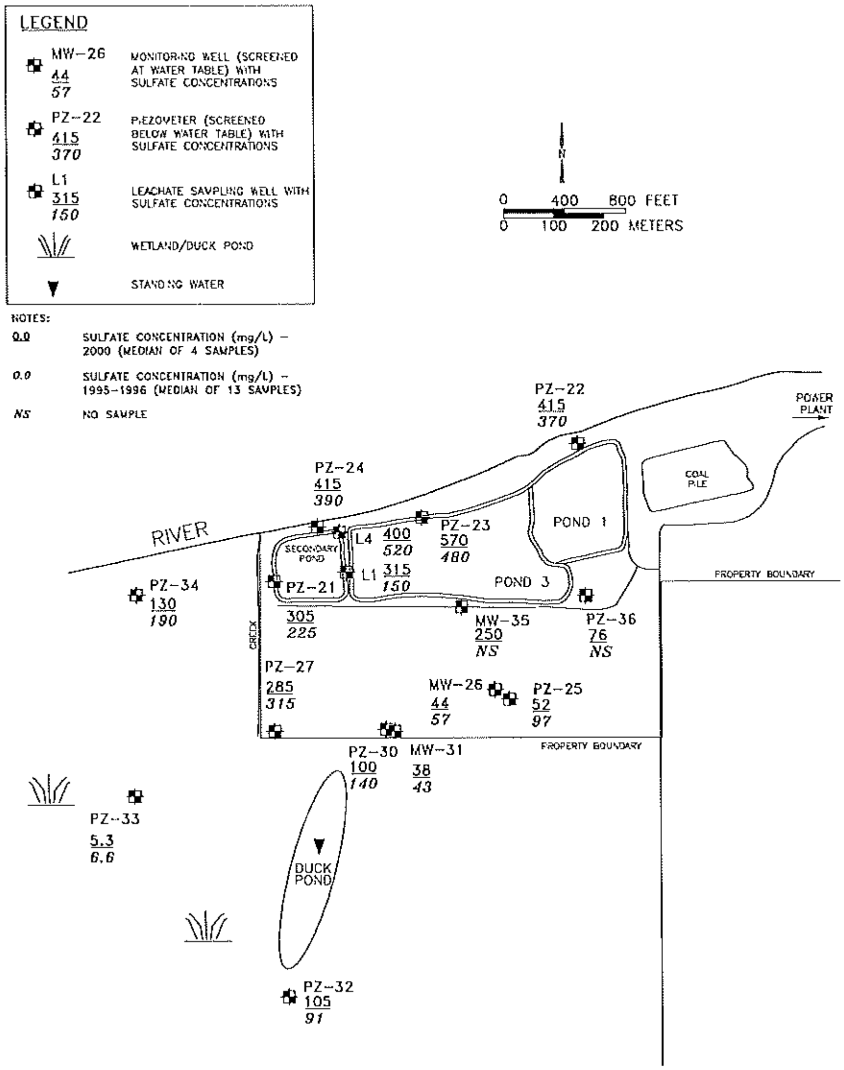


Figure 4-10
Comparison of median sulfate concentrations in 1995-96 (preclosure) and 2000 (four years after closure)

5
DISCUSSION AND CONCLUSIONS

Groundwater Quality Trends

The results of this investigation demonstrated that dewatering at two ash impoundments where all ash is situated above the water table resulted in significantly improved groundwater quality. The median boron concentration for the intermediate and downgradient wells at HA and HNE decreased by about 90 percent and median sulfate concentrations decreased by 56 percent after dewatering and closing the impoundments. Sulfate concentrations decreased by a smaller percentage than boron concentrations, because sulfate concentrations are approaching background levels. In addition, concentrations of all other analytes with downgradient concentrations higher than background, including chloride, potassium, and sodium, decreased significantly. Conversely, median boron and sulfate concentrations at HNW, as well as other analytes such as chloride and sodium, increased after that impoundment was closed. The HNW site contains ash below the water table.

All three impoundments are unlined and directly overlie aquifers. HA was covered with a native sandy soil cap one year after dewatering, while the HN impoundments were dewatered but were not capped. The similar decreases at HA and HNE suggest that the cap has little bearing on overall closure performance. The key factor for achieving concentration reduction at these two facilities was dewatering.

HNW differed from the HA and HNE impoundments because lower portions of the ash were filled below the water table. When ash in an impoundment is above the water table, dewatering of the ash greatly reduces the mass loading rate from the source. In the cases of HA and HNE, model results indicated that loading rates decreased by more than 95 percent. However, when ash remains below the water table, dewatering may be less effective because groundwater continues to leach constituents from the saturated ash, particularly if the impoundment is underlain by geologic media with relatively high rates of groundwater flow. In the case of HNW, concentrations increased because groundwater contact time with the saturated ash increased when the hydraulic gradient of the pond was removed.

Modeling suggested that mass flux from HNW was greatest beneath the eastern half of the impoundment, which is underlain by coarse-grained materials with relatively high hydraulic conductivity values. The fined grained materials underlying the western portion of the impoundment have relatively low hydraulic conductivity, which limits mass flux through, and leaching from, the saturated ash underlying that portion of the impoundment.

Use of Groundwater Modeling to Predict Closure Effectiveness

The ash indicator parameter at all three impoundments with the greatest state standard exceedance rate, whether in terms of the number of wells affected, the frequency of exceedances in a given well, or the relative magnitude by which the standard was exceeded, was boron. Therefore, boron was used as an indicator parameter in model analyses for all three impoundments.

The model simulations predicted that downgradient boron concentrations would decrease to levels below the state standard within four years of closure at HA, six years at HNE, and five years at HNW. Postclosure monitoring showed that boron concentrations followed predicted concentrations very closely at HA and HNE. Observed boron concentrations were lower than standards within two years at HA, while HNE is in the fourth year of postclosure monitoring and concentrations have decreased significantly. At HNW, observed concentrations did not follow predicted trends because the HNW model did not account for the saturated ash source, which was unknown at the time. In addition to successfully predicting the time frame in which boron concentrations at HA would meet standards, the models also successfully predicted concentration trends at HA and HNE.

The general procedure employed at these impoundments can be applied at other impoundments to evaluate alternative closure scenarios:

1. Conduct a thorough hydrogeologic investigation that identifies current aquifer conditions, including hydraulic conductivity testing, and delineates the current extent of ash indicator parameters in groundwater. If possible, the impoundment should be tested to determine leachate concentration, depth of ash (to determine whether there may be saturated ash after dewatering), and physical characteristics of the ash for input to an infiltration model such as HELP.
2. Perform a water balance on the impoundment while in service (annual inflows minus annual discharge). These data can be used to calibrate the infiltration model.
3. Perform infiltration modeling to determine leachate percolation rates from the impoundment while in service and after closure. The in-service impoundment is simulated by setting depth of water at the surface to a value roughly equal to impoundment water depth, and setting initial moisture content equal to porosity. If HELP is used, it can only simulate the last day of active impoundment operation, because it does not include a mechanism to maintain the initial surface water depth. Use the infiltration model to simulate several different cap alternatives, and record percolation from the lowest layer representing the base of the impoundment.
4. Using the initial percolation values from the infiltration model as recharge values for the impoundment, create and calibrate a preclosure groundwater flow and transport model. Then, enter the decreasing percolation rates for tested closure scenarios as decreasing impoundment recharge rates to test the effects that the closure scenarios have on downgradient groundwater quality.

Effectiveness of Alternatives to Compacted Clay or Synthetic Caps as Impoundment Closure Methods

The postclosure data collected at these former ash impoundments demonstrate that dewatering, with or without a cap, can effectively reduce downgradient concentrations of ash constituents if ash is not placed below the water table. Once the impoundment is dewatered, the mass of dissolved constituents percolating from the impoundment is greatly reduced, resulting in reduced concentrations in downgradient groundwater.

The model predictions at these sites suggested that a cap, whether compacted clay or native soil, provides little additional benefit for downgradient concentration reduction beyond that achieved by dewatering. The similarity of overall concentration reductions at HA, which was capped with native soils, and at HNE, which was not capped at all, supports the conclusions from the modeling.

A properly graded native soil cap can provide benefit by promoting storm water runoff from the facility. Without a cap, storm water collects in low areas on the ash surface and infiltrates. This situation was evident at HNE, where storm water infiltrating upgradient of MW-04 and MW-05 caused local concentrations to recover more slowly than predicted by the modeling, which assumed that runoff would not be allowed to pond and infiltrate. At HA, which was capped to facilitate runoff, concentrations decreased faster than predicted by the model.

These results demonstrate that engineered, compacted clay caps are not always necessary when closing coal ash impoundments. Factors to consider when determining the cap design include:

- The concentration of ash indicator constituents in leachate; leachate from sluiced ash will have lower indicator concentrations than leachate from fresh ash due to mass removal during sluicing.
- Impoundment liner design; lined impoundments require a cap with permeability at least as low as the liner in order to avoid the "bathtub" effect, where the closed impoundment fills with water because water infiltrates through the cap more readily than it exfiltrates through the base.
- Hydrogeology; the HA and HNE impoundments overlay thick, highly permeable aquifers and the low volume of leachate migrating from these impoundments after closure is mixed in the much higher volume of groundwater underflowing the impoundment. However, if these impoundments were located over aquifers with very limited areal extent or little thickness, or if they were located over low permeability formations, then the downgradient concentration reductions would not have been as large as observed.
- Comanagement of mill rejects; if mill rejects containing pyrites are comanaged in the impoundment, then dewatering can allow oxidation of the pyrites, resulting in pH reductions and corresponding release of sulfate and certain metals (EPRI, 1995). In these situations, a cap that limits infiltration of water and oxygen can reduce the potential for pyrite oxidation.
- Saturated ash; as shown at HNW, the existence of saturated ash will greatly reduce the effectiveness of any cap design when the facility is underlain by geologic materials with high hydraulic conductivity, because groundwater will continue to leach ash constituents.

6 REFERENCES

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A
MODEL INPUT DATA

HA Impoundment

Table A-1
HA—FOWL-GH Input Parameters

Parameter	Sensitivity Range	Prediction Value(s)
Nearest Default Climate Station	St. Louis, MO	St. Louis, MO
Latitude	Site specific	Site specific
Leaf Area Index	1, 2, 3.3	see Fig A-1
Growing Season	April 15 - Oct 20	April 15 - Oct 20
Evaporative Zone Depth	14 to 28 inches	see Fig A-1
Average Monthly Temperatures	Based on local climatological record, 1901-1962.	Based on local climatological record, 1901-1962.
Daily Rainfall Data	Synthetically generated based on local monthly averages, 1871-1960.	Synthetically generated based on local monthly averages, 1871-1960.
Site Surface Area (ft^2)	100,000 (see note below)	100,000 (see note below)
Active Facility (Y/N)	N – assumes modeling begins on day of closure.	N – assumes modeling begins on day of closure.
Open site runoff fraction	Not used for inactive facility.	Not used for inactive facility.
Number of Soil Layers	1 to 2	See Fig A-1
Soil Layer Type	Percolation	Percolation
Layer Thickness (in.)	24-60 inches (cover), 420 inches (ash)	See Fig A-1
Vegetative Cover Type	Poor	See Fig A-1
(SCS) Runoff Curve Number	FOWL-GH calculated based on soil and vegetation.	FOWL-GH calculated based on soil and vegetation.
Initial Soil Water Content	= Porosity	Based on results of calibration runs
Wilting Point (vol/vol)	0.024-0.280	See Fig A-1
Field Capacity (vol/vol)	0.062-0.378	See Fig A-1
Porosity (vol/vol)	0.430-0.501	See Fig A-1
Hydraulic Conductivity (cm/s)	Ash: 1.0E-3 to 1.7E-5 Cover: 1.0E-2 to 1.8E-7	See Fig A-1

Notes:
Actual surface area is about 770,000 ft^2, maximum value accepted by model is 100,000 ft^2; however this parameter did not affect model calculations, which were per unit area.

Model Input Data

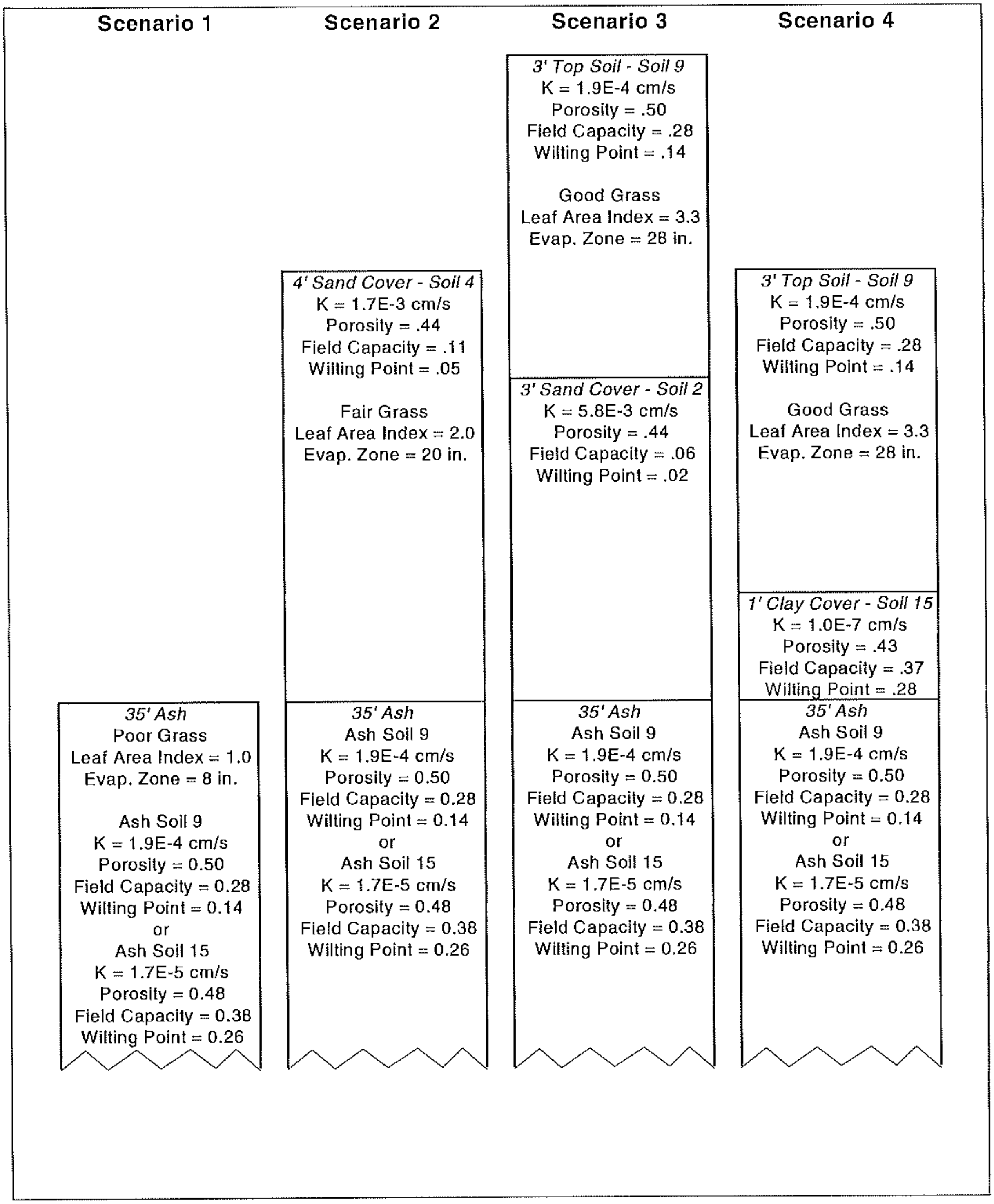


Figure A-1
HA Cap Scenarios

Table A-2
HA—PCTRANS General Input Parameters

Soil Parameters	**Dune Sands**	**Outwash**
Hydraulic Conductivity (K_x)	17 m/d	78 m/d
Hydraulic Conductivity (K_z)	1.7 m/d	7.8 m/d
Storage (S)	0.30	.30
Effective Porosity (n_e)	0.33	0.33
Longitudinal Dispersivity (α_l)	22.9 m	22.9 m
Vertical Dispersivity (α_z)	.0025 m	.0025 m
Bulk Density (ρ_B)	1.6 kg/m^3	1.6 kg/m^3
Solute Parameters	**Sulfate**	**Boron**
Distribution Coefficient (Kd)	0	0.398
Initial Concentration	250 mg/L	2.5 to 3 mg/L
Background Concentration	22 mg/L	0
Boundary Conditions	**Calibration**	**Prediction**
Constant Head – Backwater Pond	134.7 m	134.7 m
Constant Flux - Areal Recharge	7.2×10^{-4} m/d/m^2	7.2×10^{-4} m/d/m^2
Constant Flux – Flow From East	2.0×10^{-1} m/d/m^2	2.0×10^{-1} m/d/m^2
Constant Flux - Main Pond (leachate percolation)	8.9×10^{-3} m/d/m^2	variable - see Table A-3
Model Parameters	**Calibration**	**Prediction**
Number of Time Steps	1	120 – 228
Time Step Length	Steady State	3 - 30 days

Table A-3
HA—PCTRANS Impoundment Percolation Input Parameters

Time (days)	Sand Cover no topsoil	Sand Cover with topsoil	Clay Cover with topsoil
0	8.9E-03	8.9E-03	8.9E-03
30	3.7E-03	3.7E-03	3.7E-03
60	2.6E-03	2.6E-03	2.6E-03
90	1.8E-03	1.8E-03	1.8E-03
120	1.4E-03	1.4E-03	1.4E-03
150	1.1E-03	1.1E-03	1.1E-03
180	9.9E-04	9.9E-04	9.9E-04
210	8.5E-04	8.5E-04	8.5E-04
240	7.2E-04	7.2E-04	7.2E-04
270	6.7E-04	6.7E-04	6.7E-04
300	5.8E-04	5.8E-04	5.8E-04
330	5.5E-04	5.5E-04	5.5E-04
360	4.9E-04	4.9E-04	4.9E-04
720	4.0E-04	4.0E-04	3.7E-04
1080	4.0E-04	3.1E-04	2.4E-04
1440	3.5E-04	2.3E-04	1.8E-04
1800	2.5E-04	1.7E-04	1.6E-04
2160	4.2E-04	2.1E-04	1.4E-04
2520	5.9E-04	2.3E-04	1.2E-04
2880	5.4E-04	2.8E-04	1.2E-04
3240	3.8E-04	2.1E-04	1.1E-04
3600	5.5E-04	2.5E-04	1.0E-04
3960	4.0E-04	2.2E-04	1.0E-04
4320	4.3E-04	2.3E-04	9.9E-05
4680	3.8E-04	2.1E-04	9.6E-05
5040	7.6E-04	3.4E-04	9.4E-05
5400	7.3E-04	4.3E-04	9.3E-05

Notes:
All values in $m/d/m^2$

HNE Impoundment

Table A-4
HNE—HELP Input Parameters

Parameter	Sensitivity Analysis Value(s)	Prediction Value(s)
Nearest Default Climate Station	not tested	Chicago, IL
Latitude	not tested	Site specific
Leaf Area Index	0, 0.5, 1 and 2	0 (scenario A, only) and 1
Growing Season	not tested	April 15 - Oct 20
Evaporative Zone Depth (in.)	8 to 24	4 (Scenario C, only), 8 (Scenario A, only) and 20
Average Monthly Temperatures	not tested	Based on local climatological record
Daily Rainfall Data	not tested	Synthetically generated based on local monthly averages
Site Surface Area (ft^2)	not tested	1,393,920
Active Facility (Y/N)	not tested	No
Vegetative Cover Type	Bare to fair	Bare (ash, only) and poor
Surface Slope (%)	0.51, 1, 2 and 3	0.51 (all but topsoil) and 1
(SCS) Runoff Curve Number	HELP calculated	HELP calculated
Ash Layer Parameters	**Sensitivity Analysis**	**Prediction (all scenarios)**
Default Soil Type	not tested	30
Layer Thickness (in.)	1 layer @ 360" and 10 layers @ 36"	10 layers @ 36"
Porosity (vol/vol)	0.40, 0.48 and 0.541	0.541
Hydraulic Conductivity (cm/s)	1.0E-4, 7.0E-5 and 1.0E-5	7.0E-5
Field Capacity (vol/vol)	0.187, 0.25 and 0.35	0.187
Wilting Point (vol/vol)	0.047, 0.15 and 0.25	0.047
Initial Moisture Content	0.40 and 0.48 (no cap scenario)	Predicted value at end of 1997
Black Earth Parameters	**Sensitivity Analysis**	**Prediction (Scenarios C and D)**
Default Soil Texture No.	not tested	6
Layer Thickness (in.)	not tested	4
Porosity (vol/vol)	0.3, 0.35 and 0.4	0.3
Hydraulic Conductivity (cm/s)	3.0E-5, 1.0E-4 to 1.0E-3	3.0E-5
Field Capacity (vol/vol)	not tested	0.19
Wilting Point (vol/vol)	not tested	0.085
Initial Moisture Content	0.21, 0.245 and 0.28	0.21
Silty-Clay Earth Parameters	**Sensitivity Analysis**	**Prediction (Scenario B)**
Default Soil Texture No.	9 (silt), 12 and 14 (clays)	12
Layer Thickness (in.)	not tested	24
Porosity (vol/vol)	0.501, 0.471 and 0.479, for #9, #12 and #14, respectively	0.471

Hydraulic Conductivity (cm/s)	1.9E-4, 4.2E-5 and 2.5E-5, respectively	4.2E-5
Field Capacity (vol/vol)	0.284, 0.342 and 0.371, respectively	0.342
Wilting Point (vol/vol)	0.135, 0.21 and 0.251, respectively	0.21
Initial Moisture Content	0.284, 0.342 and 0.371, respectively	0.342
Clay Liner Parameters	**Sensitivity Analysis**	**Prediction (Scenario D)**
Default Soil Texture No.	not tested	29
Layer Thickness (in.)	12, 24 and 36	24
Porosity (vol/vol)	0.31, 0.35 and 0.4	0.31
Hydraulic Conductivity (cm/s)	1.0E-7, 5.0E-6 and 1.0E-5	5.0E-6
Field Capacity (vol/vol)	0.265, 0.299 and 0.342	0.265
Wilting Point (vol/vol)	0.189, 0.213 and 0.244	0.189
Initial Moisture Content	0.217, 0.245 and 0.28	0.217
Gravel Layer Parameters	**Sensitivity Analysis**	**Prediction (Scenarios B, C, D)**
Default Soil Texture No.	not tested	21
Layer Thickness (in.)	Not tested	12-24
Porosity (vol/vol)	Not tested	0.397
Hydraulic Conductivity (cm/s)	Not tested	3.0E-1
Field Capacity (vol/vol)	Not tested	0.0.32
Wilting Point (vol/vol)	Not tested	0.013
Initial Moisture Content	Not tested	0.013

Table A-5
HNE—HELP Prediction Scenarios

Cap Scenario	A - No Cap	B - Silty Earth	C - Top Soil	D - Compacted Clay
Layer 1 (top)		2 ft - silty clay earth	4 in. - black earth	4 in. - black earth
Layer 2		1 ft – gravel	2 ft - gravel	1, 2, or 3 ft - clay
Layer 3				1 ft - gravel
Slope	0.50%	0.50%	1, 2, or 3 %	0.50%
Notes	Assumes measures taken to enable runoff		Insensitive to slope – 1% modeled	Not sensitive to clay thickness > 2 ft – 2 ft modeled

Table A-6
HNE—MODFLOW/MT3D Input Parameters

Parameter	Symbol	Value	Unit	Notes
Horizontal Hydraulic Conductivity	Kx	480	ft/d	(1.7E-01 cm/s) Average value measured for sand and gravel aquifer
		24	ft/d	(8.5E-03 cm/s) Upgradient portion of aquifer, based on calibration
		0.37	ft/d	(1.3E-04 cm/s) Alluvium, based on slug test at well 29
Vertical Hydraulic Conductivity	Kz	=Kx	ft/d	
Anisotropy ratio	Kx:Ky	1		
Effective Porosity	n_e	0.2		Sand & gravel, based on Mercer & Waddell (1993)
		0.1		Alluvium, based on Mercer & Waddell (1993)
Specific Storage	Ss	1.00E-05	1/ft	Sand and gravel aquifer, based on Smith & Wheatcraft (1993)
		1.00E-04	1/ft	Upgradient sand aquifer, based on Smith & Wheatcraft (1993)
		1.00E-03	1/ft	Alluvium, based on Smith & Wheatcraft (1993)
Constant Flux	CF	0.071	$ft^3/d/ft^2$	At upgradient boundary
Recharge (ambient)	R	0.00228	ft/d	(10 in/yr) Based on calibration
Recharge (closed quarries)		0.0046	ft/d	(20.2 in/yr) Based on HELP simulation
Recharge (active ash pond)		0.198	ft/d	Average – based on water balance calculation
Recharge (closed ash pond)		varies	ft/d	See Figure A-2
Recharge (new lined pond)		0.0010	ft/d	From 0 to 2.5 years
		0.0017	ft/d	From 2.5 to 5.0 years
		0.0024	ft/d	After 5.0 years
River Conductance	Criv	0.074	$ft^2/d/ft^2$	
River Head	Hriv	varies	ft	441.45 (upstream) to 440.7 (downstream)
Source Boron Concentration	Co	22	mg/L	Highest observed boron concentration
Background Boron Concentration	Cbkg	0	mg/L	
Dispersivity (longitudinal)	α_l	30	ft	
Dispersivity (transverse)	α_t	3.75	ft	
Dispersivity (vertical)	α_z	0.03	ft	
Diffusion Coefficient	D	0	ft^2/d	
Retardation	R	1.5		

Model Input Data

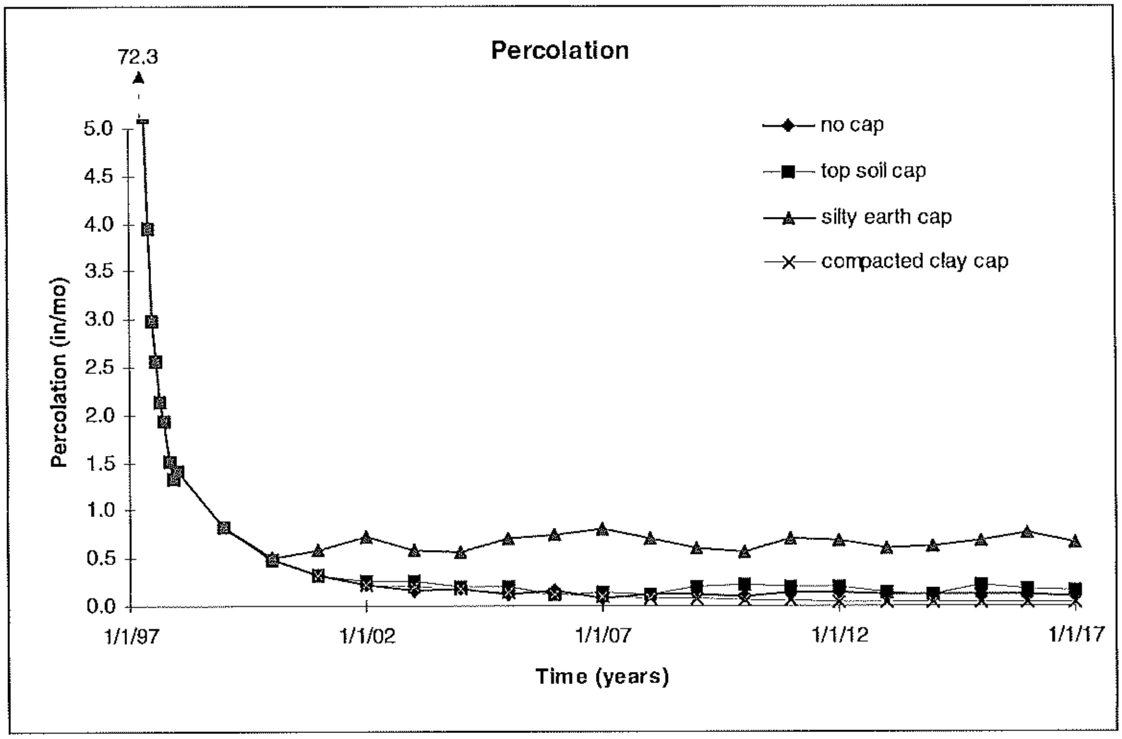


Figure A-2
HNE—HELP Predicted Percolation Results

HNW Impoundment

Table A-7
HNW—HELP Input Parameters

Parameter	Sensitivity Analysis Value(s)	Prediction Value(s)
Nearest Default Climate Station	Not tested	Chicago, IL
Latitude	Not tested	Site specific
Leaf Area Index	0, 1, and 2	1
Growing Season	Not tested	April 15 – Oct 20
Evaporative Zone Depth (in.)	14, 20, and 28	20
Average Monthly Temperatures	Not tested	Based on local climatological record
Daily Rainfall Data	Not tested	Synthetically generated based on local monthly averages
Site Surface Area (acres)	Not tested	15
Vegetative Cover Type	Poor and Fair	Poor
Surface Slope (%)	0.51 and 1	0.51
Slope Length (ft)	Not tested	200
(SCS) Runoff Curve Number	HELP calculated	HELP calculated
Ash Layer Parameters	**Sensitivity Analysis**	**Prediction (all scenarios)**
Default Soil Type	Not tested	30
Layer Thickness (in.)	Not tested	3 layers @ 60"
Porosity (vol/vol)	0.40 and 0.541	0.541
Hydraulic Conductivity (cm/s)	1.0E-4 and 1.0E-5	1.0E-4
Field Capacity (vol/vol)	0.12, 0.187, and 0.25	0.187
Wilting Point (vol/vol)	0.02. 0.047, and 0.10	0.047
Initial Moisture Content	Not tested	Predicted value at end of 1997
Black Earth Parameters	**Sensitivity Analysis**	**Prediction (Scenario C)**
Default Soil Texture No.	Not tested	6
Layer Thickness (in.)	Not tested	4
Porosity (vol/vol)	0.25, 0.30, and 0.35	0.3
Hydraulic Conductivity (cm/s)	1.0E-5, 3.0E-5, and 1.0E-4	3.0E-5
Field Capacity (vol/vol)	Not tested	0.19
Wilting Point (vol/vol)	Not tested	0.085
Initial Moisture Content	Not tested	0.21
Silty-Clay Earth Parameters	**Sensitivity Analysis**	**Prediction (Scenario B)**
Default Soil Texture No.	9 (silt), 12 and 14 (clays)	12
Layer Thickness (in.)	Not tested	24
Porosity (vol/vol)	0.501, 0.471 and 0.479, for #9, #12 and #14, respectively	0.471
Hydraulic Conductivity (cm/s)	1.9E-4, 4.2E-5 and 2.5E-5, respectively	4.2E-5

Field Capacity (vol/vol)	0.284, 0.342 and 0.371, respectively	0.342
Wilting Point (vol/vol)	0.135, 0.21 and 0.251, respectively	0.21
Initial Moisture Content	0.284, 0.342 and 0.371, respectively	0.342
Clay Liner Parameters	**Sensitivity Analysis**	**Prediction (Scenario C)**
Default Soil Texture No.	Not tested	29
Layer Thickness (in.)	12, 24 and 36	24
Porosity (vol/vol)	0.25, 0.31, 0.35	0.31
Hydraulic Conductivity (cm/s)	1.0E-7 and 1.0E-6	1.0E-7
Field Capacity (vol/vol)	0.15, 0.20, 0.25	0.20
Wilting Point (vol/vol)	0.02 and 0.10	0.02
Initial Moisture Content	Not tested	0.217
Gravel Layer Parameters	**Sensitivity Analysis**	**Prediction (Scenarios B & C)**
Default Soil Texture No.	Not tested	21
Layer Thickness (in.)	Not tested	12 (B), 24 (C)
Porosity (vol/vol)	Not tested	0.397
Hydraulic Conductivity (cm/s)	Not tested	3.0E-1
Field Capacity (vol/vol)	Not tested	0.32
Wilting Point (vol/vol)	Not tested	0.013
Initial Moisture Content	Not tested	0.013

Table A-8
HNW—HELP Prediction Scenarios

Cap Scenario	A - No Cap	B - Silty Earth	C – Compacted Clay
Layer 1 (top)		2 ft - silty clay earth	4 in. - black earth
Layer 2		1 ft - gravel	2 ft - clay
Layer 3			2 ft - gravel
Slope	0.50%	0.50%	0.50%
Notes	Assumes measures taken to enable runoff		Not sensitive to clay thickness > 2 ft – 2 ft modeled

Table A-9
HNW—MODFLOW/MT3D Input Parameters

Parameter	Value	Unit	Notes
Hydraulic Conductivity	varies	ft/d	see Figure A-5
Effective Porosity	0.2		Sand & gravel, based on Mercer & Waddell (1993)
	0.1		Silt, based on Mercer & Waddell (1993)
Storage (Specific Storage)	1.00E-05	1/ft	Sand and gravel aquifer, based on Smith & Wheatcraft (1993)
	1.00E-03	1/ft	Silt, based on Smith & Wheatcraft (1993)
Storage (Specific Yield)	0.2		Sand and gravel
	0.1		Silt
Recharge (calibration)	varies	ft/d	see Figure A-4
Recharge (ash pond-prediction)	varies	ft/d	see Figure A-3
River Conductance	0.37	$ft^3/d/ft^2$	This value is multiplied by the area of the cell
River Head	varies	Ft	441.2 (upstream) to 439.86 (downstream)
Large Pond Conductance	0.123	$ft^3/d/ft^2$	This value is multiplied by the area of the cell
Large Pond Head	447	Ft	Maximum possible pond elevation is 448 ft
Small Pond Conductance	0.123	$ft^3/d/ft^2$	This value is multiplied by the area of the cell
Small Pond Head	446.5	ft	
Creek Conductance	446	$ft^3/d/ft^2$	This value is multiplied by the area of the cell
Creek Head	0.37	ft	
Source Boron Concentration	8 to 10	mg/L	Determined during calibration
Background Boron Concentration	0	mg/L	
Dispersivity (longitudinal)	10	ft	
Dispersivity (transverse)	1.25	ft	
Dispersivity (vertical)	0.625	ft	
Diffusion Coefficient	0	ft^2/d	
Retardation	1		

Model Input Data

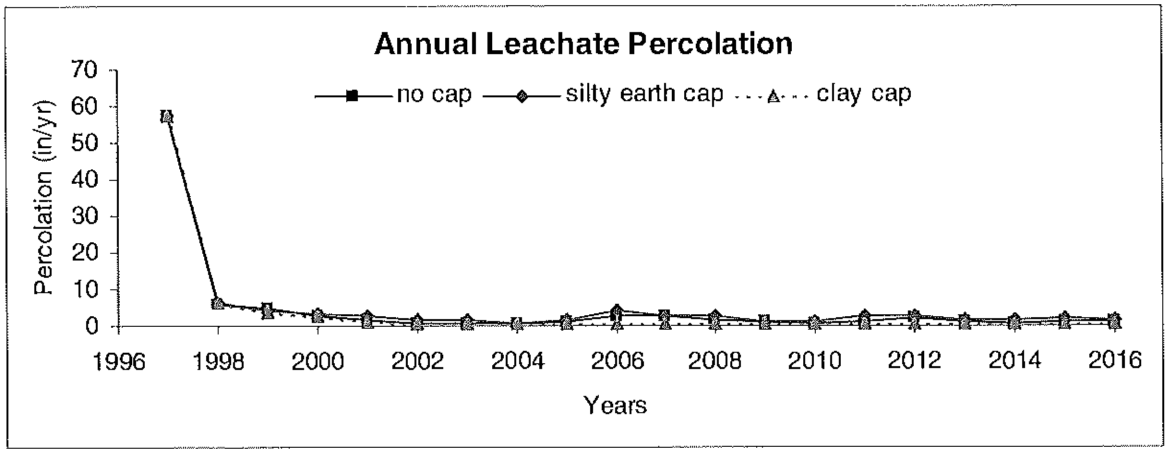


Figure A-3
HNW—HELP predicted percolation rates

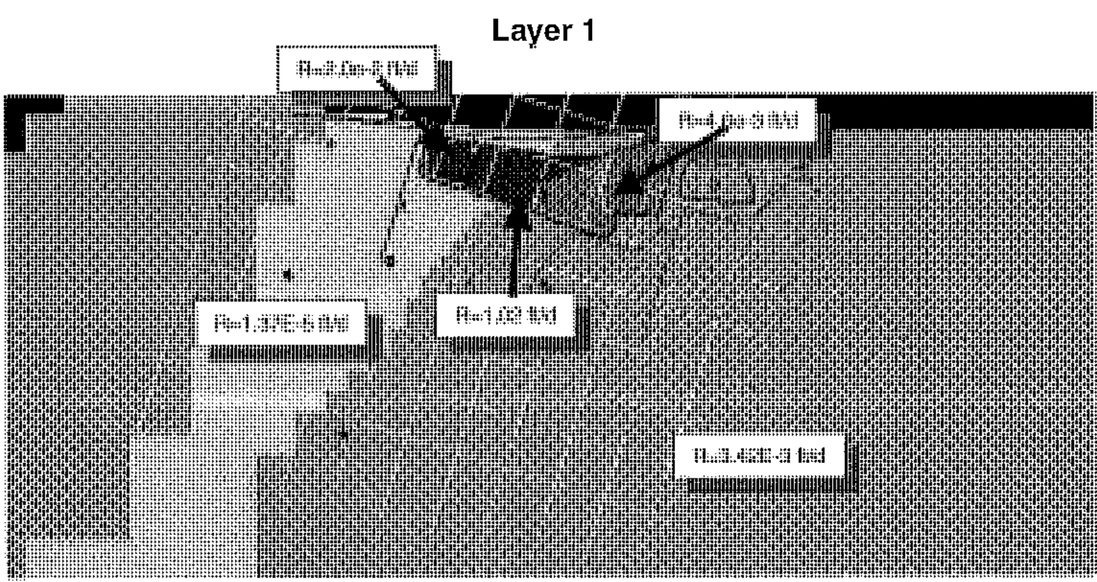
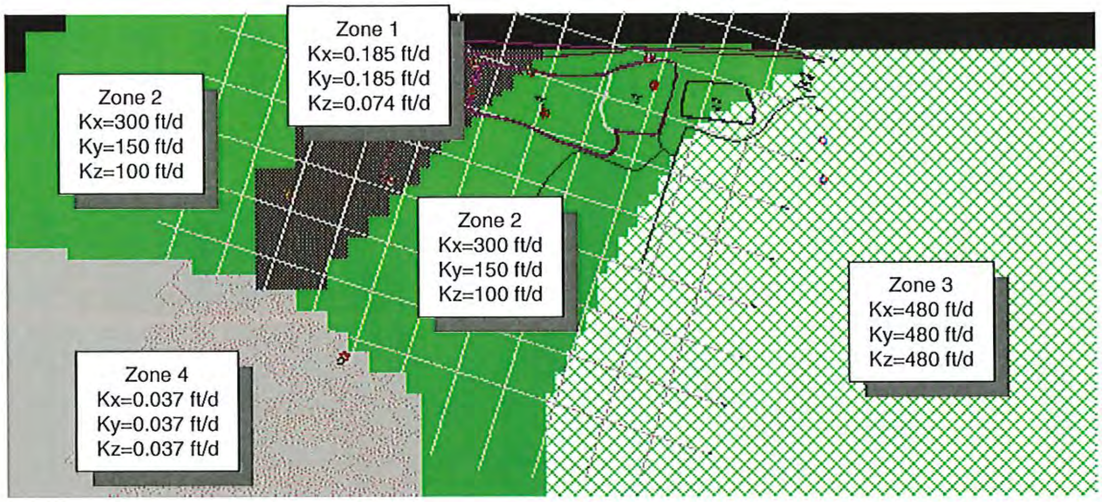


Figure A-4
HNW—Recharge values. Ash pond values are for initial conditions only.

Layer 2



Cross-Section Through Row 15

Figure A-5
HNW—Hydraulic conductivity values

Target:

Groundwater Protection & Combustion By-Products Management

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CERTIFICATE OF SERVICE

I hereby certify that copies of the foregoing *Late-Filed Exhibit No. 10* as filed in Docket No. E-2, Sub 1219, were served via electronic delivery or mailed, first-class, postage prepaid, upon all parties of record.

This, the 16th day of October, 2020.

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